THE BIOLOGICAL MONITORING OF IMPACT AND RECOVERY IN STREAMS FOLLOWING POLLUTION INCIDENTS

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SUMMARY

Monitoring benthic macroinvertebrates in rivers as a means of assessing water quality is in extensivel use. Their use in the assessment of ecosystem response to pollution incidents is less well developed.

The responses of the macroinvertebrates within the substratum to a number of actual pollution incidents was investigated. It was found that recovery following pollution incidents was most adequately represented by changes in the abundance of selected taxa; by a measure of Taxa Richness and by multivariate analysis of community similarity using nonmetric multidimensional scaling. The importance of adequate reference conditions was highlighted.

In order to learn more about the processes involved in the structural changes of the benthic macroinvertebrate community following a transient pollutant, a laboratory study was carried out to investigate the behavioural response of the amphipod, *Gammarus pulex,* to a toxicant. An increase in burrowing behaviour was found, a behaviour which may, under field conditions, result in increased survival following short term pollution. This finding gives an indication of the complexities underlying overt structural changes within the community following a transient pollution.

A novel assessment of recovery within streams: the 'Recovery Index' was developed. This attempts to numerically define and rank the differential colonisation times shown by different taxa. By allocating higher scores to taxa that are slower to recolonise sites, utilisation of the 'Recovery Index' in the assessment of recovery will result in a more sensitive assessment of recovery times following pollution incidents and the reduced likelihood of erroneous assessment of recovery.

The Control of Industrial Major Accident Hazards (CIMAH) regulations have guidelines for environmental survey, against which the effects of an accident could be judged. Recommendations for the development of these guidelines are given, incorporating the use of the 'Recovery Index'.

KEYWORDS: MACROINVERTEBRATES, BIOMONITORING, RECOVERY, STREAMS, POLLUTION INCIDENT

Dedication

This thesis is dedicated to my parents Dr Vasant G. Pisolkar and Elfriede B. Pisolkar.

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Chapter 1

Introduction

1.1 Background to Research Project

In 1989 a three year project investigating the impact of discharges from surface water sewers on receiving water quality was completed at Aston University (Payne, 1989). It was found that the discharges from surface water sewers had a measurable impact on the benthic macroinvertebrate community using the Biological Monitoring Working Party index which categorises organisms according to their sensitivity to organic pollution. The Department of Civil Engineering, in seeking to expand upon the knowledge gained during Paynes' project, instigated further research into the impact of intermittent discharges upon receiving streams through an EPSRC funded Total Technology research studentship.

Negotiations with the sponsoring company: Zeneca Ltd., relating to their particular needs and requirements, led to the development of the present research project which expands upon the use of biomonitoring in the investigation of intermittent pollution. Biological assessment is considered in quantitative terms, and the relative utility of different biological measures in the analysis of impact and recovery is compared. The focus of the study was on pollution incidents as reported to the National Rivers Authority (now the Environment Agency). The information obtained is to be used to refine the prediction of the impact of pollution incidents and to develop procedures for the assessment of recovery following incidents.

Premises within the United Kingdom involving the manufacture or storage of dangerous substances are covered by the Control of Industrial Major Accident Hazards (CIMAH) regulations (HMSO, 1984). The guidelines to the CIMAH regulations (DOE, 1991) do not give detailed information regarding the assessment or monitoring of a potential pollution impact. One of the aims of the present study is to investigate the utility of biological monitoring in assessment of the impact of pollution incidents on streams and to provide further definition to the survey work required under the CIMAH regulations.

1.2 Research Objectives

The research project has the following objectives.

1. To assess the utility of standard macroinvertebrate sampling in the examination of the ecological impact of actual pollution incidents.

2. To establish the relative merits of different measures used in the analysis of macroinvertebrate community response.

3. To ascertain whether there is any similarity in patterns of recovery in macroinvertebrate communities following pollution incidents.

4. To investigate the behavioural response of a widely distributed species of macroinvertebrate to a pollutant.

5. To develop a procedure/protocol using macroinvertebrates for :

   - The **prediction** of response following a pollution incident.
   - The **assessment** of response following an incident

The research study involved two parts;

1. Field Study
   Pollution incidents were selected from incidents reported to the National Rivers Authority. Benthic macroinvertebrate response was assessed by quantitative sampling of the macroinvertebrate community at a series of impacted sampling points downstream of the pollution incident outfall and at non impacted reference sites. Reference sites were most often situated upstream of the pollution outfall but where possible reference sites were also established on adjacent watercourses. Sampling was continued for at least 6 months to establish impact and recovery. Measures of the macroinvertebrate community were either univariate (richness, diversity, BMWP, ASPT, abundance of dominant taxa) or multivariate (based upon a similarity matrix

produced from the comparison of all replicate samples).The value of a variety of measures of the community in demonstrating impact and recovery was assessed.

2. Laboratory Study
   A laboratory study was carried out to investigate the detailed behavioural response of a selected macroinvertebrate species to a pollutant. This involved focused individual behavioural recording before and after the introduction of a pollutant.

1.3 Arrangement of Thesis

Chapter 2 provides an introduction to biomonitoring of water quality and the investigation of impact and recovery following pollution incidents. The background to the formulation of the project is given.

Chapter 3 describes the fieldwork methodology employed in the investigation of pollution incidents including the sampling and processing techniques.

Chapter 4 gives details of the methodology used to process the data obtained from investigations of the pollution incidents.

The results and analysis of the pollution incident investigations are presented in Chapter 5,. with each subsection of Chapter 5 being devoted to one of the 11 pollution incidents.

Chapter 6 is the general discussion chapter, bringing together the findings presented in Chapter 5 and discussing the utility of the biotic measures used in the investigation of the impact of pollution incidents.

Chapter 7 describes the laboratory investigation of the behavioural response of a selected macroinvertebrate to pollution and gives the results, analysis and discussion.

In Chapter 8, the results obtained in the study are used to critically appraise the utility of macroinvertebrate sampling following pollution incidents and to derive guidelines for the assessment and prediction of impact and recovery following pollution incidents. Possible directions for future work conclude the chapter.

The conclusions arising from the study are presented in Chapter 9.

Chapter 2

Pollution incidents and Biomonitoring

2.1 Introduction

This chapter begins with a general account of pollution incidents, their importance within the United Kingdom and the response of the regulatory authorities to incidents. The Control of Industrial Major Accident Hazards (CIMAH) regulations are introduced and current developments in terms of environmental protection are mentioned. Section 2.3 gives an outline of biological monitoring, particularly in relation to benthic macroinvertebrates. Section 2.4 deals with the study of pollution incidents. This begins with a brief section emphasising the intrinsically disturbed nature of the lotic environment and describes attempts to understand the ecological impact of anthropogenic disturbance through toxicity testing. The study of actual and simulated field pollution events in the literature is then described and discussed. The final section describes the research project and its aims and objectives.

2.2 Pollution incidents - General

2.2.1 The relative importance of intermittent pollution.

The pressures on the river systems associated with a highly populated, industrialised region such as the United Kingdom have been well documented (Hynes, 1960; Craig and Craig, 1989; Haslam, 1990; Mason, 1996). Historically, initial attempts at pollution control were largely directed towards improved sewage treatment facilities resulting in a cleaner discharge to rivers (Sweeting, 1994). Subsequently, standards of effluent discharge were established with regard to the dilution potential of the watercourse and, more recently, also having regard to the classification of the river and associated objectives of water quality. This has resulted in improved treatment of waste water (NRA, 1994d; Seager and Maltby, 1989). However, despite the increasing controls within a regulatory framework, the last survey of river quality in England and Wales carried out in 1995, shows a high proportion of the rivers in the 'poor' to 'bad' quality categories (EA, 1997). A proportion of this low quality can be attributed to pollution incidents, which fall outside the scope of the consenting process. As regulatory controls on discharges become more stringent, and in spite of the efforts made by pollution control departments to play a more proactive role in the identification of potential pollution problems, the relative importance of pollution incidents in their impact on the environmental quality of rivers is likely to increase (McCahon and Pascoe, 1990; EA, 1996; Sweeting, 1994).

Fig 2.1 shows the number of reported pollution incidents in England and Wales from 1981-1995 (taken from NRA 1992, 1993, 1994b; EA,1996). It is clear that there has been a steady increase over this time, although there is a suggestion that after 1993 a proportion of this increase may be attributable to the introduction and publicising of a pollution hotline for notification of incidents (EA, 1996). Over 35,000 substantiated pollution incidents were recorded in 1995. Figure 2.2 shows the types of pollutant involved in the pollution incidents in 1995. It can be seen that fuels, oils and sewage are prevalent. If, however, only Category 1 or major incidents are considered (see Appendix 1 for EA definitions of pollution incident categories), chemical sources show a relative increase in importance (Fig. 2.3).

Thus, whilst sewage remains an important source of pollution incidents in the England and Wales, chemicals, fuels and oils are also significant pollutants particularly in those incidents having a major impact on the watercourse.

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Figure 2.1: Number of reported pollution incidents in England and Wales, 1981-1995 (taken from NRA 1992, 1993, 1994b; EA,1996)

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Figure 2.2: Distribution of substantiated pollution incidents by type in 1995 (from EA,1996)

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Figure 2.3: Distribution of Category 1 incidents by type in 1995 (from EA, 1996)

2.2.2 Response to pollution incidents

Within the U.K., incidents of accidental discharge to water courses are notified to the Environment Agency and dealt with by the pollution control departments. Although no information is available on the source of notifications, it is likely that a large proportion come from members of the public, particularly since the introduction of a free telephone link to the Environment Agency in 1993 (EA, 1996).

A proportion of the incidents will be notified by the polluter, but apart from these, detection of the incidents is dependent upon an identifiable change in the parameters of the watercourse, e.g. dead fish or invertebrates or changes in the water: smells or a change in colour.

Following notification, a pollution control officer visits the watercourse to determine the extent and cause of the incident. If appropriate, water samples are taken. A variety of remediation measures are used including physical obstruction, absorption, and pumping of the polluted water, as well as chemical measures, for example the addition of neutralising agents.

The Environment Agency has protocols and guidance notes relating to the investigation of pollution incidents. These include notes for the collection of evidence for prosecution, investigating fish kills, analytical procedures, categorisation of incidents and for reporting locally and nationally. The Environment Agency have a guidance framework for investigation of the environmental impact of aquatic pollution incidents (unpublished internal guidance note). This requires the investigating officer to be on site within 2 hours of the initial report and to collect samples of fish, water and river bed sediment and to make observations on the species and numbers of fish killed and on the behaviour of any survivors. The guidance note suggests that:

> "samples of macroinvertebrates should be collected above, at and below the discharge point by the standard Environment Agency sampling techniques to demonstrate the effects of the incident."

The note also states that :

> "extent of damage to the river in terms of maximum damage and the distance downstream affected should be described."

The procedure for doing this is left to the relevant monitoring department and the guidance notes appear only to emphasise the upstream and downstream sampling points in the establishment of impact. The guidance notes acknowledge that sites used for normal monitoring purposes may not be appropriately located to give reference information and suggest the use of a sampling point as close as possible upstream of the discharge as a reference site. The constraints of cost appear to play a large part in the establishment of impact and utilising the above methods, no indication of within site variability is given, or attempts to separate chronic discharges from the outfall, from the pollution incident under investigation.

The Environment Agency do not have a procedure for monitoring recovery following incidents due to the presumption that the impact will be short-lived and that the ecology of the watercourse will recover without further intervention (J.Cocker, personal communication, 1997). Following the incident, restoration of fish stocks is often carried out if there has been a major impact on fish populations.

2.2.3 Pollution incidents and the environment

Consideration of the effects of pollution incidents needs to be viewed in the widest environmental context. The importance of having regard to the protection of the environment in all man's activities is now internationally acknowledged as evidenced by the United Nations Conference on Environment and Development (the 'Earth Summit') held at Rio de Janeiro in 1992. The Convention on Biological Diversity was an important component of the Earth Summit. This recognised the importance of maintaining and enhancing the diversity of life: at the ecosystem and habitat level, between species and the genetic diversity within species (HMSO, 1994).

Within the United Kingdom, one of the duties of the Environment Agency is to protect or enhance the environment taken as a whole, in order to play its part in attaining the objective of sustainable development (Environment Act, 1995). Current guidance with respect to the Environment Agency's contribution to sustainable development reinforces the integrated approach to protection and enhancement and also refers to the need to conserve and enhance biodiversity (EA, 1996b).

The protection and enhancement of the aquatic ecosystem and associated habitats is the underlying rationale for much of the risk management and biological assessment that is described in the following paragraphs.

Following a number of major industrial pollution incidents in Europe in the 1970's, the European Community introduced the "Seveso" Directive in 1982, the aim being the prevention and limitation of the effects of accidents arising from industrial activities involving dangerous substances. This was implemented in the United Kingdom in the form of the Control of Industrial Major Accident Hazards (CIMAH) regulations (HMSO, 1984). The regulations refer specifically to major accidents: "an occurrence leading to a serious danger to persons or the environment".

Rivers in the United Kingdom are given a water quality rating by the Environment Agency which reflects the background level of pollution and its use suitability; the General Quality Assessment, (NRA, 1991b). This classifies river stretches into 6 bands according to chemical or biological criteria. A major accident to the environment with regard to freshwater lotic habitats is defined within the CIMAH regulations as:

> "an incident that either lowers the chemical water quality by one class for more than one month or lowers the biological quality by one class for more than one year or causes long term damage to the habitat overall in a significant part of the watercourse (a ten kilometre stretch or a "reach", whichever is the lesser)" (DOE, 1991).

Future developments will probably result in the accommodation of less severe criteria and thus the inclusion of lesser accidents into the scheme (DOE, 1991).

Under the CIMAH regulations, where an accident occurs on a site, the site management is required, as part of the information to be made available to the Health and Safety Executive (now part of the Environment Agency), to provide data for assessing the effects of the accident on persons *and the environment,* as well as a statement of the steps envisaged to alleviate medium or long term effects of the accident. This includes:

> "thorough survey of the environs of the site, including surface water and groundwater or aquifer catchments", "assessment of the possible environmental consequences of the accident, with particular reference to ecotoxicity and persistence of effects" (DOE, 1991).

The DOE guidelines concerning the environmental survey suggest that detailed ecological survey work is conducted to provide a set of data for species distribution and abundance, against which the effects of an accident could be judged. Furthermore, in the event of an accident, immediate evaluation of the impact and in some cases monitoring of recovery may be necessary (DOE, 1991).

In spite of these regulations and guidelines, it has been suggested that few of these risk control measures are implemented in a way that protection of the environment is the result (Khayyat, 1993).

Future developments to the assessment of risk within the CIMAH regulations, involve greater definition of potential environmental harm in the establishment of an Environmental Harm Index (E.H.I.). This would include terms relating to the concentration of the pollutant and the size of the stretch affected relative to a reference size. In future, a term relating to recovery time may also be included within the index (B. Harland, personal communication, 1997).

The guidelines to the CIMAH regulations (DOE, 1991) do not give detailed information regarding the assessment or monitoring of a potential pollution impact. One of the aims of the present study is to investigate the utility of biological monitoring in assessment of the impact of pollution incidents on streams and to provide further definition to the survey work required under the CIMAH regulations.

2.3 Biological Monitoring

2.3.1 Biological Monitoring - A Historical Perspective

The principle underlying biological monitoring is that elements within the biota show changes in response to changes in the environment and that these changes can be measured and used to assess the environmental changes. Biological monitoring of rivers as a means of assessing water quality and, more recently, general environmental quality, has been extensively used in Europe and North America particularly in the latter half of this century (De Pauw and Hawkes, 1993; Metcalfe-Smith, 1994; Davis, 1995).

Elements of the ecosystem that have been used for biological monitoring include algae (Whitton *et al.*, 1991; Cairns *et al.*, 1972), protozoa (Cairns *et al.*, 1972), macrophytes (Haslam, 1990), fish (Yoder and Rankin, 1995), all of which have been used to supplement information gained through chemical monitoring. However, benthic macroinvertebrates are the component of the stream ecosystem most commonly chosen for monitoring for the following reasons:

- their taxonomy is well established and identification is relatively straightforward;

- their sampling methodology is relatively straightforward and has the potential for quantitative analysis;
- they form a complex structural community with taxa displaying varying habitat requirements and sensitivities;
- many elements within the trophic structure of a stream system are represented by macroinvertebrate taxa, such that changes within the macroinvertebrate community gives some indication of ecosystem changes;
- macroinvertebrates have a pivotal function in the majority of stream ecosystems; playing an essential role in productivity, nutrient cycling and decomposition (Cummins, 1975; Reice and Wohlenberg, 1993).
- in many lotic ecosystems, macroinvertebrates are particularly important in political or economic terms in that they provide food for fish;
- their relatively sedentary habits provide an indication of the present as well as recent historical conditions of the watercourse;
- there is a large body of information relating to the sensitivities of different taxa to certain pollutants, particularly organic pollution.

There is great variation on the aspects of the macroinvertebrate community that are used for biomonitoring, with some systems using the whole community and other systems concentrating on particular groups e.g. the Chironomidae (Wilson, 1987).

The structural aspects of the macroinvertebrate community have been most widely used in biological monitoring although functional properties, morphological and behavioural changes have also been shown to give valuable information (Wallace *et al.*, 1986; Warwick, 1988; Heinis *et al.*, 1990).

Organic pollution resulting mainly from sewage and agricultural effluent was for many years the major cause of river degradation and consequently resulted in a great deal of research into the effects of organic pollutants on receiving waters (Butcher, 1947; Haslam, 1990; Hynes, 1960). Less well understood is the river ecosystem response to substances that are industrial in origin. These pollutants are assuming greater importance in rivers as sewage treatment plants are improved, thus reducing the organic component in waste water. Mason (1996) indicates that 1500 substances, including those that are organic in origin, have been listed as pollutants in freshwater ecosystems. The wide range of pollutants and the multiplicity of synergistic and antagonistic processes to which they are subjected within the environment, make prediction and appraisal of their impact a challenging field of study.

Early biological assessments involved expert assessments of macroinvertebrate species abundance lists. However, the wealth of information collected over a period of time and the need to convey the information in a succinct way to other specialists and managers, required condensation of the information gathered.

The ways in which the basic taxonomic/abundance information is processed is diverse and this reflects the different levels at which the community can be viewed as well as other factors including: historical precedence; the level of understanding of dynamic components within stream ecology; the requirements of the sampling programme; the development of computational facilities; and the level of knowledge regarding the habitat requirements of taxa (Cairns and Pratt, 1993; Metcalfe, 1989).

Macroinvertebrate community information can be processed in the following four ways.

- Reduction of community information to single figures, without assumptions about the structure or function of the components, for example: number of taxa; number of individuals; biomass.

- Reduction of the community information into single figures dependent upon assumptions regarding the structural aspects of the community: e.g. the range of diversity indices which incorporate the idea that an unstressed community will show a large number of taxa with relatively even distribution of abundance levels. Structural analysis of this kind does not take the kinds of taxa into account.

- Reduction of the community information into single figures based upon the known sensitivities of the taxa to known pollutant (usually organic) - a biotic score system: e.g. Biological Monitoring Working Party (BMWP) score (Chesters, 1980), Trent Biotic index (Woodiwiss, 1964), Chandler Biotic Score (Chandler, 1970), Saprobien index (Kolkwitz and Marsson 1908;1909).

- Community comparison indices (similarity indices) assess changes in community composition without regard to sensitivity or structure. Thus, quantitative community comparison indices simultaneously compare either the relative or absolute abundance of each species within two samples, and can therefore measure changes in community composition due to pollution if reference samples are compared with impacted samples. Recent multivariate techniques enable the preservation of a high proportion of the original data without making assumptions about structure and relative sensitivity.

Some biological monitoring incorporates a combination of the above analyses: e.g. the Invertebrate Community Index (DeShon, 1995) and Rapid Bioassessment Scores (Resh and Jackson, 1993).

2.3.2 Biological Monitoring - current situation

Biological monitoring of water quality is used extensively in Europe and North America and to varying degrees in other parts of the world. There is, however, a great deal of variation in the sampling, processing and analysis procedures used such that it is fair to conclude that although the value of biological monitoring has been established, the methodology is continuously being refined and developed.

Diversity indices, which calculate the ratio of abundance of species to the number of species, are still widely used in certain countries (Metcalfe-Smith, 1994). Biotic indices are prevalent in Europe including the United Kingdom (Davis, 1995). In the USA, the adoption of a multiple measures to assess the macroinvertebrate communities are currently being developed (DeShon, 1995).

In the United Kingdom, biological monitoring is carried out on a routine basis at a large number of sites on all major river systems to a standard format, although there are regional differences. As a minimum, taxa are identified to family level, enabling the calculation of BMWP scores (Chesters, 1980: see Chapter 4.3.4.4. for description). The scores are compared to hypothetical scores that would be predicted from a number of the environmental characteristics found on the site utilising the River Invertebrate Prediction and Classification System (RIVPACS) (Wright *et al.*, 1989). In addition to the above information, taxa are usually given abundance categories, and in most regions, a proportion of the macroinvertebrates are identified to genus or species level.

2.3.3 Biological Monitoring - the future

Our understanding of the changes within the macroinvertebrate community brought about by a range of natural and anthropogenic factors has increased rapidly in the last forty years, with concomitant changes in the ways in which the community has been analysed and the use to which the information has been put. Biologists have long been concerned about the loss of information resulting from the use of indices and scores, information that is nevertheless still collected and stored. In the UK, recent developments to address this problem include the development of Expert Systems and

Artificial Neural Networks (Walley, 1993). Expert systems aim to improve upon the subjective assignment of macroinvertebrate taxa into levels of pollution sensitivity by utilising expert assessments of pollution status based on macroinvertebrate communities to generate probability distributions of their sensitivity and mathematically combining this evidence. The application of artificial neural networks to the interpretation of benthic data allows for the development of a system that acquires knowledge from new data sets and does not require total input of knowledge from an expert. In this case, as in the expert systems, subjectivity is still required on the part of the operator in the assignment of pollution status to patterns generated by the analysis or to data sets used in the 'training' process (Walley, 1993).

Predictive modelling is another avenue of approach. Based to a large degree on ecotoxicological information, the outcome of a release of a particular pollutant into the water system can be predicted with definable degrees of probability (US EPA, 1996; Mitchell, 1994).

The development of multivariate analysis of communities within the ecosystem is an important approach, but not used to a great degree by the regulatory authorities. This was initially developed for the assessment of patterns in terrestrial plant communities and is now increasingly being applied to both marine and freshwater systems for the assessment of pollution (Clarke and Warwick, 1993; Wright *et al.*, 1989; see Chapter 4.3.5).

The development of in situ bioassays involving sublethal responses as 'early warning systems', for example, the feeding rate of *Gammarus pulex,* is another avenue of development (Johnson *et al.*, 1990). However the extent to which these responses relate to the impact on the community present within the watercourse is not yet fully understood.

In the USA, invertebrates are used in the development of rapid assessment protocols of water and habitat quality (DeShon, 1995). Where earlier methods of biomonitoring tended to concentrate on a single aspect of the macroinvertebrate community, the protocols utilise the information gained from a range of measures.

Biological monitoring of streams utilising macroinvertebrates has hitherto concentrated on long-term situations and background water quality. Relatively little information exists on the utility of biological monitoring in the assessment of impact and recovery following pollution incidents.

2.4 The study of pollution incidents

2.4.1 Pollution incidents and disturbance in lotic ecosystems

An understanding of the ecological properties of streams is essential to the study of disturbance and recovery resulting from anthropogenic factors. Streams are essentially dynamic systems. The variability of lotic systems is evident at a number of different levels.

- Resources available within the ecosystem may be patchily distributed e.g. leaf packs, stands of emergent vegetation, substratum-related distribution of periphyton. This is related to the physical and temporal variability of lotic systems and requires movement by organisms to utilise those resources (Townsend and Hildrew, 1976; Peckarsky, 1980).

- The water flow within streams can be viewed as a chronic source of disturbance, and organisms have developed various mechanisms to prevent and compensate for the downstream relocation of populations resulting from this (e.g. Muller, 1982; Townsend and Hildrew, 1976; Soderstrom, 1987).

- In addition, lotic systems may have larger scale disturbances of a seasonal or unpredictable nature: increased and decreased flow; scour due to drift ice during ice break-up; frazil and anchor ice formation in high latitude streams; highly variable organic input (Hynes, 1970; Mcauliffe, 1983; Reice *et al.*, 1990; Niemi *et al.*, 1990).

Reice *et al.*(1990) suggest that the composition of the benthic community is related to the disturbance regime of the site, and that many stream communities exist in a state of perpetual recovery from frequent disturbances. The organisms associated with streams show life history features, as well as physiological, behavioural and morphological adaptations which increase the chance of survival in a changing environment.

The fact that variability at all levels is an important part of most stream ecosystems, has considerable implications for the understanding of the system response to anthropogenic disturbances. In particular, the colonisation mechanisms of invertebrates are well developed, and this influences the timing and mode of recovery following pollution incidents.

Disturbance has been described in a number of ways, all of which relate to a change within the system from an expected or predictable situation. For example, Stanford and Ward (1983) describe disturbance as

> "any stochastic event which forces normal system environmental conditions substantially away from the mean".

Wallace, (1990) defines disturbance with respect to the impact on the ecosystem as

> "any event that results in a significant change (either positive or negative) in macroinvertebrate community structure (species, abundance, biomass, or production) beyond that expected over the annual cycle within a particular habitat".

Succession can be defined as the changes that occur on a site after a disturbance (Fisher, 1990), which can, in this study, be linked to the term 'recovery'. Recovery following a pollution incident can be variously defined depending on the level of ecosystem analysis. Cairns *et al.*, (1977) recognise this in their three definitions of recovery:

1. restoration to usefulness as perceived and defined by the general public;
2. restoration to original functional and structural conditions, although the elements (species) that comprise this structure may be significantly different than those present originally;
3. restoration to the original functional and structural condition with original elements present.

The latter two definitions, in their use of the concept of 'original condition', do not take the dynamic nature of stream ecosystems into account as discussed above. Wallace (1990) defines recovery as that which:

> "constitutes the re-establishment of community structure to within the range expected over the annual cycle within a particular habitat prior to the initial disturbance."

Gore (1982) gives greater definition to the comparative processes required in an assessment of recovery by describing it as:

> "the return to an ecosystem which closely resembles unstressed surrounding areas", or "predisturbance levels" (Gore *et al.*, 1990).

Assessment of recovery, therefore, includes two components:

1. the selection of a reference condition;
2. movement of the impacted community towards similarity to the reference condition.

The point at which recovery can be said to be complete (i.e. the recovery endpoints) is dependent upon the subjective judgement of the observer as well as the degree of knowledge of the structure and function of the relevant ecosystem. The choice of reference condition and the continuum of features of the community by which similarity is measured, are as diverse as the studies investigating recovery from disturbance (refer to section 2.4.3).

One of the main focus points of the present study is an assessment of the ways in which recovery can be described and defined following pollution incidents.

2.4.2 Ecotoxicology and the prediction of impact/risk assessment

Ecotoxicological studies have provided a great deal of information regarding the response of a number of components of the stream ecosystem to a range of toxic effluents. The majority of these studies have been carried out in the laboratory under replicated and highly controlled conditions, and their applicability to real-life situations continues to be questioned (Cummins, 1975; Seager and Maltby, 1989; Pascoe, 1989).

Conventional bioassays involve continuous exposure of animals to a fixed concentration of a toxicant. The ecological response to a discrete event can not necessarily be predicted from the response to continuous pollution as has been shown in laboratory studies (Maltby *et al.*, 1987; Gammeter and Frutiger, 1988; Green *et al.*, 1988; Pascoe, 1988) as well as in field situations (Dowson *et al.*, 1996; Turner, 1992). In addition, local environmental conditions in streams result in variability and attenuation of the pollutants (Dowson *et al.*, 1996; Turner, 1992). The synergistic or antagonistic actions of compounds comprising complex spills or their action with other compounds in the river can influence their acute toxicity.

An example of the difficulties in transferring information about laboratory derived predictions of toxicity to actual events is demonstrated in a study of the impact of a fire at a tyre dump (Best and Brookes, 1981). The prediction of the effects of a variety of organic compounds and cyanide based on the known toxicity levels of phenol and cyanide, was that there would be no mortality. However, in reality, fish and other organisms showed extensive mortality following the incident. This may have been the result of synergisms resulting from multiparameter exposure which are difficult to predict from laboratory bioassays.

In recent years toxicity testing has taken a more ecological approach incorporating a greater degree of realism, ranging from the provision of habitat features pertinent to the test organism e.g. substratum and food, to the testing of complete mesocosms and assessing the effects of toxicants at a number of levels (Hermanutz *et al.*, 1987; Mathes and Wiedemann, 1990; Brock *et al.*, 1992; Mitchell *et al.*, 1993; Matthews *et al.*, 1996). In addition, sublethal effects of pollutants and the impact of pulse or intermittent exposure rather than chronic exposure has received greater attention (Pascoe, 1988; Borlakoglu and Kikuth, 1990; Heinis *et al.*, 1990; Maltby *et al.*, 1990). It has been demonstrated that under certain conditions animals can recover fully following brief exposure to levels of a pollutant which, under more prolonged exposure, would result in mortality (Wright, 1976; Green *et al.*, 1988).

In addition to these controlled studies, investigation of the impact of actual pollution incidents, although more limited in nature, is required to fully understand the factors contributing to impact and recovery and to enable limits on the extent of predictability within any given situation to be defined.

2.4.3 Overview of the study of pollution incidents

The study of pollution incidents that have impacted natural watercourses can be divided into two kinds: those that are essentially opportunistic studies of actual pollution incidents, and those that involve simulated pollution incidents.

The experimental simulation of a pollution incident allows investigation of the biota prior to, as well as following the incident, combined with simultaneous assessment of water quality. Field studies have investigated the impact of simulated incidents of a variety of toxicants including acid (McCahon *et al.*, 1989; Bernard *et al.*, 1990), oil (Rosenberg and Snow, 1975; Barton and Wallace, 1979; Lock *et al.*, 1981; McCart and Denbeste, 1987), biocides (Jeffrey *et al.*, 1986; Sibley *et al.*, 1991; Yasuno *et al.*, 1982; Wallace *et al.*, 1986) and organic farm waste (Turner, 1992). Simulated pollution studies, whilst enabling detailed investigations of the ecological impact of episodic pollution to be carried out, are of course limited in the scale and extent to which they can be applied to natural situations because of the risks of causing environmental harm.

Opportunistic studies have tended to concentrate on the impact of a pollution incident on various aspects of the biota, often in relation to the impact that would be predicted from ecotoxicological studies. In the majority of such studies, attention has been on the initial impact of the incident, and subsequent changes in the biota have not been investigated (Niemi *et al.*, 1990).

The literature was examined for field studies of both actual and simulated pollution incidents or studies following the cessation of chronic pollution in lotic aquatic systems. Only those studies that included assessment of impact on the macroinvertebrate community and gave some information about recovery of the community were included. References describing 40 pollution incidents were examined, 13 of which were simulated and 27 were actual pollution incidents (see reference list in Appendix B).

Studies encompassed a wide range of pollutants, with the majority being concerned with oil and refined oil products such as kerosene and other forms of fuel oil (Table 2.1). Biocides comprising a variety of pesticides were the next most important group.

Table 2.1: Pollution incident studies - pollution type

Pollutant type	Number of Studies
Industrial - mixed /composition unknown	7
Oil and oil derivatives	14
Acid	4
Cyanide and waste	2
Chlorine	1
Biocide	8
Organic factory waste/sewage	2
Mine waste	1
New channel	1

The choice of reference conditions is an important part of the assessment of impact and recovery within a disturbed system. It is surprising, therefore, that a number of studies did not mention any reference conditions, relying instead on judgement when assessing biotic changes (Table 2.2).

The most commonly used reference conditions were those found at sites situated on the impacted watercourse, but located upstream of the point of impact. The use of the upstream sites as a 'control' is limited in experimental terms, as they and the impacted sites are not independent. However, the use of adjacent channels is not widespread, presumably because of the difficulties in finding sites that have similar environmental conditions. Historical data is not often quoted and is usually only available for the studies investigating simulated pollution incidents. The advantage of choosing upstream sites as reference sites, is that environmental conditions, for example, ambient conditions of water quality, are likely to be similar and both sites would be subjected to the same seasonal changes in environmental factors and macroinvertebrate population cycles. However, a number of studies found that, for various reasons, the upstream sites were inadequate as reference sites in that their environmental parameters were not similar to those of the impacted sites (Victor and Ogbeibu, 1986; Crunkilton and Duchrow, 1990). In addition, although reference sites were used in the majority of studies, a number did not utilise them in the assessment of impact or recovery.

Table 2.2: Pollution incident studies - Reference conditions

Location of Reference condition	Number of studies
No reference sites	6
Pre- impact	4
Upstream sites	25
Downstream sites (presumed to have avoided pollution impact)	2
Nearby channel	3

The ways in which recovery was defined within the pollution incidents were varied and are listed in Table 2.3. Most commonly, recovery is recorded in terms of the identity of taxa or first reappearance of taxa and this is usually, but not always, in comparison with a reference site. Commonly, some measure of abundance of the individual taxon is included, and often, the studies identified the macroinvertebrates to a higher level than that used in the analysis of recovery due to the difficulties of assessing the recovery of individual taxa when abundance levels were low. As a consequence, information was lumped e.g. to families, total density of individuals, diversity indices etc. and did not make complete use of all the macroinvertebrate information that had been collected.

Recovery, whilst being described in the studies, is not always given a timescale in terms of completeness and was most often based on subjective judgements.

Niemi *et al.*(1990) identify recovery endpoints according to the following criteria:

- recovery to average individual size;
- recovery to previous density;
- recovery of species or genera richness (at least 80% of original number of taxa);
- recovery of total biomass;
- first reappearance;
- return to relatively stable population level as determined from a seasonal population curve which was judged to be similar to pre-stressor levels.

Table 2.3: Pollution incident studies - biological measures used to assess recovery

Biological measure	Number
Biotic index	1
Taxa richness	4
Identity of taxa	13
Abundance of individual taxa	11
Total density	2
Trophic structure	2
Diversity index	3
Drift changes	2
BMWP score	1
Density of dead animals	1

Studies were generally less than 12 months in time (Table 2.4) and the recovery was often defined as being complete by the end of the study, with no further information to enable possible seasonal factors to be assessed. The recovery times shown in Table 2.4 illustrate the speed with which organisms can repopulate an area after defined mortality, with the majority showing recovery within a year.

Table 2.4: Pollution incident studies; timescale of studies

Time of pollution study (generally equivalent to recovery time as assessed by each study)	Number of studies
1 month or less	4
1-2 months	7
2-6 months	10
7-12 months	7
13-24 months	6
25-36 months	1
37-48 months	0
more than 48 months	5

The colonisation patterns utilised by taxa following pollution incidents is an important factor in the understanding and prediction of recovery, although this was not generally considered in the pollution studies investigated. A few studies hypothesise about the probable route of colonisation based upon the known community characteristics of unimpacted sites. Drift from unaffected tributaries or upstream sites is often quoted as the most important route of colonizing taxa (e.g. Crunkilton and Duchrow, 1990; Cairns and Dickson, 1977; Douglas and Maccreanor, 1990; Bernard *et al.*, 1990; Gore, 1979; Heckman, 1983). This is supported by more detailed studies of colonisation following disturbance of small areas or of artificial substrates (Doeg *et al.*, 1989; Turner, 1992; Williams and Hynes, 1976; Townsend and Hildrew, 1976).

Given the diverse nature of the study of actual pollution incidents, it was considered that a study of selected pollution incidents, utilising similar sampling, processing and analysis methods would contribute to our understanding, both of the processes occurring within streams following anthropogenic disturbance, and the relative utility of the biological measures used in the assessment of recovery.

<u>The Assessment and Prediction of Recovery</u>

Various attempts have been made to elucidate similarities in the response of stream ecosystems to disturbance of different kinds such that a predictive capability is created (Cairns and Dickson, 1975; Neuhold, 1981; Minshall *et al.*, 1983). One of the most notable is that of Cairns (1990) who suggests that the ability of an ecosystem to recover following a disturbance can be predicted in broad terms through the use of an 'Ecosystem Recovery Index'. This includes terms relating to the colonisation sources, the mobility of dissemules, habitat condition and residual toxicity following the disturbance and the remediation activities carried out by management agencies. Cairns (1990) emphasises that this a first step in a predictive approach to the assessment of recovery. Further development of a predictive approach to ecosystem recovery is one of the aims of the present study.

2.5 Project formulation

As discussed above, the use of macroinvertebrates in assessment of ecosystem response to chronic situations involving water quality and habitat alterations is well established in the developed world. However, the use of macroinvertebrates to study pollution incidents is fragmentary and less well developed.

As the regulations regarding chronic discharges to watercourses are made more stringent, the relative importance of pollution incidents with respect to environmental quality continues to increase. Where pollution incidents have been investigated, the response, in general, has been described in terms of initial impact whilst temporal impact and recovery have not received a great deal of attention. Higher order lotic systems support a fauna that is adapted to disturbance and it is widely accepted that recovery will be rapid. However, knowledge of the factors contributing to recovery and the universality of patterns of recovery is at a relatively poor state of development.

Although toxicological studies have been the major tool in the prediction of response of the macroinvertebrate community to a variety of known toxicants, their utility in the prediction of responses to pollution events of short duration is less clear.

For these reasons, this study, initiated in October, 1993, was designed to concentrate on the response of the benthic macroinvertebrates to a number of actual pollution incidents as reported to the National Rivers Authority. The drawbacks to this approach included: potential lack of, or limited pre-impact data; lack of control over variables and replication such that only *apparent* response to pollution incident can be determined; and a multiplicity of factors influencing macroinvertebrate response, both watercourse related and pollutant related. However, notwithstanding these problems, the study has advantages in that real situations are being investigated and although the findings can not be specific, their generality would increase their applicability to a range of situations. In addition, one of the aims of the study was to produce information that would be of practical application in the prediction of environmental impact and the environmental survey required by the CIMAH regulations.

A large body of knowledge exists concerning the impact of organic pollution, particularly sewage-related, whereas pollution from industrial, particularly chemical sources, is less well researched and has been shown to produce a more variable and less predictable ecological response (Milner, 1994). For this reason, emphasis within the study was given to those pollution incidents not associated with sewage or agricultural effluent.

Investigation of ecosystem response to disturbance entails analysis of the initial impact as well as recovery over time. The macroinvertebrate response is taken to be indicative of the ecosystem response, with the response of the community being measured in a variety of ways. Quantitative samples of benthic macroinvertebrates were taken as soon as possible following a pollution incident at both impacted and reference sites. Further samples were taken at various time intervals following the pollution incident. The species/abundance data obtained was analysed in various ways to elucidate the macroinvertebrate community response to the pollution incident.

The structural response of the macroinvertebrate community to change has formed the basis of much biomonitoring of water quality. Similarly, assessment of benthic community change following pollution incidents has concentrated on structural changes with the assumption that these have been caused by mortality-induced drift (e.g. Turner, 1993). However, few studies have investigated the processes leading to the measured changes, for example, the relative importance of mortality and behavioural avoidance. To gain further insight into the processes involved in the response to a pollutant, a laboratory study was carried out to determine the detailed individual behavioural response of a commonly found macroinvertebrate to a pollutant. *Gammarus pulex* was chosen for study as it forms an important structural component of many temperate stream communities and the requirements of this species is well understood, making the maintenance of a laboratory population relatively straightforward.

Chapter 3

Investigation of Pollution Incidents

Data Collection Methods

This chapter describes the data collection methodology adopted in the study of the pollution incidents.

3.1 Pilot Studies

Pilot studies were carried out on two pollution incidents reported to the N.R.A. in order to evaluate different sampling methodologies: a spill of hydrochloric acid to the Mousesweet Brook and a spill of heating oil to the Callow Brook. Details of the sampling methodologies used are given in Section 3.4.1. The results from these pilot studies were considered to contribute to the aims and objectives of the project and they were therefore included within the results section in Chapter 5.

3.2 Selection of Pollution Incidents

The project required access to information about the pollution incidents occurring within an accessible geographical location. Arrangements were made with the pollution control departments of three different regions within the Severn Trent Area of the National Rivers Authority: Upper Severn, Lower Severn and Upper Trent. The method of acquiring information from each region varied according to their operational procedures but consisted essentially of regular, weekly telephone contact with office-based staff. If a pollution event was considered potentially appropriate for the project, follow-up telephone contact was made with the pollution control officers involved with the incident. The decision to follow up the incident was then made, based upon the following criteria.

1. Pollution source likely to be industrial (farm, sewage and related organic pollution events were not considered).
2. Likely or evident effect on the biota. This involved a certain amount of conjecture. If dead macroinvertebrates or fish were seen downstream of the outfall, this was presumed to have been the result of the incident.
3. The watercourse should have a natural bed.
4. Suitable sampling stations both up and downstream of the source of the incident or reference sites available in an adjacent watercourse.
5. Watercourse accessible to sampling using a Surber Sampler (depth < 40cm).
6. Geographically accessible.

3.3 Incident Response and Choice of Sampling Points

When the decision had been made to visit the location of a pollution incident, this was done within 24 hours whenever possible and the relevant pollution control officer was met on site.

At least four sampling points were chosen according to the following criteria.

1. All sampling points were selected to have similar environmental conditions e.g. depth, substrate, current velocity. Riffle sites were chosen where possible. Stratification i.e. restricting the sampling to habitat types within the stream, is commonly used to reduce the intersite variability due to environmental factors other than the pollution incident itself, particularly substratum type (Harper, 1990; Resh and McElravy, 1993). Riffle sites are the most common biotope to be sampled in impact studies (Resh and McElravy, 1993). Riffles support a large number of macroinvertebrate species from a wide variety of higher taxa (Brinkhurst, 1965; Brown and Brussock, 1991; Hynes, 1970). In addition, it was considered that riffle sites would experience greater exposure to a transient pollutant than pool areas.

2. Selection of a reference site. Reference conditions are required against which impact and recovery can be assessed. The selection of a suitable reference site in field studies necessarily involves compromise from the ideal due to the lack of control over differential effects of parameters other than the pollution under investigation (Norris *et al.*, 1992). Resh and McElravy (1993), in reviewing quantitative lotic studies, found that the majority of reference sites were spatial

(upstream of the outfall on the same watercourse). This was also found in the pollution incident literature discussed in Chapter 2. Fewer studies utilised the impacted sites before impact, or sites on different watercourses within the same catchment as reference sites. The present study used at least one upstream sampling point as a reference site and, where possible, adjacent watercourses were also sampled to provide reference conditions. Upstream sites were chosen to be close to the incident source, whilst having regard to the potential for influence from the outfall due to mixing.

3. At least three sampling points were chosen downstream of the pollution source to identify any decrease in impact with increased distance from the pollution incident.

3.4 Biological Data Collection

3.4.1. Sampling Method

The object of sampling at each station was to discover the species of benthic macroinvertebrates present and their relative abundance.

The kick-sampling methods utilised by the Environment Agency during biological monitoring programmes are semi-quantitative in that they are limited by time, and are designed to sample a range of biotopes within a defined area (BSI, 1988). One of the pilot studies (Mousesweet Brook) utilised this sampling technique although sampling was restricted to a single biotope (riffles) for the reasons outlined above (Section 3.3). However, the lack of replication which is required to indicate the degree of within-site and between-site variability, made assessment of impact in this case difficult. A more quantitative approach was therefore considered to be appropriate to the present study. The method chosen has the disadvantage that rare species are less likely to be collected as only a relatively small area of the substratum is sampled ($0.3m^2$) at each station.

The operation of two methods of quantitatively sampling the substratum: the Aston Cylinder Sampler (HMSO, 1982) and the Surber sampler (HMSO, 1982; BSI, 1988), were compared during pilot studies. Both samplers define an area to be sampled, but the Aston Cylinder Sampler improves upon the Surber sampler in that it consists of an enclosed tube thus reducing the chance of loss of organisms through drift. However, it proved difficult to use in very shallow water and coarse substrates. The Surber sampler was chosen for the present study for its greater flexibility in a wide range of substratum types.

A Surber sampler is designed to collect benthic macro-invertebrates from a defined area of substratum; 30cm by 30 cm, to give a sampling area of 0.1 m^2. Benthic macroinvertebrates were collected according to a standard procedure (HMSO, 1982; BSI, 1988). The substratum is disturbed by hand to a depth of 5-10 cm for a period of one minute. Dislodged materials are washed into the 1mm mesh net by the current. If the current is less than 0.1 m/s, collection involves alternately disturbing the substratum and agitating the water towards the net.

Quantitative sampling was carried out to give an estimate of numbers per unit area and an estimate of within-site variability. Benthic macroinvertebrates often display a contagious or clumped distribution (Elliott, 1977) thus requiring a large number of sampling units for accurate sampling, however only three to five replicates are commonly used in benthic biomonitoring studies in streams (Resh and McElravy, 1993). The use of a low number of samples in detecting a given difference between sites or times results in a high proportion of Type II errors (concluding that all means are derived from the same population when in fact they are not). However, the use of the number of samples that would be required to detect differences in samples that show low values of means or high variability can be prohibitively high and impractical in most biomonitoring surveys (Norris *et al.*, 1992; Resh and McElravy, 1993). The constraints involved in much biomonitoring require that cost-saving techniques predominate. As one of the aims of the study was to recommend a sampling protocol for use in detecting potential impact, the minimum number of replicates (3) that can be taken to detect a quantitative change was selected for the study. One of the pilot studies (Callow Brook) involved the use of five replicates at each sampling point. However, it was considered that the decrease in within site variability found with five as opposed to three samples had to be balanced against the greater logistical difficulties involved in the transportation and sorting of the extra samples at pollution incidents that involved more than the four sampling points used in Callow Brook. Girton (1980) investigated the effect of replication on BMWP scores, and found that 75-85% of the BMWP score produced by 10 samples was achieved by 3.

McElravy *et al.*, (1989) describe a 7-year study that found statistically significant differences in univariate measures of a benthic community associated with extremes of precipitation, despite selecting a sampling design that exaggerated variability: four samples were taken at each sampling point; two samples from riffles and two samples from pools. By limiting the sampling through stratification by habitat type, the present study aimed to reduce the variability among replicates.

The problems of pseudoreplication (Hurlbert, 1984; Cooper and Barmuta, 1993) are associated with the method of impact assessment employed in this study, a point which is considered further in Chapter 4.

The Surber Sampler was initially placed at a random point within the riffle site and two further samples were taken within 10 metres in an upstream direction. Samples were placed into separate buckets and covered with stream water. An initial inspection was carried out to remove and record fish and other large predators. The samples were packed into coolboxes containing freezer packs for transportation to the laboratory where they were stored at 10^{o}C prior to processing.

3.4.2. Sampling Times

The initial sampling time was as soon after the pollution incident as possible. Subsequent sampling times were initially closely spaced at intervals of weeks and subsequently more widely spaced at intervals of months. Ideally, sampling times would have been evenly spaced following the pollution incident, thus making no prior assumptions about patterns of colonisation. However, studies have shown that colonisation after disturbance generally follows a lognormal pattern with rapid inital changes (Crunkilton and Duchrow, 1990; Cairns and Dickson, 1977; Turner, 1992). Therefore, because of the time constraints involved in monitoring change following a number of pollution incidents at the same time, sampling was most frequent during the time when changes were most likely to occur. Sampling continued for a period of at least one year to allow for seasonal changes in invertebrate communities to be apparent.

3.4.3. Sorting

Sorting was required to remove the macroinvertebrates from the debris and preserve them for subsequent identification. Sorting was carried out within 24 hours to ensure that the majority of the invertebrates were still alive. Previous experience had indicated that the advantages of live sorting are that movement of the animals aids identification amidst the debris/algae, thus reducing the time required for sorting and reducing the likelihood of missing certain small invertebrates e.g. cased caddis larvae and snails.

Disadvantages are that a large amount of sorting needs to be done within a limited time period and there is the possibility that predators, e.g. leeches, feed on other invertebrates within the container. In the case of erpobdellid leeches, the most commonly found predator, these would have been Oligochaeta and Chironomidae in particular. The storing of the containers at a temperature of 10^{o}C, which reduces the activity of leeches, would have reduced predation to a certain extent.

The samples were washed into a 2.00mm sieve stacked on top of a 0.5 mm sieve. Coarse material was retained by the 2.00 mm sieve and animals passing through were retained by the 0.5mm sieve. The coarse material was carefully searched to remove any attached animals. These animals were added to the contents of the 0.5mm sieve and emptied into a white tray containing a 2 cm depth of tapwater. The tray was divided into sections to enable systematic sorting and estimates of abundance to be carried out where necessary. The macroinvertebrates were removed by hand and most were preserved in 70% industrial methylated spirit (IMS) for subsequent identification. When large amounts of substrate material were present (e.g. algae), the sample was divided and sorted in sections.

Where particular macro-invertebrate taxa were abundant, the first 50 individuals were picked out and preserved and an estimate was made of the abundance of the remaining animals. The extracted animals were removed from defined tray squares to eliminate possible bias towards larger, more visible specimens (Brinkhurst and Kennedy, 1965).

3.4.4. Identification

The level of taxonomic identification (species, genus, family) required for different groups was investigated. It is generally acknowledged that identification to species will yield greater information and that use of higher taxonomic levels can result in a loss of a portion of this information because of differences in the ecological requirements of related species (Resh and Unziger, 1975; Resh and McElravy, 1993). Organisms were therefore identified to species level wherever possible.

Following consultations with NRA and Water Company pollution biologists, the decision was made to identify the following groups to family or order level only: Hydracarina, Sphaeriidae, Simuliidae. This decision was based upon the assumption that the amount of additional information to be gained regarding environmental factors would not be sufficient to justify the amount of time required for identification. Two other groups of organisms: the Chironomidae and the Oligochaeta presented difficulties in identification and were therefore not identified to a lower taxonomic level, although where Oligochaeta consisted largely of Tubificidae, this was noted. Identification was carried out to species level in the following groups (with identification keys used):

Tricladida (Reynoldson, 1978);
Hirudinea (Elliott and Mann, 1979);
Mollusca (Macan, 1977);
Crustacea (Gledhill *et al.*, 1993);
Ephemeroptera (Elliott *et al.*, 1988);
Trichoptera (Wallace *et al.*, 1990; Edington and Hildrew, 1995);
Coleoptera (Friday, 1988);
Hemiptera (Savage, 1989).
Dipteran Larvae (Smith, 1989): generic level.

Due to the difficulties involved in identifying preserved specimens of Hirudinea and Tricladida, these were identified live (Reynoldson, 1978; Elliott and Mann, 1979). All other groups were preserved in 70% industrial methylated spirit (IMS) prior to identification. Samples were stored to enable confirmation or further identification if required.

3.5 Environmental Data Collection

Selected environmental measures were taken at each sampling point and on each sampling occasion. The measures were intended to give a general indication of the environmental parameters on the site and their variability over time. Although in certain cases, the measures may have been related to the pollution incident, direct measurement of the pollutant involved was not undertaken due to the transient nature of the pollutant and in some cases, the uncertainty as to exact nature of the pollutant.

Pollutant

Obvious indications of the pollutant, e.g. the presence of oil during substratum disturbance, and remedial action undertaken by the statutory authority were recorded. NRA information regarding the pollutant concentrations within the watercourse over time was included where this was available.

Current Velocity (m/s)

Water velocity both directly and indirectly represents one of the most important environmental factors affecting organisms of lotic waters (Hynes, 1970; Cummins, 1975; Erman and Erman, 1984; Allen, 1995). However, the current speed to which benthic invertebrates are actually exposed can vary within a short distance, particularly in riffle habitats, where flow is turbulent (Statzner and Higler, 1986). It was not considered necessary, therefore, to give more than a rough estimate of the current velocity within the channel at a sampling point to enable comparisons between sampling points to be made.

Pilot studies were carried out which involved the measurement of the velocity using a 'Sensa - RC2' velocity meter which operates by means of an electro-magnetic sensor. Velocity measurements were taken at various points within a selected riffle and compared with surface velocity as estimated from the time taken for a standard floating object (a 3 cm^2 card disc) to travel a given distance at the centre of the channel. It was found that the surface velocity as measured by the disc was less variable and gave a relatively accurate measure of velocity when compared with the electro-magnetic velocity meter. The floating object method was therefore chosen due to its greater ease of transportation and flexibility of use in a range of water depths. The mean of three measurements was taken and, to provide an estimate of the mean velocity of the channel, the surface velocity thus obtained was multiplied by 0.8 (Allan, 1995).

Channel characteristics

Width (W) and depth (D) measurements were used with mean current velocity (U) to estimate discharge (Q= WDU).

The nature of the substratum was estimated by visual inspection in terms of percentages of silt/sand/gravel/pebble/cobble/boulder/other.

pH

A measure of the concentration of hydrogen ions, hence the acidity or alkalinity of the water, was measured on site using a portable Whatman meter (No. 6602 4220)

Conductivity ($\mu S\ cm^{-1}$ at 25^{o})

A measure of electrical conductance of water and an approximate indicator of total dissolved ions. This was measured on site using a portable Whatman meter (No. 6602 4220). Differences in conductivity result primarily from the concentration of the charged ions in solution, and to a lesser extent from temperature and the composition of ions.

Watershed Characteristics

An estimate of the relative proportions of urban and rural land surface within the catchment of each stream was made using maps. This was included as land cover has an important influence on both the hydrology and the background water quality of the stream (Mason, 1991). Additional environmental and biological information was obtained for each watercourse including location of tributaries, catchment characteristics as estimated from maps, prior pollution history and biological and environmental data from the NRA.

Chapter 4

Investigation of Pollution Incidents

Data Analysis Methods

Analysis of data was carried out on Microsoft Excel, Versions 4 and 5 and Packages of multivariate analysis within Primer (Plymouth Routines In Multivariate Ecological Research, Clarke and Warwick, 1994).

4.1. Introduction

The data collected in the manner described in Chapter 3 is analysed with the following main aims:

1) to ascertain evidence of impact following the pollution incident;
2) to elucidate subsequent changes which may be attributed to recovery following the incident.

The organisation and treatment of the data is described and then an account is given of the overall strategy of analysis adopted.

4.2. Prediction of Impact

A general prediction of the likely impact of the pollution incident on the different biotic measures used was made. This was done by consulting relevant literature on toxicity data and studies following pollution spills of related substances.

4.3. Organisation of data.

4.3.1. Raw data

The data were initially organised into species/samples tables using the spreadsheet software package; Microsoft Excel, Versions 4 and 5. Fig. 4.1 shows a section of a results table with the names of the taxa in the left hand column and the numbers of individuals found in each sample shown in the subsequent columns. The sample code refers to the sampling time, the location of the sampling point and the replicate number. Thus, sample number 2Aa refers to the first replicate taken on the second sampling occasion at Site A. A sample data set for one of the pollution incidents investigated is presented in Appendix C. The assemblage of taxa within a sample is termed the 'community' although this does not imply any structuring of the community.

Figure 4.1: Section of a Results Table

Hewell Brook
31 October 1994

	2Aa	2Ab	2Ac	2Ba	2Bb	2Bc	2Ca	2Cb	2Cc	2Da	2Db	2Dc	2Ea	2Eb	2Ec
Polycelis ten/nigra	0	0	0	0	1	2	3	0	0	0	0	0	4	0	0
Dugesia polychroa	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dendrocoelum lacteum	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Bithynia tentaculata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Potamopyrgus jenkinsi	1	0	0	28	22	106	0	0	0	0	0	0	25	30	17
Lymnaea peregra	1	0	0	10	9	3	1	0	0	1	0	0	12	0	6
Lymnaea stagnalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis vortex	15	19	8	1	1	3	0	0	0	0	0	0	0	0	0

4.3.2. Abundance changes of selected taxa

Ideally, the response of every species of macroinvertebrate to the pollution incident should be investigated to fully understand the ways in which the community responds to short -term anthropogenic disturbance. However, if numbers of a particular taxon are too low throughout the study period, the response of individual taxa is difficult to establish (Norris *et al.*, 1992). The most abundant taxa within each stream investigated were therefore selected for individual analysis for the following reasons:

1. they form a dominant part of the stream fauna in numerical terms and changes in their abundance are likely to have a relatively greater effect upon the stream ecology;

2. changes in their abundance can be assessed statistically;

3. patterns of abundance changes within these taxa following pollution incidents may indicate their utility as indicator organisms;

The drawbacks to choosing the most abundant taxa within each stream are that:

1. the most numerous organisms within a stream are often those having wide tolerances for different environmental conditions, decreasing their potential value as indicator organisms;

2. it is possible that the taxa chosen for individual analysis on the basis of their abundance through the study were not the taxa most susceptible to the pollution incident.

The abundance of an organism within the stream systems studied may be due to a variety of factors: they may form an important component of the standard stream fauna, or their presence or increased abundance may be a direct or indirect result of the pollution incident.

To ensure that the taxa chosen do not exclude those taxa that are a standard component of the pre-pollution ecosystem, but have been reduced in abundance by the incident, the taxa that were most abundant at the reference sites were also selected for analysis.

4.3.3. Proportion and abundance within functional feeding groups

An alternative to the analysis of community structure is the evaluation of energetic relationships within the community. Currently, the primary approach in this area is the analysis of the patterns of functional feeding groups (Cummins, 1975; Cummins and Klug, 1979). The functional component of the benthic community response to a pollution incident therefore can be elucidated by assessing changes in the relative proportion of different functional feeding groups (e.g. filter feeders, shredders, scrapers etc.). However functional analysis presumes a greater understanding of the ecosystem under study than is generally the case. In addition, pilot studies showed that where the impacted streams supported a limited taxa, consideration of individual taxa would provide as much information as ascribing the taxa to a functional feeding group. That is to say, in many cases, each functional feeding group would only be represented by a single taxon. The emphasis within this study is therefore on structural changes within the community.

4.3.4. Univariate Measures

The complexity of multivariate information can be reduced in a number of ways to convey various aspects of the community within a sample. The univariate measures of a sample that were considered for the study are summarised below.

4.3.4.1. Total number of individual organisms

This has been used in a number of studies as a simple way of assessing macroinvertebrate community change (Resh and McElravy, 1993). However, as it can be dominated by variations in the abundance of relatively few taxa, it was considered too crude a measure for the present study.

4.3.4.2. Total number of taxa

Taxa richness, i.e. the number of taxa in a sample, is a simple measure that is widely used in biomonitoring studies (Resh and McElravy, 1993; Reice and Wohlenberg, 1993). The underlying principle in its use in the assessment of impact, is that stable biological communities in streams have high species richness (DeShon, 1995). Its value in discriminating between sites depends firstly upon the level of identification employed within the study, with greatest discrimination found when all taxa are identified to species level, and secondly, upon an understanding of the species richness that would be expected on a site in the absence of pollution impact.

4.3.4.3. Diversity

Diversity indices have been commonly used to contract a wealth of species abundance data into a single figure that has some ecological meaning. Diversity incorporates a measure of the number of species present and a measure of their equitability, i.e. how evenly the individuals are distributed among the different species. Hellawell (1986) gives accounts of the relative values of the different diversity indices. It is acknowledged that diversity values decrease with decreasing water quality (Norris and

Georges, 1993; Mason, 1996) and it is the case that undisturbed environments are characterised by high diversity or richness (Davis, 1995). Pollution often results in a reduction in species numbers and an increase in the abundance of tolerant taxa (Hynes, 1974). This is manifested in a concomitant reduction in the diversity index.

Although a number of workers have suggested that diversity is a descriptive term that does not have ecological validity in many situations (Metcalfe-Smith, 1994; Davis, 1995; Resh and McElravy, 1993; Green, 1979), diversity indices are still widely used in pollution studies (Resh and McElravy, 1993; Norris and Georges, 1993; Metcalfe-Smith, 1994; Victor and Ogbeibu, 1986; Woodward and Riley, 1983). Diversity indices are relatively independent of sample size and unlike biotic indices, can be treated statistically (De Pauw and Hawkes, 1993).

Due to the continuing widespread use of diversity indices, it was decided to include a diversity index in the analysis of the data. The most commonly used index is the Shannon-Wiener diversity index - H' (Shannon and Weaver, 1949; Resh and McElravy, 1993; Norris and Georges, 1993).

$$H' = -\Sigma_i \, p_i(\log p_i)$$

(where p_i is the proportion of the total count arising from the *i*th species).

The value of H' reaches its maximum value when all species are distributed evenly. Biologically this is assumed to be indicative of favourable environmental conditions.

4.3.4.4. BMWP and ASPT

Biotic indices combine a measure of taxa richness, with the tolerance to pollution of the taxa found, and are usually presented as a single number or score. The tolerance to pollution incorporated within the score is based upon known or estimated tolerance to a particular pollutant, usually organic, although as knowledge of organism responses to other pollutants progresses, scores based upon taxa responses to pollutants most pertinent to a particular region are being developed e.g. acid pollution in Norway (Fjellheim and Raddum, 1990). All biotic scores rely upon the knowledge of a species' ecological requirements and its responses to different pollutants.

In Britain, the Biological Monitoring Working Party (BMWP) score is in widespread use within the regulatory authority and water industry as well as in academic studies concerned with pollution. The score is the outcome of the Biological Monitoring

Working Party which was set up in 1976 to develop a standardised system for assessing the biological quality of rivers in England and Wales (DOE Standing Committee of Analysts, 1983).

It is a pollution index based upon the presence or absence of macro-invertebrate families and is designed to give a measure of organic pollution. Each family is given a score between 1 and 10 depending upon their perceived susceptibility to organic pollution (as subjectively allocated by a panel of experts), with the most susceptible, such as stoneflies, given the highest scores and Oligochaeta given the lowest score (see Appendix D). The BMWP score is the sum of the scores within the sample. The Average Score Per Taxon (ASPT) is the BMWP score divided by the number of contributing taxa to give a score that is independent of sample size (a larger sample is likely to include more families thus inflating the BMWP score) and shows less variation over time than the BMWP score (Jones, 1973; Balloch *et al.*, 1976).

The BMWP score does not take abundance into account or incorporate the effects of biotope type although these factors are considered in a recent development of the BMWP score, which also incorporates allocation of an indicator value to each family present within a sample (Walley and Hawkes, 1997). Thus the Oligochaeta and Chironomidae which are frequent across all classes of water quality, have low indicator values which implies that in their present form, (i.e. without further sub-division), they have little value as indicators of water quality.

Although the BMWP score system was developed utilising the response of macroinvertebrates to organic pollution, it is widely used in the United Kingdom for water quality monitoring, including the recently developed Environmental Quality Index. (NRA, 1991) . In addition, it is frequently considered to indicate general water and habitat quality (e.g. IEA, 1995).

Although the majority of the pollution incidents investigated were not organic in origin, because the BMWP score is in widespread use as a measure of general pollution, it was included in the analysis of the data. to provide information regarding the response of this score in the assessment of recovery following pollution incidents.

The replicates from each sample were combined to calculate the BMWP score.

4.3.5. Multivariate Analysis

Univariate indices are appropriate when a measured attribute of the biotic system appears to vary monotonically or uniformly with changes in environmental parameters e.g. pollutant concentration gradients, or time following a pollution incident, and can therefore utilise a linear response model. However, the response of each taxon to environmental changes is not necessarily linear or predictable. The multivariate approach utilises each taxon as a variable and makes no assumptions about response to pollution. Multivariate techniques preserve more of the species information than univariate methods and subtle changes in the species composition or the abundance of particular species between samples are not masked by summarising the community composition into a single value (Norris and Georges, 1993).

Multivariate methods are based upon similarity coefficients calculated between every pair of samples i.e. the extent to which samples share taxa at the same level of abundance, and these are set out in a similarity matrix.

The measures of association between pairs of samples as calculated by the similarity coefficient, can then be used to cluster the samples into groups which share similarities, or to map samples on an ordination plot such that the distance between samples reflects their relative similarities of biotic composition.

The aim of the present study was the assessment of the relationship between samples on a continuous scale, in particular the understanding of patterns of community change over time following pollution impact. Ordination of the samples was therefore considered the most appropriate approach.

Several methods of ordination exist, including the following,

1. Principal Components Analysis (PCA) ; one of the founding techniques of multivariate statistics (e.g. Chatfield and Collins, 1980).
2. Principal Co-ordinates Analysis (PCoA: Gower, 1966);
3. Detrended Correspondence Analysis (DECORANA: Hill, 1979);
4. Multi-Dimensional Scaling (MDS: Shepard, 1962; Kruskal, 1964).

The application of the different methods of multivariate analysis to community ecology is discussed in various texts (e.g. Jongman *et al*, 1995; Norris and Georges, 1993). The present study has utilised non-metric MDS for the following reasons.

1. Non-metric MDS makes few assumptions about the data because only rank order information is used. Studies that have compared various ordination methods recommend the use of MDS for community data (Clarke and Warwick 1994, Kenkel and Orloci 1986).
2. MDS has greater flexibility in the definition and conversion of dissimilarity to distance and the preservation of these relationships in the ordination space than PCA and PCoA. With PCA and PCoA, the solution is optimised in terms of squared distances, so that larger distances between samples are given disproportionate weight (Norris and Georges, 1993). Thus, the first few axes will emphasise distances between natural groupings at the expense of distances between samples within the groupings. This is useful if the study seeks to find clusters within the samples. However, the present study requires a representation of <u>changes</u> between samples rather than the elucidation of clusters.
3. PCA is more suited to multivariate analysis of environmental data than species abundance data as it requires the exclusion of variables which are less common. Abiotic variables are usually few in number (compared with numbers of species), are continuously scaled and do not have a preponderance of zero values (Clarke and Warwick, 1994). With MDS, species deletions are unnecessary as the similarity matrix forms the base rather than the original data assemblage as in the other methods.

The aim of MDS is the construction of a configuration or sample map in which the distances between the samples have the same rank order as the corresponding (dis)similarities between the samples. Thus if sample A has greater similarity to sample B than to sample C, then sample A will be placed closer on the configuration to sample B than to sample C. It is not possible to arrange the sites such that the mutual distances between the sites in the two dimensional configuration are all equal to the calculated dissimilarity values. The 'stress value' is the number that expresses how well or how badly the distances in the MDS configuration correspond to the dissimilarity values. This is calculated by a 'Shepard diagram'; a scatter diagram of the dissimilarities calculated from the species data against the distances between the sites in the ordination diagram. The extent to which the MDS plot is a usable summary of the sample relationships can be indicated by the stress values as follows (taken from Clarke and Warwick, 1994):

- stress < 0.05 gives an excellent representation with no prospect of misinterpretation;

- stress < 0.2 gives a potentially useful 2-dimensional picture, though for values at the upper end of this range too much reliance should not be placed on the detail of the plot;
- stress 0.2-0.3 values should be treated with scepticism and discarded if in upper range;
- stress > 0.3 indicates that the points are close to being arbitrarily placed in the 2-dimensional ordination space.

Nonmetric Multidimensional Scaling is so-called because only the rank order of the dissimilarities or similarities between samples is preserved in the geometric representation. A disadvantage is that the analyst must provide the dimension of the solution in advance and the most appropriate dimension may not be evident.

In the present study, the dimension is the degree of pollution impact (spatial) and time following pollution impact (temporal) so this technique is appropriate particularly if other environmental dimensions or variables are relatively constant.

Similarity matrix The similarity of species/abundance information between two samples can be calculated such that if two samples have no species in common, S = 0%, and if two samples have identical species then S = 100%. Abundance can be used in various ways (absolute numbers or transformed) as well as simple presence or absence.

The similarity coefficient used is the Bray Curtis similarity coefficient which is commonly used in ecological work (Bray and Curtis, 1957):

$$Sjk = 100 \times \left\{1 - \frac{\sum pi=1\ |yij - yik|}{\sum pi=1\ (yij + yik)}\right\}$$

where Sjk is the similarity between the jth and kth sample.

This similarity coefficient varies linearly with changes in species numbers and abundance (Norris and Georges, 1993) and depends upon species which are present in one or both samples and not on species which are absent from both.

The data can be considered in a number of forms, as follows, each giving different emphasis to infrequent species;

1. Untransformed - this was considered the basic data set and, as a known area of the substratum was sampled on each occasion, standardisations such as relative

abundance of each species was not considered necessary. Similarity coefficients calculated from these values may give a disproportionate importance to a small number of highly abundant species.

2. Root transformation, Root/root transformation, log transform (log(1+y)) - these transformations increase the relative importance of the rarer taxa, thus retaining information on the abundance of various taxa whilst giving less importance to the more abundant taxa.
3. Presence/absence - rare and abundant taxa are given the same weighting. This loses valuable information about the abundance of a taxon.

This classification of the samples aims to identify groupings such that samples within a group are generally more similar to each other than samples in different groups. An initial analysis that is important prior to considering any differences related to pollution impact is that replicates within a site form a cluster that is distinct from replicates within other sites.

4.4. Analysis of data

4.4.1. Introduction

This study involved investigation of a single impact on a single stream so that the design is, as is the case with most field studies, is an example of pseudoreplication (Hurlbert,1984). It is not possible for treatments to be allocated randomly such that variables other than the pollution impact, such as flow rate or substrate can be controlled for. To provide independence of samples, separate channels need to be used to provide both reference and unimpacted sites. However, this is not practicable in the situation of this project where the impact of real and unpredictable pollution incidents is being investigated. Thus, although useful indications of possible effects can be determined from this design, it is not sufficiently rigorous to determine cause and effect.

Graphical representations of all spatial and temporal changes of the various measures used, are given. Where patterns are evident, further analysis has been carried out, however it must be emphasised that the problems of design mentioned above: lack of adequate controls, pseudoreplication (Hurlbert, 1984; Cooper and Barmuta, 1993; Norris and Georges, 1993) limit the statistical testing that can be carried out. For this reason it has been suggested that interpretation based on scientific judgement rather than statistical hypothesis testing may carry more weight with regard to rivers than other habitat types (Norris *et al.*, 1992).

4.4.2. Univariate data

For each site, the univariate information derived from taxon counts within a sample can be displayed with means and standard error bars. With replicate samples at different sites or times, it would be possible to test for differences using an Analysis of Variance (ANOVA) after transformation of the data to validate statistical assumptions for parametric tests. However the basic form of the data includes a large numbers of zero entries for most species, making transformation to a normally distributed data set difficult. Analysis of variance and the '*t*' test are not very resistant to outliers (Jongman *et al.*, 1995). In addition, the sampling distribution of many indices is unknown (Norris and Georges, 1993). Norris and Georges. (1993) also point out that the numbers of animals collected at successive samplings in time and space may be correlated highly with the numbers collected in previous samples. Such autocorrelation (Hurlbert, 1984; Stewart-Oaten *et al.*, 1986) may invalidate the use of many parametric statistical tests because the assumption of independence is violated. Nonparametric statistical tests, however, make no assumptions about data distribution (Green, 1979; Clarke and Warwick, 1994; Norris and Georges, 1993), although their use does result in a loss of 'significant efficiency' (Cummins, 1975).

For the reasons described above, in the majority of instances, testing for differences between samples involved the use of a distribution-free method (e.g. the Mann-Whitney test) thus enabling small groups to be compared without any assumptions about their distributional form although assumes the form to be the same.

4.4.3. Multivariate data

As no pre pollution information is available for most of the sites, it is difficult to ascertain with certainty that any apparent changes downstream of the pollution outfall were due to the pollution incident. Even with pre-impact biological information, the assignment of changes to anthropogenic influences as opposed to natural variation, is problematic (Underwood, 1992).

The approach adopted in this study is essentially one of descriptive statistical analysis such that the temporal pattern of change on impacted sites differs from the temporal pattern of change on unimpacted sites in various ways (this assumes that the timescale of the study is such that recovery from pollution impact will be apparent - a valid assumption based upon analysis of the pollution incident data in the literature in Chapter 2)..

Three hypotheses of community change within the impacted sites are proposed;

1. Impacted sites will show a progressively greater similarity to the reference sites over time . This information will be used to determine a time of recovery within a site.

2. Reference sites will show less variability over time than impacted sites (e.g. Wallace *et al.*, 1986).

3. If one assumes that unimpacted sites show greater stability over time than disturbed sites, then within a site that has been impacted, samples should show progressively greater similarity over time. This assumption does not require a reference site for comparison, the spatial configuration of samples within each site over a period of time following the incident is all that is required.

The above mentioned changes in the relationships between samples can initially be assessed by visual examination of the MDS plots based on data that has undergone various degrees of transformation. If required, further analysis can be carried out to assess the statistical difference between samples and the components within those samples that are responsible for the differences.

To test for statistical differences between groups of samples, the ANOSIM (analysis of similarities) test is used (Clarke and Warwick, 1994). This is a test based on a non-parametric permutation procedure applied to the rank similarities between samples in the similarity matrix underlying the ordination of samples. It involves the computation of a test statistic: R, reflecting the observed differences between sites, contrasted with differences among replicates within sites. The R statistic itself can be used as a measure of the degree of separation of sites. Thus, a high value of R indicates that the differences between replicate samples within sites are smaller than the differences between replicate samples from different sites. Thus in this case the replicates show greater within-sample similarity than between-sample similarity and can be averaged and treated as a single sample when constructing the MDS configuration, resulting in a clearer visual representation.

The R statistic is recalculated under different permutations of the sample labels (the programme ANOSIM selects a random sample of the full set of permutations, the default of 5000 permutations is used in this study). The significance level is then calculated by comparing the actual value of R to the distribution of the values of R obtained under permutations of the sample labels: if only t of the T simulated values of R are as large (or larger than) the observed R, then the Null Hypothesis (of no difference in community composition between sites) can be rejected at a significance level of $100(t+1)/(T + 1)\%$.

The weakness of the ANOSIM test is that it is not as powerful as parametric tests e.g. MANOVA in detecting differences and requires a higher degree of replication.

However, as described in the previous section on MDS, nonparametric methods of display and testing make no distributional assumptions and are therefore more suited to the analysis of multispecies assemblages.

After carrying out analyses demonstrating differences between samples, the original data matrix can be re-examined to identify the taxa responsible for the observed differences. This is done by computing the average dissimilarity between all pairs of intergroup samples. The separate contributions from each taxon to the average dissimilarity can then be computed with a measure of how consistently the taxon contributes to the dissimilarity in inter-comparisons of all samples in the groups, i.e. the degree to which it is a good discriminator. These computations are carried out by the PRIMER programme: SIMPER.

Chapter 5

Pollution Incident Investigations - Results

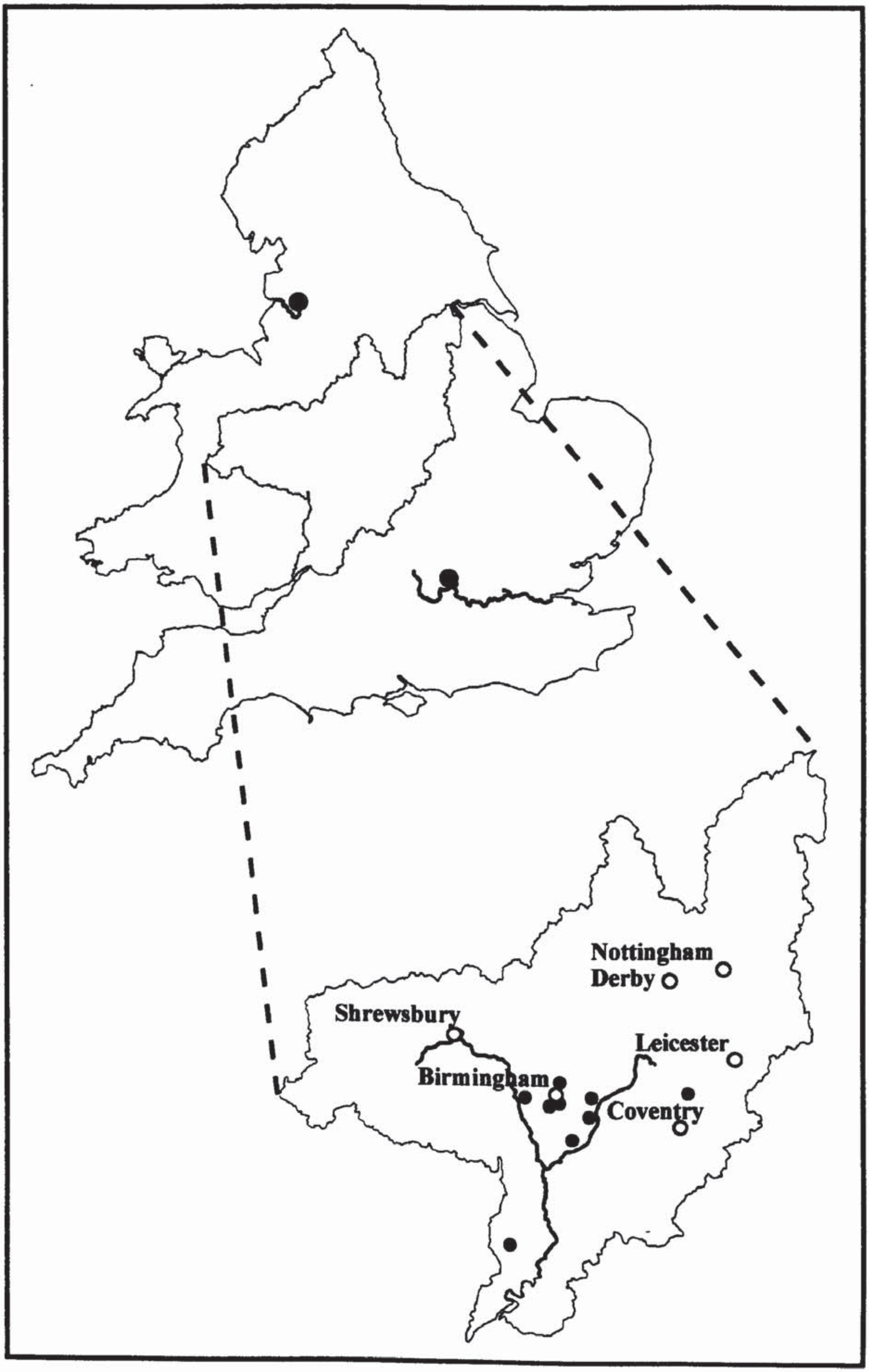
5.1 Introduction

This chapter gives the results of the biotic measures taken following eleven pollution incidents. The pollution incidents are summarised in Table 5.1 and are hereinafter referred to by the name of the watercourse impacted. Figure 5.1 shows the location of the pollution incidents within England.

Table 5.1: Summary of pollution incidents investigated

Name	Pollutant	Date of Incident	First sampling date	Number of visits	Period of sampling
Mousesweet Brook	Hydrochloric Acid	20.3.94	27.3.94	6	9 months
Callow Brook	Heating Oil	16.5.94	18.5.94	2	1 week
Spark Brook	Sewage	14.7.94	21.7.94	3	3 months
Preston Bagot Brook	Hydrochloric Acid	25.8.94	12.10.94	8	12 months
Chinn Brook	Unknown	3.8.94	9.8.94	6	9 months
Hewell Brook	Kerosene	2.10.94	17.10.94	9	20 months
River Wye	Cyanide	2.12.94	13.12.94	4	15 months
Canley Brook	Lubricating Oil	25.3.95	17.11.94	7	13 months
Sketchley Brook	Fuel Oil	10.7.95	13.7.95	7	12 months
Mythe Tributary	Diesel Oil	10.11.95	14.11.95	5	12 months
River Lostock	Pesticide	13.1.96	23.2.96	4	14 months

Figure 5.1: Map of England and Wales to show location of pollution incidents



Chapter 5 is divided into sections 5.2-5.12, each describing the results found with one of the pollution incidents. The sections of the chapter are alphabetically ordered according to the name of the affected watercourse. Three pollution incidents were investigated as part of a pilot project for the study (Callow Brook, Mousesweet Brook, Spark Brook). Although the sampling methodology used in two of these incidents differed from the methodology subsequently used, the results are included as it was considered that the data obtained contributed towards the aims of the project.

The source, immediate impact and any remedial action, is described for each pollution incident. A description of the impacted watercourse is given with the sampling procedure and timing. A general prediction of the kinds of changes in the biotic measures that would be expected from the particular pollutant investigated is made. A sample data set is presented in Appendix C and a full data set is available from the School of Engineering, Aston University. The analysis of the results is presented in the sections of this chapter and the interpretation and discussion of the results is covered in Chapter 6.

5.2 Callow Brook

5.2.1 Description of Incident

On the 17 May, 1994, a spill of heating oil from a hospital was released into a surface water sewer discharging into the Callow Brook. Remedial action by NRA involved the placement of booms downstream of the outfall with associated absorbent mats. The amount of oil released was not known.

5.2.2 Site Description and History

Callow Brook is a first order stream which joins the River Rea in Birmingham, forming part of the River Tame Catchment. The catchment of the Brook is urban in nature, comprising residential and retail estates and public open space.

5.2.3 Sampling Procedure

Four sampling sites were selected, two upstream of the outfall and two downstream of the outfall (Fig. 5.2.1). Five Surber samples were taken at each sampling site. The site was first visited one day after the spill was reported to the NRA, when oil was still found to be trickling from the outfall. The second sampling visit occurred one week later. Imminent and extensive building works directly adjacent to the investigated length of the brook resulted in the necessary abandonment of further investigations. The environmental parameters of the sampling sites are summarised in Table 5.2.1.

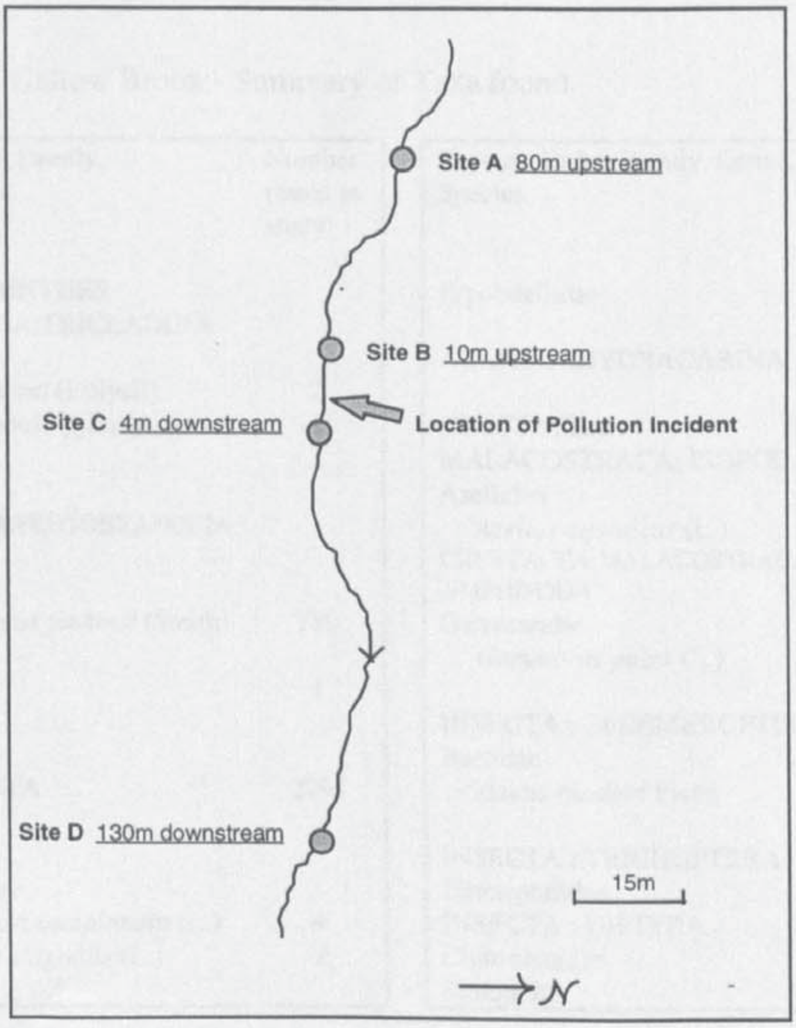
5.2.4 Predicted Impact of Pollution

A small amount of oil was released, and although it caused a slick for some distance downstream of the incident, little was observed on the surface of the substratum. At sampling Site C, however, oil was released from the substratum during disturbance.

Table 5.2.1: Callow Brook - Summary of Environmental Parameters at each sampling point

Site Name	Location (G.R.)	Distance from pollution incident	Substrate	Depth (cm)	Width (cm)	Flow m/sec (18/5/94)	Q (cu.m/sec) (18/5/94)	Temp. ^oC
Site A	SP 275 776	80m upstream	90%pebble/ cobble, 10%sand	5	175	0.33-.255	0.0217	9-9.3
Site B	SP 311 774	10m upstream	90% pebble/ cobble, 10%sand	5-7	200	0.3-.288	0.027	8.8-9.3
Site C	SP 291 777	4m downstream	90% pebble, 10% cobble	3-7	200	0.3-.278	0.0225	8.1-9.3
Site D	SP 296 775	130m downstream	90% pebble, 10% cobble	3-7	200	0.33-.24	0.0248	8.6-9.4

Figure 5.2.1: Callow Brook - Map to show location of Pollution Incident and Sampling Points



5.2.5 Results and Analysis

5.2.5.1. Selected Taxa

Table 5.2.2 is a summary of the taxa found and their total abundance. 15 taxa were found throughout the study. The six most abundant taxa are listed below and changes in their presence and abundance throughout the study are shown in Fig. 5.2.2. Significance tests (Mann-Whitney U test) were carried out to compare abundance levels at each impacted site at each sampling time with the combined reference sites (Table 5.2.3). A summary of the impact on each selected taxon is given in Table 5.2.4.

Asellus aquaticus	*Gammarus pulex*
Baetis rhodani	Oligochaeta
Chironomidae	*Potamopyrgus jenkinsi*

Table 5.2.2: Callow Brook - Summary of Taxa found

Phylum, Order, Family, Genus, Species	Number found in study
PLATYHELMINTHES	
TURBELLARIA:TRICLADIDA	
Planariidae	
Polycelis felina (Dalyell)	2
Polycelis tenuis (Ijima)/*nigra* (Muller)	1
MOLLUSCA	
GASTROPODA:PROSOBRANCHIA	
Hydrobiidae	
Potamopyrgus jenkinsi (Smith)	780
BIVALVIA	
Sphaeriidae	1
ANNELIDA	
OLIGOCHAETA	2295
HIRUDINEA	
Glossiphoniidae	
Glossiphonia complanata (L.)	4
Helobdella stagnalis (L.)	2

Phylum, Order, Family, Genus, Species	Number found in study
Erpobdellidae	34
ACARINA:HYDRACARINA	16
CRUSTACEA: MALACOSTRACA: ISOPODA	
Asellidae	
Asellus aquaticus (L.)	96
CRUSTACEA:MALACOSTRACA: AMPHIPODA	
Gammaridae	
Gammarus pulex (L.)	83
INSECTA : EPHEMEROPTERA	
Baetidae	
Baetis rhodani Pictet	242
INSECTA : TRICHOPTERA	
Limnephilidae	4
INSECTA : DIPTERA	
Chironomidae	130
Simuliidae	1

Table 5.2.3: Callow Brook - Summary of Significance tests (Mann-Whitney U-test) carried out to compare the abundance of selected taxa at combined reference sites (A and B) with each impacted site (C and D) (s = significant difference $p<0.05$, ns = not significant)

Taxa selected	A1,B1 compared with C1	A1,B1 compared with D1	A2,B2 compared with C2	A2,B2 compared with D2
Asellus aquaticus	ns	ns	s	s
Baetis rhodani	ns	ns	s	ns
Chironomidae	ns	ns	ns	s
Gammarus pulex	s	ns	ns	s
Oligochaeta	ns	ns	ns	ns
Potamopyrgus jenkinsi	ns	s	ns	ns

Asellus aquaticus

The incident does not appear to have an immediate impact on the abundance of *A.aquaticus* in that no significant difference is found when the impacted sites are compared with the reference sites one day after the incident. However, after 7 days, the impacted sites are significantly different to the reference sites with an increase in *A. aquaticus* seen at the site close to the outfall (C) and a complete absence of *A.aquaticus* at Site D.

Baetis rhodani

The abundance of *B.rhodani* at the impacted sites is not significantly different to that at the reference sites one day after the incident. Seven days after the incident, the abundance at C is significantly different to that of the reference sites due to higher abundance seen at the impacted sites.

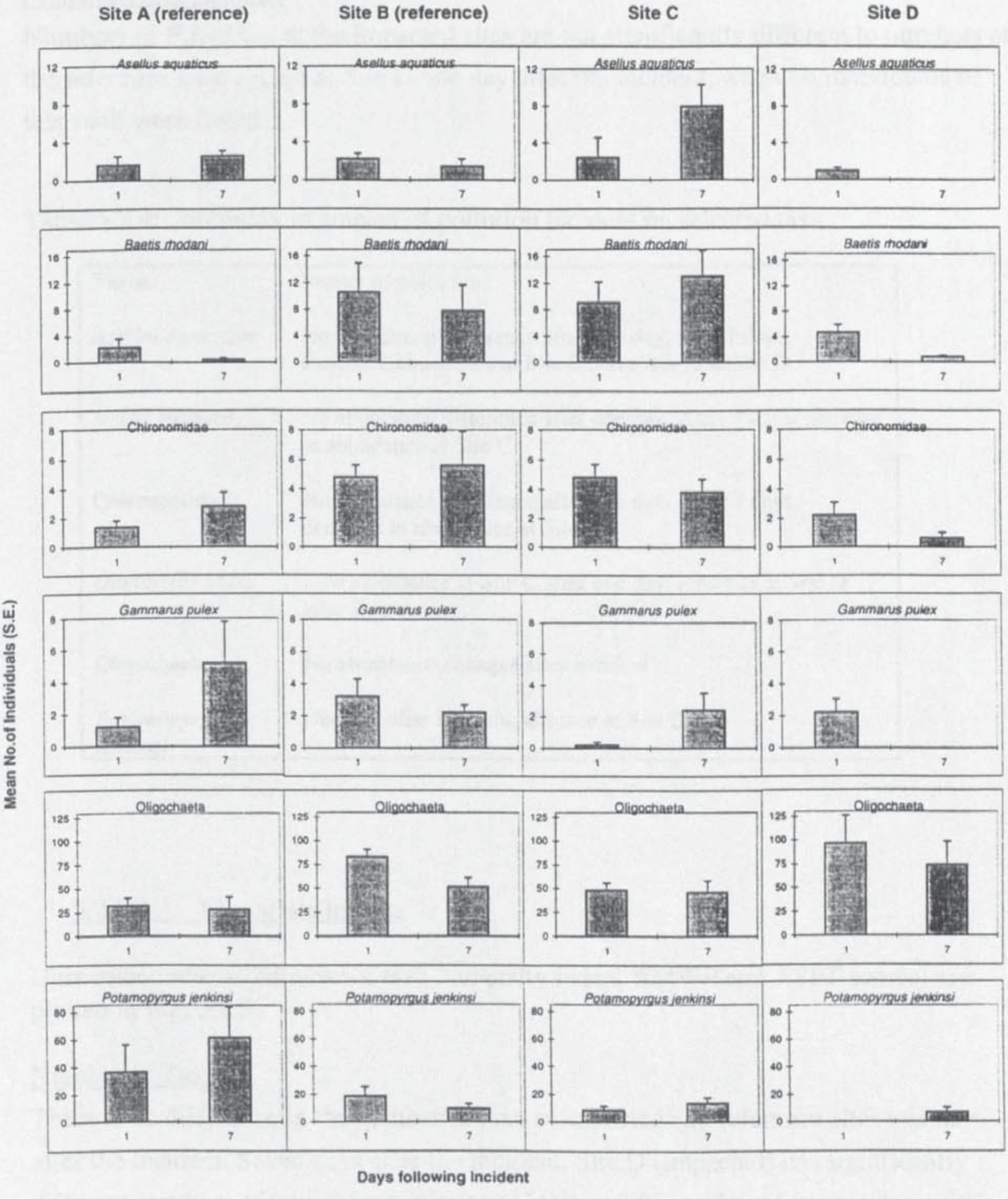
Chironomidae

The abundance of Chironomidae at the impacted sites is significantly different to that of the reference sites at Site D, seven days following the incident (lower abundance at the impacted site).

Gammarus pulex

The abundance of *G. pulex* at Site C is significantly different (lower abundance) to the reference sites one day after the incident, showing a possible immediate impact of the pollution. At impacted Site D, there is an absence of *G. pulex* 7 days after the incident.

Figure 5.2.2: Callow Brook: Abundance Changes in Selected Taxa following Pollution Incident



Oligochaeta

The abundance of Oligochaeta at the impacted sites shows no difference to the reference sites at any time.

Potamopyrgus jenkinsi

Numbers of *P.jenkinsi* at the impacted sites are not significantly different to numbers at the reference sites except at Site D one day after the incident, when no individuals of this snail were found.

Table 5.2.4: Summary of Impact of pollution incident on selected taxa

Taxon	Impact of pollution
Asellus aquaticus	No abundance difference after one day, after 7 days, increased abundance at Site C and absence at Site D
Baetis rhodani	No abundance difference after one day. After 7 days, increase in abundance at Site C.
Chironomidae	No abundance difference after one day. After 7 days, decrease in abundance at Site D.
Gammarus pulex	Low abundance at Site C after one day. Absence at Site D after 7 days.
Oligochaeta	No abundance changes after incident
Potamopyrgus jenkinsi	One day after incident, absence at Site D.

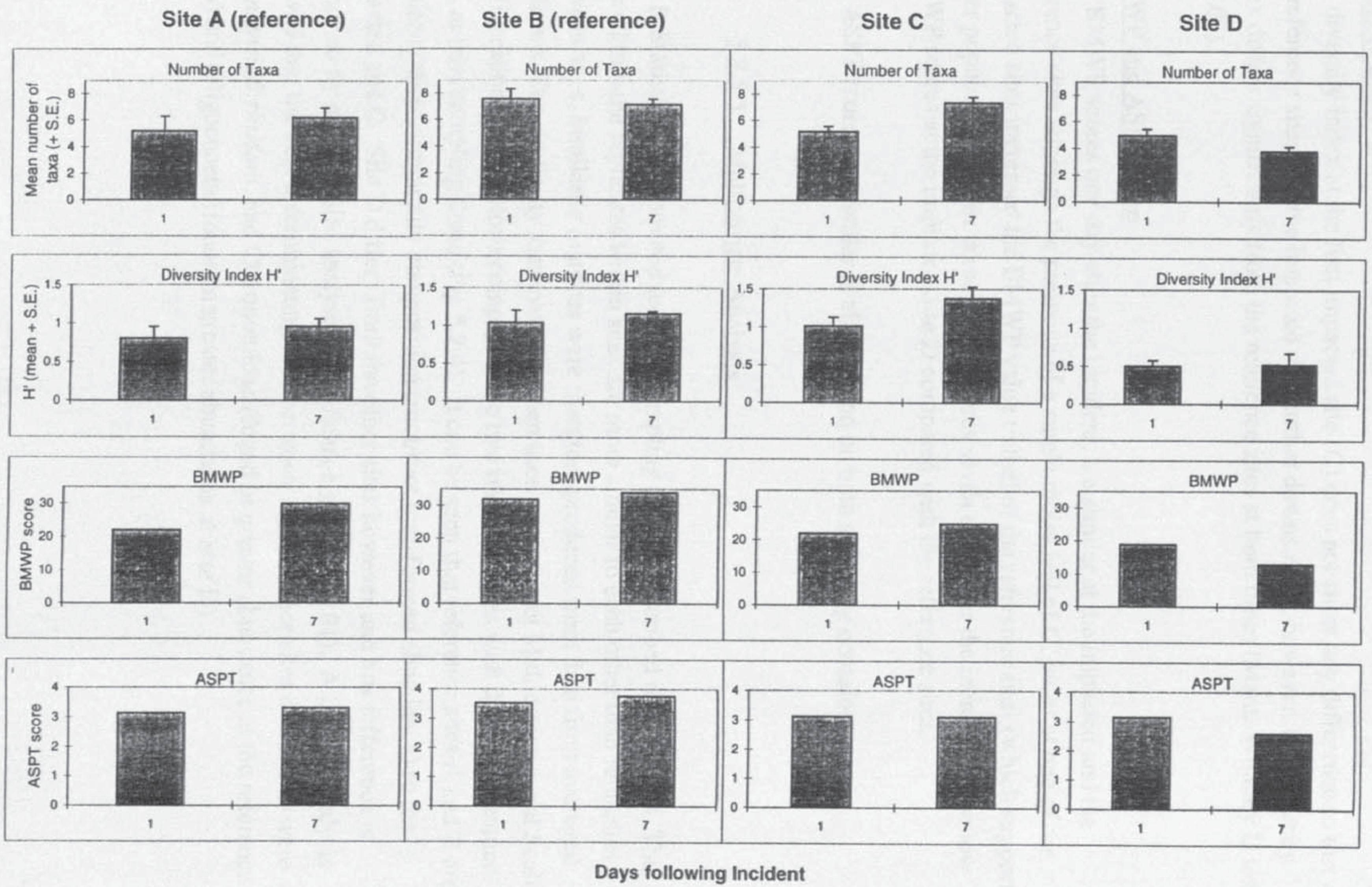
5.2.5.2 Univariate Indices

Univariate indices (Number of taxa, Diversity index, BMWP and ASPT scores) are plotted in Fig. 5.2.3.

Number of Taxa

There is no difference in the number of taxa at impacted and reference sites one day after the incident. Seven days after the incident, Site D (impacted) has significantly different numbers of taxa (Mann-Whitney U test $p<0.05$) to the reference sites, with fewer taxa present.

Figure 5.2.3: Callow Brook - Changes in Univariate Measures following Pollution Incident



Diversity Index

The diversity index at the first impacted site (C) does not show any difference to that of the reference sites. At the impacted site further downstream, however, the diversity index differs significantly from the reference sites at both times (Mann-Whitney U test $p<0.05$).

BMWP and ASPT score

The BMWP scores one day after the incident, are similar at the impacted and the reference sites although the presence of a single individual of *G. pulex* at one of the impacted sites increased the BMWP value to that of the reference sites (which supported larger populations of this taxon). The scores seven days after the incident, show low BMWP scores at the impacted Site D compared with the reference sites.

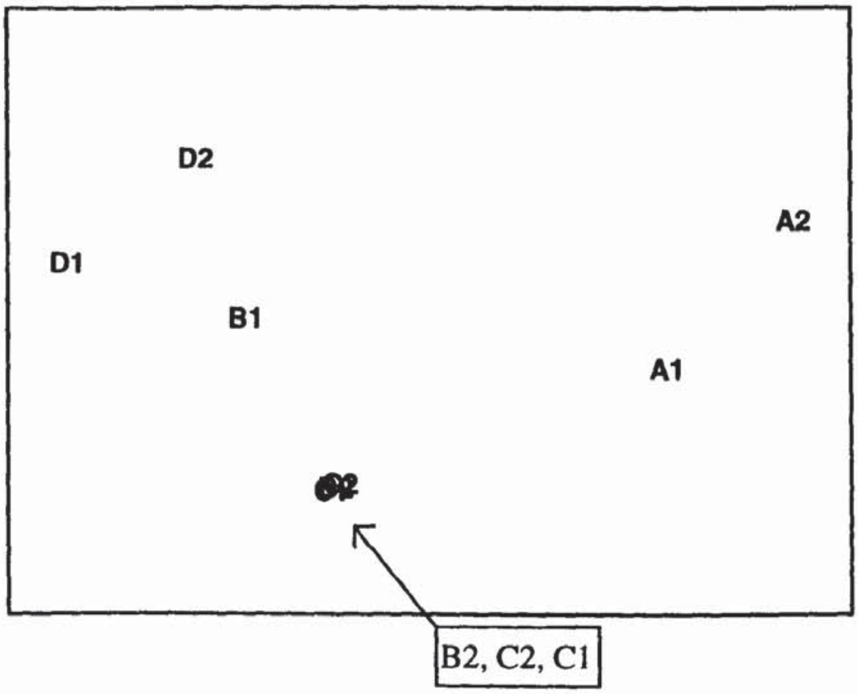
The ASPT scores are similar at all sites and on both sampling occasions.

5.2.5.3. Multivariate Analysis

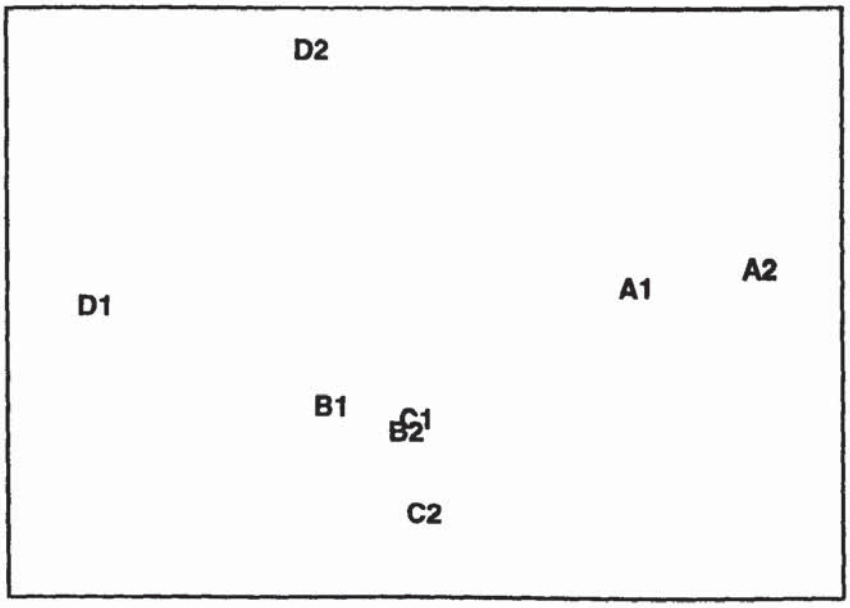
The R Statistic was computed for each sampling time as described in Chapter 4. This showed that the replicates within sites are more similar to each other than replicates between sites. Similarity matrices were therefore produced from data from averaged replicates. The similarity matrices were then used to construct Multidimensional Scaling configurations (MDS plots) comparing the two impacted sites with the two reference sites at both sampling times (Fig 5.2.4). It can be seen that reference sites A and B are similar in their community composition (root/root transformed abundance) to the impacted site C. Site D differs from the other sites however and this difference is supported by an ANOSIM analysis (Significance level $p < 0.01$). A Simper analysis showed that the best discriminating taxa between the reference sites and Site D were *P.jenkinsi*, *B.rhodani*, and Chironomidae (found in greater abundance at the reference sites) and Oligochaeta (found in greater abundance at Site D).

Figure 5.2.4: Callow Brook - MDS configuration of Sites A, B (reference) and Sites C and D

No transformation. Stress value = .01



Root/root transformation. Stress = .05



5.2.6 Summary

Callow Brook is a stream of relatively poor biological quality with intermittent pollution from surface water sewers draining urban areas. Table 5.2.5 summarises the pollution impact on all the biotic measures used.

A varied pattern of impact is shown by changes in abundance of the selected taxa. Oligochaeta show no evidence of pollution impact. *Gammarus pulex* shows a decrease in abundance at Site C, the most impacted site, one day after the incident with only one individual present and no impact further downstream at Site D. After seven days, however, the situation is reversed and Site D is found to support no *G. pulex*, with Site C showing numbers similar to reference sites. *Asellus aquaticus* shows no initial impact but a decrease in abundance at Site D after seven days and an *increase* in abundance at Site C. *Baetis rhodani*, similarly shows no initial impact after one day but after seven days shows an increase in abundance at Site C when compared with reference sites. Chironomidae show a relative decrease in abundance at the downstream impacted site (Site D) after seven days.

Univariate measures present a more simplified picture of pollution impact with lower numbers of taxa, lower diversity (H') and lower BMWP scores at Site D, 7 days following the incident. Diversity indices at Site D are also lower one day after the incident when compared with the reference sites. Multivariate measures show a similar impact at Site D, with the communities at this site being significantly different to the reference communities on both sampling occasions following the incident.

The above analysis shows that both univariate and multivariate assessment mask the relative increases or decreases in abundance of each taxon, which may vary according to site.

The decreases in abundance of selected taxa may have been due to the toxic fractions of the oil. Disturbance of the substratum at Site C released oil into the water on both sampling occasions. The origin of this oil is not known: it may have come from the present oil spill or from previous, unrecorded oil spills.

The increase in abundance of *A.aquaticus* and *B.rhodani* may have been the result of a potential food source resulting from biodegradation of the oil.

Table 5.2.5: Callow Brook - Summary of pollution Impact on all measures

Measure	Type of impact	Spatial Impact	Temporal Impact
Individual Taxa	No impact after one day then an increase in abundance at seven days (*A.aquaticus, B. rhodani*)	C	
	No impact after one day then a decrease in abundance at seven days (*A.aquaticus*,chironomidae, *G.pulex*)	D	
	Decrease in abundance after one day (*G.pulex*)	C	
	Decrease in abundance after one day (P.jenkinsi)	D	
	No evidence of impact (Oligochaeta)		
Number of Taxa	Low numbers of taxa	D	at 7 days
Diversity (H')	Low diversity index	D	at 7 days
BMWP, ASPT	Low BMWP score No change in ASPT scores	D	at 7 days
Multivariate Analysis	Significant difference in communities when compared with reference sites.	D	3 and 7 days

5.3 Canley Brook

5.3.1 Desciption of Incident

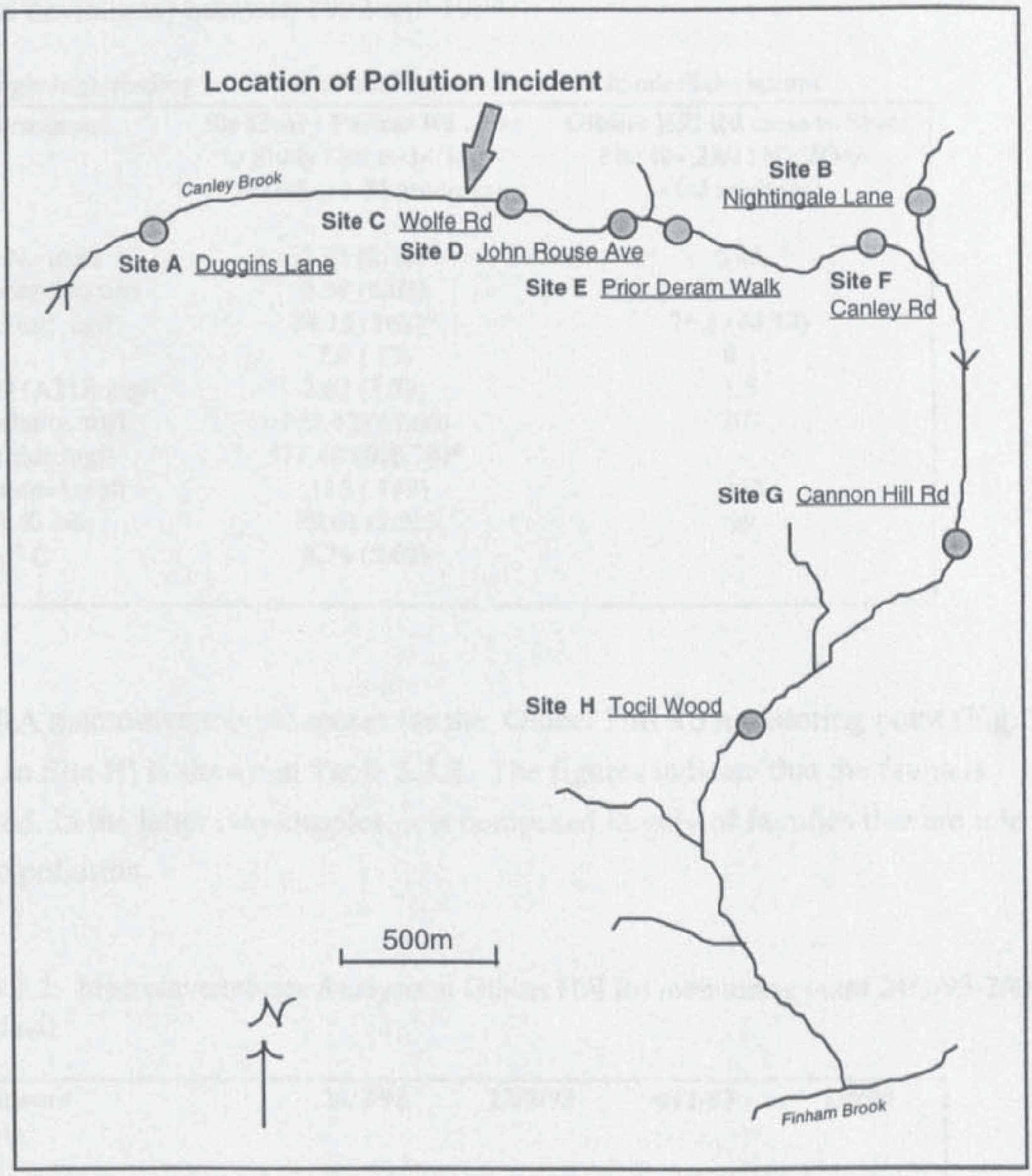
On 25 March 1995, the Canley Brook in Coventry, West Midlands suffered a spill of suds oil. The spill was transitory in nature and had entered a culverted section of the brook passing through an industrial estate, although the company responsible could not be located (Fig. 5.3.1). The amount of pollutant entering the brook and its exact nature could not therefore be determined. The assumption was that the highly visible milky emulsion reported by members of the public was suds oil as this had been a previous cause of pollution in the brook.

A fish mortality investigation carried out by the NRA on 26 March revealed 900 dead stone loach, 3 sticklebacks and 1 gudgeon over a length of 5 km. In addition, further dead fish were seen from A429 road bridge, a distance of 5.7 km from the pollution source. During the first sampling visit on 31 March, 1995, 6 days after the incident, dead Stone Loach (*Neomachilus barbatulus*) and Three-spined Stickleback (*Gasterosteus aculeatus*) were seen at the first sampling point downstream of the incident (Site C, refer to Fig. 5.3.1). Dead invertebrates were also seen here: erpobdellid leeches, *Gammarus pulex* and *Lymnaea peregra*. At Site E, dead Three-spined Sticklebacks were seen 6 days after the incident.

5.3.2 Site Description and History

Canley Brook is a first order stream, 7.8 km in length before its confluence with Finham Brook which forms part of the River Avon catchment in Warwickshire. It arises in farmland to the southwest of Coventry and flows through arable, pasture and gardens for less than a kilometre before entering a 700 m culvert through an industrial estate. The brook emerges from the culvert to flow within an urban area consisting of residential and industrial estates, largely through open areas comprising public open space and a golf course. The brook then flows through agricultural land before joining Finham Brook close to Kenilworth. The underlying geology comprises Mercia Mudstones, a predominantly impermeable geology which, with the urban nature of the catchment leads to fast runoff after heavy rainfall (NRA, 1994c).

Figure 5.3.1: Canley Brook - Map to show location of Pollution Incident and Sampling Points



Water quality information collected at two monitoring points by the NRA is shown in Table 5.3.1. The General Quality Assessment for the upper 4 km reach of the stream is Grade C (1994) and the River Quality Objective is RE4 (NRA, 1995). The low quality of the stream is due to urban surface water discharges and intermittent pollution incidents (NRA pollution control officer, personal communication).

Table 5.3.1: NRA Water Quality Information at two monitoring points (Mean and standard deviations) between 1992 and 1994.

*a single high reading in 1996 increased mean values and standard deviations

Determinand	**Sir Henry Parkes Rd** close to Study Site F -14/1/92-29/2/96 (11-31 readings)	**Gibbet Hill Rd** close to Study Site H - 23/11/93-2/3/94 (1-2 readings)
T.O.N. mg/l	5.95 (2.15)	6.45
Copper(filt) ug/l	6.38 (6.08)	-
Zinc(tot) ug/l	74.15 (162)*	76.8 (44.12)
pH	7.9 (.17)	8.1
BOD (ATU) mg/l	2.67 (3.73)	1.5
Alkalinity mg/l	175.42 (45.66)	207
Chloride mg/l	171.48 (608.78)*	-
Ammonia mg/l	.113 (.149)	.155
D.O. % satn	89.61 (7.98)	99
temp° C	8.74 (3.62)	-

The NRA macroinvertebrate scores for the Gibbet Hill Rd monitoring point (Fig. 5.3.1 - close to Site H) is shown in Table 5.3.2. The figures indicate that the fauna is restricted. In the latter two samples, it is composed largely of families that are tolerant of organic pollution.

Table 5.3.2: Macroinvertebrate Analysis at Gibbet Hill Rd monitoring point 24/3/93-7/4/94 (NRA data)

Measure Date	**24/3/93**	**13/7/93**	**4/11/93**	**7/4/94**
Number of Taxa	15	14	6	5
Number of BMWP Taxa	14	13	6	5
BMWP score	56	53	22	16
ASPT score	4.00	4.08	3.67	3.2

5.3.3 Sampling Procedure

Three replicate Surber samples were taken from each of 8 sites (see Fig. 5.3.1). Two reference sites were chosen; one upstream of the pollution incident outfall on the Canley Brook (Site A) and one on a small tributary of the Canley Brook (Site B). The 6 impacted sampling sites (Site C-H) covered a distance of 4.6 km from the source of the pollution incident.

The environmental parameters of the sampling sites are summarised in Table 5.3.3. All sampling points were situated in riffle areas. The two reference sites had lower flow and discharge values than the other sites due to their upstream locations.

Table 5.3.3: Canley Brook - Summary of Environmental parameters at each sampling point

Site Name	Location (G R)	Distance from pollution incident (m)	Substrate	Ave. Depth (cm)	Width (cm)	Flow m/sec (26/3/96)	Q (cu.m/sec) (26/3/96)
Site A (reference) - Duggins Lane	SP 275 776	1700 m upstream	50% sand, 40% pebble, 10% gravel	4	60	0.085	0.00612
Site B (reference) - Nightingale Lane	SP 311 774	tributary	50% pebble, 30% sand, 20% gravel	4	70	0.05	0.0042
Site C - Wolfe Rd	SP 291 777	50m downstream	40% pebble, 20% cobble, 20% boulder/brick, 20% sand/silt	6	80	0.125	0.0168
Site D - John Rouse Ave	SP 296 775	600m downstream	80% pebble, 10% boulder, 10% gravel/sand	12.5	80	0.225	0.036
Site E - Prior Deram Walk	SP 298 775	750m downstream	80% pebble, 10% gravel, 10% sand	6.5	200	0.1	0.0368
Site F - Canley Rd	SP 309 773	2000m downstream	50% sand, 40% pebble, 10% boulder	4	200	0.125	0.044
Site G - Cannon Hill Rd	SP 312 762	3400m downstream	40% pebble, 30% gravel/sand, 20% boulder	10	300	0.1	0.048
Site H - Tocil Wood	SP 301 754	4600m downstream	80% pebble, 10% sand/gravel, 10% boulder	25	100	0.225	0.054

Canley Brook was known by the pollution control department of the NRA for its frequency of reported pollution incidents. In 1994, pollution control officers suggested that it would be worth sampling the brook at various points to ascertain 'background' levels of macroinvertebrate fauna. One sampling visit to certain sites (A, B, D, F, G, H) was therefore undertaken in November 1994, 5 months *before* the pollution incident. Five further sampling visits were distributed over a period of 367 days following the pollution incident (Table 5.3.4). Samples 1 and 5 are seasonally comparable, being taken in March of consecutive years.

Table 5.3.4: Summary of Sampling Times

Sampling time Code	Date		Days following incident
pre impact	11 November	1994	127 days before incident
Ca1	31 March	1995	6
Ca2	16 May	1995	52
Ca3	7 July	1995	101
Ca4	29 September	1995	188
Ca5	26 March	1996	367

5.3.4 Predicted Impact of the Pollution Incident

Suds oil is a water extendable cutting fluid used for lubrication in metalworking. It is manufactured from mineral oils, alkaline surfactant emulsifiers and additives and forms a stable emulsion when added to water. Hexylene Glycol (2-methylpentan-2,4-diol) and bactericides may be present. When released to water, the minerol oil will disperse as an emulsion and the components will not evaporate to any great extent. Dissolved components may be absorbed on to sediment and will biodegrade in aerobic conditions. Mineral oil is described as 'not toxic to aquatic organisms but all components have a high potential to bioaccumulate'(Shell, 1993).

As little is known of specific toxic effects of Suds oil, the generalised predictions of the impact of oil pollution on the measures used in this study, as discussed in Section 5.5 (Hewell Brook), are applied to this incident and summarised in Table 5.3.5.

5.3.5 Results and Analysis

The data is organised as described in Chapter 4. Impact of the pollution incident can be assessed by comparison with the pre impact samples and with the two upstream reference samples. In addition, changes between the samples taken 6 days after the incident and 367 days after the incident are compared as they are seasonally comparable.

5.3.5.1. Water Quality

Table 5.3.6 gives the physicochemical measurements taken at selected sampling times. Temperature varies seasonally. The pH measurements are variable at all sites with a mean of pH 7.79 which is comparable to NRA data (Table 5.3.1). Conductivity measurements are relatively high and variable at both reference and impacted sites.

Table 5.3.5: Predicted Impact of Pollution Incident on Biotic Measures

Measure	Predicted effect
Individual taxa	Initial toxicity causes loss of sensitive taxa and decrease in abundance of less sensitive taxa. This is followed by a rapid increase in those taxa able to benefit from the changed conditions and then showing a decline, and a more gradual increase in other taxa as conditions resemble those prior to the incident.
Number of Taxa	Initial low numbers followed by a gradual increase.
Diversity	Initial low diversity followed by a gradual increase.
BMWP and ASPT	Initial low BMWP score and ASPT score as taxa are lost and any taxa are dominated by tolerant taxa, followed by an increase over time.
Multivariate Analysis of Community Similarity	1. Upstream 'control' sites will show less variability over time than impacted sites (impact). 2. Impacted sites will show a progressively greater similarity to the 'control' sites over time (temporal recovery). This information can be used to determine a time of recovery within a site. 3. There will be progressively less variability and less movement towards similarity between impacted and control sites, the further downstream of the impact the sampling point is located (Spatial recovery). 4. Progressive increase in similarity within impacted site i.e. within an impacted site those samples closer in time to the pollution incident will show less similarity to each other than those samples further in time.

5.3.5.2. Selected Taxa

Table 5.3.7 is a summary of the taxa found and their total abundance. 33 taxa were found throughout the study. Six taxa were selected and changes in their presence and abundance through the study are shown in Fig. 5.3.2 and described below.

Asellus aquaticus	Oligochaeta
Chironomidae	*Potamopyrgus jenkinsi*
Gammarus pulex	Sphaeriidae

Numbers of taxa were compared in 3 ways:

- between the pre-incident sample and the sample taken 6 days after the incident,
- between impacted sites and reference sites,
- between impacted sites 6 days and 367 days after the incident (i.e. seasonally comparable).

Table 5.3.6: Physicochemical Conditions in Canley Brook.
(Conductivity is in $\mu S/cm^3$)

		Site A			Site B		
Date	Days following Disturbance	Temp(°C)	pH	Cond.	Temp (°C)	pH	Cond.
16-May-95	52	9.4	8.2	518	9.8	7.8	491
07-Jul-95	101	13.5	8	603	n/a	n/a	n/a
29-Sep-95	188	10.2	7.4	296	10.4	7.5	640
26-Mar-96	367	5.2	8.4	509	5.9	8.1	436

		Site C			Site D			Site E		
Date	Days following Disturbance	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond.
16-May-95	52	10.9	7.8	549	10.8	7.7	573	11	8.3	564
07-Jul-95	101	14.4	7.9	635	14.4	7.3	645	14.2	6.8	618
29-Sep-95	188	13	7.7	628	12.1	7.8	645	11.9	7.9	540
26-Mar-96	367	5.8	n/a	499	5.5	n/a	506	5.3	n/a	501

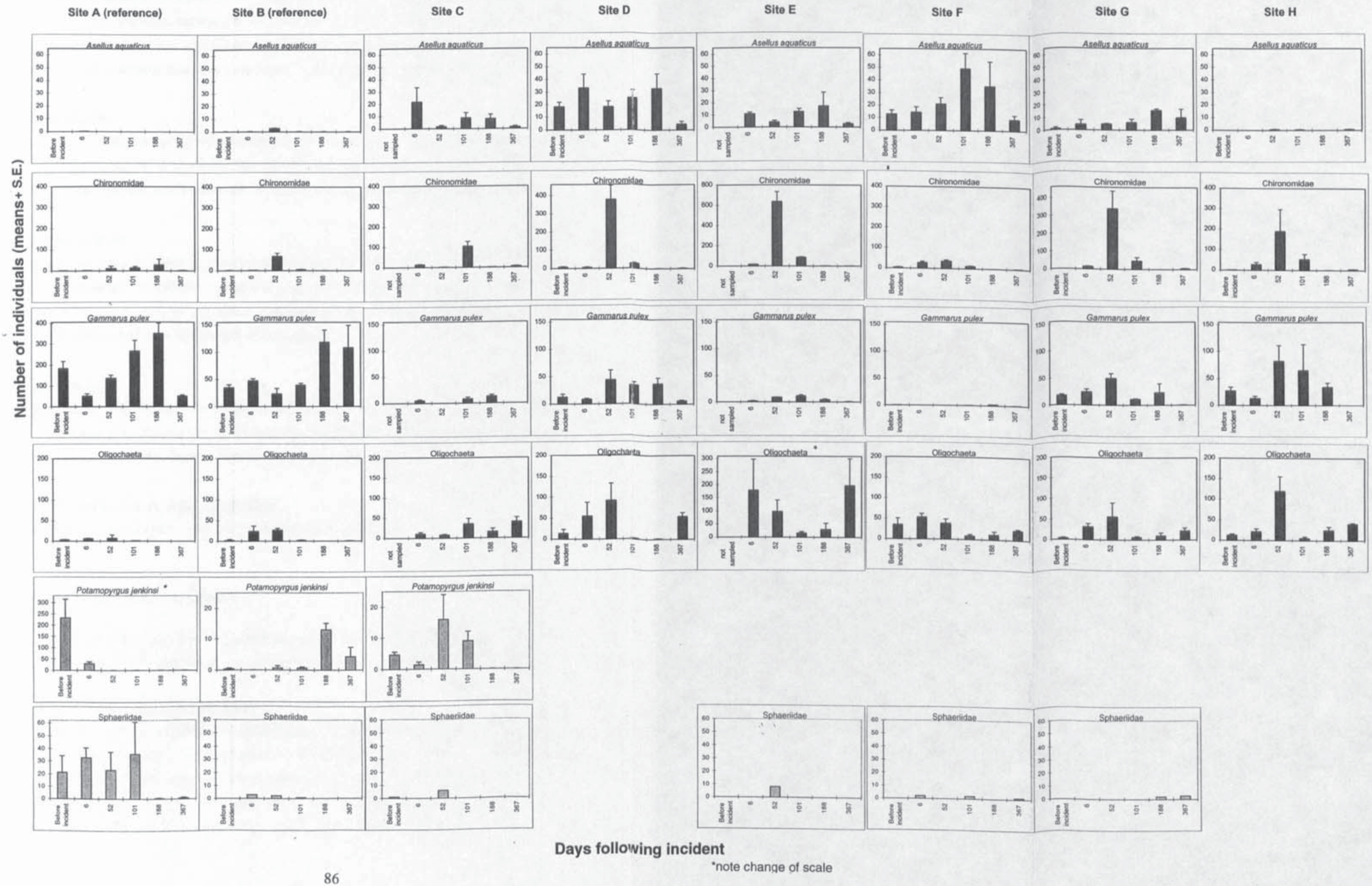
		Site F			Site G			Site H		
Date	Days following Disturbance	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond.
16-May-95	52	10.2	8	551	10.6	n/a	524	10.1	8.2	504
07-Jul-95	101	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
29-Sep-95	188	11	7.4	646	10.7	7.7	478	10.5	7.6	634
26-Mar-96	367	4.8	n/a	436	5.1	n/a	514	5.1	8.1	491

Table 5.3.7: Canley Brook - Summary of Taxa found

Phylum, Order, Family, Genus, Species	Number found in study
PLATYHELMINTHES	
TURBELLARIA:TRICLADIDA	
Planariidae	
Polycelis felina (Dalyell)	26
Polycelis tenuis (Ijima)/*nigra* (Muller)	7
MOLLUSCA	
GASTROPODA:PROSOBRANCHIA	
Hydrobiidae	
Potamopyrgus jenkinsi (Smith)	1191
GASTROPODA:PULMONATA	
Lymnaeidae	
Lymnaea palustris (Muller)	1
Lymnaea peregra (Muller)	615
Lymnaea truncatula (Muller)	1
Ancylidae	
Ancylus fluviatilis Muller	24
Physidae	
Physa sp.	3
BIVALVIA	
Sphaeriidae	389
ANNELIDA	
OLIGOCHAETA	4357
HIRUDINEA	
Glossiphoniidae	
Glossiphonia complanata (L.)	367
Erpobdellidae	43
Erpobdella octoculata (L.)	2
ACARINA:HYDRACARINA	93
CRUSTACEA: MALACOSTRACA: ISOPODA	
Asellidae	
Asellus aquaticus (L.)	1230
CRUSTACEA:MALACOSTRACA: AMPHIPODA	
Gammaridae	
Gammarus pulex (L.)	5909

Phylum, Order, Family, Genus, Species	Number found in study
INSECTA : MEGALOPTERA	
Sialis lutaria (L.)	1
INSECTA : EPHEMEROPTERA	
Baetidae	
Baetis rhodani Pictet	173
Ephemerellidae	
Ephemerella ignita (Poda)	1
INSECTA : COLEOPTERA	
Dytiscidae	1
Dytiscidae larva	46
Hydroporus sp.	5
Elmidae	
Elmis sp. adult	1
Elmis aenea Muller	1
Haliplidae	
Haliplus sp.	3
Haliplus larva	2
INSECTA : TRICHOPTERA	
Hydropsychidae	
Hydropsyche angustipennis (Curtis)	58
Limnephilidae	34
INSECTA: HEMIPTERA, HETEROPTERA	
Corixidae	2
Gerris sp.	1
INSECTA : DIPTERA	
Dipteran larvae	11
Ceratopogonidae	5
Chironomidae	6581
Simuliidae	77
Tipulidae	35

Figure 5.3.2: Canley Brook - Changes in abundance of selected taxa following pollution incident



Asellus aquaticus

Few individuals of *Asellus aquaticus* were found at the reference Sites A and B but this species was a standard component of the fauna at the impacted sites C-G. Comparisons between pre-incident and post incident samples show no impact of the pollution. However, comparisons between 6 days and 367 days at the most impacted sites (C,D and E combined) show a significant difference in abundance, with numbers higher 6 days after the incident than one year later (Mann-Whitney U test, $p < 0.05$).

Chironomidae

Numbers of Chironomidae are generally low at reference and impacted sites. A peak in abundance occurs at 52 days after the incident at impacted sites D, E, G and H. A similar peak occurs at reference site B so this can not be attributable to the incident.

Gammarus pulex

Gammarus pulex is found in greatest numbers at the reference sites. At the impacted sites no evidence of pollution impact is apparent when the pre- and post- incident samples are compared, the reference and control sites are compared and the impacted sites are compared over the period of one year.

Oligochaeta

Oligochaeta (Tubificidae) are found in greatest numbers at the impacted sites. Changes in abundance are variable through the study period but no apparent impact of the pollution is seen in the three comparisons mentioned above.

Potamopyrgus jenkinsi and Sphaeriidae

These taxa, were largely confined to the reference sites.

5.3.5.3. Univariate Indices

Univariate indices (Number of Taxa, Diversity index, BMWP and ASPT scores) are plotted in Fig. 5.3.3, and the results of significance tests are presented in Table 5.3.8.

Number of Taxa and Diversity Index

The number of taxa and diversity indices found 6 days after the incident is compared with the same measures 367 days after the incident at both reference sites (A,B) and the impacted sites closest to the source of pollution (C,D,E). At the *reference* sites, the results are significant with *fewer* taxa and lower diversity indices found 367 days after the incident (Mann-Whitney U-test $p < 0.05$). At the impacted sites, the number of taxa shows no significant difference between these times, although there is a significant difference in the diversity index with lower indices being found 367 days after the incident.

When all samples are considered, there is no difference between reference and impacted sites in the mean number of taxa.

A comparison between reference and impacted sites (C,D,E) with all sampling times compared shows no difference in the number of taxa but a significant difference in the diversity indices with lower indices being found at the reference sites.

The number of taxa and diversity indices found five months before the pollution incident is compared with the number of taxa 6 days after the incident at reference and impacted sites. No significant difference was found at the impacted sites although the reference sites showed a significant difference in the diversity index with high diversities following the impact.

Table 5.3.8: Number of Taxa and Diversity Index - Summary Table of comparisons between samples (Mann-Whitney U Test, ns = not significant; s = significant $p < 0.05$)

	6 days and 367 days (reference)	6 days and 367 days (impacted C,D,E)	reference and impacted (C,D,E)	Before incident and 6 days after incident (reference)	Before incident and 6 days after incident (impacted)
Number of Taxa	s	ns	ns	ns	ns
Diversity Index	s	s	s	s	ns

BMWP and ASPT scores

Both reference and impacted sites show low BMWP and ASPT scores, with the highest scores being found at Site H (furthest downstream).

At both reference sites and impacted sites, BMWP scores are variable and with so few taxa present at each site, the score can be changed by the presence of a single individual of a high scoring taxon e.g. Limnephilidae. The ASPT scores remain relatively constant over time within sites.

Figure 5.3.3: Canley Brook - Univariate measures - changes following pollution incident

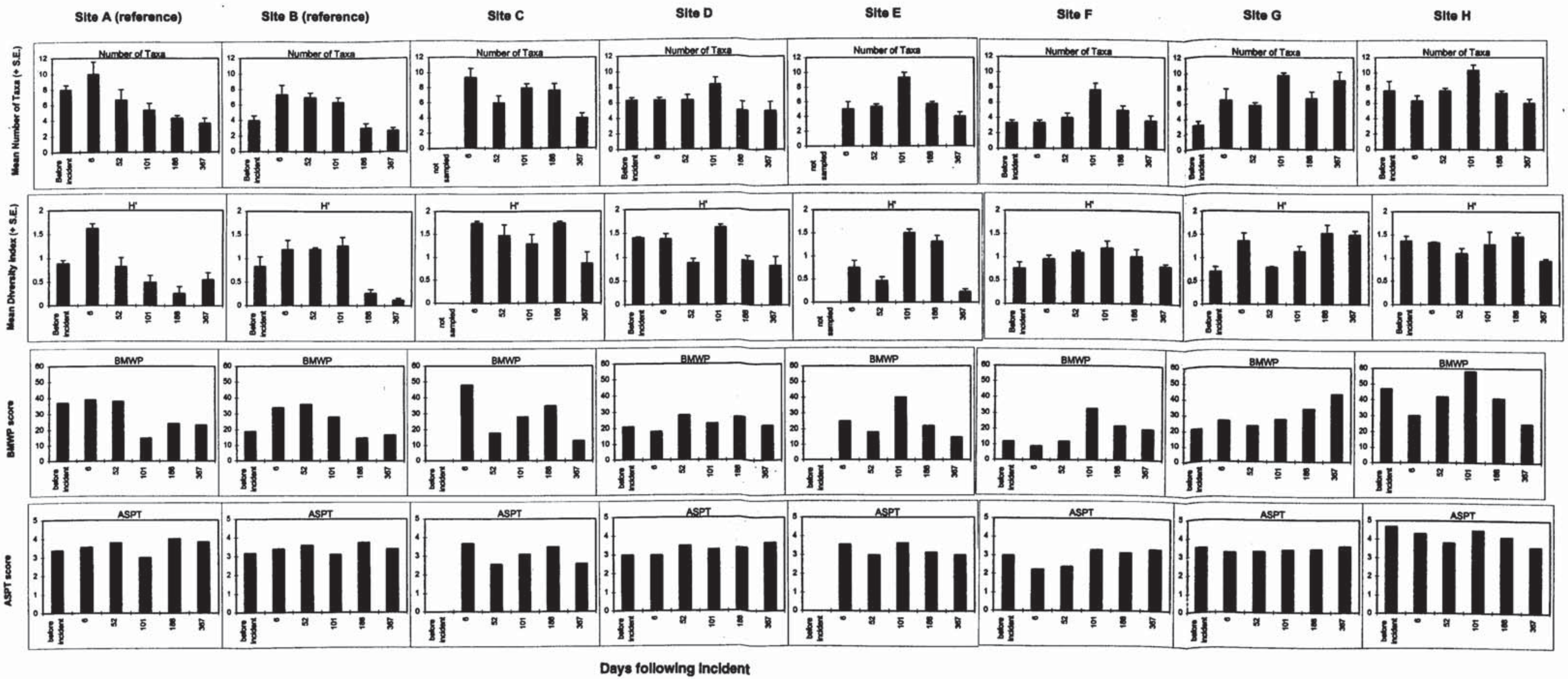
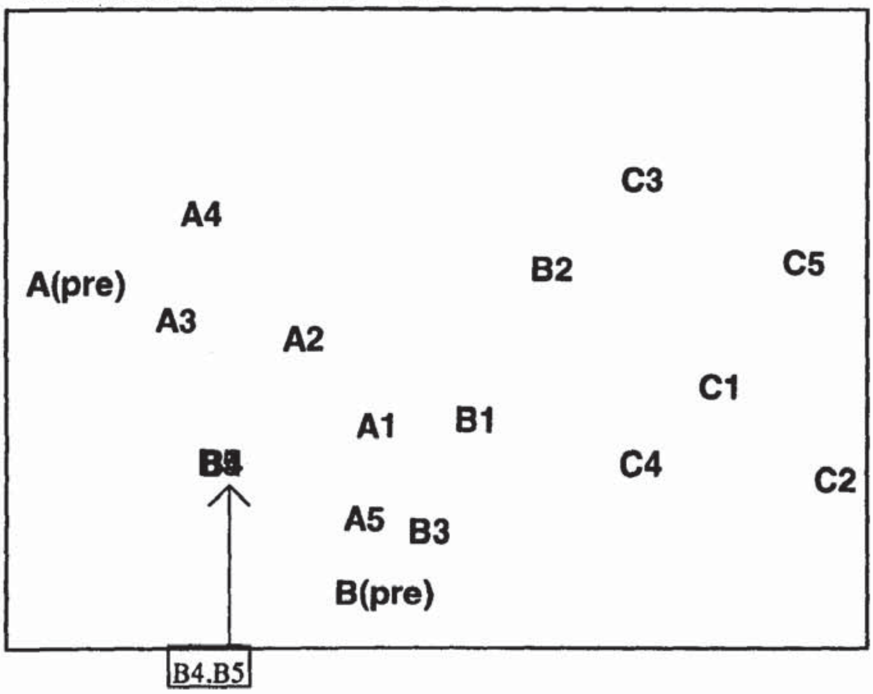


Figure 5.3.4a: Canley Brook - MDS configuration of Sites A and B (reference) and Site C

No transformation. Stress = .10



Root/root transformation. Stress = .14

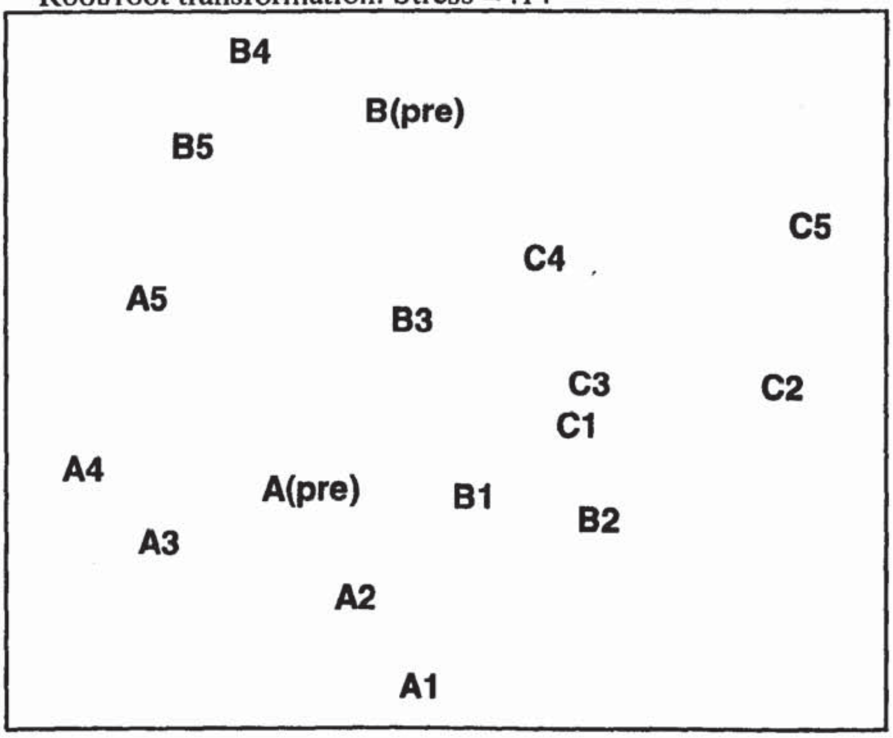
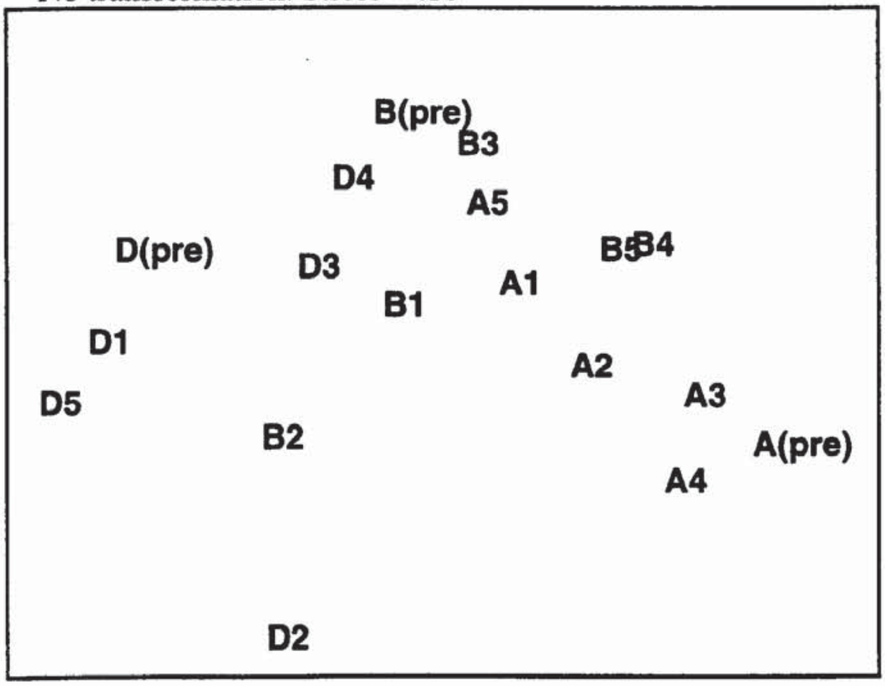


Figure 5.3.4b: Canley Brook - MDS configuration of Sites A and B (reference) and Site D

No transformation. Stress = .10



Root/root transformation. Stress = .18

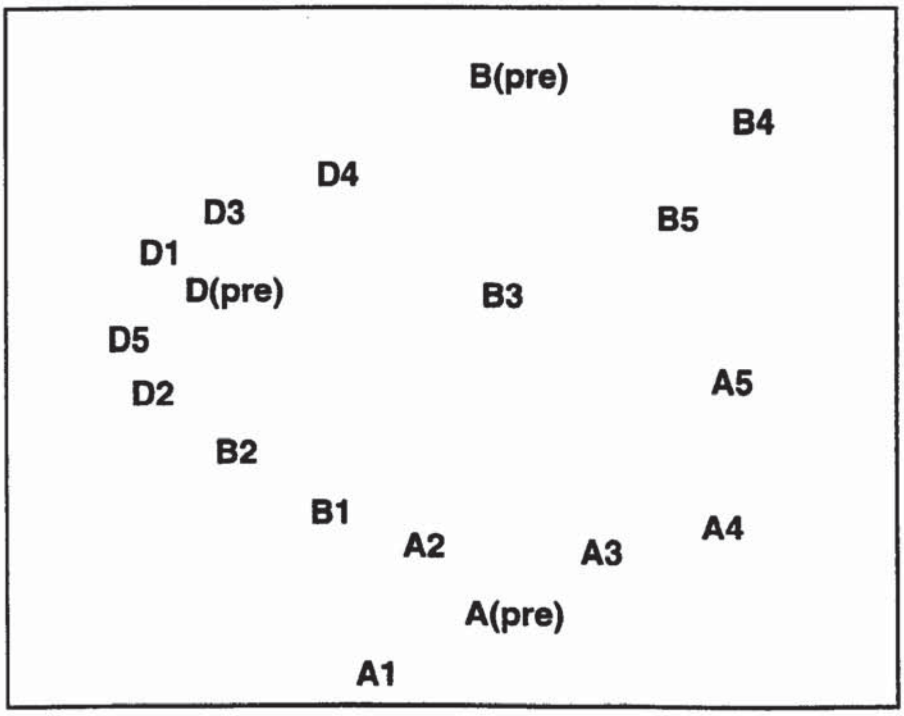
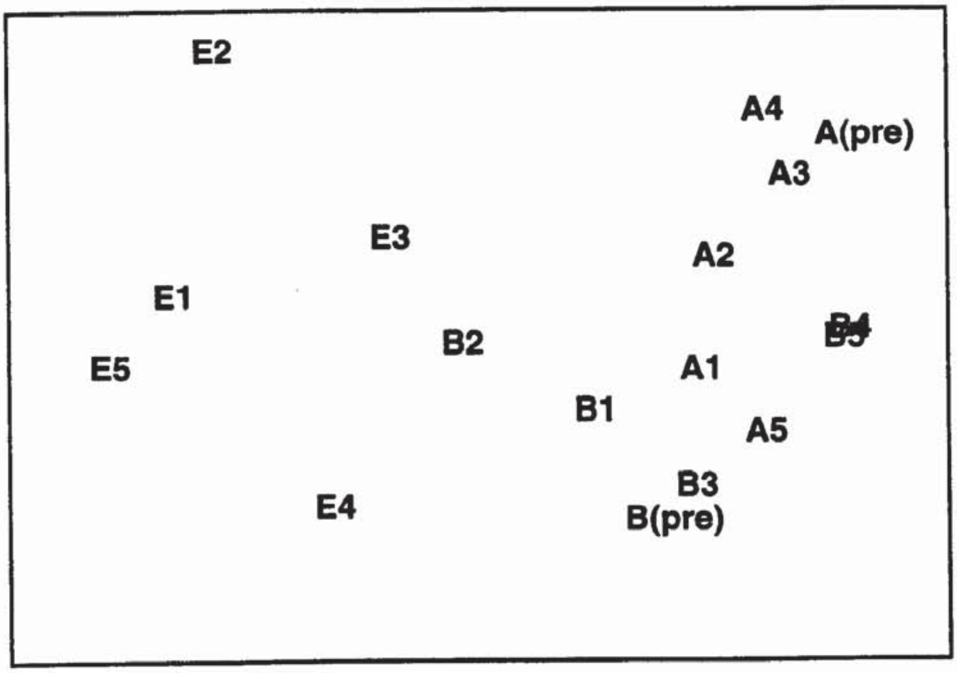


Figure 5.3.4c: Canley Brook - MDS configuration of Sites A and B (reference) and Site E

No transformation. Stress = .08



Root/root transformation. Stress = .15

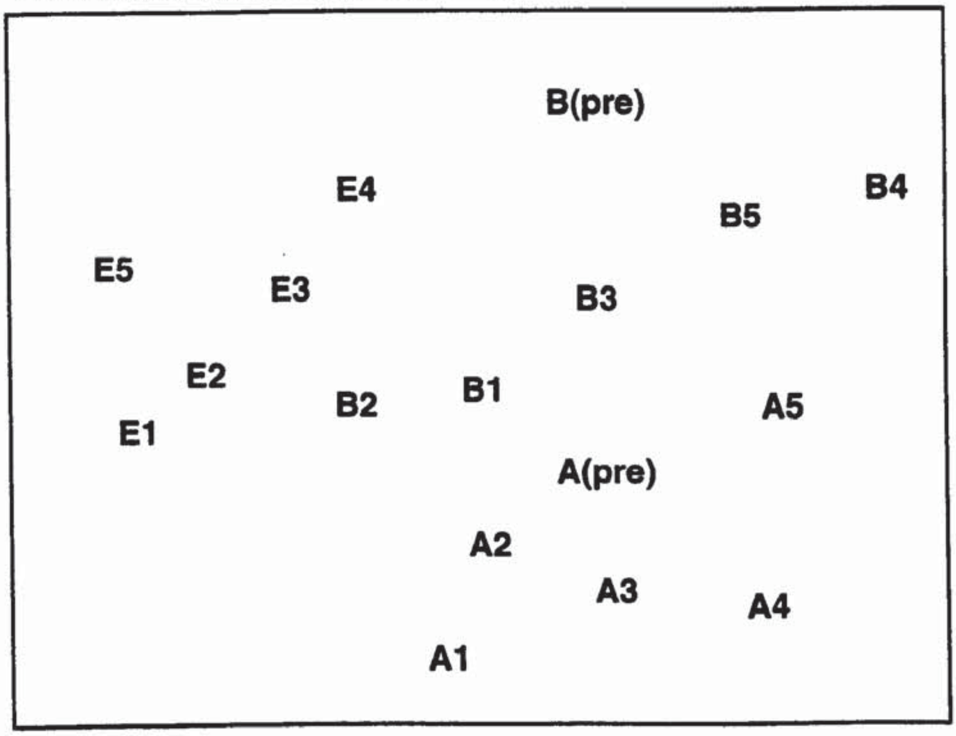
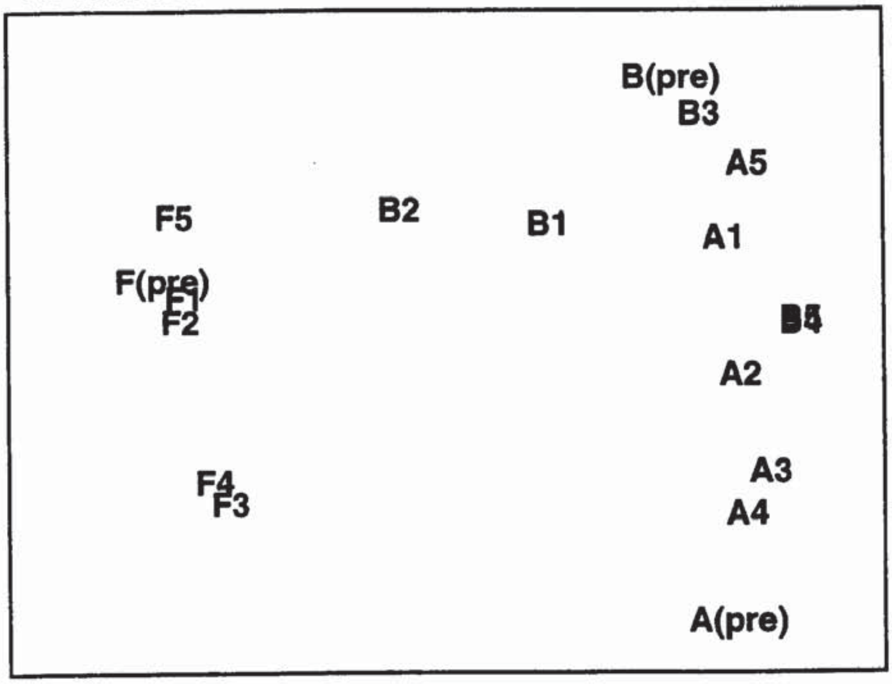


Figure 5.3.4d: Canley Brook - MDS configuration of Sites A and B (reference) and Site F

No transformation. Stress = .09



Root/root transformation. Stress = .14

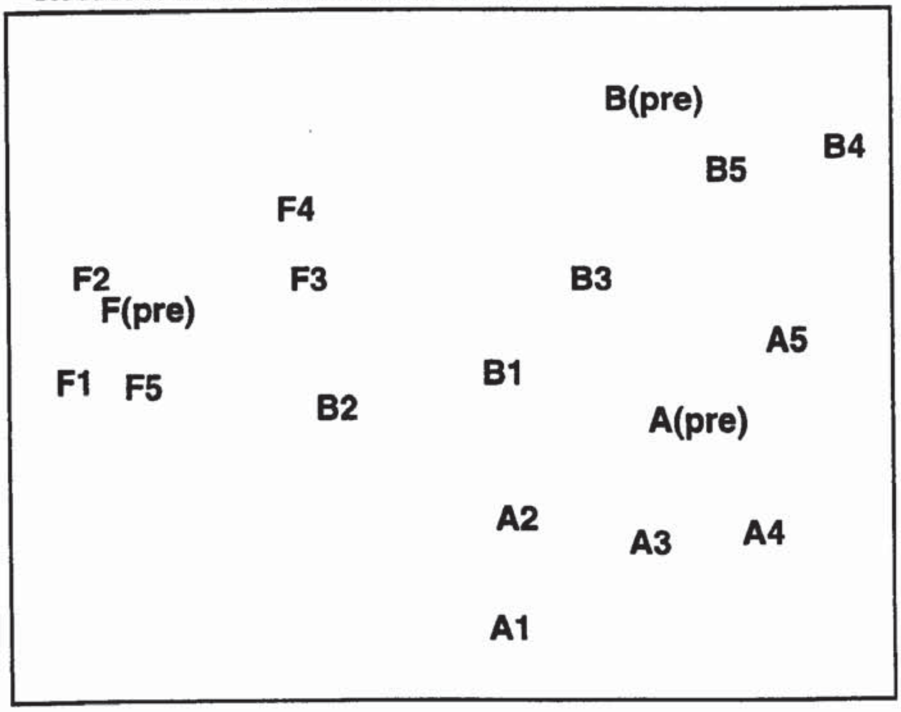
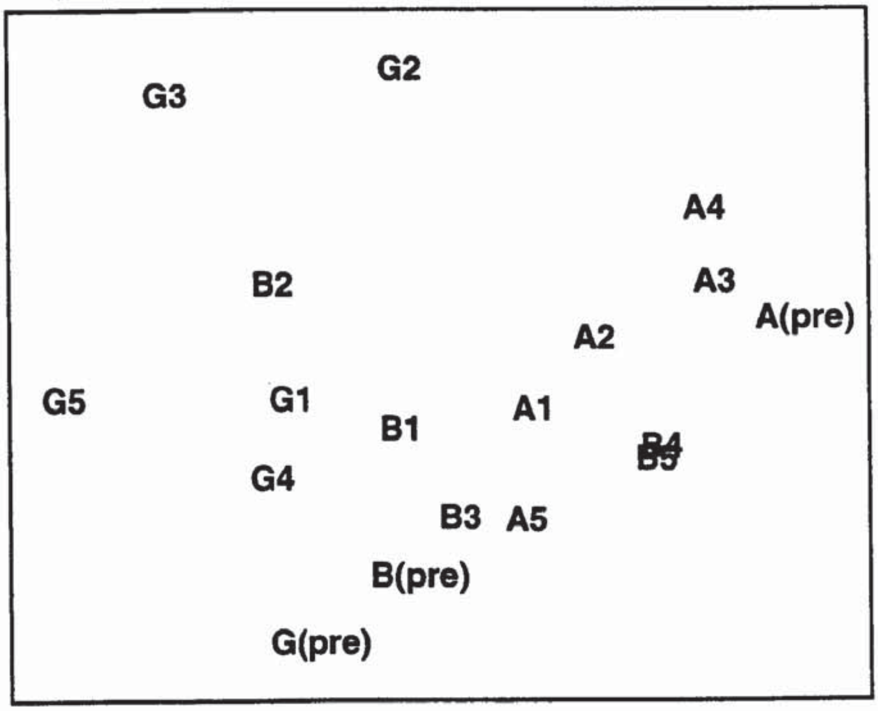


Figure 5.3.4e: Canley Brook - MDS configuration of Sites A and B (reference) and Site G

No transformation. Stress = .11



Root/root transformation. Stress = .19

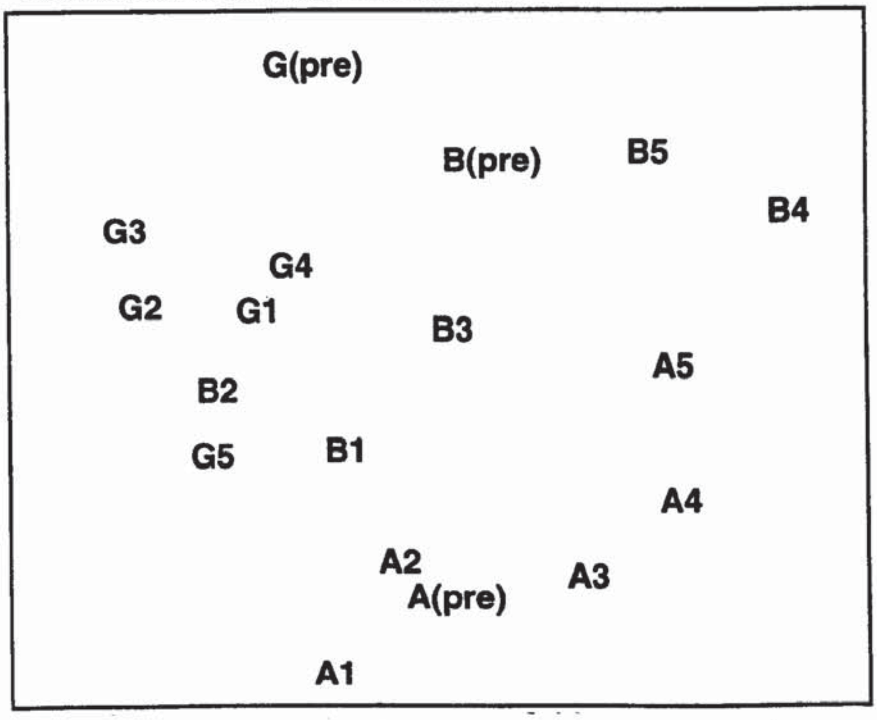
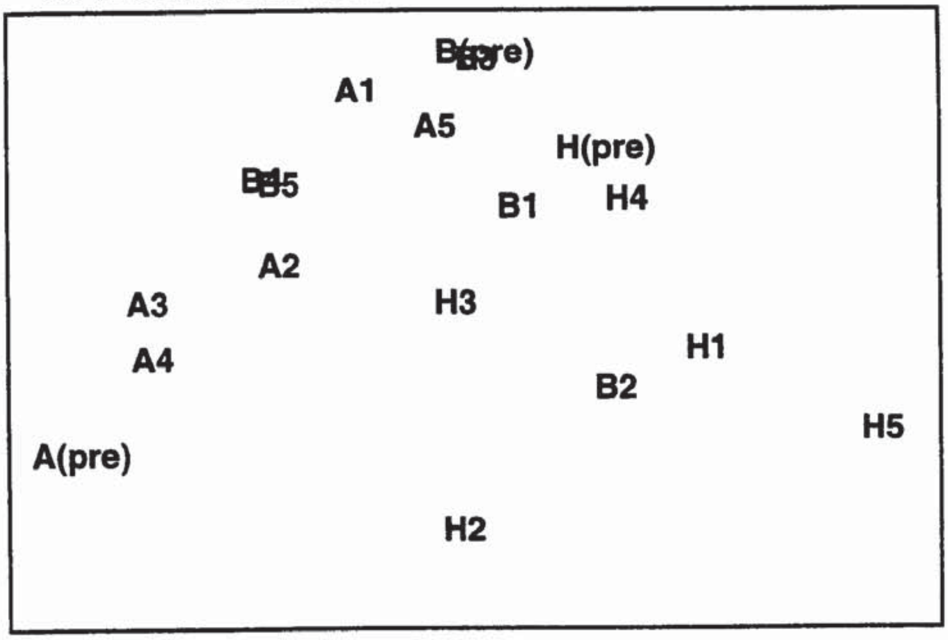
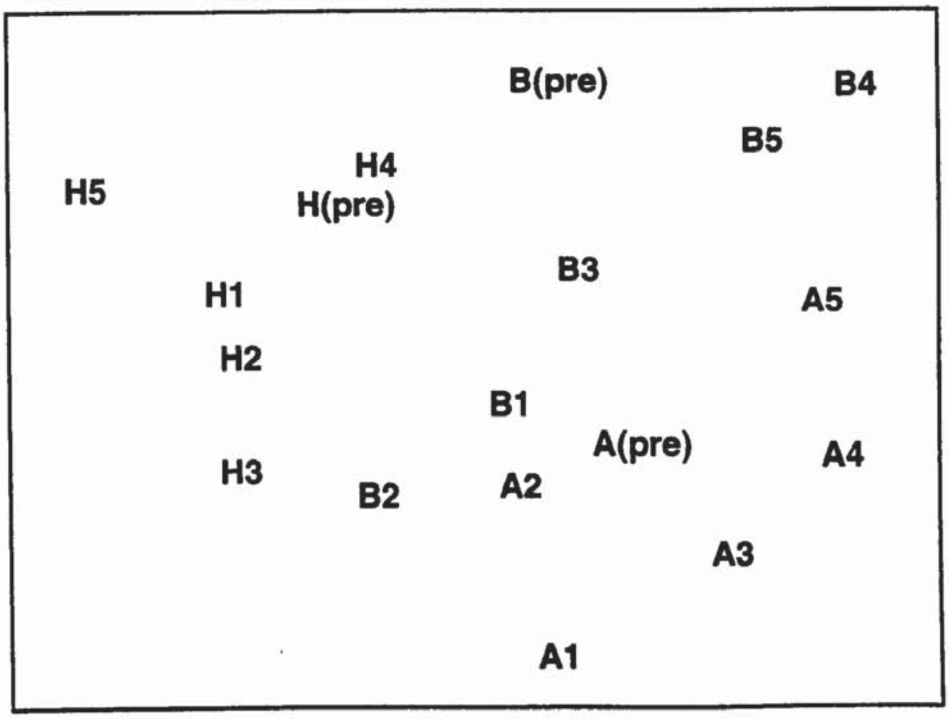


Figure 5.3.4f: Canley Brook - MDS configuration of Sites A and B (reference) and Site H

No transformation. Stress = .10



Root/root transformation. Stress = .16



5.3.5.4. Multivariate analysis

The R Statistic was computed for each sampling time as described in Chapter 4. This showed that the replicates within sites are more similar to each other than replicates between sites. Similarity matrices were therefore produced from data from averaged replicates. The similarity matrices were used to construct Multidimensional Scaling configurations (MDS plots) comparing each impacted site with the reference sites. (Figs 5.3.4 a-f).

The samples from each site are more similar to each other than to samples from other sites. The configurations do not show any evidence of impact that was predicted in Table 5.3.5.

5.3.6. Summary

Table 5.3.9 summarises the pollution impact on all the biotic measures used.

Canley Brook is a stream of poor chemical and biological quality with a history of pollution incidents. As a major part of the stream flows through public open space and a golf course, it is highly visible to the public and visible pollution incidents are more likely to be reported by the public.

In spite of the relatively poor quality, the stream supported an abundant, although not a taxonomically rich fauna, which included *Baetis rhodani*, *Gammarus pulex* and the Stone Loach, *Neomachilus barbatulus*. The pollution incident investigated had a more serious impact than previous incidents in causing a large fish kill. Dead *Gammarus pulex*, *Lymnaea peregra*, *Asellus aquaticus* were also seen. The only indication of impact on the measures used, however, was on abundance of *Asellus aquaticus*, with greater numbers of this species found 6 days after the incident than one year later. The fact that dead individuals of this species were seen during the sampling occasion 6 days after the incident, showed that only a proportion of the population had been affected and indeed with the absence of a major predator (Stone Loach), conditions may have been more favourable for this taxon. No evidence of impact was found in any of the measures used when comparing impacted communities to reference communities or to pre-impact communities at the same site.

Table 5.3.9: Summary of pollution impact on all measures

Measure	Type of impact	Spatial Impact	Temporal Impact
Initial impact of incident	Dead fish (900+), dead invertebrates	Fish (5km) Invertebrates (50m)	6 days following incident
Individual Taxa	Impact on abundance of *Asellus aquaticus* (greater abundance 6 days after the impact than 367 days after impact)	Sites C,D,E	6 days after the impact
Number of Taxa	No evidence of impact		
Diversity (H')	No evidence of impact		
BMWP, ASPT	No evidence of impact		
Multivariate analysis	No evidence of impact		

5.4 Chinn Brook

5.4.1 Description of Incident

On 2 August, 1994, a pollution incident of Fluoroscene dye accompanied by a noxious smell in a small tributary of the Chinn Brook was reported to the NRA. A visit was made 2 days after the incident and dead Oligochaeta and Chironomidae were seen on the surface of the substratum. The exact source and nature of the pollution was not discovered although it was known to come from an industrial estate, the surface water of which drained into the tributary. Pollution control officers suspected illegal tipping of waste into the drain by a lorry from outside the industrial estate.

5.4.2 Site Description and History

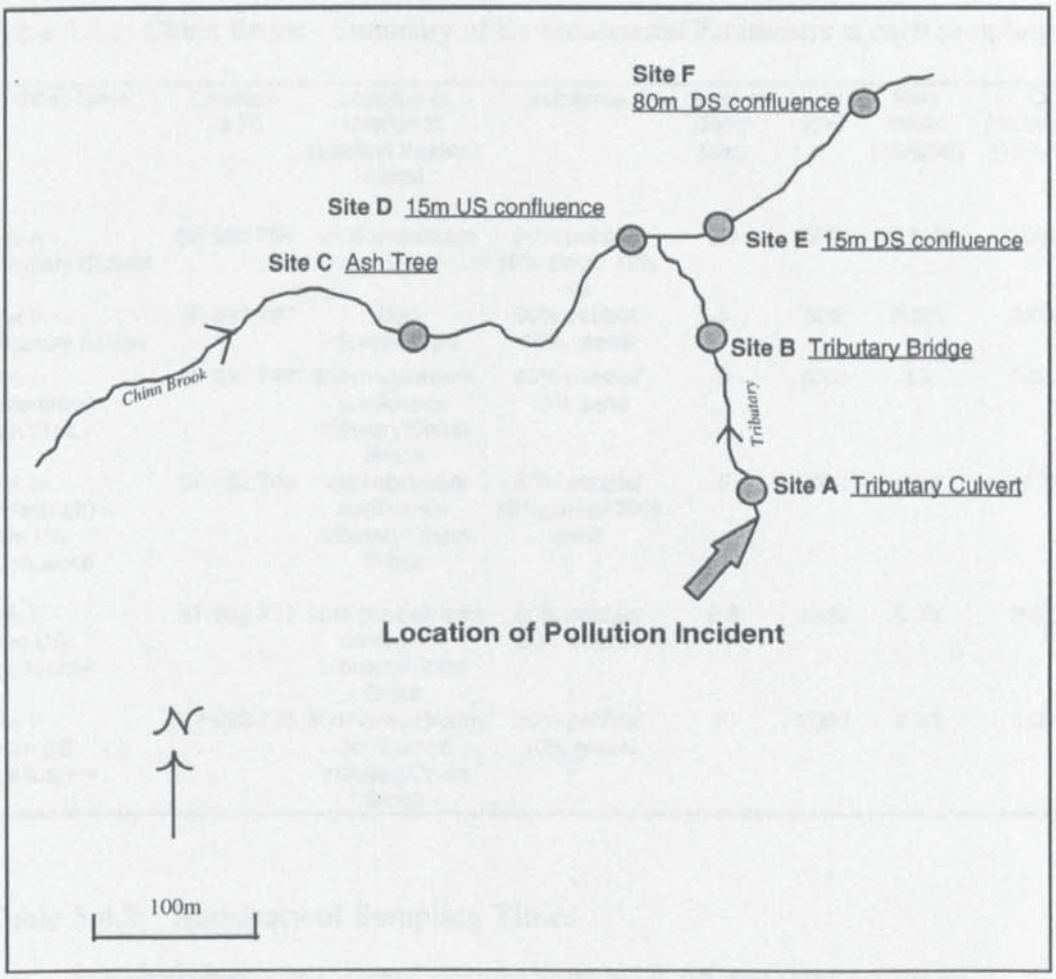
The Chinn Brook is a tributary of the River Cole in Birmingham which forms part of the River Tame catchment. The Chinn Brook is about 5 km in length. The tributary into which the spill occurred is less than 1 km in length and takes the surface water drainage from a small industrial estate, flowing through a culvert beneath the Stratford-on-Avon canal before emerging into a golf course where it joins the Chinn Brook (Fig. 5.4.1). The land-use within the catchment of the Chinn Brook is urban comprising residential and industrial estates and public open space.

Relatively impermeable Mercia Mudstones form much of the underlying geology. Sand and gravel deposits underlie these and groundwater discharges from these deposits contribute a baseflow to the River Cole.

Rainfall is lower than average for the region with an annual mean of 714 mm. A spray irrigation licence for the golf course abstracts water from the Chinn Brook.

The NRA do not hold water quality information on the Chinn Brook itself. The River Cole, into which the Chinn Brook flows, has a River Quality Objective of 1B in the upper reaches and Class 2 for the remainder. It is currently Class 2 for all of its length. The Biological Water Quality of the River Cole is moderate with typical BMWP scores of 36-65 (NRA, 1993).

Figure 5.4.1: Chinn Brook - Map to show location of Pollution Incident and Sampling Points



5.4.3 Sampling Procedure

Three replicate Surber samples were taken from 6 sites (see Fig. 5.4.1). Given the upstream location of the pollution incident, it was not possible to have upstream reference sites. Two sites were situated on the impacted tributary (A and B). Two reference sites (C, D) were located upstream of the tributary/Chinn Brook confluence and 2 sites within 400 m downstream of the confluence.

The environmental parameters of the sampling sites are summarised in Table 5.4.1. All sampling sites were situated in riffle areas. The two tributary sites had low discharge and flow values.

Six sampling visits were distributed over a period of 287 days following the pollution incident as shown in Table 5.4.2. All sites were sampled on each occasion except for 24 August when only Sites D-F were sampled.

Table 5.4.1: Chinn Brook - Summary of Environmental Parameters at each sampling point

Site Name	Location (G R)	Location in relation to pollution incident outfall	Substrate	Average Depth (cm)	Width (cm)	Flow m/sec (15/5/95)	Q (cu.m/sec) (15/5/95)
Site A - Tributary Culvert	SP 082 796	3m downstream	80% pebble/ 10% Brick/ 10% silt	2.5	500	0.143	0.0014
Site B - Tributary Bridge	SP 082 797	100m downstream	90% pebble/ 10% gravel	5	800	0.125	0.002
Site C (reference) Ash Tree	SP 081 797	200m upstream confluence tributary/Chinn Brook	90% pebble/ 10% sand	6	1000	0.2	0.012
Site D (reference) - 15m US confluence	SP 082 798	15m upstream confluence tributary/Chinn Brook	80% pebble/ 10%gravel/ 20% sand	6	2000	0.19	0.0228
Site E - 15m DS confluence	SP 082 798	15m downstream confluence tributary/Chinn Brook	80% pebble/ 20% cobble	8.5	1800	0.25	0.027
Site F - 80m DS confluence	SP 083 798	80m downstream confluence tributary/Chinn Brook	90% pebble/ 10% gravel	10	2000	0.33	0.032

Table 5.4.2: Summary of Sampling Times

Sampling time Code	Date		Days following incident
chi1	9 August	1994	7
chi2	24 August	1994	22
chi3	4 November	1994	95
chi4	19 December	1994	139
chi5	2 February	1995	184
chi6	15 May	1996	287

5.4.4 Predicted Impact of the Pollution Incident

The nature of the pollutant involved in this incident is not known although members of the public reported unpleasant smells. It is known that the pollution comprised a toxic element, as dead Oligochaeta and Chironomidae were seen close to the outfall two days after the incident had been reported. The predicted impact of a toxic event would thus be a reduction in numbers and abundance of taxa within the tributary. The toxic effects may have extended into the Chinn Brook, thus similar changes should be seen downstream of the tributary/Chinn Brook confluence when compared with the upstream sites.

5.4.5 Results and Analysis

The data is organised as described in Chapter 4. No pre- impact measures exist and there are no upstream reference sites on the tributary itself. Impact of the pollution incident must therefore be assessed by temporal changes within the tributary and at the Chinn Brook by spatial and temporal changes within Sites E and F compared with reference Sites C and D.

5.4.5.1 Water Quality

Table 5.4.3 gives the physicochemical measurements taken at selected sampling times. Temperature is seen to vary seasonally. The temperature of the tributary water emerging from the culvert at Site A is consistently higher than the temperatures further donwstream and within the Chinn Brook. Values of pH are variable at all sites with a mean value of 7.8. Conductivity measurements are relatively high and variable at all sites although at the last sampling visit, 287 days after the pollution incident, the values at the tributary are much lower.

5.4.5.2. Selected Taxa

Table 5.4.4 is a summary of the taxa found and their total abundance. 28 taxa were found throughout the study. The five most abundant taxa are listed below and changes in their presence and abundance throughout the study are considered in this section, shown in Figure 5.4.2, and summarised in Table 5.4.5.

Ancylus fluviatilis	*Gammarus pulex*
Baetidae	Oligochaeta
Chironomidae	

Ancylus fluviatilis

This species was not found in the tributary and was found in significantly lower numbers upstream of the confluence (Sites C,D) than downstream (Sites E,F) (Mann-Whitney U test $p<0.05$). Different patterns of abundance changes over time are seen at Sites E and F and are therefore presumed to be the result of factors other than the pollution incident.

Table 5.4.3: Physicochemical Conditions in Chinn Brook.
(Conductivity is in µS/cm³)

		Site A - Tributary Culvert			Site B - Tributary Bridge		
Date	Days following Disturbance	Temp(°C)	pH	Cond.	Temp(°C)	pH	Cond.
09-Aug-94	7	15.9	7.6	387	15.5	7.4	349
24-Aug-94	22	n/a	n/a	n/a	n/a	n/a	n/a
02-Feb-95	184	6.3	7.5	383	5.2	7.5	417
15-May-95	287	10.6	7.7	292	8.8	7.8	196

		Site C - Ash Tree			Site D - 50m US confluence		
Date	Days following Disturbance	Temp(°C)	pH	Cond.	Temp(°C)	pH	Cond.
09-Aug-94	7	15.6	8.3	490	15.5	8.3	486
24-Aug-94	22	n/a	n/a	n/a	14.3	8.2	325
02-Feb-95	184	5.6	7.7	353	5.1	7.5	348
15-May-95	287	9.6	7.4	428	9.6	8.3	406

		Site E - 50m DS confluence			Site F - 80m DS confluence			Site G - Warstock lane/BMX		
Date	Days following Disturbance	Temp(°C)	pH	Cond.	Temp(°C)	pH	Cond.	Temp(°C)	pH	Cond.
09-Aug-94	7	15.3	8.2	473	15.4	7.5	485	15.6	8.2	490
24-Aug-94	22	14.1	8.1	452	14	8	448	n/a	n/a	n/a
02-Feb-95	184	5.4	7.4	349	5	7.4	330	n/a	n/a	n/a
15-May-95	287	9.2	8.3	417	9.5	8	413	8.6	7.7	402

Baetidae

All those identified in this family were *Baetis rhodani*. Only one individual was found in the tributary (Sites A,B). Otherwise Baetidae were evenly distributed through the sites within the Chinn Brook with a peak in abundance at 184 days following the incident at both reference and impacted sites which is therefore presumed to be seasonal. The finding that *B.rhodani* has not colonised the tributary may be due to unfavourable environmental conditions for this species for example chronic pollution from the outfall.

Chironomidae

Chironomidae were found to the greatest extent in the tributary (Sites A,B). Highest numbers were found 95 days following the incident at Site A.

Gammarus pulex

Within the tributary, a few individuals of *Gammarus pulex* were found from 139 days following the incident but not before this. *Gammarus pulex* was abundant at all the Chinn Brook sites with significantly higher numbers upstream of the confluence than downstream (Mann-Whitney U test $p<0.05$). Relative changes in abundance over time were the same up and downstream of the confluence and could not be attributed to any effect of the pollution incident.

Oligochaeta

The Oligochaeta found comprised largely Tubificidae worms. Most of the Oligochaeta found within the study were in the tributary (Sites A,B) with peak abundances found 184-287 days after the pollution incident.

Species differences up vs downstream of the confluence.

Differences were found in the abundance of particular taxa up and downstream of the tributary entering the Chinn Brook. The sampling sites were chosen for the purpose of elucidating any effect of the pollution incident on the Chinn Brook fauna, and all sampling sites were similar in their environmental characteristics. The differences found, however, may not be due to any effects of the incident as no temporal progression in the differences is seen and the differences are not those that would be expected resulting from a pollution incident (initial decrease in taxa, increase in sensitive taxa).

Ancylus fluviatilis is more abundant downstream of the tributary. This species is an indiscriminate grazer of the algal mat (Geldiay, 1956). The downstream sites may have a higher density of algal covering on the substratum as there are fewer bankside trees. *Gammarus pulex*, on the other hand, although abundant throughout the Chinn Brook, is significantly more abundant upstream of the tributary. This could similarly be explained by variations in food resources. In terms of feeding ecology, *Gammarus* can be described as a facultative shredder (Cummins and Klug, 1979), feeding primarily on decomposing plant material, usually allochthonous in origin (Kaushik and Hynes, 1971). The greater abundance of riparian trees at Sites C and D might therefore account for the higher numbers of *Gammarus*.

Table 5.4.4: Chinn Brook - Summary of Taxa found

Phylum, Order, Family, Genus, Species	Number found in study	Phylum, Order, Family, Genus, Species	Number found in study
PLATYHELMINTHES		ACARINA:HYDRACARINA	12
TURBELLARIA:TRICLADIDA			
Planariidae	1	INSECTA : EPHEMEROPTERA	
Dugesia sp.		Baetidae	
Polycelis tenuis (Ijima)/*nigra* (Muller)	2	*Baetis rhodani* Pictet	1275
Dendrocoelidae		Ephemerellidae	
Dendrocoelum lacteum (Muller)	11	*Ephemerella ignita* (Poda)	51
MOLLUSCA			
GASTROPODA:PROSOBRANCHIA		INSECTA : COLEOPTERA	
Hydrobiidae		Dytiscidae	
Potamopyrgus jenkinsi (Smith)	65	Dytiscidae larva	4
GASTROPODA:PULMONATA			
Lymnaeidae		INSECTA : TRICHOPTERA	
Lymnaea peregra (Muller)	90	Hydropsychidae	
Ancylidae		*Hydropsyche angustipennis* (C.)	42
Ancylus fluviatilis Muller	555	Psychomyiidae	
BIVALVIA		*Tinodes waeneri* (L.)	1
Sphaeriidae	33	Limnephilidae	5
Physidae		*Chaetopteryx villosa* (Fab.)	44
Physa sp.	84	INSECTA: HEMIPTERA, HETEROPTERA	
ANNELIDA		Veliidae	
OLIGOCHAETA	3986	*Velia* sp.	1
HIRUDINEA			
Glossiphoniidae		INSECTA : DIPTERA	
Glossiphonia complanata (L.)	132	Ceratopogonidae	4
Erpobdellidae	115	Chironomidae	1668
Hirudinidae		Simuliidae	28
Haemopis sanguisuga (L.)	1	Syrphidae	
		Eristalis sp.	3
CRUSTACEA: MALACOSTRACA		Tipulidae	3
Asellidae			
Asellus aquaticus (L.)	37		
Gammaridae			
Gammarus pulex (L.)	12438		

Figure 5.4.2: Chinn Brook - Changes in abundance of selected taxa following pollution incident

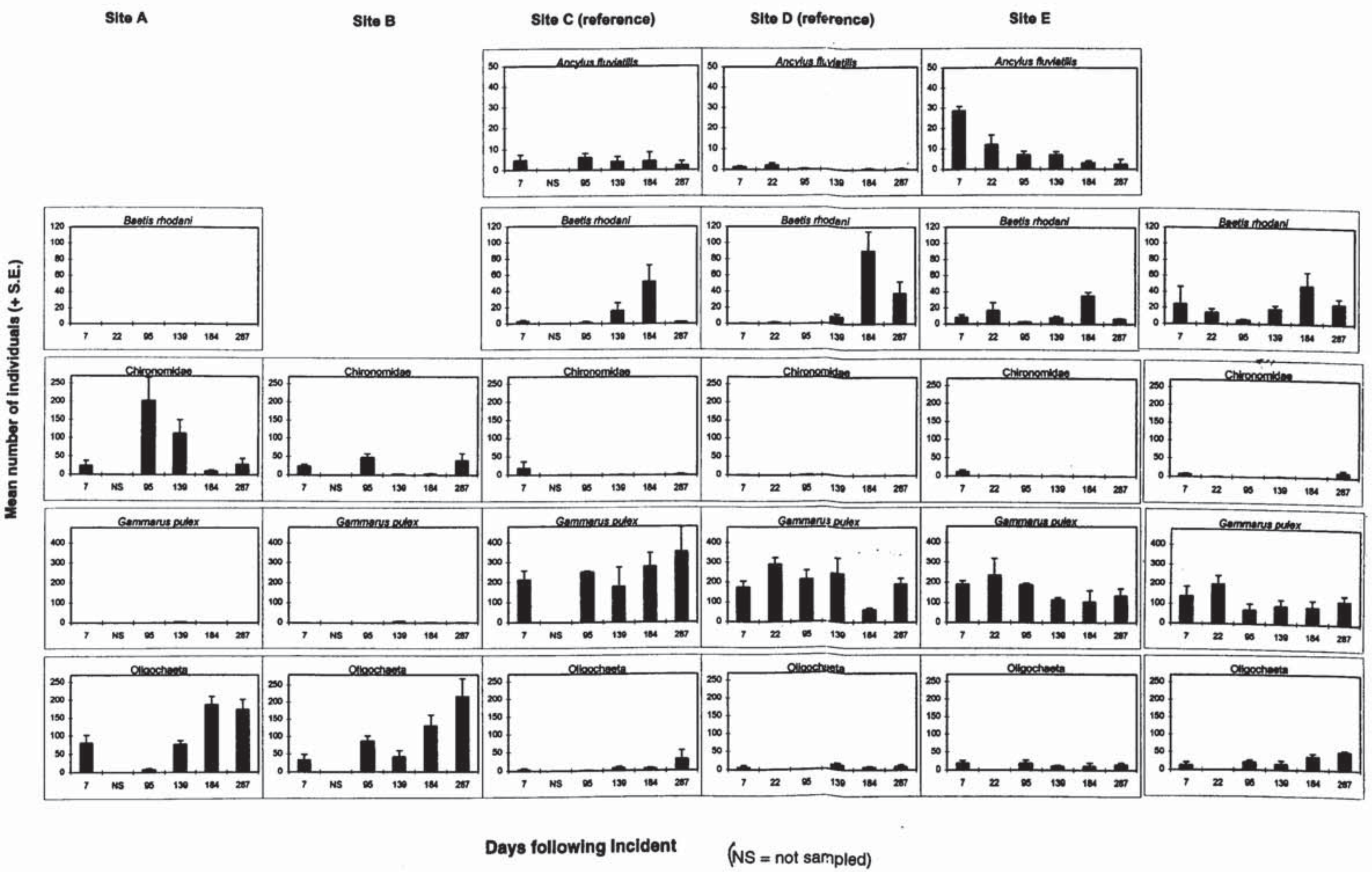


Table 5.4.5: Summary of Impact of pollution incident on selected taxa

Taxon	Possible Impact of pollution
Ancylus fluviatilis	Greater numbers at impacted Sites E and F
Baetidae	No apparent impact
Chironomidae	Increase in abundance at tributary sites 95 days following pollution incident
Gammarus pulex	Absence at Sites A and B for 95 days. Lower numbers at impacted Sites E and F
Oligochaeta	Increase in abundance at tributary sites 184-287 days following the pollution incident

5.4.5.3. Univariate Indices

Univariate indices (Number of taxa, Diversity index, BMWP and ASPT scores) are plotted in Fig 5.4.3.

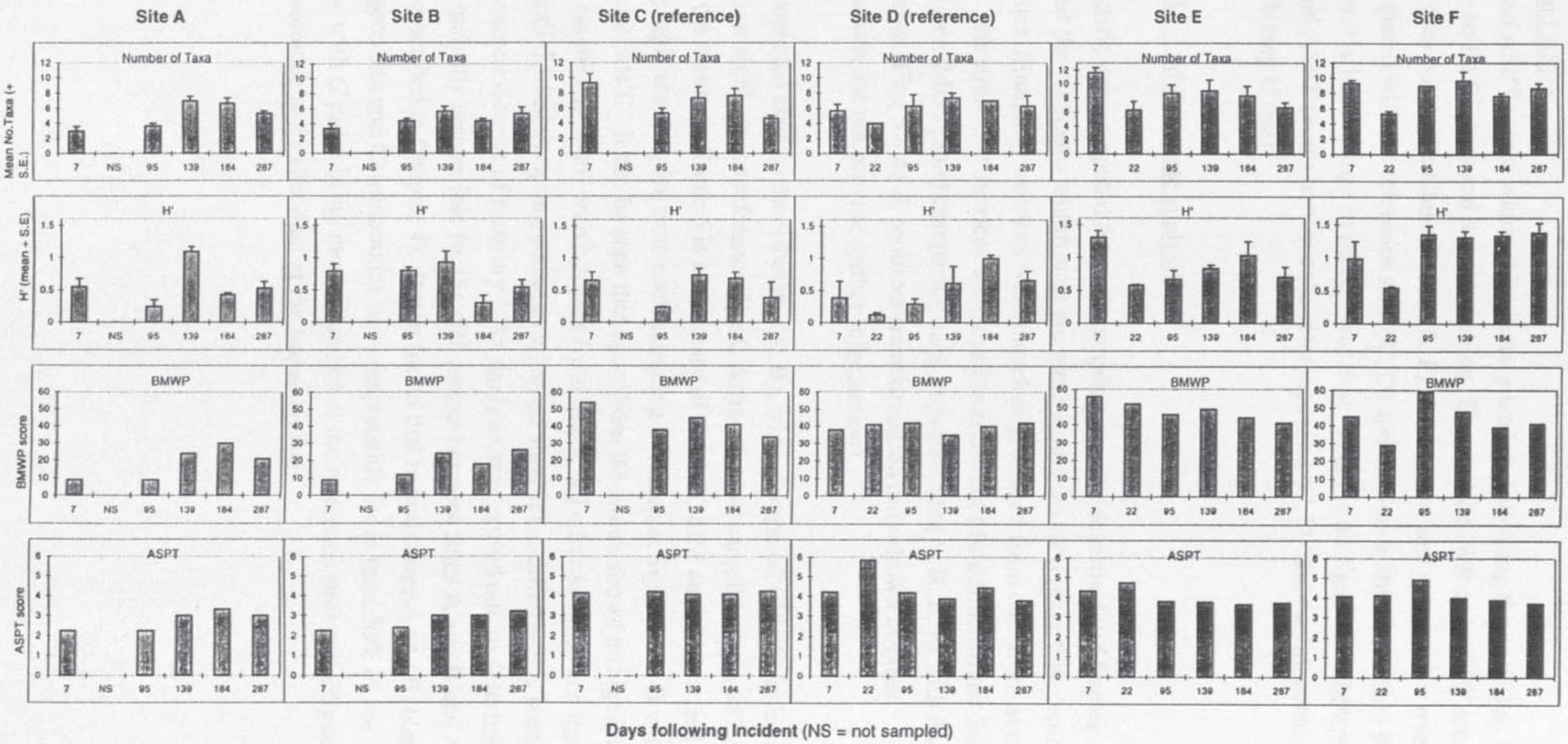
Number of Taxa

The tributary sites (A, B) show initially lower numbers of taxa and an increase in numbers 139 days after the incident (Mann-Whitney U tests comparing tributary sites with the two reference sites C and D, show significant differences up to 95 days following the incident and no significant differences thereafter). The sites on the Chinn Brook support higher numbers of taxa than the tributary. There is a significant difference in the mean numbers of taxa between the reference sites (C, D) and the impacted sites (E, F) within the Chinn Brook (Mann Whitney U test, $p<0.05$) with higher numbers of taxa downstream of the confluence. As discussed above, these differences are likely to be due to factors other than the pollution incident.

Diversity Index

Diversity index measures within the tributary are low with no clear pattern of recovery. The mean diversity indices at the impacted sites within the Chinn Brook (E, F) do not show any progression over time when compared with the reference sites (C, D). Diversity is significantly higher downstream of the confluence than upstream (Mann-Whitney U test, $p<0.05$).

Figure 5.4.3: Chinn Brook - Univariate measures - changes following pollution incident



BMWP and ASPT

BMWP and ASPT scores within the tributary are low following the pollution incident, remaining so for 95 days and then increasing. The total BMWP and ASPT scores at the impacted sites within the Chinn Brook (E, F) do not show any progression over time when compared with the reference sites (C, D) and all sites within the Chinn Brook show relatively little change in scores over time. There is no significant difference in the BMWP and ASPT scores between the reference sites (C, D) and the impacted Sites E, F (Mann-Whitney U test).

5.4.5.4. Multivariate analysis

The R Statistic was computed for each sampling time as described in Chapter 4. This showed that the replicates within sites are more similar to each other than replicates between sites. Similarity matrices were therefore produced from data from averaged replicates. The similarity matrices were used to construct Multidimensional Scaling configurations (MDS plots) comparing each impacted site (A, B, E, F) with the reference sites (Figs 5.4.4a-d, root/root transformation only shown as other transformations did not provide further information).

Both the impacted sites on the tributary (A, B), when compared with the reference sites, show greatest similarity to reference Site C during the last sampling visit, 287 days following the incident. Table 6 is a summary of the ANOSIM analyses carried out comparing replicate samples from each sampling occasion at Sites A and B with all the samples from Site C. It can be seen that apart from B3 which shows no significant difference (at the 5% level) with samples from Site C, all other samples are significantly different until the last sampling occasion when no significant difference is seen indicating some evidence of recovery. An analysis was carried out to determine which taxa are principally responsible for the differences between Sites A and B and reference Site C (as described in Chapter 4). It was found that in both Sites A and B, *Gammarus pulex*, Oligochaeta and Chironomidae were responsible for at least 50% of the differences, with *G.pulex* being more abundant at the reference sites and Oligochaeta and Chironomidae more abundant at the impacted sites.

Figure 5.4.4a: Chinn Brook - MDS configuration of Site A (impacted) and Sites C and D (reference)

Root/root transformation. Stress = .13

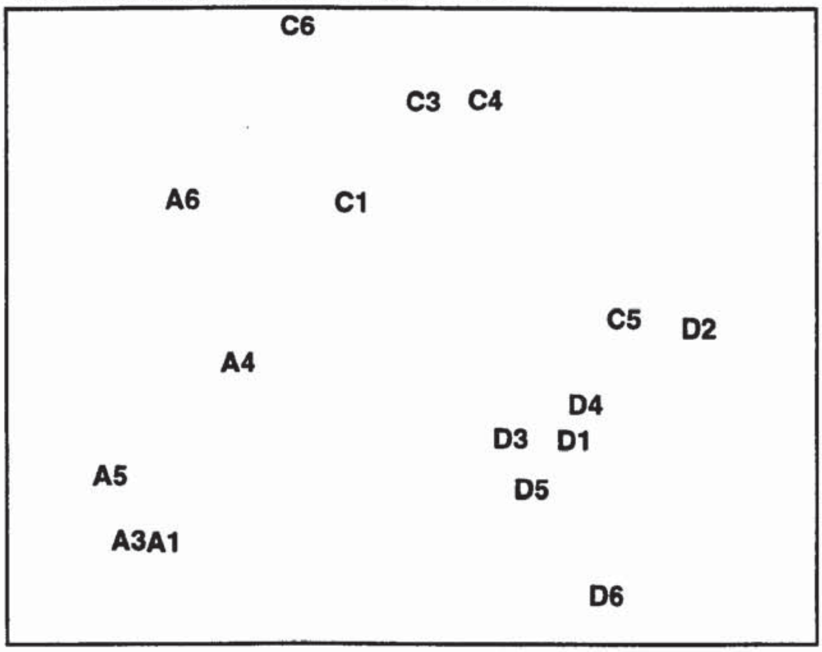


Figure 5.4.4b: Chinn Brook - MDS configuration of Site B (impacted) and Sites C and D (reference)

Root/root transformation. Stress = .13

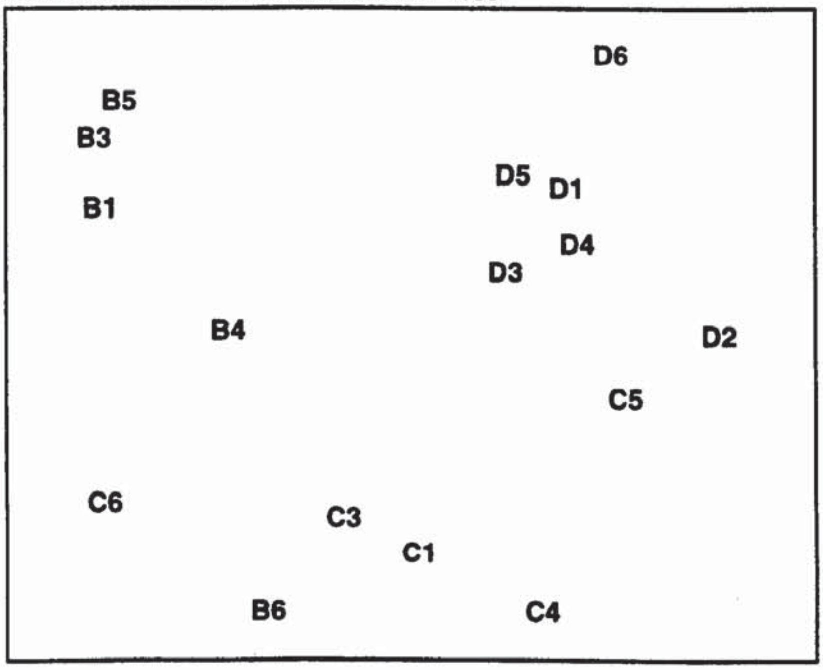


Figure 5.4.4c: Chinn Brook - MDS configuration of Site E (impacted) and Sites C and D (reference)

Root/root transformation. Stress = .19

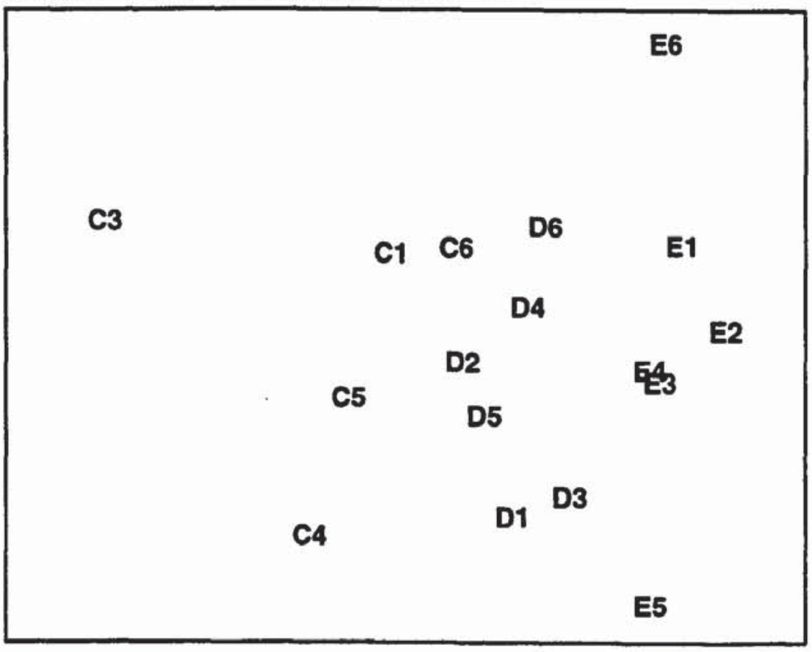


Figure 5.4.4d: Chinn Brook - MDS configuration of Site F (impacted) and Sites C and D (reference)

Root/root transformation. Stress = .19

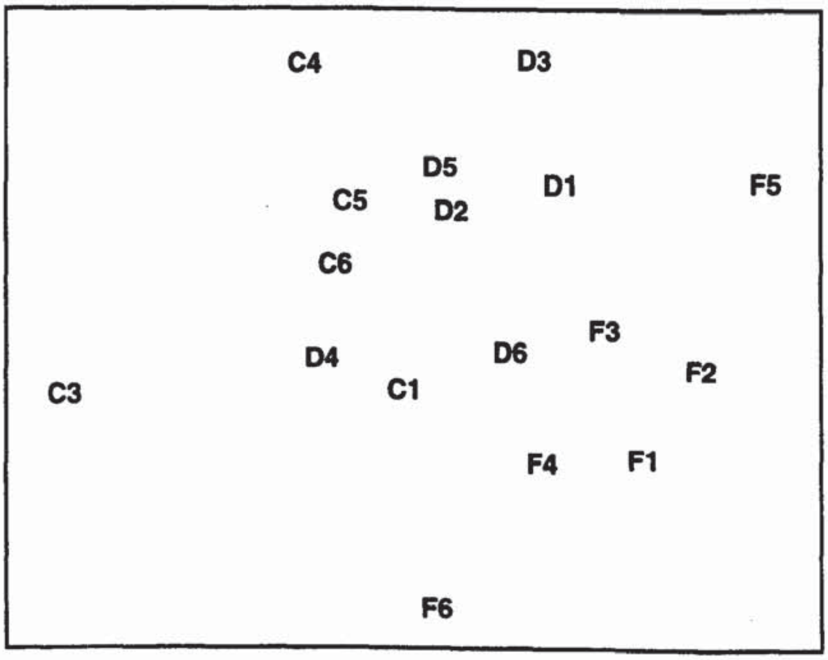


Table 5.4.6: Summary of ANOSIM analyses comparing impacted Sites A, B, E, F, with reference Sites C and D (root/root transformation of abundance: other transformations show similar findings; ns = not significant, s = significant $p<0.05$)

Comparisons between:					
A1 and C1-C6		A3 and C1-C6	A4 and C1-C6	A5 and C1-C6	A6 and C1-C6
s		s	s	s	**ns**
B1 and C1-C6		B3 and C1-C6	B4 and C1-C6	B5 and C1-C6	B6 and C1-C6
s		**ns**	s	s	**ns**
E1,F1 and C1,D1	E2,F2 and C2,D2	E3,F3 and C3,D3	E4,F4 and C4,D4	E5,F5 and C5,D5	E6,F6 and C6,D6
s	**ns**	s	s	s	**ns**

The impacted sites on the Chinn Brook (E, F) do not show any increase in similarity over time with the two reference sites (C, D) (Figs 5.4.4c, d). Table 5.4.6 shows the results of ANOSIM analyses carried out comparing samples from Sites E and F with C and D on each sampling occasion. It can be seen that in four out of six sampling occasions, a significant difference can be seen.

Investigations of the taxa contributing to the differences between Sites C and D, and E and F show that Oligochaeta, *Baetis*, *Ancylus fluviatilis*, *Gammarus pulex*, *Glossiphonia complanata* account for 55% of the dissimilarity. None are particularly good discriminators although *Ancylus fluviatilis* is better than the others.

5.4.6. Summary

The pollution incident had a toxic impact on the macroinvertebrates as was evident from the dead invertebrates seen during the first sampling visit.

The sites on the tributary showed an impact using all measures apart from the diversity index. Few taxa, low BMWP and ASPT scores were seen following the pollution incident. The freshwater shrimp *Gammarus pulex* did not colonise the tributary until 95 days following the incident. The use of MDS configurations showed significant evidence of community similarity between reference sites and impacted sites A and B, 287 days following the incident. Prior to this, reference and impacted Sites A and B are significantly different. Of the five best discriminating taxa between the reference and impacted tributary sites, *G. pulex* was the taxon present in highest numbers at the reference sites.

The analysis of selected taxa showed that Oligochaeta and Chironomidae may have shown an increase in abundance as a result of the pollution, with *G. pulex* and possibly *B. rhodani* demonstrating an initial absence, followed by recolonisation in the case of *G.pulex*.

No impact was apparent on the diversity index. This may have been due to the continued low diversity of the stream with a few taxa showing relatively high abundance. Both taxa richness and BMWP/ASPT scores show a similar temporal impact following the pollution.

The longest impact was shown by the multivariate analysis of the data at184 days. At the following sampling visit, 287 days after the incident, the communities at the tributary sites were not significantly different to those at the reference sites.

The different measures considered give a range of different recovery times, using the non-impacted sites on the Chinn Brook as reference conditions. However it has been shown that these sites support communities that differ significantly from those sites on the Chinn Brook, downstream of the tributary. Their utility as a reference for the tributary, which has quite different flow and discharge figures requires consideration of the changing conditions.

With these caveats in mind, the different estimates of recovery can be viewed as stages along a continuum with total or complete recovery not defined.

Sources of colonisation within the tributary must be either aerial or from downstream as the pollution impact arose from the culvert draining the source of the tributary. The difficulty of assessing recovery in a situation with no adequate reference site is apparent. However temporal changes within the tributary can be tentatively attributed to effects of the pollution incident.

No evidence of impact from the pollution incident was evident in the Chinn Brook. This is probably the result of the dilution potential of the Chinn Brook (see Table 5.4.1). Up and downstream variations in the macroinvertebrate community are probably due to factors other than any influence of the pollution incident.

Table 5.4.7 Summary of Pollution Impact on All Measures

Measure	Type of impact	Spatial Impact	Temporal Impact
Individual Taxa	Chironomidae -increase in abundance Oligochaeta - increase in abundance *Gammarus pulex* - absence	Sites A and B Sites A and B Sites A and B	95 days 184-287 days >95 days
Number of Taxa	Low number of taxa	Sites A and B	95 days
Diversity (H')	No apparent impact		
BMWP, ASPT	Low BMWP and ASPT scores	Sites A and B	95 days
Multivariate analysis	Presumed decrease in similarity of community when compared with controls	Sites A and B	184 days

5.5 Hewell Brook

5.5.1. Description of Incident

On 2 October 1994 a discharge of oil was reported from one of the surface water sewers draining an industrial estate and discharging into the Hewell Brook (Figure 5.5.1). This was found to be kerosene from a burst subterranean pipe supplying an emergency electricity board generator which normally operates for 150-200 hours per year to reinforce the standard electricity supply to Redditch. Security procedures precluded direct site investigation but the NRA pollution control officer conjectured that the kerosene may have acted as a solvent during its passage to the surface water sewer causing a number of unknown compounds to enter the brook with the kerosene. The electricity station was shut down and the pipe disconnected from the fuel supply. NRA estimates were that 9100 litres of kerosene had leaked into the brook prior to action being taken, possibly over a period of 2 days.

Over the next 2 days 4500 litres of water were tankered from the brook, the flow at this point being about 20 l/s (0.02 m^3/s). Three absorbent booms were placed downstream of the outfall and left in place for 3 weeks. Twenty metres downstream of the outfall, a 50 metre bed of Branched Bur-reed (*Sparganium erectum*) had trapped oil and the NRA decided not to disturb this.

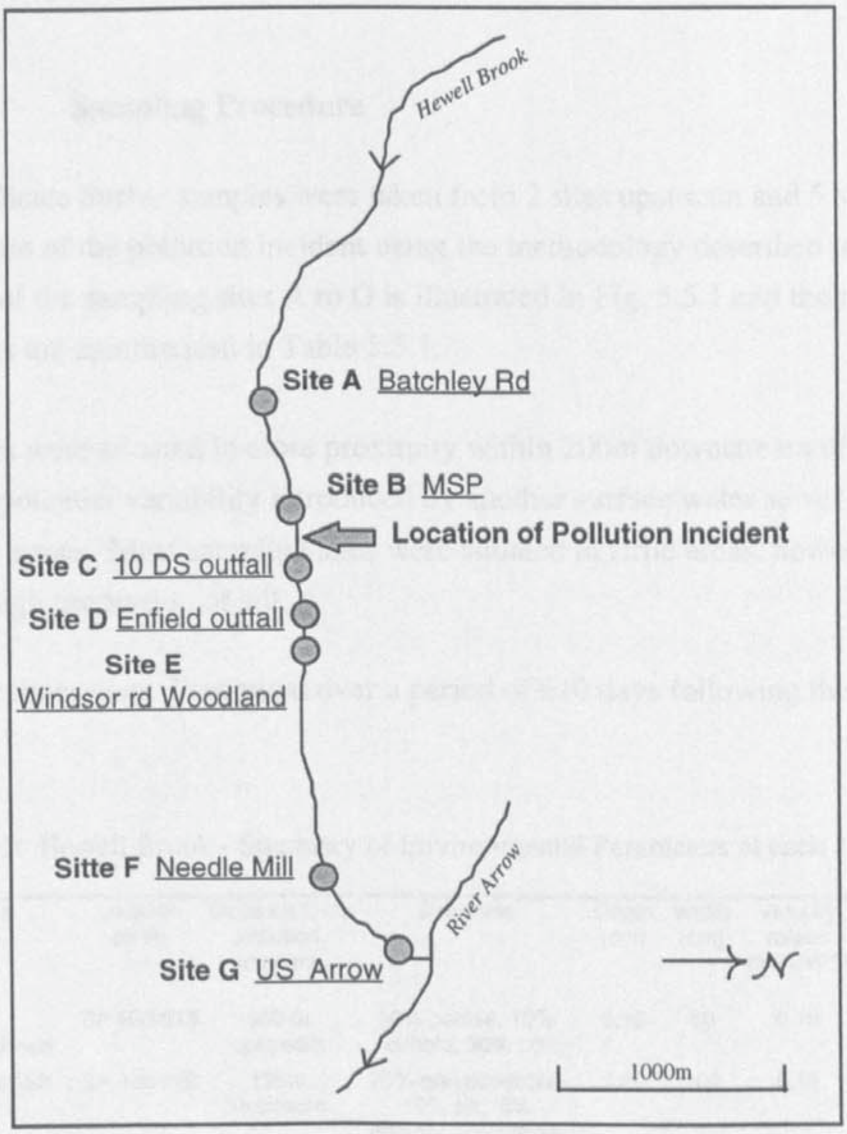
Quantities of oil continued to issue from the pipe for some days and small quantities were seen on the second sampling occasion 29 days following the incident.

5.5.2. Site Description and History

Hewell Brook, located on the northern outskirts of Redditch in Worcestershire, is a first order stream, about 4 km in length before its confluence with the River Arrow (Fig. 5.5.1). The catchment is varied, consisting of agricultural land, housing and industrial estates and a country park. Catchment characteristics as determined from maps is 60 % urban, 30% rural (pasture) and 10% park land.

The brook initially drains agricultural land before entering a residential estate in Redditch. It flows into an ornamental pool (Batchley Pond) which supports fish and waterfowl.

Figure 5.5.1: Hewell Brook - Map to show location of Pollution Incident and Sampling Points



Hewell Brook then receives drainage and surface water effluent from a swimming pool and two industrial estates which have all been the cause of minor pollution incidents in the past. Surface water drainage from residential and commercial areas continues to flow into the brook. On entry into a country park, comprising mainly pasture land, the brook divides into two channels, one of which drains into a marshy area, before rejoining the main channel which flows into the River Arrow.

The NRA does not have information on the water quality of Hewell Brook. There were reports of freshwater crayfish (*Austropotamobius pallipes* (Lereboullet)) in the lower sections of the brook in 1988 (A.Jones: Countryside Ranger), but none have been seen since the construction of a large retail and commercial area with associated highways, the surface drainage from which, discharges into the watercourse above Site F (Fig 5.5.1). This species is known to be particularly sensitive to organic pollution (Holdich and Reeve, 1991).

5.5.3. Sampling Procedure

Three replicate Surber samples were taken from 2 sites upstream and 5 sites downstream of the pollution incident using the methodology described in Chapter 3. Location of the sampling sites A to G is illustrated in Fig. 5.5.1 and their environmental parameters are summarised in Table 5.5.1.

Three sites were situated in close proximity within 200m downstream of the outfall to ascertain potential variability introduced by another surface water sewer from an industrial estate. Most sampling sites were situated in riffle areas, however Sites C and D had a high proportion of silt.

Sampling times were distributed over a period of 610 days following the incident (Table 5.5.2).

Table 5.5.1: Hewell Brook - Summary of Environmental Parameters at each sampling point

Site Name	Location (G R)	Distance from pollution incident	Substrate	Depth (cm)	Width (cm)	Velocity m/sec (31/10/95)	Discharge m^3/s (31/10/95)
Site A - Batchley Road	SP 030 678	550 m upstream	60% pebble, 10% cobble, 30% silt	5;10	80	0.16	0.0069
Site B - MS&P	SP 035 682	125m upstream	70% gravel/pebble, 10% silt, 5% boulder, 5% cobble	5;20	100	0.15	0.0102
Site C - 10 m DS outfall	SP 036 682	10 m downstream	80% silt, 20% gravel	20	250	0.14	0.0243
Site D - US Enfield Outfall	SP 036 682	130m downstream	50% silt, 25% cobble/brick, 25% gravel	20	200	0.08	0.0227
Site E - Windsor Rd Woodland	SP 037 684	200m downstream	65% gravel/pebble, 20% boulder/cobble, 10% sand, 5% debris	10;15	200	0.21	0.0257
Site F - Needle Mill	SP 044 685	925m downstream	50% gravel, 40% cobble, 10% sand	10	150	0.13	0.0161
Site G - US Arrow	SP 049 688	1400m downstream	70% pebble/gravel, 30% cobble	20-30	150	0.15	0.03

Table 5.5.2: Summary of sampling times

Sampling time Code	Date		Days following incident
he1	17 October	1994	15
he2	31 October	1994	29
he3	24 November	1994	53
he4	11 January	1995	101
he5	1 March	1995	150
he6	12 May	1995	222
he7	17 July	1995	288
he8	31 October	1995	394
he9	5 June	1996	610

5.5.4. Predicted Impact of Oil Pollution

The impact of refined hydrocarbon spills on stream ecology has been found to be variable (Guiney *et al.*, 1987; Hoehn *et al.*, 1974; Nauman and Kernodle, 1975; Bury, 1972). The heterogeneity of response appears to reflect the complexity of petroleum compounds (Mason, 1996) and the consequences of their degradation (Harrel, 1985; Foght and Westlake,1987), as well as the different responses of the species found in each stream (Lock *et al.*, 1981; Guiney *et al.*, 1987; McCauley, 1966).

When oil is spilled into water, volatile elements escape quickly, soluble fractions enter the water column, and the remaining insoluble fraction floats or combines with silt and organic matter and sinks leading to gradual chemical and bacterial degradation (Harrel, 1985).

Toxicity can occur as a result of direct contact with the oil and as consequence of the products of oil degradation (Hoehn *et al*, 1974). Representative toxic compounds present in crude oil include the organics benzene, toluene, naphthalene, fluorine, phenanthrene, fluoranthrene, pristane, phytane, pyrene and the inorganics nickel and vanadium (Crunkilton & Duchrow, 1990). The acute toxic effects are mainly due to water-soluble components (Siron *et al.*, 1991; Mackay, 1987), however the effects are complex and can change during degradation. For example, Siron *et al.* (1991) investigating the effects of the water soluble fractions of crude oil on 2 species of marine algae: the diatom *Phaeodactylum tricornutum* and the chlorophyte *Dunaliella tertiolecta*, found both stimulation and inhibition of growth at different stages which they suggest may have been a reaction to chemical changes within the oil fractions over time. Stimulation of bacterial growth has been reported following a kerosene spill

(Guiney *et al.*, 1987) although whether this was a primary effect of the oil, or a secondary effect due to the toxic impact on other elements within the ecosystem, is not clear. The toxicological significance of the different components of the oil product to the biota is therefore difficult to analyse and is likely to change over time as the oil weathers (Crunkilton and Duchrow, 1990).

Oil can also cause mechanical damage to invertebrates due to its high viscosity and adherent properties (Guthrie, 1989) and reduce the supply of oxygen by the prevention of oxygen absorption through the surface membrane, although the latter property is more apparent in still water.

Degradation is caused by bacteria and by ultraviolet light. Macroinvertebrates have also been observed to feed on the oil (Jahn, 1972; Harrel, 1985).

Lock *et al* (1981), studying experimentally oiled substrates, suggested that the greatest impact of an oil spill would be on the communities of the river edge where oil contaminated dry rocks are subsequently submerged and where lower flow rates and vegetation would assist trapping. Least impact would be upon rocks in mid-channel having well established periphyton coating, as experimentally oiled bricks that had first been colonised with periphyton were found to show little effect of oil.

From the above information the following effects on measurements of the biota can be predicted (Table 5.5.3).

Table 5.5.3: Predicted Impact of Pollution Incident on Biotic Measures

Measure	Predicted effect
Selected taxa	Initial toxicity causes loss of sensitive taxa and decrease in abundance of less sensitive taxa. This is followed by a rapid increase in certain taxa benefiting from the changed conditions and then showing a decline and a gradual increase in other taxa as conditions resemble those prior to the incident.
Number of Taxa	Initial low numbers followed by a gradual increase.
Diversity	Initial low diversity followed by a gradual increase.
BMWP and ASPT	Initial low BMWP score and ASPT score as taxa are lost and any taxa are dominated by tolerant taxa, followed by an increase over time.
Multivariate Analysis of Community Similarity	Initial low similarity to upstream 'control' sites followed by an increase in similarity over time.

5.5.5. Results and Analysis

The data is organised as described in Chapter 4. As no pre-impact measures exist, both impact and recovery have to be inferred by comparison with the two unimpacted upstream sites.

5.5.5.1. Water Quality

Table 5.5.4 gives the physicochemical measurements taken at each sampling time. The temperature changes appear to be seasonal and unrelated to the spill. The conductivity measurements tend to be high throughout the brook. The first scouring floods occurred at the end of October 1994, one month after the spill, accounting for some dissipation of the oil.

Table 5.5.4: Physicochemical Conditions in Hewell Brook (Conductivity: µS cm-1 at 25°)

		Site A			Site B			Site C			Site D		
Date	Days following Disturbance	Temp (°C)	pH	Cond	Temp (°C)	pH	Cond	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond
17-Oct-94	15	10.4	NA	NA	10.5	NA	NA	10.5	NA	NA	10.6	NA	NA
31-Oct-94	29	11.2	7.8	315	11.2	7.6	399	11	NA	392	11	NA	383
11-Jan-95	101	7.8	7.7	452	5.7	7.5	407	6	7.7	420	5.9	7.9	402
01-Mar-95	150	6.9	7.6	311	6.7	7.5	329	6.6	7.6	320	6.6	7.7	316
12-May-95	222	11	8.2	462	9.5	7.9	512	9.3	8	525	9	7.7	540
17-Jul-95	288	17.7	7.8	744	15.1	8	559	15.1	8	570	15.2	8	574
31-Oct-95	394	11.2	7.8	563	10.5	7.6	610	10.5	7.7	608	10.6	7.9	642
05-Jun-96	610	16.1	NA	344	13.9	NA	376	15.5	NA	419	14.6	NA	400

		Site E			Site F			Site G		
Date	Days following Disturbance	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond	Temp (°C)	pH	Cond.
17-Oct-94	15	10.6	NA	NA	10.1	NA	NA	NA	NA	NA
31-Oct-94	29	11.1	NA	318	11.6	7.7	409	11.2	7.8	398
11-Jan-95	101	6.5	7.8	433	6	7.7	471	6.2	7.8	473
01-Mar-95	150	6.5	7.8	309	6.7	7.1	424	7.2	7.6	341
12-May-95	222	9.2	8.1	554	9.3	7.5	643	9.6	8.3	546
17-Jul-95	288	15.5	8	590	16.3	7.9	570	17.2	7.9	575
31-Oct-95	394	10.4	7.8	637	10.5	7.7	641	9.5	7.8	619
05-Jun-96	610	14.8	NA	430	15.3	NA	526	16.6	NA	430

Visible Oil within the Water Column and Substratum

Qualitative observations were made of the presence of oil as follows:

- Outfall - oil was still being released from the surface water pipe 15 days after initial reporting of the incident, the amount was reduced 29 days after the incident and no further oil was seen coming from the outfall pipe 53 days after the incident.
- surface film - oil was present as a surface film at Sites C, D, E, and F until 53 days after the incident.
- oil in the substratum - for 29 days following the pollution incident, oil was visible when the substratum was disturbed at Sites D and E . At Site C, the site just downstream of the outfall, some oil was released during disturbance throughout the sampling period of 610 days. The released oil took the form of adhesive balls as well as a surface film. Site C differs from the other sites in having a large proportion of silt in the substratum (Table 5.5.1).

5.5.5.2. Macrophytes

The stream bed between Site C, just downstream of the outfall and Site D supported patches of emergent and submerged vegetation. The vegetation was dominated by a stand of Branched Bur-reed (*Sparganium erectum*) which was starting to die back for the winter at the time of the pollution incident. Oil effluent was trapped within this stand of vegetation. Changes in the vegetation were subjectively assessed during sampling as follows.

Table 5.5.5: Summary of qualitative changes in vegetation cover between Sites C and D

Date	Days following Incident	Observations
17.10.94	15	Dead *Sparganium* leaves downstream of the outfall coated with oil. Some *Potamogeton crispus*.
24.11.94	53	Dead *Sparganium* leaves with trapped oil.
1.3.95	150	Little new growth of *Sparganium* for 15m downstream of the outfall, extensive new growth at 40m downstream of the outfall
17.7.95	288	Sparse *Sparganium* within 15m but dense beds at 20m downstream of the outfall
5.6.96	610	Increase in *Potamogeton crispus* and 2 stands of *Sparganium* within 15m downstream of the outfall.

From the above table it can be seen that recovery of the extent of the emergent vegetation stands did not occur in the season following the incident. Two years after the incident, the vegetation had recovered further but still not to its pre-pollution extent as estimated by the amount of vegetation cover present at the time of the incident. The species present, however, resembled the macrophyte community upstream of the outfall 610 days following the incident, indicating qualitative if not quantitative recovery. The impact on the vegetation was, however, localised and 40 m downstream of the outfall, the macrophyte vegetation was not visibly affected.

5.5.5.3. Selected Macroinvertebrate Taxa

Table 5.5.6 is a summary of the taxa found and their total abundance. 49 taxa were found throughout the study. The seven most abundant taxa are listed below and changes in their presence and abundance through the study are considered in the next section and shown in Fig. 5.5.2.

Asellus aquaticus	Oligochaeta
Chironomidae	*Potamopyrgus jenkinsi*
Gammarus pulex	Sphaeriidae
Hydropsyche angustipennis	

Asellus aquaticus

At impacted Sites C, D and E, no individuals of *A. aquaticus* are found following the pollution incident but numbers gradually increase over the following year and are similar to upstream site B 101-150 days following the incident although continue to rise until 394-610 days following the pollution; an increase which matches that seen at reference Site B. Upstream sources probably account for the majority of the colonising individuals as increases in abundance are first evident at the more upstream impacted sites.

Chironomidae

The pollution incident occurred during a period of seasonally low abundance for this family (late autumn/winter). Numbers increased in the summer following the incident at both the upstream control sites and the impacted sites. Thus no obvious impact of the oil to Chironomidae could be detected. However it is known that different species of Chironomidae tolerate different environmental conditions (Cranston, 1982; Wilson, 1987; Rosenberg and Snow, 1975; Lenat, 1983). A more defined response to the presence of the oil may have been apparent if the Chironomidae had been identified to species level.

Table 5.5.6: Hewell Brook - Taxa found in study

Phylum, Order, Family, Genus, Species	Number found in study
PLATYHELMINTHES TURBELLARIA:TRICLADIDA	
Planariidae	
Dugesia sp.	94
Polycelis tenuis (Ijima)/*nigra* (Muller)	584
Dendrocoelidae	
Dendrocoelum lacteum (Muller)	45
MOLLUSCA	
GASTROPODA:PROSOBRANCHIA	
Valvatidae	
Valvata piscinalis (Muller)	4
Hydrobiidae	
Bithynia tentaculata (L.)	39
Potamopyrgus jenkinsi (Smith)	6017
GASTROPODA:PULMONATA	
Lymneidae	
Lymnaea peregra (Muller)	1540
Lymnaea stagnalis (L.)	3
Physidae	
Physa sp.	3
Planorbidae	
Planorbis sp.	1
Planorbis carinatus (L.)	4
Planorbis vortex (L.)	155
Planorbis albus (Muller)	3
Ancylidae	
Acroloxus lacustris (L.)	1
Ancylus fluviatilis Muller	330
BIVALVIA	
Sphaeriidae	5181
ANNELIDA	
OLIGOCHAETA	8596
HIRUDINEA	
Piscicolidae	
Piscicola geometra (L.)	5
Glossiphoniidae	
Glossiphonia complanata (L.)	543
Glossiphonia heteroclita (L.)	11
Helobdella stagnalis (L.)	
Theromyzon tessulatum Muller	6
Erpobdellidae	80
Erpobdella octoculata (L.)	51
Trocheta bykowskii Gedroyc	1

Phylum, Order, Family, Genus, Species	Number found in study
ARTHROPODA	
ACARINA:HYDRACARINA	133
CRUSTACEA: MALACOSTRACA: ISOPODA	
Asellidae	
Asellus aquaticus (L.)	2117
CRUSTACEA:MALACOSTRACA: AMPHIPODA	
Gammaridae	
Crangonyx pseudogracilis Bousfield	1
Gammarus pulex (L.)	9102
CRUSTACEA : BRANCHIURA	
Argulus foliaceus	1
INSECTA : EPHEMEROPTERA	
Baetidae	
Baetis rhodani Pictet	740
INSECTA : COLEOPTERA	
Dytiscidae larva	3
Elminthidae	
Elmis aenea Muller larva	1
Oulimnius tuberculatus Muller	1
INSECTA : TRICHOPTERA	
Polycentropidae	
Plectonemia conspersa (Curtis)	1
Hydropsychidae	
Hydropsyche angustipennis (Curtis)	1659
Hydropsyche pellucidula (Curtis)	1
Hydropsyche siltalai Dohler	7
Hydroptilidae	
Hydroptila sp.	14
Limnephilidae	10
Limnephilus extricatus McLachlan	2
Limnephilus lunatus Curtis	6
Micropterna sequax McLachlan	3
Leptoceridae	
Athripsodes sp.	26
INSECTA: HEMIPTERA, HETEROPTERA	
Corixidae	6
INSECTA : DIPTERA	
Ceratopogonidae	11
Chironomidae	2167
Simuliidae	347
Tipulidae	95

Gammarus pulex

Numbers of *G. pulex* at the impacted Sites C, D and E are low for 53 days following the incident and then increase in numbers; an increase which is maintained apart from low numbers after 288 days in July, 1995. This occurred throughout the stream, including the reference sites and may have been due to seasonal factors (high temperature resulting in low oxygen levels).

The reduction in *G. pulex* at sites F and G during the latter part of the study (from 288 days to 610 days) can not be explained by the oil incident and may have been related to potential effluents coming from an outfall situated just upstream of Site F.

Oligochaeta

Oligochaeta (mostly Tubificidae) were found to be variable at the reference sites. Sites C, E, F and G show a very low abundance of Oligochaeta 15 days after the incident. Site C supported no Oligochaeta 15 days after the incident but from 29 days, a consistent increase was seen.

A great increase in abundance of Oligochaeta was seen at the furthest downstream sites (Sites F and G) during the latter part of the study, at 610 days.

Potamopyrgus jenkinsi

This species is found in quantity only at Sites B (reference), C, D and E but is absent from the furthest downstream Sites F and G and is present in low numbers at the upstream reference Site A. At Sites C, D and E, *P. jenkinsi* is not present immediately after the pollution. The incident may have caused an initial impact on *P. jenkinsi*, eliminating numbers just downstream of the outfall for 15-29 days. At the reference Site B, *P.jenkinsi* is present throughout the study with an abundance pattern showing 2 peaks at 53 and 222 days following the incident. Population peaks were also found at Sites D and E 53-222 days after the pollution incident. The rapid increase of this species at these impacted sites may have been associated with reduced competition from other species combined with a relatively greater ability to take advantage of favourable habitats in terms of reproductive ability, mobility and eclectic food preferences (Haynes and Taylor, 1984; Haynes *et al.*, 1985).

Sphaeriidae

Sphaeriidae were only frequent at upstream Site A and found in consistent but small numbers at Site G. Sphaeriidae were initially absent at Site C and then showed an increase in numbers 150 days following the incident, with a continued increase to a peak at 394 days.

Hydropsyche angustipennis

The majority of the *H. angustipennis* in the stream are found at one of the reference sites (Site A). This species requires substrate of a certain size on which to spin its nets and is tolerant of a moderate degree of organic pollution (Edington and Hildrew, 1995). The absence of this species at all downstream sites is not due to lack of suitable substrate and may therefore be indicative of a chronic pollution source downstream of Site A, to which *H. angustipennis* may be sensitive.

Table 5.5.7 is a summary of the response of the most abundant taxa to the pollution incident. Certain distinct patterns of response can be elucidated:

- low numbers initially, followed by a gradual increase as shown by *Asellus aquaticus*, *Gammarus pulex*, Sphaeriidae, Oligochaeta. This is the commonly expected response to a pollution incident;
- low numbers initially, followed by an increase that is greater than expected from comparison with control sites, as demonstrated by *Potamopyrgus jenkinsi*;
- no discernible response e.g. Chironomidae.

Figure 5.5.2: Hewell Brook - Changes in abundance of selected taxa following pollution incident (means S.E.)

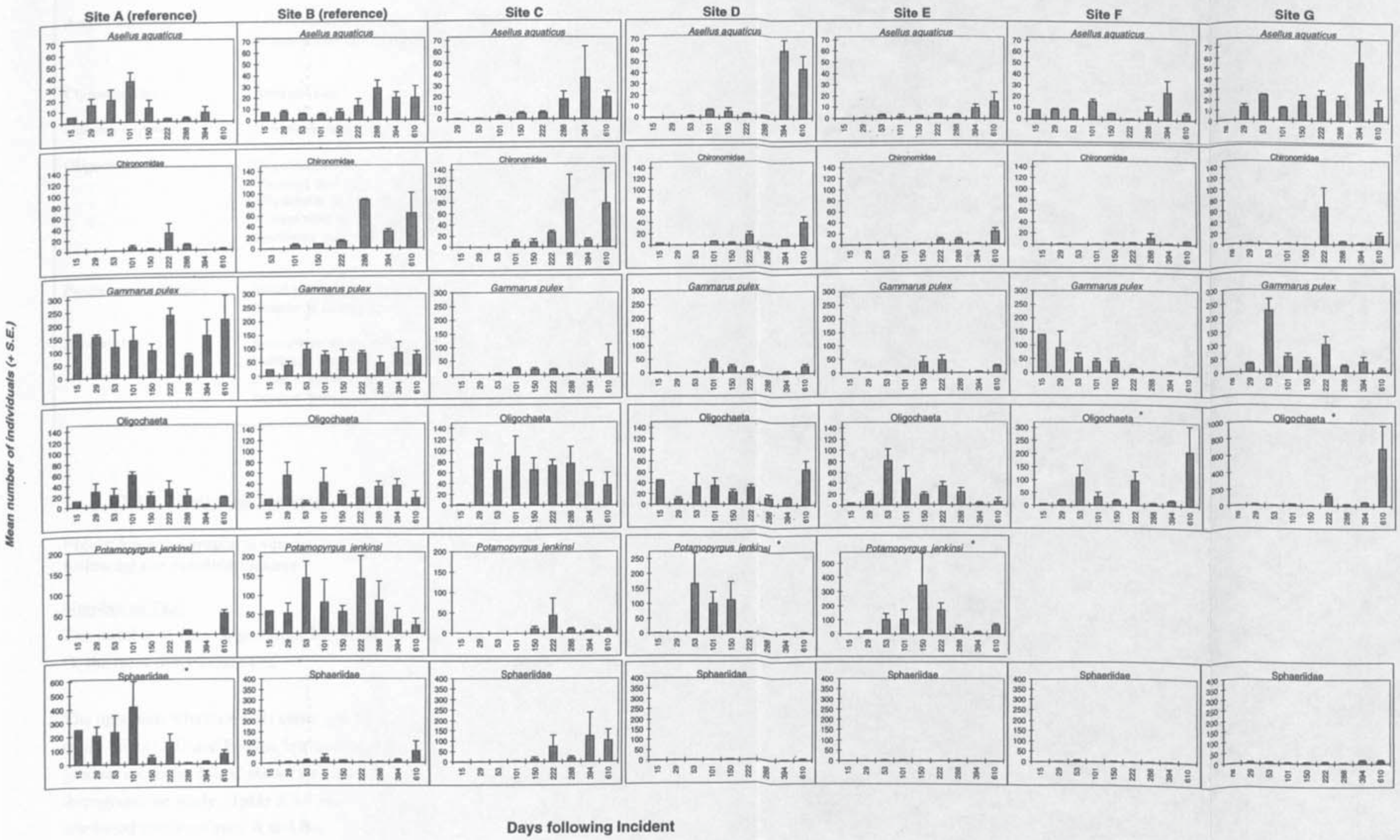


Table 5.5.7: Summary of Impact of pollution incident on the most abundant taxa

Taxon	Impact of pollution
Asellus aquaticus	Low numbers at C,D,E, initially, increasing to 110/150 days, the increases after this matching increase at reference site B
Chironomidae	None obvious
Gammarus pulex	Low numbers at C,D,E for 53 days
Oligochaeta	Oligochaeta (mostly Tubificidae) were found to be variable at the reference sites. Sites C,E,F and G show a very low abundance of Oligochaeta 15 days after the incident. Site C supported no oligochaeta 15 days after the incident but after 29 days, an increase was seen and this abundance was maintained.
Potamopyrgus jenkinsi	Initial low numbers at C,D,E Increase in numbers after 53-222 days at D and E
Sphaeriidae	Sphaeriidae were only frequent at upstream site A and found in consistent but small numbers at Site G. Sphaeriidae were initially absent at Site C and then showed an increase in numbers, 150 days following the incident. with a continued increase to a peak at 394 days

5.5.5.4. Univariate Measures

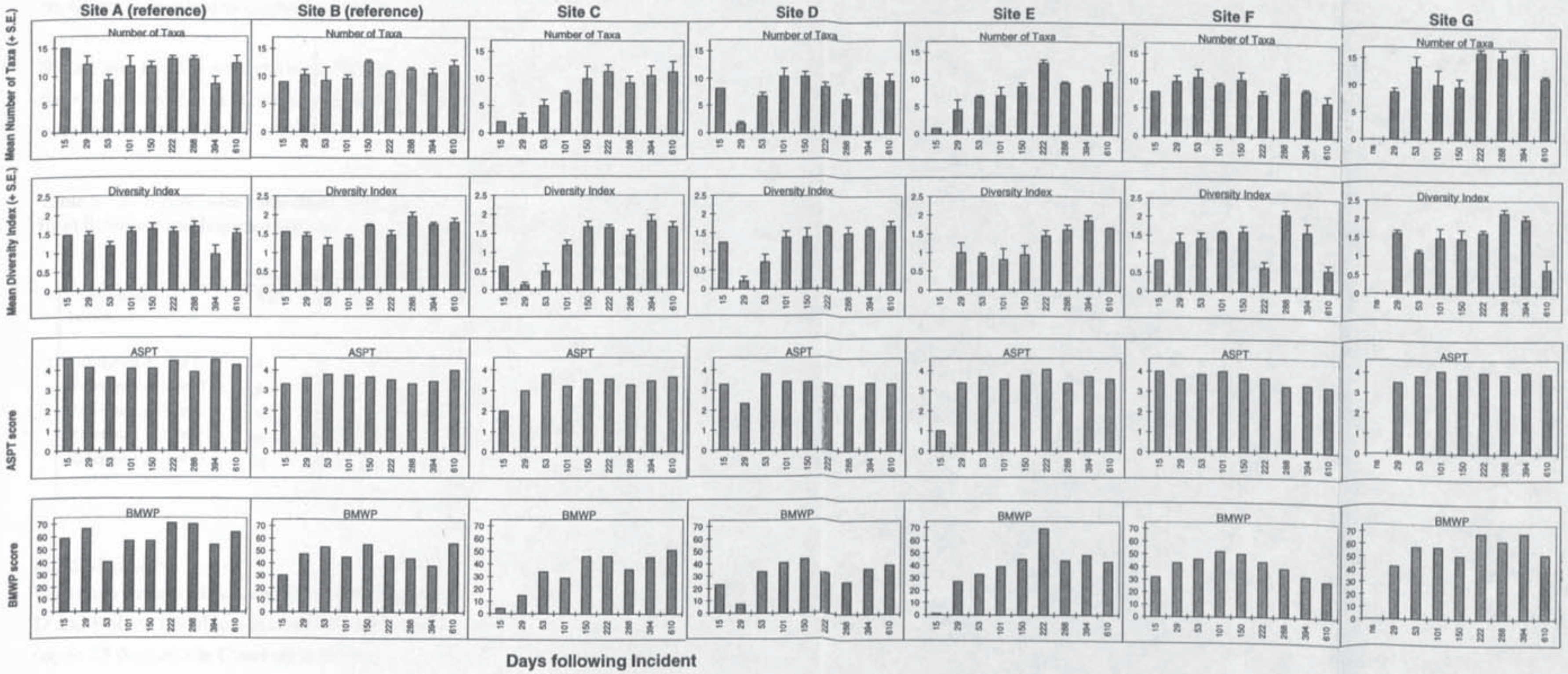
Figure 5.5.3 is a graphical summary of the changes in univariate biotic measures taken following the pollution incident.

Number of Taxa

The number of taxa ranged from 2 at Site C, 15 days following the incident, to 23 at Site G, the most downstream site.

The upstream reference sites show relatively even numbers of taxa throughout the study. Sites C, D and E show low numbers of taxa after the incident followed by a gradual increase. Site C shows the clearest effect with an increase in taxa found throughout the study. Table 5.5.8 shows the results of significance tests comparing combined reference sites A and B with each impacted site (unless otherwise stated, a significant difference involves fewer taxa at the impacted sites). An impact is shown in sites C, D and E with low numbers of taxa following the pollution incident (up to 101 days in Sites C and D and up to 29 days at Site E), however significant differences are

Figure 5.5.3: Hewell Brook - Univariate measures - changes following pollution incident



also found after this, which indicates that either a stable recovery has not occurred, or that the variability in taxa within each replicate is such that a greater number of replicates is required to establish a stable change.

Sites F and G show no significant difference following the incident, although after 222 days, numbers of taxa at Site G exceed those of the reference sites. At 610 days numbers of taxa at Site F are low.

Table 5.5.8: Significance Table-Mann-Whitney U test applied to Number of Taxa in replicates from Reference and Impacted Sites (ns = not significant; s = significant, $p<0.05$)

	Days following Pollution Incident							
comparisons between:	15&29	53	101	150	222	288	394	610
Reference & Site C	s	s	s	ns	ns	s	ns	ns
Reference & Site D	s	s	s	ns	s	s	ns	ns
Reference & Site E	s	ns	ns	s	ns	s	ns	ns
Reference & Site F	ns	ns	ns	ns	s	ns	ns	s
Reference & Site G	ns	ns	ns	ns	s (more in G)	s (more in G)	s (more in G)	ns

Diversity Index

Upstream control sites A and B show relatively consistent diversity over time. Sites C, D and E show initial low diversity which is significantly different to upstream figures (up to 53 days at Site C and up to 29 days at Sites D and E). Diversity at Sites F and G is not significantly different to control sites except at the last sampling period.

Table 5.5.9: Significance Table - Mann-Whitney U test applied to Diversity Indices of replicates from Reference and Impacted Sites (ns = not significant; s = significant, $p<0.05$)

	Days following Pollution Incident							
comparisons between:	15&29	53	101	150	222	288	394	610
Reference & Site C	s	s	ns	ns	ns	ns	ns	ns
Reference & Site D	s	ns	ns	ns	ns	ns	ns	ns
Reference & Site E	s	ns	s	ns	ns	ns	ns	ns
Reference & Site F	ns	ns	ns	ns	s	ns	ns	s
Reference & Site G	ns	ns	ns	ns	ns	ns	ns	s

BMWP and ASPT

The upstream control sites A and B show relatively stable BMWP and ASPT scores. At Sites C, D and E, BMWP scores are initially low and rise to 110-150 days following the incident. Sites F and G also show lower BMWP scores after the pollution but the effect at these sites is less marked.

Sites C and E show lower ASPT scores for 29 days following the pollution but no such effect is seen at the other sites.

This indicates that, as well as a simple loss of taxa, the initial impact of the pollution was a loss of those taxa that are less tolerant of organic pollution.

A relatively greater effect is upon BMWP score than on ASPT which indicates that loss of taxa is more important an impact than loss of taxa sensitive to organic pollution.

Summary of Univariate Measures

Table 5.5.10 summarises the impact of the pollution incident on the univariate measures. An impact is seen at Sites C, D and E with lower numbers of taxa, diversity indices and BMWP and ASPT scores following the pollution incident. The time taken for these sites to show similar scores to the reference sites is variable depending upon the site and the measure used but the number of taxa shows impact at Site C for the greatest length of time.

In Site D, a number of taxa were found 15 days following the incident which were not seen in the next few sampling occasions. These were presumed to have been drifting individuals and not necessarily taxa supported by the site at the time. It is known that unfavourable conditions (flooding, lack of food, toxic substances) can cause an increase in the drift behaviour of macroinvertebrates (Wiley and Kohler, 1984; Bohle, 1978; Bernard et al, 1990). The presence of these taxa, although not numerous, is enough to increase measures of 'number of taxa', diversity index, BMWP and ASPT scores.

Table 5.5.10: Summary of Pollution Impact on Univariate measures

Measure	Impact
Number of Taxa	An impact is shown in sites C, D and E with low numbers of taxa following the pollution incident (up to 101+ days in Sites C and D and up to 29 days at Site E),
Diversity Index	Sites C, D and E show initial low diversity which is significantly different to upstream figures (up to 53 days at Site C and up to 29 days at Sites D and E).
BMWP and ASPT	At sites C, D and E, BMWP scores are initially low for about 53 days following the incident. Sites F and G also show lower BMWP scores after the pollution but the effect at these sites is less marked. Sites C and E show lower ASPT scores for 29 days following the pollution but no such effect is seen at the other sites.

5.5.5.5. Multivariate Measures

The R Statistic was computed for each sampling time as described in Chapter 4. This showed that the replicates within sites are more similar to each other than replicates between sites. Similarity matrices were therefore produced from data from averaged replicates.

Initially a comparison was made between the two reference sites; A and B. An ANOSIM analysis carried out on the samples from each site showed a significant difference between them, with *Hydropsyche angustipennis*, *Potamopyrgus jenkinsi* and *Ancylus fluviatilis* being the best discriminators. *H. angustipennis* and *A. fluviatilis* were more abundant at Site A, and *P.jenkinsi* more abundant at Site B. The difference in community composition may be related to chronic discharges coming from an outfall pipe situated upstream of Site B. Given the proximity of Sites C, D and E to Site B, the influence of any effluent is likely to be similar and therefore for these sites, Site B is considered to be a more valid reference site than Site A. For the furthest downstream Sites F and G, both A and B are used as reference sites.

The similarity matrices were used to construct Multidimensional Scaling configurations (MDS plots) comparing each impacted site with the reference sites (Figs 5.5.4 a-e).

Figure 5.5.4a: Hewell Brook - MDS configuration of Site B (reference) and Site C (impacted)

Root/root transformation. Stress = .08

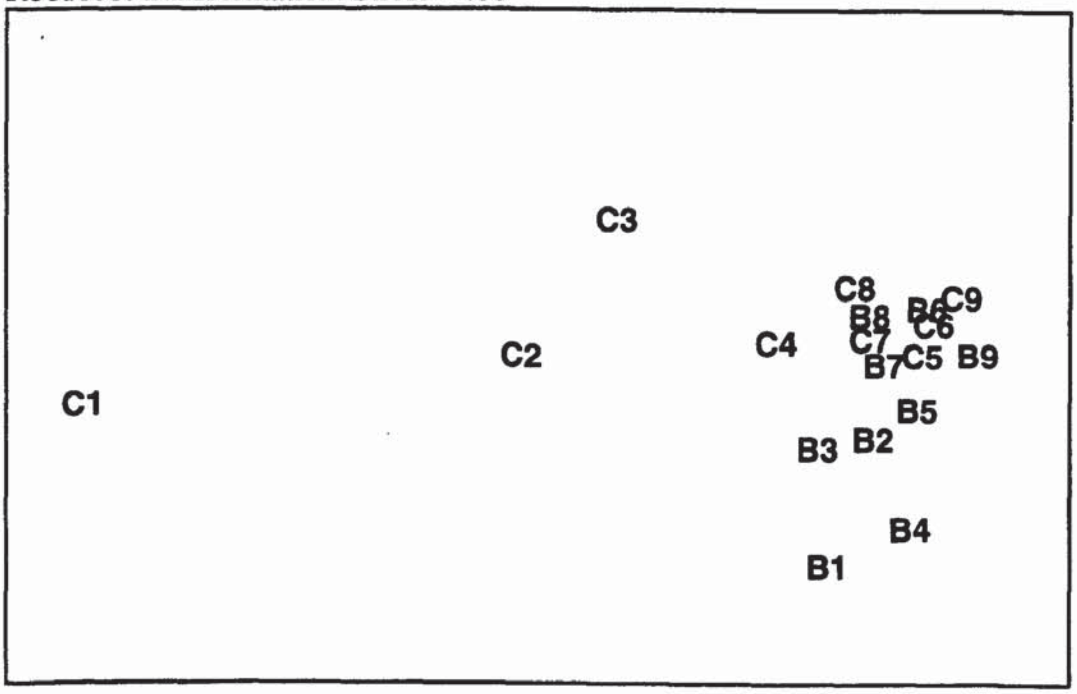


Figure 5.5.4b: Hewell Brook - MDS configuration of Site B (reference) and Site D(impacted)

Root/root transformation. Stress = .09

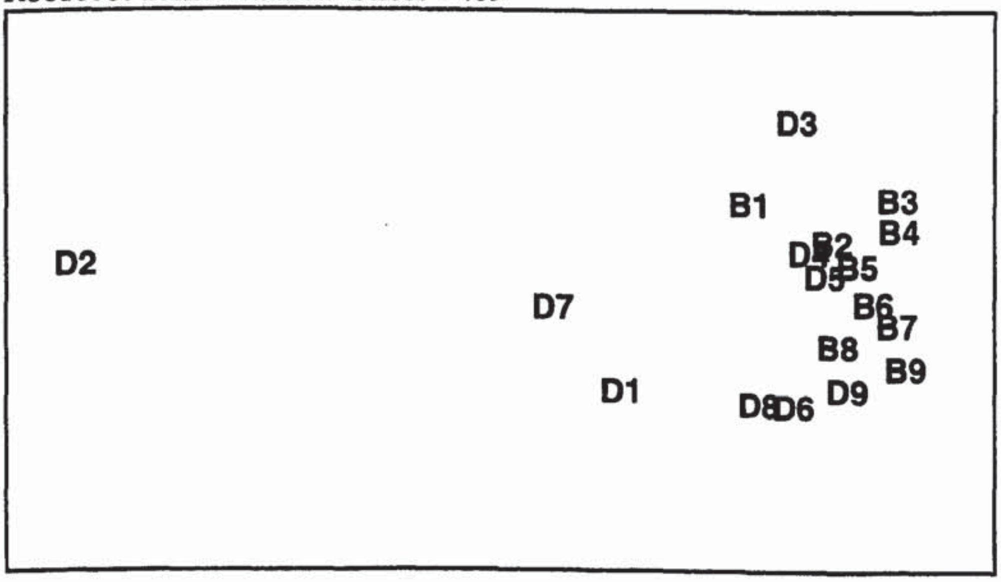
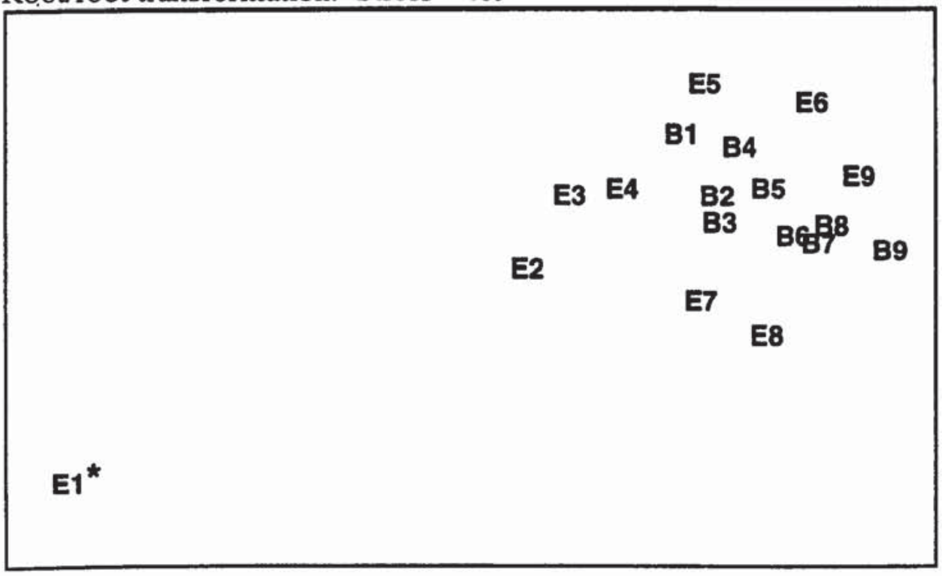


Figure 5.5.4c: Hewell Brook - MDS configuration of Site B (reference) and Site E(impacted)

Root/root transformation. Stress = .09



*MDS configuration placed E1 off the scale so this sample has been repositioned manually to enable the relationships between the other samples to be seen.

Figure 5.5.4d: Hewell Brook - MDS configuration of Sites A, B (reference) and Site F (impacted)

Root/root transformation. Stress = .05

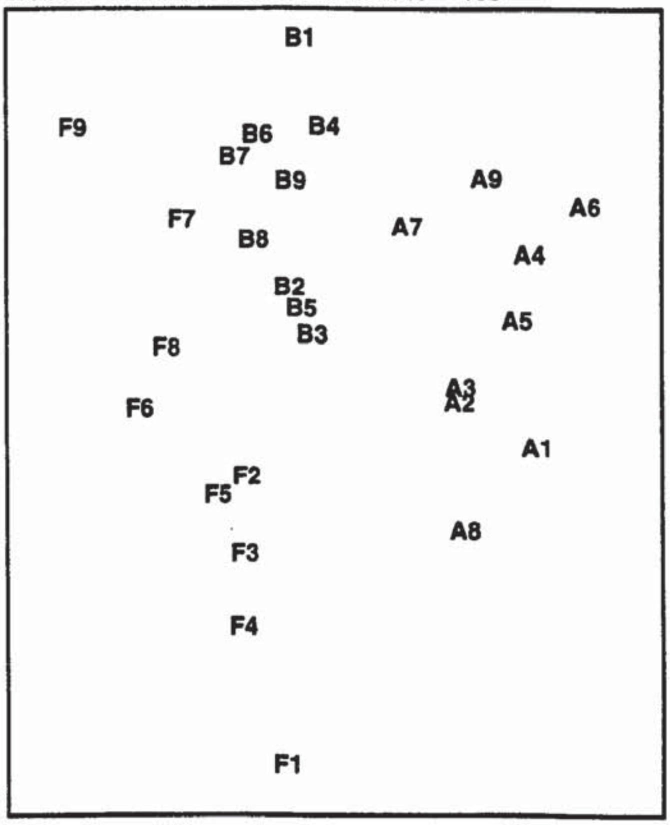


Figure 5.5.4e: Hewell Brook - MDS configuration of Sites A, B (reference) and Site G (impacted)

Root/root transformation. Stress = .19

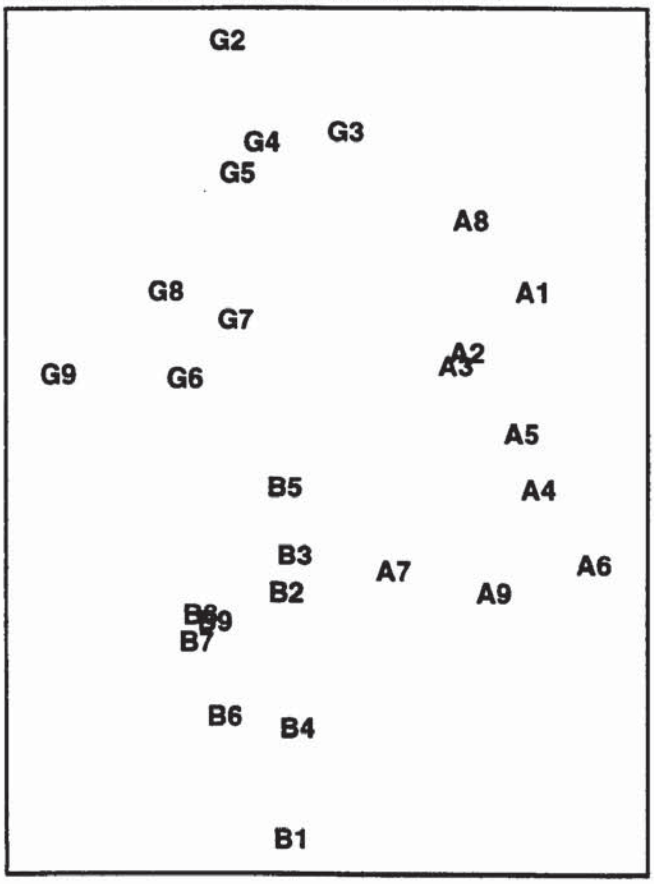


Table 5.5.11: Summary of Multivariate Analysis

MDS configurations of Sites;	Summary of Visual examination of MDS configurations
Site B and Site C	Site C shows progressive similarity to Site B with C5 (150 days) showing greatest similarity
Site B and Site D	D1 was similar to B sites, then D2 showed least similarity with progressively greater similarity shown by subsequent samples, D4(101 days) showing greatest similarity
Site B and Site E	Site E shows progressive similarity to Site B with E4 (101 days) showing greatest similarity
Sites A , B and Site F	Some evidence of impact in that later samples (F6-F8) show greater similarity to B samples than F1-F5. However F9 shows a decrease in similarity
Sites A, B and Site G	No evidence of impact

Visual examination of the MDS plots showed that Sites C, D, E and F showed evidence of pollution impact in that the first samples taken after the incident were least similar to the upstream control sites and there was a progression in time with samples becoming more similar to the upstream controls.

The differences between samples at impacted sites and reference sites were analysed using the ANOSIM procedure described in Chapter 4 (Table 5.5.12). The samples taken at the most downstream impacted sites F and G were significantly different to the reference sites at all times. At impacted sites C, D and E, which were compared with reference site B for the reasons outlined above, the results varied according to the degree of transformation used. If data are not transformed, all the samples taken from Site C differ significantly from the samples at reference Site B. If, however, presence/absence data is used, then communities found on or after C4 (101 days) do not differ significantly from the reference sites.

Sites D, E, F and G do not show a clear pattern of increase in similarity over time to the reference site when significance tests are used and, on the whole, the samples from these sites remain significantly different to the reference site. However, evidence of a progressive increase in similarity is found on the MDS plots. Several explanations can be put forward to explain these findings:

- the variability in environmental conditions at each sampling site are such that there will always be a significant difference between their respective communities even after the effects of the pollution incident are over;
- the pollution incident has had a longer term effect than the time period of this study, i.e. longer than 610 days;
- the use of the ANOSIM procedure requires more replicates and the value of the multivariate approach used lies in the examination of the trends evident in the MDS plots;
- there may be seasonal factors governing the appearance and abundance of certain taxa. The use of all the samples within the reference site as a comparison for each impacted sampling period does not take account of this.

Table 5.5.12: ANOSIM analyses to compare the impacted sites with reference sites at each sampling time (NS = not significant; S = significant, $p<0.05$)

ANOSIM comparisons between:	No Transformation	Root/root transformation	Presence/absence transformation
C1, C2 and B	S	S	S
C3 and B	S	S	S
C4 and B	S	S	NS
C5 and B	S	NS	NS
C6 and B	S	S	NS
C7 and B	S	S	NS
C8 and B	S	S	NS
C9 and B	S	S	NS
D1,D2 and B	S	S	S
D3 and B	S	S	S
D4 and B	NS	NS	NS
D5 and B	NS	NS	NS
D6 and B	S	S	S
D7 and B	S	S	S
D8 and B	S	S	NS
D9 and B	S	NS	NS
E1,E2 and B	S	S	S
E3 and B	S	S	S
E4 and B	S	S	S
E5 and B	S	S	NS
E6 and B	NS	S	S
E7 and B	S	S	S
E8 and B	S	S	S
E9 and B	NS	NS	S
F and A	S	S	S
G and A	S	S	S
F and A,B	S	S	S
G and A,B	S	S	S

5.5.6. Summary

Table 5.5.12 is a summary of the changes in different measures used at each site for the period of 610 days following the pollution incident. The spatial extent of the impact, according to most measures, was within 200 m of the pollution outfall, at Sites C, D and E. Only subjective assessment of the MDS plots showed an impact at Site F, further downstream.

The temporal extent of the impact varied according to the measure used. The qualitative assessments of the vegetation and the substratum showed that, even at the end of the study, 610 days following the incident, evidence of the oil pollution was apparent. Simple observations of the presence of oil following substrate disturbance has been shown in other studies to be a simple field indicator of adverse impacts on the invertebrate community (Harrel, 1985; Crunkilton and Duchrow, 1990). In Hewell Brook, Site C was the only sampling site to show oil within the substratum after 610 days, this being possibly related to the partly depositional nature of the sampling site. Biodegradation of hydrocarbons can be reduced by sorption to particulate matter and sedimentation, particularly when the oil is present as tar balls (Foght and Westlake, 1987).

In terms of the univariate measures used, the macroinvertebrate community may have recovered to reference site levels 610 days after the incident. However, at Site C, a further increase in the total number of taxa was seen on this sampling occasion, and a great increase in the abundance of a dominant species within the brook: *Gammarus pulex*. It is possible that recovery is not complete and that continued changes within the macroinvertebrate community would have been apparent had the sampling period been extended.

Multivariate comparisons of the communities highlighted the importance of choice of a valid reference site. In streams in highly populated areas, it is difficult to find sampling sites upstream of an outfall that are similar to downstream sites prior to impact as even surface water sewers can impact the macroinvertebrate community (Payne and Hedges, 1990). The two reference sites chosen, although close in proximity, showed consistent differences in community composition. In addition, all the impacted sites showed consistent differences with the reference sites. Two main explanations can be put forward for this finding.

Firstly, the conditions at each site, apart from the pollution incident under investigation, are such that the communities at each site would be significantly different. Although this was not demonstrated in the environmental parameters measured, chronic influences on water quality may be present. This may be the case at the downstream site G which, as with the reference sites, does not show the progressive community changes over time seen in the other impacted sites.

Secondly, impact resulting from the incident may still be affecting the community and an extension of the sampling period would have resulted in further movement towards similarity to the reference sites. This may be the case in the most impacted sites C, D and E. Site C shows recovery in terms of 'no significant difference of communities between impacted site and reference site' at 101 days (see Section 5.5.5.5), but only if the abundance is ignored, thus giving equal weight to all the taxa found.

The data can not distinguish between the above two factors but it appears that the first may be important for the furthest downstream sites and a combination of the explanations may be important for sites closest to the impact.

At 610 days, a number of changes within the community were seen at sites F and G: an increase in the abundance of Oligochaeta; a reduction in *G. pulex* and low univariate indices (diversity and richness). These findings do not appear to be related to the pollution incident under investigation and may be the result of unreported pollution coming from an outfall upstream of Site F.

Table 5.5.13: Summary of Pollution Impact on All Measures

Measure	Type of impact	Spatial Impact	Temporal Impact
Individual Taxa	Absence or low abundance (*A.aquaticus*, *G.pulex*, Oligochaeta, *P.jenkinsi*, Sphaeriidae)	Sites C, D, E	29-394 days
Number of Taxa	Reduction in number of taxa	Sites C, D, E	29-101 days
Diversity (H')	Reduction in diversity	Sites C, D, E	29-53 days
BMWP, ASPT	Low BMWP scores	Sites C, D, E	53 days
	Low ASPT scores	Sites C and E	29 days
Multivariate analysis	Decrease in similarity of community to upstream control	Sites C, D, E, F	101-150+ days
Visual assessment of macrophytes	Emergent vegetation coated with oil and subsequent lack of regrowth	Between Sites C and D	288/610 days
Visual assessment of substratum	Oil released when substratum is disturbed	Sites C, D, E	29 -610 days

5.6 River Lostock

5.6.1. Description of Incident, Remedial Action

Following a poor result at a routine biological monitoring site on the River Lostock in Lancashire, further biological samples were taken at sites further upstream on the 1 December, 1995 to investigate the cause.

At a sampling point downstream of a surface water sewer draining an industrial estate, the fauna was found to be sparse in comparison with a sampling point upstream of the outfall, with only *Tubificid* worms present.

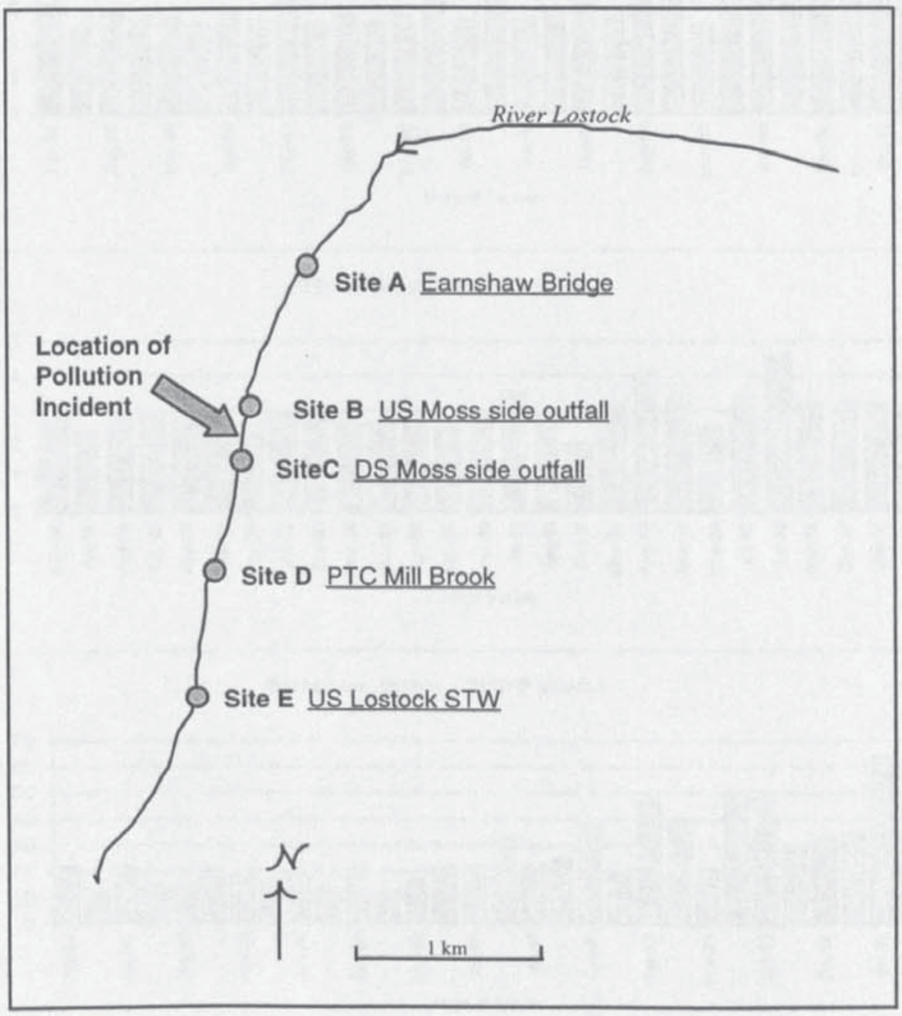
Investigations at the industrial site revealed a leak in a storage tank containing the insecticide Cypermethrin. The length of time the Cypermethrin had been leaking into the surface water drain is not known but the NRA pollution control officer suspected that the leak may have been present throughout the previous summer. As conditions had been abnormally dry during the summer, the insecticide did not reach the watercourse until the rains in the autumn. The levels of Cypermethrin found on the 2 January 1996 was 244.0 μg/l at the outfall and 1.31 μg/l downstream of the outfall. The proposed Environmental Quality Standard for Cypermethrin is 1.0 ng/l - Maximum Acceptable Concentration (WRc, 1994). The leak was repaired in the week prior to the 12 January, 1996.

5.6.2. Site Description and History

The River Lostock forms part of the River Douglas Catchment in Lancashire (Fig. 5.6.1) It is about 25 km in length and initially receives drainage from Rivington Moor and the town of Leyland as well as agricultural areas, comprising mostly pasture, before joining the River Yarrow. The catchment is underlain by sedimentary rock, primarily Millstone Grits (NRA, 1995b).

Water quality on the River Lostock is variable throughout its length, being Class 1A-1B in the upper reaches, and is then subject to runoff from the M6 motorway, urban surface water runoff and effluent from STW, resulting in lower water quality (Class 2) further downstream (NRA data; NRA, 1995b). Historical data from the NRA shows a gradual improvement in BMWP and ASPT scores over a 10 year period at two sampling points (Fig. 5.6.2).

Figure 5.6.1: River Lostock - Map to show location of Pollution Incident and Sampling Points



5.6.3. Sampling Procedure

Samples of the benthic macroinvertebrate fauna were taken by the Environment Agency (NW Region, Central Area). Sampling involved the standard NRA procedure of a 3 minute kick sample (HMSO, 1979), and were sorted and counted in the field.

Samples were taken from 2 sites upstream and 3 sites downstream of the pollution incident. The location of the sampling points A-E is illustrated in Fig. 5.6.1 and their environmental parameters are summarised in Table 5.6.1.

Figure 5.6.2: River Lostock - ASPT and BMWP scores at two sampling point from 1984-1995 (from NRA monitoring data)

All sampling points were situated in riffle areas. The two reference sites had lower flow and discharge values than the other sites due to their upstream locations.

Table 5.6.1: River Lostock - Summary of Environmental Parameters at each sampling point

Site Name	Location (G R)	Distance from pollution incident (m)	Substrate	Average Depth (cm)	Width (cm)
Site A - Earnshaw Bridge (reference)	SD 529 228	1200m upstream	Boulder/cobble 60%,Gravel/pebble 30%, sand5%, Silt/clay5%	25	500
Site B - US Moss Side outfall (reference)	SD 526 225	2m upstream	Boulder/cobble5%,gravel/ pebble80%, sand5%, silt/clay10%	30	300
Site C - DS Moss Side outfall	SD 526 225	10m downstream	Boulder/cobble5%,gravel/ pebble80%, sand5%, silt/clay10%	30	300
Site D - ptc Mill Brook	SD 524 215	1000m downstream	Boulder/cobble80%, gravel/pebble10%, sand10%	55	500
Site E - US Leyland STW	SD 524 209	2000m downstream	Boulder/Cobble10%, gravel/pebble60%, sand20%, silt/clay10%	30	600

Eight sampling times were distributed over a period of 465 days following the incident (Table 5.6.2) Sampling times 1-4 were taken before abatement of the cypermethrin leak which is presumed to have occurred on January 12 1996.

Table 5.6.2: Summary of sampling times (s = sampled)

Sampling Occasion	Date	Time before or after abatement of leak	Sites sampled				
			A	B	C	D	E
1	01-Dec 1995	42 days before	s	s	s	s	s
2	12-Dec 1995	30 days before		s	s	s	s
3	20-Dec 1995	22 days before		s	s	s	s
4	05-Jan 1996	8 days before		s	s	s	s
5	23-Feb 1996	42 days after		s	s	s	s
6	07-Aug 1996	86 days after	s	s	s	s	s
7	27-Nov 1996	318 days after	s			s	s
8	17-Mar 1997	423 days after		s	s		

5.6.4. Predicted Impact of Pollution

The predicted impact of pesticide pollution can be determined from a combination of toxicological data and information on the impacts resulting from accidental discharges of insecticides. A summary of the predicted impact is shown in Table 5.6.3.

Cypermethrin is a pyrethroid insecticide which is highly toxic to aquatic biota, particularly invertebrates, the levels recorded in the stream being far higher than the M.A.C proposed by the WRc (see Section 5.6.2). Stephenson (1982) showed that the 24 hr LC_{50} values ranged from 0.05 µg/l for Hydracarina to 0.2 µg/l for *Asellus aquaticus.* Field experiments indicated high mortality of fish and crustaceans following the introduction of Cypermethrin but no loss of fish and most species had recolonised the pond after 10 weeks (Crossland, 1982). Levels of 2.52 µg/l for 10 days resulted in the death of fish and invertebrates (Davies *et al.*, 1994). Sibley *et al.* (1991) found application of permethrin, another pyrethrin analogue, to a forested stream caused catastrophic drift of all invertebrate taxa (only insects present) within minutes of application. The proportion of silt in the substratum was low and permethrin levels decreased rapidly. Most taxa recovered their abundance levels within 6 weeks, however the individual species were differentially affected depending on the life cycle position at the time of application. One year later, all species were back to reference conditions. Cypermethrin is known to lose its toxicity after 80 days (Environment Agency, Pollution control staff).

The levels of Cypermethrin found on the 2 January, 1996, of 244.0 µg/l at the outfall and 1.31 µg/l downstream of the outfall show that levels were toxic to biological life close to the outfall, and would probably have remained within the sediment for up to 80 days following removal of the leak.

Invertebrate taxa that can burrow within the substratum may escape the effects of a transient pollution. This was found for example, by Victor and Ogbeibu (1986) when monitoring a pesticide impact on a tropical stream. Here, large populations of Chironomidae and Oligochaeta were found two weeks following the incident.

Table 5.6.3: Predicted Impact of Pollution Impact on Biotic Measures

Measure	Predicted effect
Individual taxa	Initial toxicity causes loss of aquatic arthropods and decrease in abundance of less sensitive taxa; oligochaeta and mollusca This is followed by a rapid increase in certain taxa benefiting from the changed conditions and then showing a decline and a gradual increase in other taxa as conditions resemble those prior to the incident.
Number of Taxa	Initial low numbers followed by a gradual increase.
Diversity	Initial low diversity followed by a gradual increase.
BMWP and ASPT	Taxa sensitive to insecticide are not necessarily the most sensitive to organic pollution making predictions of impact on this score impossible.
Multivariate Analysis of Community Similarity	Initial low similarity to reference sites followed by an increase in similarity over time.

5.6.5. Results and Analysis

The data is organised as described in Chapter 4. Impact of the pollution incident can be assessed by comparison with the pre-impact samples taken at the standard NRA monitoring points and with the two upstream reference samples.

5.6.5.1. Selected Taxa

Table 5.6.4 is a summary of the taxa found and the number of times each taxon falls into the abundance categories used by the NRA. 30 taxa were found throughout the study. The 6 most abundant taxa are listed below and changes in their presence and abundance through the study are considered in the next section, summarised in Table 5.6.5 and graphically represented in Fig. 5.6.3.

Asellus aquaticus
Baetis rhodani
Chironomidae
Lymnaea peregra
Tubificidae
Sphaeriidae

Figure 5.6.3: River Lostock - Changes in abundance of selected taxa following pollution incident

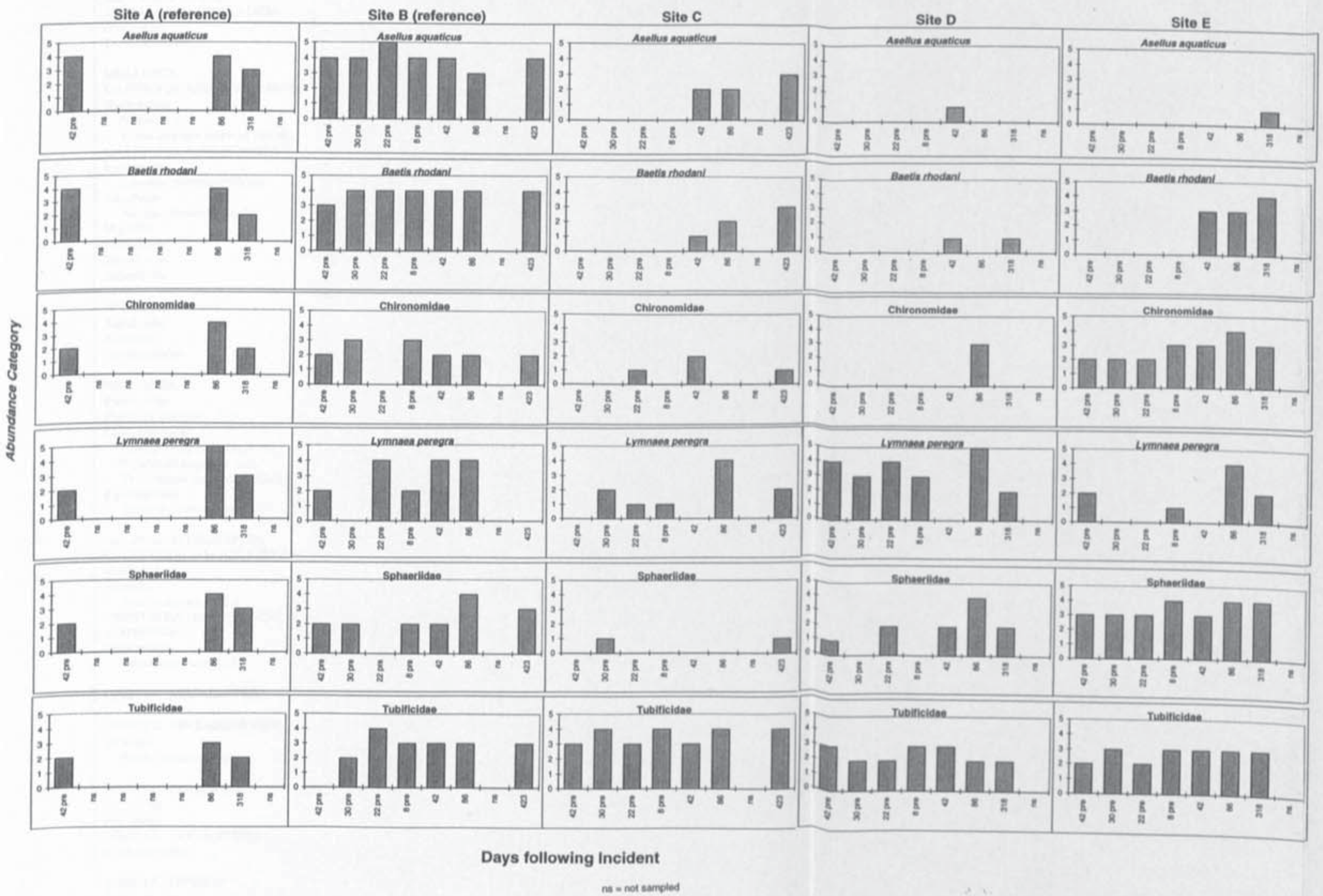


Table 5.6.4: River Lostock - Summary of Taxa found and numbers in each abundance class

Phylum, Order, Family, Genus, Species	*Abundance Class 1	Abundance Class 2	Abundance Class 3	Abundance Class 4	Abundance Class 5
PLATYHELMINTHES					
TURBELLARIA:TRICLADIDA					
Planariidae	6	7	5		
Dendrocoelidae	6				
MOLLUSCA					
GASTROPODA:PROSOBRANCHIA					
Hydrobiidae					
Bithynia sp.	1	1			
Potamopyrgus jenkinsii (Smith)	4	6	5	3	
GASTROPODA:PULMONATA					
Lymnaeidae	1	1	1	1	
Lymnaea peregra (Muller)	3	8	3	7	1
Ancylidae					
Ancylus fluviatilis Muller	4	3			
Physidae	6	4	0	1	
Planorbidae	6	3	3	1	
BIVALVIA					
Sphaeriidae	5	8	6	6	
ANNELIDA					
Tubificidae		6	16	8	
Naididae	4	4	3		
Lumbriculidae	4	1			
HIRUDINEA					
Piscicolidae					
Piscicola geometra (L.)	2				
Glossiphoniidae					
Glossiphonia complanata (L.)	6	7	4		
Helobdella stagnalis (L.)	6	3			
Theromyzon tessulatum (Muller)	2				
Erpobdellidae	1	3	5		
Erpobdella octoculata (L.)	1	9	8	3	
ACARINA:HYDRACARINA	1	3	4	4	
CRUSTACEA: MALACOSTRACA: ISOPODA					
Asellidae					
Asellus aquaticus (L.)	2	2	3	7	1
CRUSTACEA:MALACOSTRACA: AMPHIPODA					
Gammaridae					
Gammarus pulex (L.)	2				
INSECTA : MEGALOPTERA			1		
INSECTA : EPHEMEROPTERA					
Baetidae					
Baetis rhodani Pictet	5	2	4	9	
INSECTA : COLEOPTERA					
Dytiscidae			1		
Haliplidae	1	1			
INSECTA : TRICHOPTERA					
Hydropsychidae	1				
INSECTA : DIPTERA					
Chironomidae	2	9	6	1	
Simuliidae	1		2	1	
Tipulidae	8	9	5		

* Abundance Class 1 = 1 individual per sample
2 = 2-5 "
3 = 6-20 "
4 = 21-100"
5 = 101-500 "

Asellus aquaticus

A. aquaticus is abundant at the reference sites throughout the study. During the pollution leak, this species is absent at Site C but 42 days later it is found in low numbers. 453 days later the abundance is similar to reference Site B. Sites D and E show few *A. aquaticus* during and after the pollution incident.

Baetis rhodani

Baetis rhodani is an abundant component of the fauna in the River as seen in the two upstream reference sites. At Site C, however, *B.rhodani* is absent during the pollution, is found in low numbers 42 days after the pollution and increases in abundance on each sampling occasion after this. At Sites D and E, *B.rhodani* is absent during the pollution and only found in low numbers after this.

Chironomidae

Chironomidae are consistently present in the communities of Sites A, B and the downstream Site E. Occurrence at the most impacted Sites C and D is sporadic both during and after the incident. At Site C recovery of Chironomidae populations to the levels found at site B had not occurred by 423 days.

Lymnaea peregra

L. peregra is found at both the reference sites and is also found at the impacted sites during and after the incident. Some depression of abundance in Site C when compared with Site B may be apparent but as the difference is similar both during and after the incident, any differences may be the result of other environmental differences between the two sites e.g. differences in abundance of algae and *Potamogeton pectinatus* .

Sphaeriidae

Sphaeriidae are found consistently at the reference sites and the impacted Site E. Only two individuals are seen at Site C however, and Site D shows abundance similar to reference Site B.

Tubificidae

Tubificidae are present in abundance at the reference and impacted sites throughout the study.

Table 5.6.5: Summary of Impact of pollution incident on 6 most abundant taxa

Taxon	Impact of pollution
Asellus aquaticus	Low abundance at Site C. numbers similar to reference B after 423 days
Baetis rhodani	Absent at Site C at 42 days, then progressive increase to almost Site B level after 423 days. Sites D and E few found throughout study.
Chironomidae	Sporadic presence at impacted sites both during and after incident.
Lymnaea peregra	No apparent effect
Sphaeriidae	Possible reduction of numbers at sites C and D
Tubificidae	No apparent effect

5.6.5.2. Univariate Data

Fig. 5.6.4 is a graphical summary of the univariate biotic measures in the River Lostock.

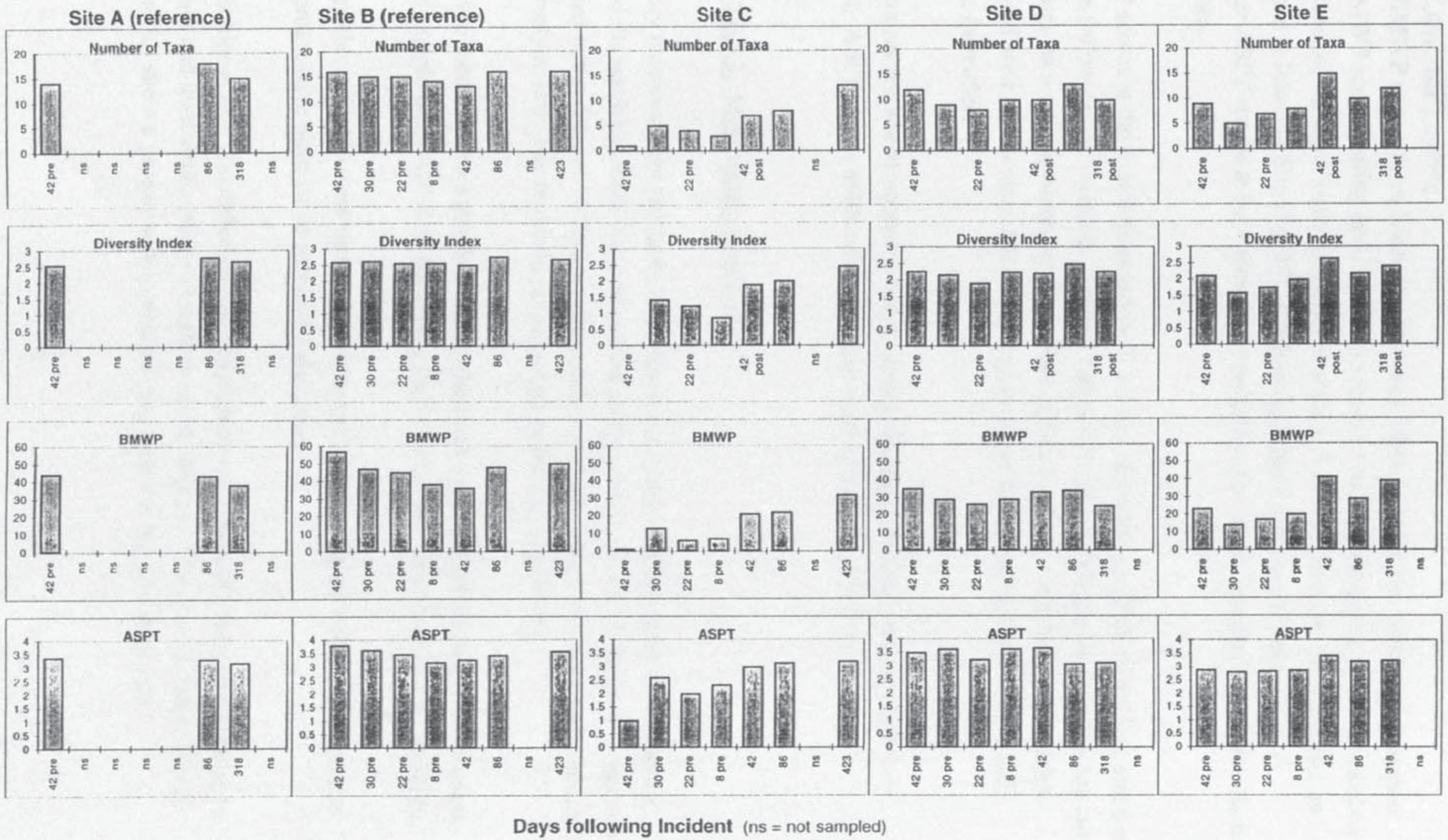
Number of Taxa

The number of taxa at the reference sites is similar and varies between 15 and 20. At the impacted Site C the number of taxa during the incident is low, dropping to one taxon at 42 days prior to cessation of the incident. Following the incident, the number of taxa shows a steady increase with numbers close to the reference site at 423 days following the incident. Numbers of taxa at the downstream Sites D and E are generally lower than reference sites throughout the study and variable.

Diversity Index

The diversity index at the reference sites remains consistent over the sampling period. The diversity index at the impacted Site C is low during the incident but increases thereafter to levels close to reference levels. At the impacted Sites D and E, the index is variable throughout the sampling period.

Fig. 5.6.4 River Lostock - Changes in univariate indices following pollution incident



BMWP and ASPT scores

When BMWP scores from NRA data between 1993 and October 1995 are compared with BMWP scores during and after the incident at the two sampling points utilised on each occasion, no significant difference is seen at Site A (reference) but a significant difference is found at Site D (PTC Mill Brook) (Mann-Whitney U Test, $p<0.05$). However ASPT scores at both sites show no difference when compared with historical NRA data.

BMWP scores at the first impacted Site C are low during the pollution incident and then increase following the incident. BMWP scores 423 days following the incident are still not as high as scores upstream of the outfall at Site B. Scores at Site D are variable during and after the incident and scores at Site E are lower during the incident and increase thereafter.

ASPT scores are low at impacted Site C during the incident and increase after the incident. ASPT scores at the other sites are relatively consistent over time.

5.6.5.3. Multivariate Analysis

Similarity matrices were produced to compare each impacted site with the reference sites and the resulting MDS plots are shown in Figs 5.6.5a-c. As abundance categories were used in the data, no further transformation was carried out. Similarity was assessed subjectively as only one replicate per sampling point had been taken.

Site C was seen to show a progressive similarity in communities to the reference sites with the sample taken 423 days following the incident showing the greatest similarity.

Communities at Site D were more variable over time than the reference communities but no progressive increase in similarity was seen.

Communities at Site E sampled during the pollution incident (E1-E4) were similar to each other and least similar to the reference communities. After the incident, Site E communities show a progressive increase in similarity to the reference site.

Figure 5.6.5a: River Lostock - MDS configuration of Site A,B (reference) and Site C

No transformation. Stress = .08

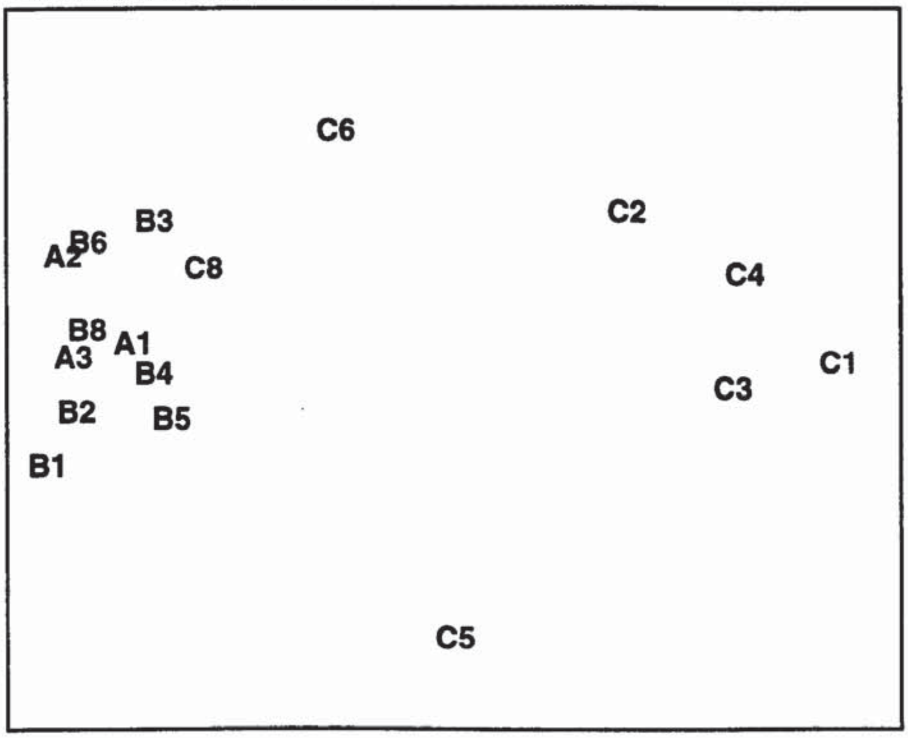


Figure 5.6.5b: River Lostock - MDS configuration of Site A,B (reference) and Site D

No transformation. Stress = .15

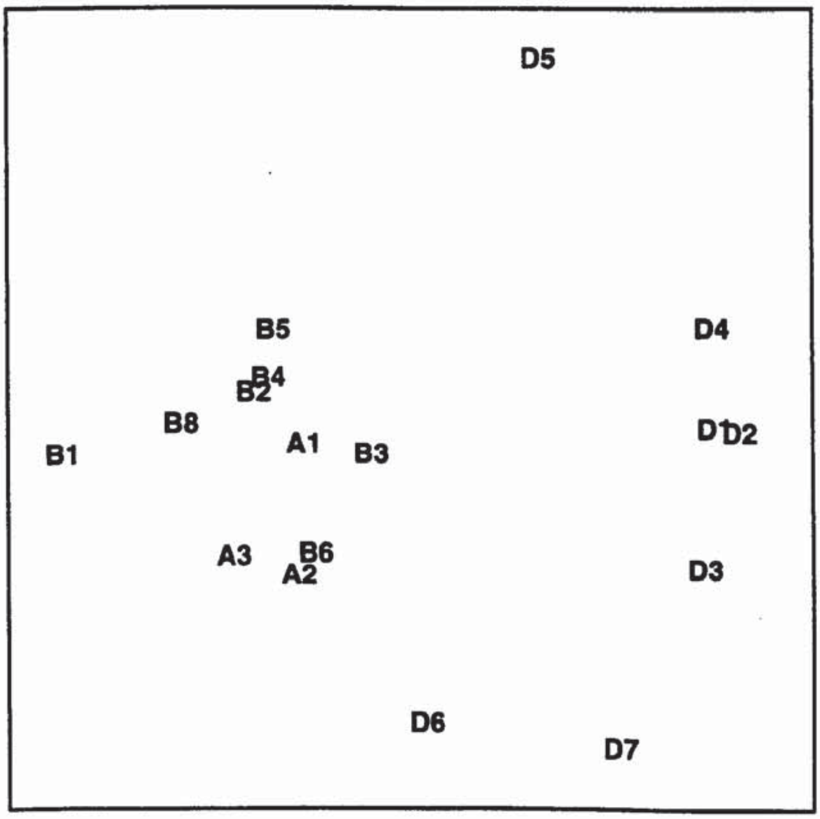
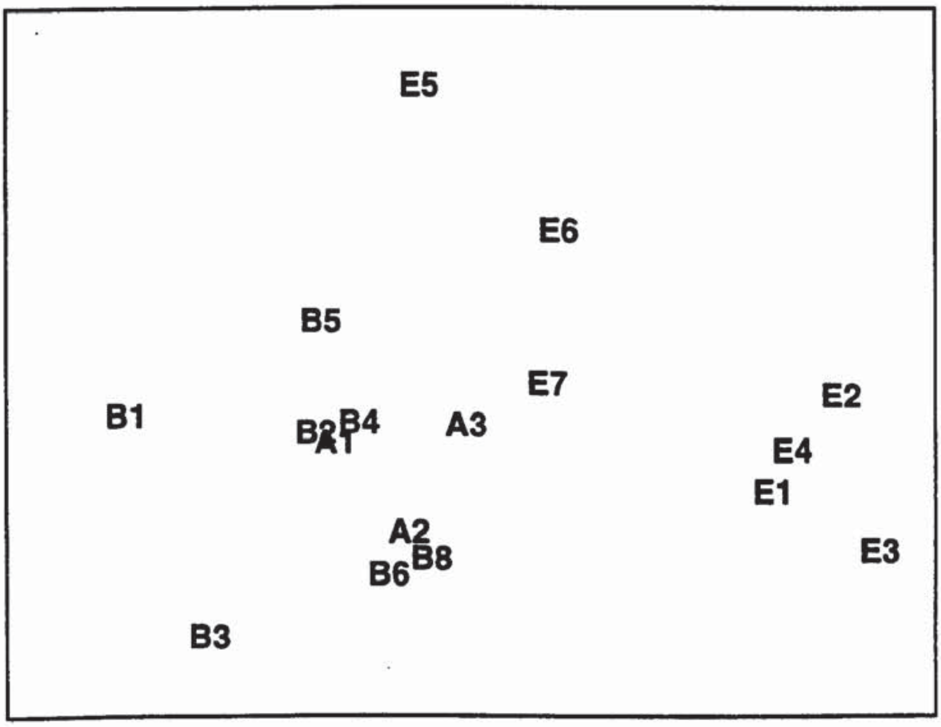


Figure 5.6.5c: River Lostock - MDS configuration of Site A,B (reference) and Site E

No transformation. Stress = .15



5.6.6. Summary

Table 5.6.6 summarises the pollution impact on all the biotic measures used.

In spite of the variety of polluting influences within the River Lostock, the impact of a pesticide pollution in a stream was clear when comparisons of the impacted biotic community were made with upstream reference sites. The site just downstream of the outfall showed the greatest impact with low numbers of taxa, and low measures on univariate indices and low community similarity to reference sites as shown by the MDS configurations. All measures show a gradual increase in time following abatement of the incident with the greatest similarity to the reference sites shown 423 days following the incident. Subsequent samples would need to be taken to show whether full recovery had occurred by this time. At the sites further downstream, the picture is more variable. There is high possibility of intermittent pollution sources between Sites C and E which may contribute to the relative variability seen in these sites when compared with the upstream reference sites. Historical data from the NRA shows that BMWP scores (although not ASPT scores) at Site D during and after the incident were lower than scores from 1992-1995.

Table 5.6.6: Summary of Pollution Impact on All Measures

Measure	Type of impact	Spatial Impact	Temporal Impact
Individual Taxa	Asellus aquaticus - low abundance	Site C	86 days
			86-423 days
	Baetis rhodani "	Site C, Sites D and E	+
	Chironomidae "	Sites C, D, E	423 days
	Sphaeriidae "	Sites C, D, E	423 days
Number of Taxa	Low numbers of taxa	Site C	423 days+
Diversity (H')	Low diversity index	Site D	423 days+
BMWP, ASPT	Low BMWP	Site C,D	423 days+
	Low ASPT	Site C	423 days+
Multivariate analysis	Presumed decrease in similarity of impacted community to reference	Site C	86 days
		Site D	86 days+

5.7 Mousesweet Brook

5.7.1. Introduction

The incident at Mousesweet Brook was investigated as part of the pilot project. Sampling was limited to timed kick sampling at each site with no replication, making assessment of impact on the abundance of taxa more difficult to determine. However, the data provides some useful information regarding the measurement of the impact of pollution incidents and is therefore included within the present study.

5.7.2. Description of Incident

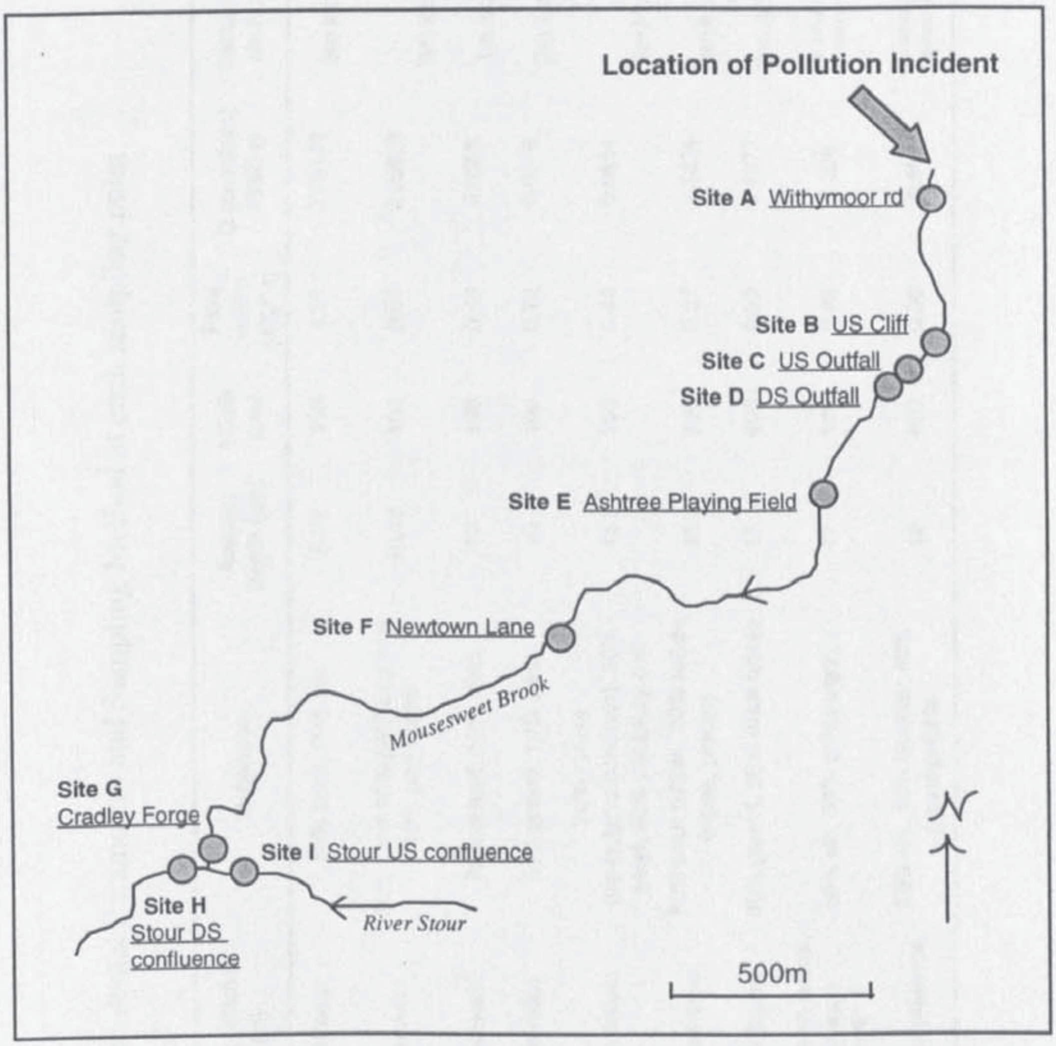
On 20 March, 1994, approximately 9000 litres of Hydrochloric acid was released from a metal plating firm into a culvert draining surface water and comprising the upstream section of the Mousesweet Brook in Dudley, West Midlands (Fig. 5.7.1). Dead sticklebacks were seen by local residents 600 m from the outfall. The pH within the brook to its confluence with the River Stour, 3.5 km downstream, fell to pH 1.0. but had returned to normal levels within one day. On 20 December 1994, another similar spill occurred in the brook. Again pH levels fell to 1.0 within the Mousesweet Brook to its confluence with the River Stour. On this occasion, the NRA introduced neutralising agent (lime) to the brook and pH levels were back to normal after one day.

5.7.3. Site Description and History

The Mousesweet Brook is a first order watercourse about 3.5 km in length. It is a tributary of the River Stour which joins the River Severn at Stourport, Worcestershire. The underlying geology comprises the relatively impermeable Carboniferous strata. The average annual rainfall is 700 mm which is lower than average for the region (NRA, 1992).

The catchment of the Mousesweet Brook is largely urban, draining industrial and residential estates and public open space. The brook is subject to discharges from combined sewer overflows, surface water sewers, and drainage from contaminated land (Patterson, 1993; NRA, 1992). These contribute to high BOD values during storm

Figure 5.7.1: Mousesweet Brook - Map to show location of Pollution Incident



conditions and heavy metal contamination. Water quality data supplied by the NRA in Table 5.7.1 shows high BOD and ammonia levels. The River Ecosystem Classification for the Mousesweet Brook is RE4 (NRA, 1995). The chemical water quality GQA grade in 1994 was Grade E in the upper reaches of the brook and Grade C close to its confluence with the River Stour.

Table 5.7.1: Water Quality Data in Mousesweet Brook 1992-94 (means and standard deviations of 36 samples) (NRA data)

Location	BOD (5 day BOD ATU inhibited, mg/l)	NH3 (Total ammonia as N mg/l)	DO (% saturation mg/l)	NH3 (unionised ammonia as NH3, mg/l)
SO 953 871 (Halesowen rd)	3.86 (1.84)	1.79 (1.26)	89.18 (4.74)	0.0356 (0.0318)
SO 934 846 (Forge Lane)	2.75 (2.86)	0.3 (0.37)	92.43 (4.24)	0.0038 (0.0036)

Table 5.7.2: Mousesweet Brook - Summary of Environmental Parameters and Sampling Method at each sampling point

Site Name	Location (G R)	Distance from pollution incident (m)	Substrate	Average Depth (cm)	Width (cm)	Flow m/sec (3/8/94)	Q (cu.m/sec) (3/8/94)	Sampling Method
Site A - Withymoor rd	SO 954 876	30m downstream	80% brick, 20% silt	7,10	150	0.61	0.05124	3m kick
Site B - US cliff	SO 954 874	500m downstream	50% sand,40% pebble/gravel, 10% part bricks	10,15	100	0.57	0.05928	3m kick
Site C - US Outfall	SO 954 872	550m downstream	90% gravel, 10% sand	10	150	0.49	0.0588	1m kick
Site D - DS Outfall	SO 953 871	600m downstream	90% gravel, 10% sand	10	100	0.72	0.0576	3m kick
Site E - Ashtree Playing Field	SO 952 868	1000m downstream	20% refuse (pipes,tyres.concrete), 20% sand, 60% pebble/gravel	15	150	0.48	0.0864	3m kick
Site F - Newtown Lane	SO 946 864	1800m downstream	80% brick rubble, 20% refuse (pipes, plastic)	16	200	0.57	0.1824	3m kick
Site G - Cradley Forge	SO 934 856	3500m downstream	80% gravel, 20% brick rubble	13	400	0.99	0.4277	3m kick
Site H - Stour DS confluence	SO 934 856	3550m downstream outfall, 50m downstream confluence	50% silt, 50% pebble/gravel	17	500	0.99	0.396	2m sweep in *Ranunculus*
Site I - Stour US confluence	SO 935 856	50m upstream confluence	50% silt, 10% boulder, 40% gravel/pebble	12	400	0.99	0.693	2m sweep in *Ranunculus*

5.7.4. Sampling Procedure

Samples were taken from nine sites (see Fig. 5.7.1). Seven sites were located on the Mousesweet Brook itself (Sites A-G) and two sites on the River Stour, up and downstream of the Mousesweet Brook/Stour confluence (Sites H and I). Site A was situated close to the culvert from which the spill issued. Sites B,C and D were situated close together to account for possible influences on the water quality from an outfall situated between Site B and C and another outfall comprising seepage of chromium from a contaminated landfill site between Sites C and D.

Three-minute kick samples were taken from each site on the Mousesweet Brook and two-minute sweep samples through submerged aquatic vegetation were taken from the two sites on the River Stour. Given the upstream location of the pollution incident, it was not possible to have upstream reference sites. The environmental parameters of the sampling sites and the sampling methods used are summarised in Table 5.7.2.

Six sampling visits were distributed over a period of 294 days following the pollution incident as shown in Table 5.7.3. Sampling was then discontinued because the brook had been subjected to another spill of hydrochloric acid delaying recovery.

Table 5.7.3: Summary of Sampling Times

Sampling time Code	Date		Days following incident
msw1	27 March	1994	7
msw2	4 May	1994	38
msw3	26 May	1994	60
msw4	7 July	1994	102
msw5	3 August	1994	129
msw6	9 January	1995	294 days following original incident but 20 days following another pollution incident

5.7.5. Predicted Impact of Pollution

Table 5.7.4 shows the minimum pH levels known to cause severe damage or loss to a range of macroinvertebrate taxa, and Table 5.7.5 gives information on the relative sensitivities of selected macroinvertebrates to acidity resulting from acid rain. From this information, it can be seen that the pH of the brook following the incident fell to levels low enough to cause damage to all macroinvertebrates. Snails (*Lymnaea peregra*, *Planorbis* spp.) and leeches (*Glossiphonia complanata* and *Theromyzon tessulatum*) are particularly sensitive and oligochaeta, Ephemeroptera (not *Baetis rhodani*) and Trichoptera are less sensitive.

Table 5.7.4: Minimum levels of pH known to cause severe damage or loss to a range of taxonomic groups (information from Jeffries, 1988)

Table 5.7.5: The relative sensitivities to acidity of selected macroinvertebrate taxa (taken from data in Fjellheim and Raddum, 1990)

The predicted impact of an acid pollution is outlined in Table 5.7.6. It is expected that there would be an initial reduction in all the taxa present with spatial presence of taxa reflecting the differential sensitivities of the taxa to pH.

5.7.6. Results and Analysis

The data is organised as described in Chapter 4. Due to the upstream location of the pollution incident, there were no upstream reference sites within the Mousesweet Brook. The only site not affected by the pollution was Site I on the River Stour, upstream of the Mousesweet Brook/River Stour confluence. Impact of the pollution incident can be assessed by changes in the community within impacted sites over time.

Although these changes are not necessarily attributable to the pollution incident, they can nevertheless be compared with data gathered from other pollution incidents and examined for common patterns.

Table 5.7.6: Predicted Impact of Pollution Incident on Biotic Measures

Measure	Predicted effect
Individual taxa	Initial toxicity causes loss of sensitive taxa and decrease in abundance of less sensitive taxa. This is followed by a rapid increase in those taxa able to benefit from the changed conditions and then showing a decline, and a more gradual increase in other taxa as conditions resemble those prior to the incident.
Number of Taxa	Initial low numbers followed by a gradual increase.
Diversity	Initial low diversity followed by a gradual increase.
BMWP and ASPT	Initial low BMWP score and ASPT score as taxa are lost and any taxa are dominated by tolerant taxa, followed by an increase over time.
Multivariate Analysis of Community Similarity	1. Reference sites will show less variability over time than impacted sites (impact). 2. Impacted sites will show a progressively greater similarity to the reference site over time (temporal recovery) followed by a decrease in similarity after the second incident. 3. There will be progressively less variability and less movement towards similarity between impacted and control sites, the further downstream of the impact the sampling point is located (Spatial recovery). 4. Progressive increase in similarity within impacted site i.e. within an impacted site those samples closer in time to the pollution incident will show less similarity to each other than those samples further in time. Hypotheses 2-4 assume that the sampling period is long enough for a certain degree of movement of impacted communities towards reference communities to occur.

5.7.6.1. Water Quality

Table 5.7.7 gives the physicochemical measurements taken at selected sampling times. The conductivity levels are seen to be particularly high at all sites, including the most upstream sites.

5.7.6.2. Selected Taxa

Table 5.7.8 is a summary of the taxa found and their total abundance. 24 taxa were found throughout the study. The 5 most abundant taxa are listed below. In addition, a further taxon, *Baetis rhodani* was included as it was one of the 5 most abundant taxa at the reference site. Changes in the presence and abundance of these taxa through the study are shown in Fig. 5.7.2. and outlined below.

Asellus aquaticus
Baetis rhodani
Chironomidae
Lymnaea peregra
Oligochaeta
Potamopyrgus jenkinsi

Table 5.7.7: Physicochemical conditions in Mousesweet Brook (Dissolved Oxygen-DO is % saturation, Conductivity: µS cm-1 at 25^{o})

		Site A Withymoor rd				Site B US Cliff				Site C US Outfall			
Date	Days following Incident	Temp (°C)	pH	DO	Cond.	Temp (°C)	pH	DO	Cond.	Temp (°C)	pH	DO	Cond.
07-Jul-94	102	15.8	7.4	n/a	1153	15.8	7.8	n/a	1191	15.4	7.8	n/a	1170
03-Aug-94	129	17.7	7.3	53.3	927	18	7.6	55.8	930	17.9	7.7	63.04	917
09-Jan-95	294	8.9	6.6	n/a	1154	8.2	7.6	n/a	1048	8	7.6	n/a	996

		Site D DS Outfall				Site E Ashtree Playing Field				Site F Newtown Lane			
Date	Days following Incident	Temp (°C)	pH	DO	Cond.	Temp (°C)	pH	DO	Cond.	Temp (°C)	pH	DO	Cond.
07-Jul-94	102	15.1	7.8	n/a	1172	15.8	7.9	n/a	722	16	7.8	n/a	579
03-Aug-94	129	19	7.6	63	910	18.1	7.7	n/a	847	17.7	7.7	76.77	349
09-Jan-95	294	8	7.6	n/a	979	8.2	7.7	n/a	973	8.4	7.7	n/a	910

		Site G Cradley Forge				Site H Stour DS confluence				Site I Stour US confluence			
Date	Days following Incident	Temp (°C)	pH	DO	Cond.	Temp (°C)	pH	DO	Cond.	Temp (°C)	pH	DO	Cond.
07-Jul-94	102	15.9	7.7	n/a	1131	15.2	7.8	n/a	560	15.3	7.9	n/a	639
03-Aug-94	129	18.3	7.4	60.9	620	17.8	7.5	61	450	17.9	7.7	65.64	471
09-Jan-95	294	9	7.3	n/a	970	7.6	7.4	n/a	728	7.1	7.6	n/a	507

Table 5.7.8: Mousesweet Brook - Summary of Taxa found during study

Phylum, Order, Family, Genus, Species	Number found in study
PLATYHELMINTHES	
TURBELLARIA:TRICLADIDA	
Planariidae	
Dugesia sp.	4
Polycelis felina (Dalyell)	1
Polycelis tenuis (Ijima)/*nigra* (Muller)	1
MOLLUSCA	
GASTROPODA:PROSOBRANCHIA	
Hydrobiidae	
Potamopyrgus jenkinsi (Smith)	341
GASTROPODA:PULMONATA	
Lymnaeidae	
Lymnaea peregra (Muller)	13049
Planorbidae	
Planorbis sp.	1
Physidae	
Physa sp.	2
BIVALVIA	
Sphaeriidae	7
ANNELIDA	
OLIGOCHAETA	2286
HIRUDINEA	
Glossiphoniidae	
Glossiphonia complanata (L.)	41
Erpobdellidae	165
Trocheta bykowskii Gedroyc	2

Phylum, Order, Family, Genus, Species	Number found in study
ACARINA:HYDRACARINA	1
CRUSTACEA: MALACOSTRACA	
Asellidae	
Asellus aquaticus (L.)	5326
Gammaridae	
Gammarus pulex (L.)	2
Crangonyctidae	
Crangonyx pseudogracilis Bousfield	5
CRUSTACEA: BRANCHIURA	
Argulus foliaceus	1
INSECTA : EPHEMEROPTERA	
Baetidae	
Baetis rhodani Pictet	66
INSECTA : COLEOPTERA	
Dytiscidae	1
INSECTA : TRICHOPTERA	
Limnephilidae	1
INSECTA : DIPTERA	
Chironomidae	2340
Simuliidae	73
Tipulidae	12
Diptera larvae	2

Asellus aquaticus

Sites A, B and G show initially low numbers of *A. aquaticus* with a subsequent increase to a peak at 129 days and then a decrease at 294 days.

Baetis rhodani

This taxon is only found on the River Stour sites (H and I) with a similar pattern of abundance, thus revealing no impact of the pollution. The pollution incident occurred during a period of low seasonal abundance of this species.

Chironomidae

Site A, B, C and D show initially low numbers of Chironomidae and a subsequent increase with peak numbers 38-60 days following the incident followed by a decrease.

Lymnaea peregra

L. peregra is one of the most sensitive taxa to low pH (Fjellheim and Raddum, 1990). Following the pollution incident, few individuals were found on the Mousesweet Brook and none at Sites A- E. Over the following period of time, there was an increase in numbers although they remained low when compared with the sites on the River Stour. At the last sampling time, numbers within the Mousesweet Brook had again fallen to zero.

Sites H and I on the River Stour have the largest numbers of *L. peregra.* Both the impacted Site H and the reference Site I have similar peaks in abundance. Numbers throughout the study are highly variable with abundance peaks comprising large numbers of juvenile *L.peregra.* However 7 days following the incident, the numbers of *L.peregra* at Site H are lower than Site I. Due to the sampling procedure, no significant difference can be demonstrated and indeed the difference may be due to natural variability. It coincided, however, with browning of the submerged macrophytes (*Ranunculus fluitans*) at Site H but not at Site I (reference). After the second pollution incident, numbers of L. peregra were low at both Site H and Site I but with a greater decrease at Site H.

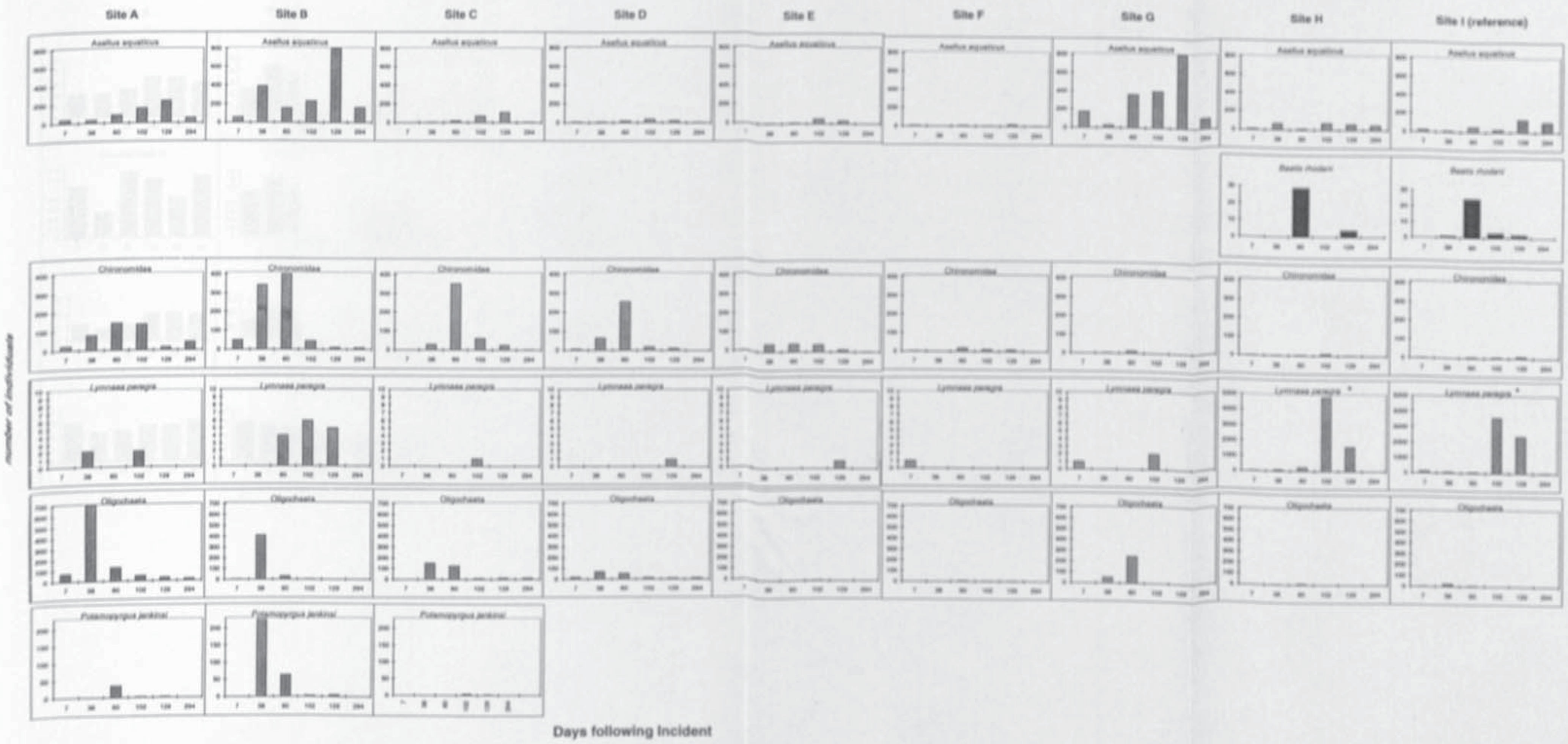
Oligochaeta

Sites A, B, C, D, G showed initially low numbers of Oligochaeta, a peak in numbers 38-60 days following the incident and subsequent decline in numbers.

Potamopyrgus jenkinsi

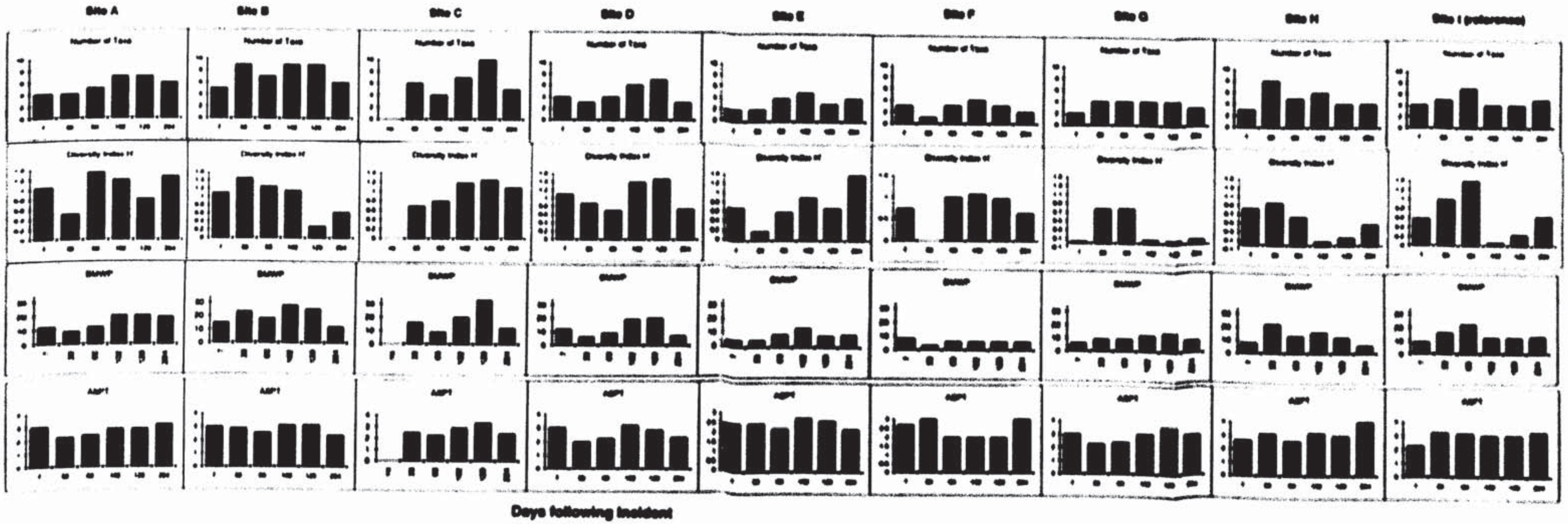
This species was only found at the sites furthest upstream on the Mousesweet Brook, at Sites A, B and C. Initially low numbers are apparent then a peak in abundance 38-60 days following the incident, followed by a decrease.

Figure 5.7.2: Mousesweet Brook - Changes in abundance of selected taxa following pollution incident



*note change of scale

Figure 5.7.3: Mousesweet Brook - Univariate measures - changes following pollution incident



5.7.6.3. Univariate Indices

Univariate indices (Number of Taxa, Diversity index, BMWP and ASPT scores) are plotted in Fig 5.7.3.

Number of Taxa

Taxa richness is low at all sites. Sites A, B, E, G and H show lower numbers of taxa 7 days after the incident.

Diversity Index

Very low diversity values are apparent throughout the stream. No clear pattern is evident.

BMWP and ASPT

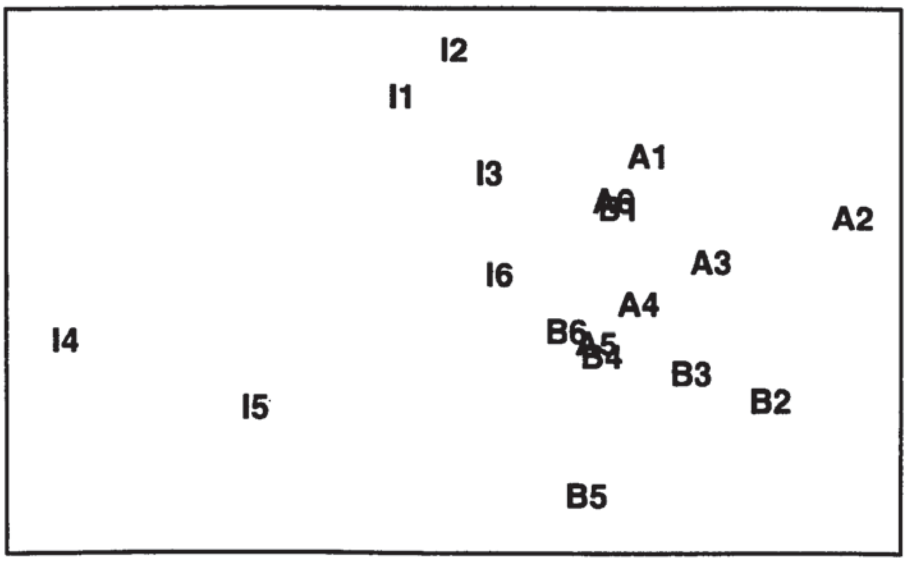
BMWP and ASPT scores are low at all sites. BMWP scores at most of the impacted sites and the reference site are initially low and then show an increase. ASPT scores show no clear pattern of changes following the incident.

5.7.6.4. Multivariate Analysis

Similarity matrices were produced to compare reference Site I with the impacted sites A-H and the resulting MDS plots are shown in Figs 5.7.4a-d. There is no evidence of impact in the affected sites (i.e. greater variability, progressive increase in similarity to the reference Site I). Only Site E, and to a certain extent Site H show a progressive similarity to the reference site followed by a decrease in similarity after the second pollution incident.

Figure 5.7.4a: Mousesweet Brook - MDS configuration of Sites A, B (impacted) and Site I (reference)

No transformation. Stress = .09



Root/root transformation. Stress = .12

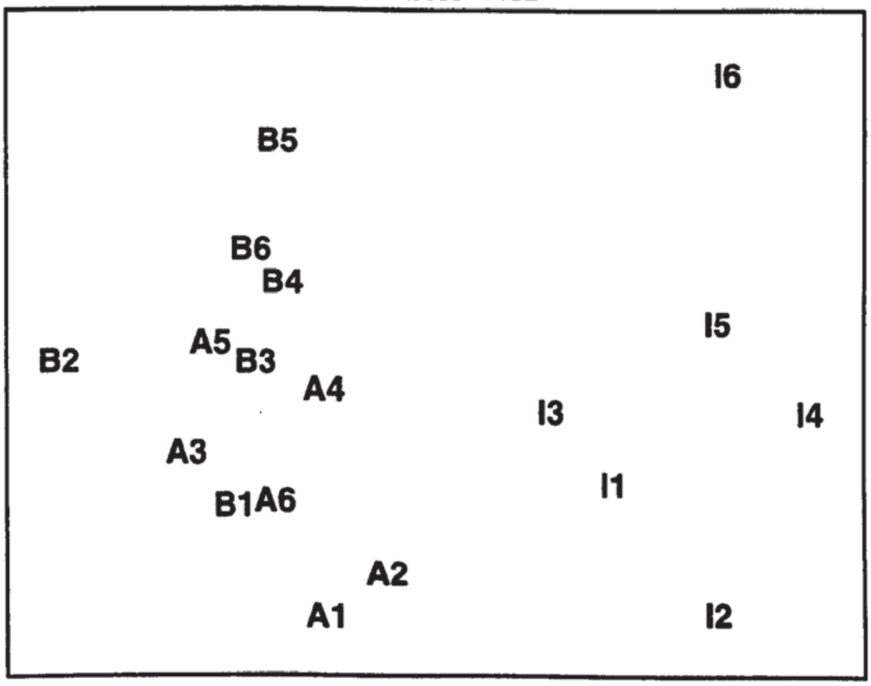
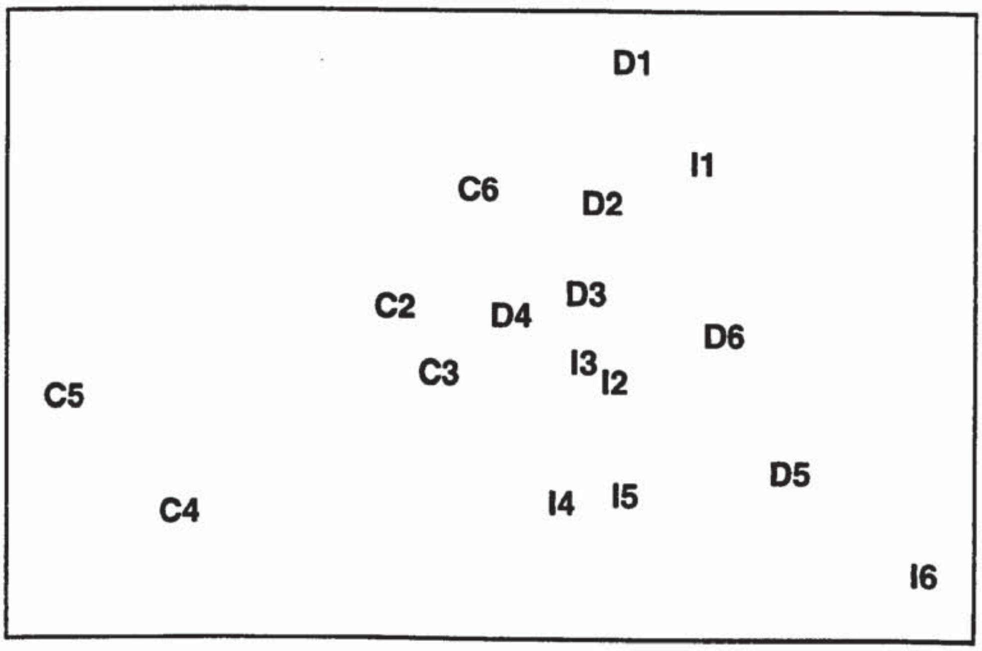


Figure 5.7.4b: Mousesweet Brook - MDS configuration of Sites C, D (impacted) and Site I (reference)

No transformation. Stress = .12



Root/root transformation. Stress = .12

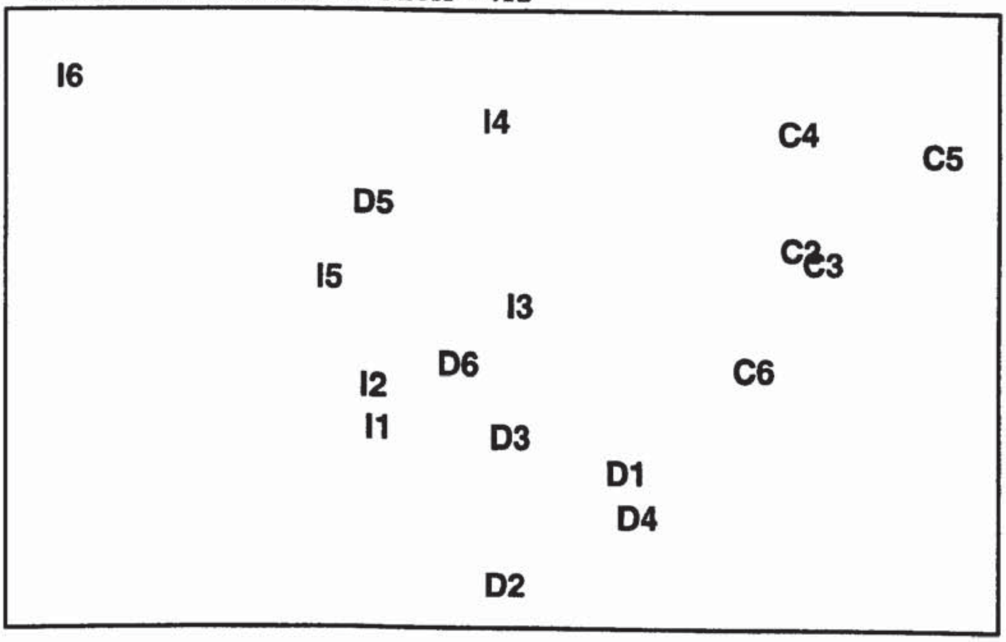
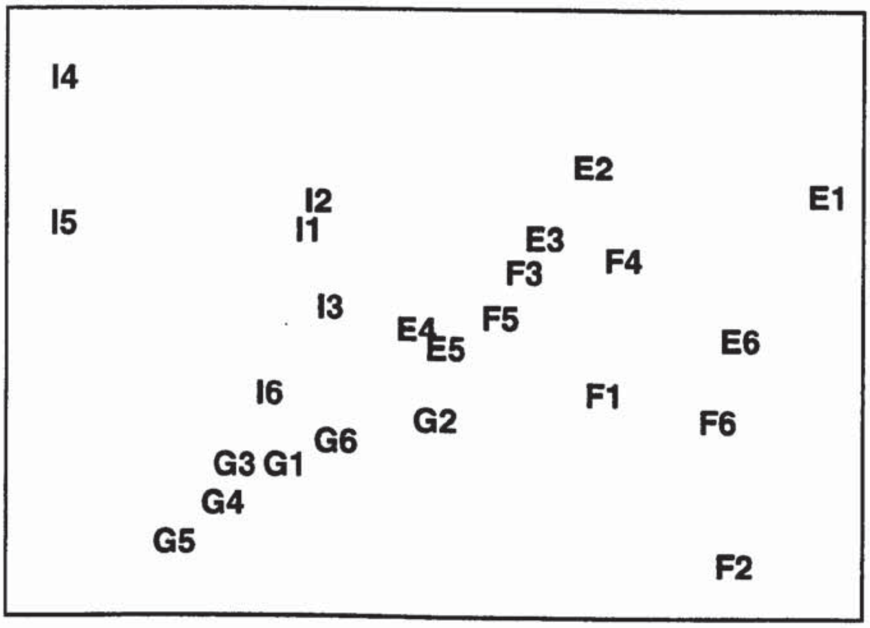


Figure 5.7.4c: Mousesweet Brook - MDS configuration of Sites E, F, G (impacted) and Site I (reference)

No transformation. Stress = .13



Root/root transformation. Stress = .17

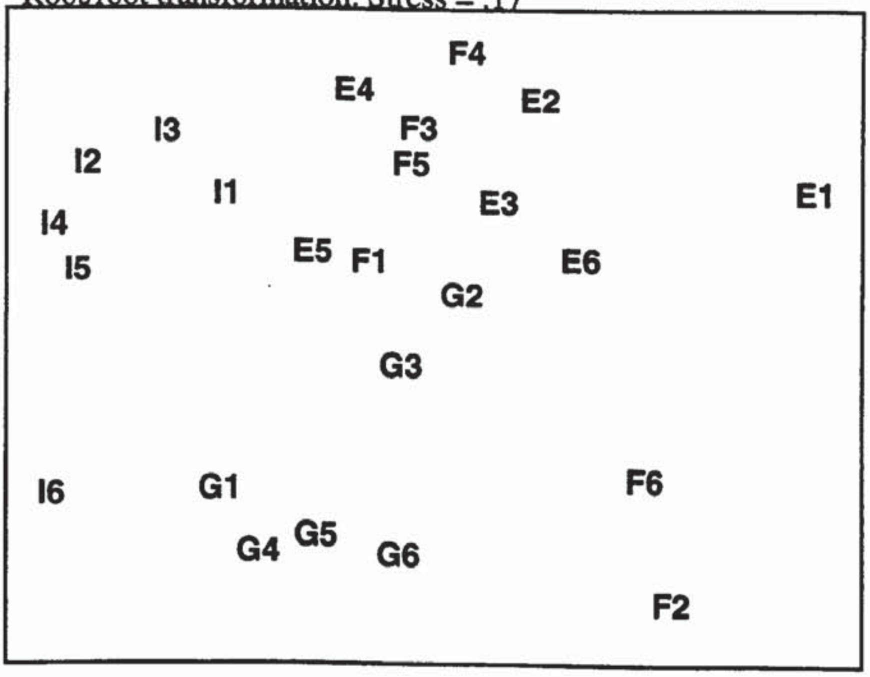
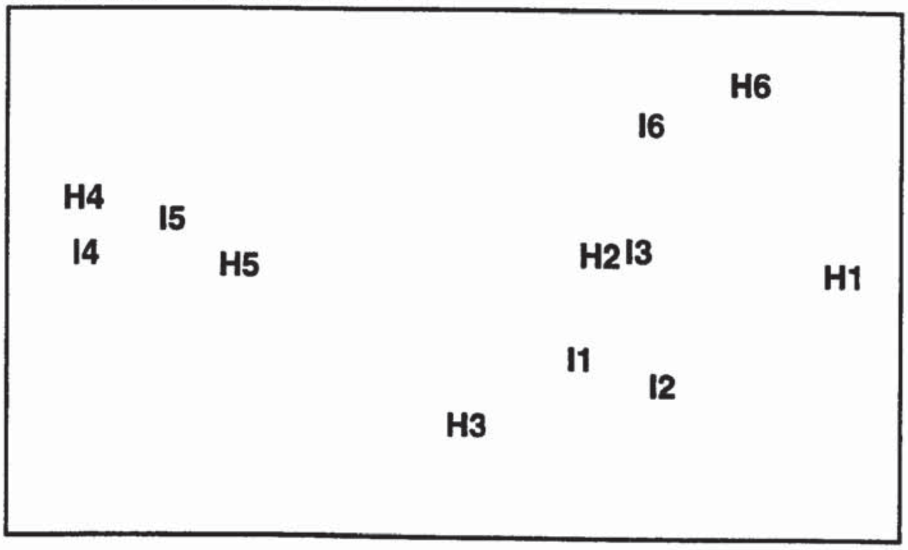
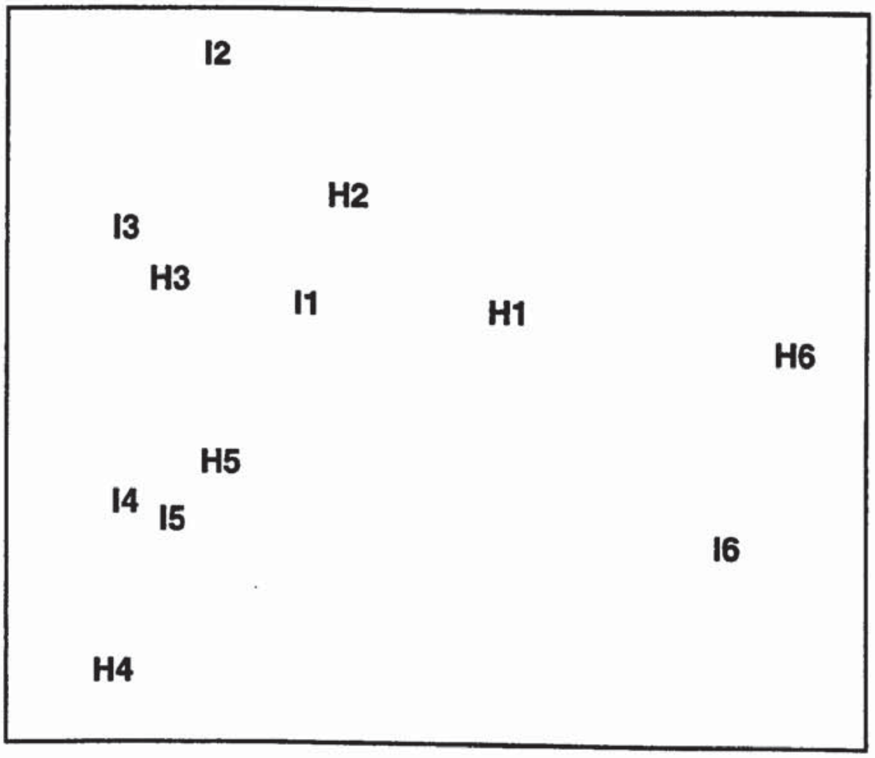


Figure 5.7.4d: Mousesweet Brook - MDS configuration of Site H (impacted) and Site I (reference)

No transformation. Stress = .03



Root/root transformation. Stress = .07



5.7.7 Summary

Table 5.7.9 summarises the measured pollution impact on the Mousesweet Brook.

Mousesweet Brook is highly contaminated by a variety of chronic and intermittent outfalls. The initial incident caused a drastic decrease in pH levels, with the most visible impact being dead Sticklebacks. The last sampling occasion took place 20 days after another pollution incident of Hydrochloric acid. Where individual taxa are concerned, *Lymnaea peregra* appeared to have been impacted by the two pollution incidents both within the Mousesweet Brook and within the River Stour, 3.6 km from the outfall, which supported high snail populations. A similar pattern was seen with *A. aquaticus* in the Mousesweet Brook: low numbers following the incident, a gradual increase and then low numbers following the second pollution incident. Chironomidae and *P. jenkinsi* also show lower numbers following the incident with a subsequent rise to a peak at 38-60 days and then decreasing in numbers. It is possible, therefore, that *L.peregra*, *A.aquaticus*, Chironomidae, Oligochaeta and *P.jenkinsi* were impacted by the pollution incident causing an initial decrease in numbers. In the case of *L.peregra*, and *A. aquaticus*, numbers then increased over time to a peak at 129 days and then decreased following a subsequent pollution incident. In the case of Chironomidae, Oligochaeta and *P. jenkinsi*, numbers increased to a peak at 38- 60 days and then decreased. Mollusca and benthic macrocrustaceans (*Gammarus, Asellus)* are particularly intolerant of low sodium and chloride concentrations resulting from low pH (Sutcliffe and Hildrew, 1989; Vangenechten *et al.*, 1989)

A pattern of lower numbers of taxa is seen following the pollution incident at sites A,B and G with a subsequent increase and another decrease following the second pollution. This pattern is also seen in the BMWP scores although ASPT scores remain relatively unchanged throughout the study. However, with so few taxa within the watercourse, changes are difficult to attribute to environmental factors.

When multivariate analysis is carried out, comparing each impacted site to the reference site, the impacted sites do not show greater variability than the reference site. Only Site E, and to a certain extent Site H show a progressive similarity to the reference site followed by a decrease in similarity after the second pollution incident. The measurement of community similarity in this watercourse with few taxa represented, may not convey more information than consideration of each taxon separately. Measures of community impact are largely dependant upon comparison with reference sites. The choice of reference site in this instance, although having similar environmental parameters to the impacted site on the River Stour, did not share environmental characteristics with any of the sites on the Mousesweet Brook and involved a different method of sampling. A nearby watercourse had been considered as a reference but was found to suffer from chronic heavy metal pollution and therefore rejected.

Table 5.7.9: Mousesweet Brook - Summary of Pollution Impact on All Measures

Measure	Type of impact	Spatial Impact	Temporal Impact
Initial impact of incident	Dead fish (sticklebacks; *Gasterosteus aculeatus*)		
Individual Taxa	*Asellus aquaticus*,	A,B,G	peak abundance at 129 days, decrease at 294 days.
	Oligochaeta,	C,D,G	peak abundance at 38 days
	Potamopyrgus jenkinsi,	A,B	peak abundance at 38-60 days
	Chironomidae (All show initial low abundance followed by an increase and a subsequent decrease at 294 days)	A,B,C,D	peak abundance at 60-102 days
	Lymnaea peregra: Initial low abundance when compared with reference	H	7 days and again at 294 days
Number of Taxa	Initially low numbers of taxa	A,B,E,G, H	7 days
Diversity (H')	No evidence of impact		
BMWP, ASPT	Initially low BMWP scores ASPT scores show no change	A,B,E	7 days
Multivariate analysis	Progressively greater similarity to reference site and decrease in similarity at 294 days	E,H	

5.8 Mythe Tributary

5.8.1. Description of Incident, Remedial Action

On 10 November, 95 a spill of 2200-4400 litres of red diesel (heating oil) was accidentally released into a tributary of the Mythe Brook. Initial remedial action involved placement of absorbent booms along the length of the tributary. Before the confluence of the tributary and the Mythe Brook, a stand of *Glyceria maxima* had trapped oil (Fig 5.8.1). The NRA removed the majority of the stand with a digger, leaving the vegetation on the side of the bank. One section of this vegetation stand was left in situ. for the purpose of this study to enable monitoring of recovery. Within the Mythe Brook, which is impounded along its length and displays low flow, the oil had been trapped by the large mats of duckweed (*Lemna* sp.) on the surface which was removed and further oil was pumped from the water surface and removed from the site.

Dead Lumbricidae worms were seen throughout the tributary four days after the incident.

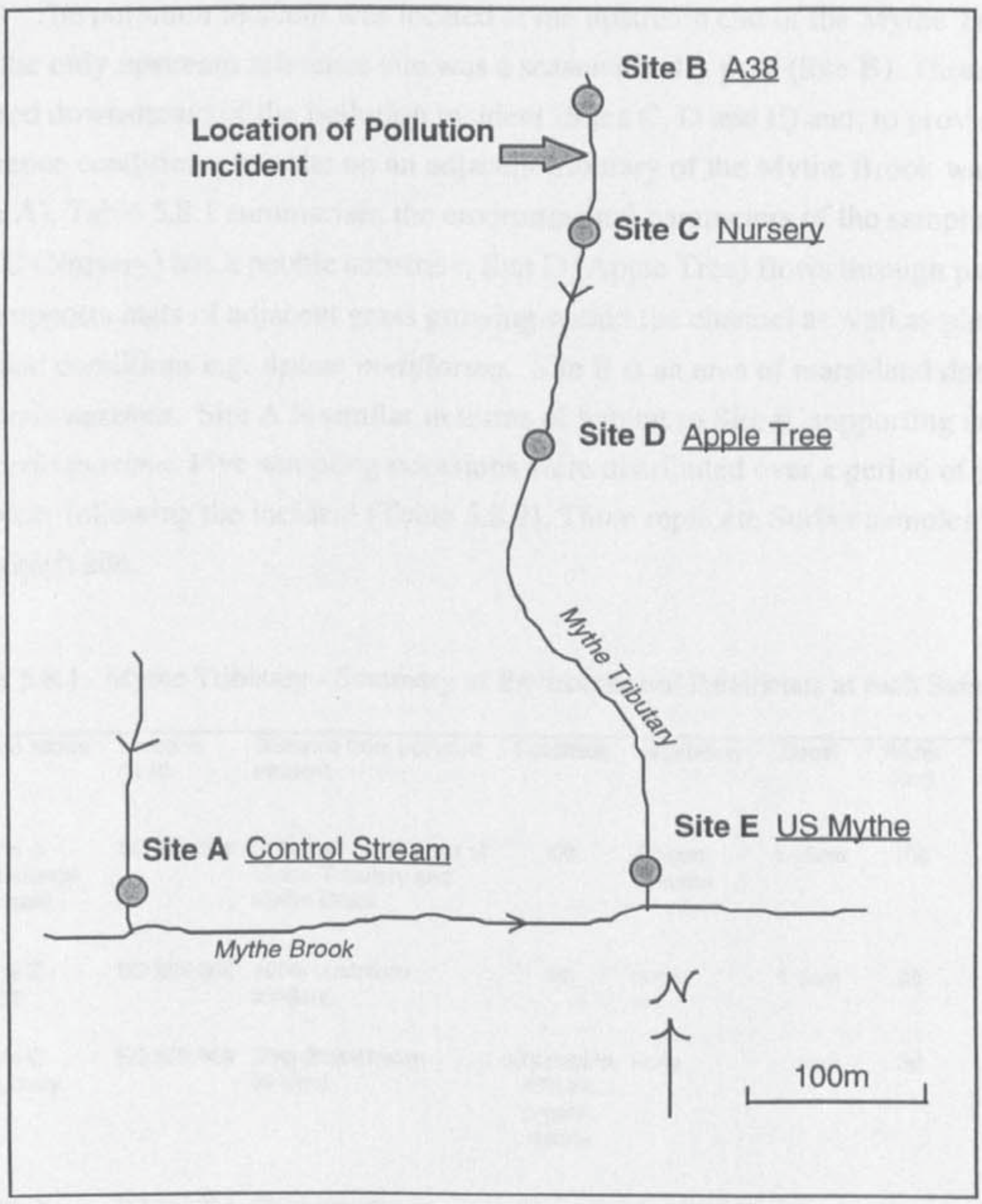
In addition the following sampling sites showed a number of dead taxa during the first sampling visit, 4 days following the pollution incident, which were identified as follows.

Site D (Apple Tree)	Site E (US Mythe)
Oligochaeta	*Bithynia tentaculata*
Helophorus grandis	*Gammarus pulex*
Chironomidae	*Agabus* sp.
Tipulidae	Dytiscidae larvae
	Helophorus sp.
	Hydroporus sp.

5.8.2. Site Description and History

The Mythe Tributary is a first order clay stream one kilometre in length and drains agricultural land (market gardens, nursery, arable and pasture). It flows into the Mythe Brook which is a tributary of the River Severn near Tewkesbury. Flow within the tributary is low and can be dry during the summer. A drainage pipe is situated downstream of Site E on the Mythe tributary and has no influence on the sampling sites chosen.

Figure 5.8.1: Mythe Tributary - Map to show location of Pollution Incident and Sampling Points



No information is available on the historical water quality of the Mythe tributary although local sources maintained that a pool between the Sites B and Site C supported large beds of watercress (*Nasturtium officinale*) that was harvested by local people. This species is generally found in unpolluted waters (Haslam *et al*, 1986). Sites E and A comprise swamp dominated by *Glyceria maxima* which is characteristic of eutrophic waters (National Vegetation Classification Community Type S5, Rodwell *et al*, 1995).

5.8.3. Sampling Procedure

Sampling was restricted to the affected tributary and a nearby unimpacted tributary (Fig. 5.8.1) The pollution incident was located at the upstream end of the Mythe Tributary and the only upstream reference site was a seasonally dry pool (Site B). Three sites were located downstream of the pollution incident (Sites C, D and E) and, to provide further reference conditions, one site on an adjacent tributary of the Mythe Brook was chosen (Site A). Table 5.8.1 summarises the environmental parameters of the sampling sites. Site C (Nursery) has a pebble substrate, Site D (Apple Tree) flows through pasture land and supports mats of adjacent grass growing within the channel as well as plants of wetland conditions e.g. *Apium nodiflorum*. Site E is an area of marshland dominated by *Glyceria maxima*. Site A is similar in terms of habitat to Site E, supporting an area of *Glyceria maxima*. Five sampling occasions were distributed over a period of just over one year following the incident (Table 5.8.2). Three replicate Surber samples were taken from each site.

Table 5.8.1: Mythe Tributary - Summary of Environmental Parameters at each Sampling Point

Site Name	Location (G R)	Distance from pollution incident	Substrate	Vegetation	Depth	Width (cm)	Flow m/sec
Site A - reference stream	SO 880 359	200m US confluence of Mythe Tributary and Mythe Brook	silt	*Glyceria maxima*	1-15cm	100	< .1
Site B - A38	SO 889 359	100m upstream incident	silt	none	1-3cm	30	< .1
Site C - Nursery	SO 888 359	50m downstream incident	60%pebble, 40%silt, organic debris	none	1-5cm	30	< .1
Site D - Apple Tree	SO 884 357	200m downstream incident	silt, grass	*Agrostis* sp., *Apium nodiflorum*	1-5cm	20-50	< .1
Site E - US Mythe	SO 881 355	850m downstream incident	silt	*Glyceria maxima*	1-15cm	100-150	< .1

Table 5.8.2 Summary of sampling times

Sampling time Code	Date		Days following incident
my1	14 November	1995	4
my2	14 December	1995	34
my3	19 February	1996	100
my4	15 May	1996	187
my5	13 November	1996	369

5.8.4. Predicted Impact of Oil Pollution

Red Diesel is a fraction of oil, and the impact of various kinds of oil was discussed in Chapter 5.5 (Hewell Brook). The predicted impact of red diesel on the biota can be summarised in Table 5.8.3.

Table 5.8.3 - Predicted Impact of Pollution Impact on Biotic Measures

Measure	Predicted effect
Individual taxa	Initial toxicity causes loss of sensitive taxa and decrease in abundance of less sensitive taxa. This is followed by a rapid increase in certain taxa benefiting from the changed conditions and then showing a decline and a gradual increase in other taxa as conditions resemble those prior to the incident.
Number of Taxa	Initial low numbers followed by a gradual increase.
Diversity	Initial low diversity followed by a gradual increase.
BMWP and ASPT	Initial low BMWP score and ASPT score as taxa are lost and remaining taxa are tolerant of organic pollution, followed by an increase in scores over time as colonisation of more taxa and less tolerant taxa occurs.
Multivariate Analysis of Community Similarity	Initial low similarity to 'control' sites followed by an increase in similarity over time.

5.8.5. Results and Analysis

The data is organised as described in Chapter 4. As no pre-impact measures exist, both impact and recovery have to be inferred by comparison with the two unimpacted sites: one upstream and one adjacent tributary.

5.8.5.1. Water Quality

Table 5.8.4 gives the physicochemical measurements taken at each sampling time. Temperature varies with season and conductivity measurements are not high within the watercourse. A record was kept of the presence of oil when the substratum was disturbed during sampling. One of the reference sites showed some oil on the first sampling occasion but not thereafter. Two of the impacted sites (D and E) showed oil throughout the period of study and the impacted Site C showed no release of oil 369 days following the incident.

Table 5.8.4: Mythe Tributary - Physicochemical Conditions at each sampling site (Conductivity: µS cm-1 at 25^{o}; * - Oil released when substratum disturbed)

		Site A				Site B				Site C			
Date	Days following Disturbance	Temp (°C)	pH	Conduct-ivity	Oil in substrate	Temp (°C)	pH	Conduct-ivity	Oil in substrate	Temp (°C)	pH	Cond-uctivity	Oil in substrate
19-Feb-96	100	1.8	8.4	110	*	1.3	7.6	114		2.9	8.1	88	*
15-May-96	187	9.1	8.3	81		13.9	7.8	50		11.5	N/A	14	*
13-Nov-96	369	4.3	N/A	N/A		N/A	N/A	N/A		6	N/A	N/A	

		Site D				Site E			
Date	Days following Disturbance	Temp (°C)	pH	Conduct-ivity	Oil in substrate	Temp (°C)	pH	Conduct-ivity	Oil in substrate
19-Feb-96	100	2.2	8.4	110	*	1.9	8.2	184	*
15-May-96	187	14.2	N/A	30	*	10.1	8.1	125	*
13-Nov-96	369	4	N/A	N/A	*	5.4	N/A	N/A	*

5.8.5.2. Selected Taxa

Table 5.8.5 is a summary of the taxa found and their total abundance. 59 taxa were found throughout the study. The five most abundant taxa are listed below and changes in their presence and abundance are shown in Fig. 5.8.2 (sites with insufficient numbers are not shown) and summarised in Table 5.8.6.

Asellus aquaticus
Chironomidae
Crangonyx pseudogracilis
Oligochaeta
Sphaeriidae

Fig. 5.8.2: Mythe Tributary - Changes in abundance of selected taxa following pollution incident (means + S.E.)

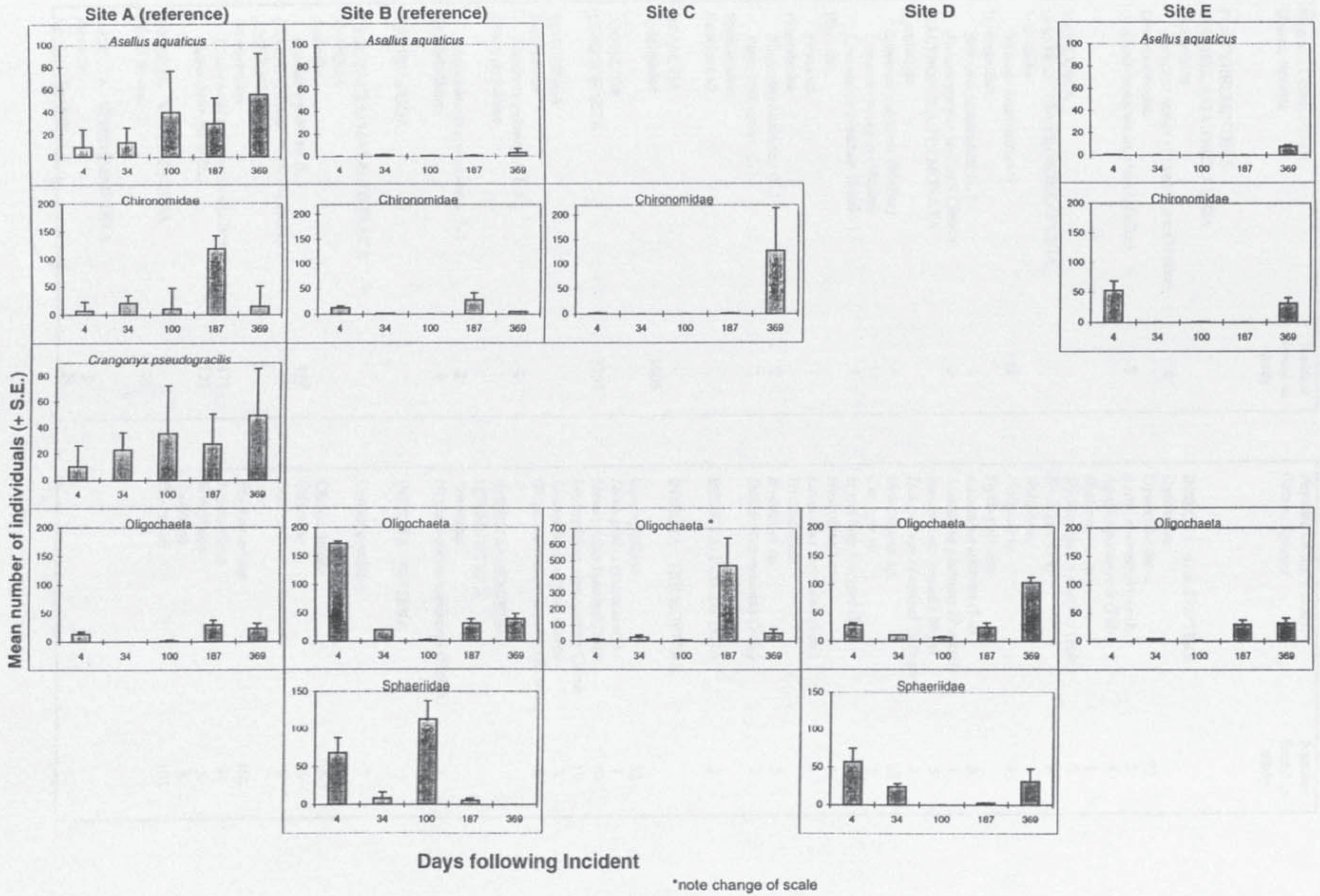


Table 5.8.5: Mythe Tributary - Summary of Taxa found in study

Phylum, Order, Family, Genus, Species	Number found in study
PLATYHELMINTHES	
TURBELLARIA:TRICLADIDA	
Planariidae	
Polycelis tenuis (Ijima)/*nigra* (Muller)	76
Dendrocoelidae	
Dendrocoelum lacteum (Muller)	8
MOLLUSCA	
GASTROPODA:PROSOBRANCHIA	
Valvatidae	
Valvata (macrostoma)	3
Hydrobiidae	
Bithynia tentaculata (L.)	4
Potamopyrgus jenkinsii (Smith)	2
GASTROPODA:PULMONATA	
Lymneidae	
Lymnaea palustris (Muller)	6
Lymnaea peregra (Muller)	14
Lymnaea truncatula (Muller)	1
Physidae	
Physa sp.	1
Planorbidae	
Planorbis carinatus (L.)	2
Planorbis vortex (L.)	1
Succineidae	
Succinea sp.	
BIVALVIA	
Sphaeriidae	1005
ANNELIDA	
OLIGOCHAETA	3357
HIRUDINEA	
Piscicolidae	
Piscicola geometra (L.)	1
Glossiphoniidae	
Glossiphonia complanata (L.)	2
Erpobdellidae	4
ARTHROPODA	
CRUSTACEA: MALACOSTRACA: ISOPODA	
Asellidae	
Asellus aquaticus (L.)	418
CRUSTACEA:MALACOSTRACA: AMPHIPODA	
Gammaridae	
Crangonyx pseudogracilis Bousfield	478
Gammarus pulex (L.)	121
INSECTA : MEGALOPTERA	
Sialis lutaria (L.)	36
INSECTA : EPHEMEROPTERA	
Baetidae	5
Cloeon dipterum (L.)	29
INSECTA : COLEOPTERA	
Dytiscidae	
Dytiscidae larva	23
Agabus bipustulatus (L.)	3
Agabus paludosus (Fab.)	1
Dytiscus marginalis (L.)	1
Hydroglyphus pusillus (Fab.)	2
Hydroporus sp.	8
Haliplidae	
Haliplus sp.	1
Hydrophilidae	
Anacaena limbata (Fab.)	3
Anacaena globulus (Paykull)	1
Helophorus grandis Illiger	5
Helophorus obscurus Mulsant	1
Helophorus sp.	13
Cercyon sp.	1
Hydrobius fuscipes (L.)	3
Hydrobius larva	1
Laccobius striatulus (Fab.)	1
Hydraenidae	
Hydraena sp.	3
Ochthebius minimus (Fab.)	2
INSECTA: PLECOPTERA	3
INSECTA : TRICHOPTERA	
Limnephilidae	52
Limnephilus (flavicornis)	1
Limnephilus lunatus Curtis	10
Limnephilus marmoratus Curtis	10
Limnephilus stigma Curtis	2
Glyphotaelius pellucidus (Retsius)	6
INSECTA: HEMIPTERA, HETEROPTERA	
Corixidae	
Hesperocorixa sahlbergi (Fieber)	1
INSECTA : DIPTERA	
Ceratopogonidae	3
Chironomidae	1295
Culicidae	59
Dixidae	6
Ptychopteridae	101
Psychodidae	94
Simuliidae	5
Tipulidae	8
Fly larvae	101

Asellus aquaticus

This species is found consistently at Site A (reference). Few individuals are found at the impacted sites, but at Site E an increase in numbers is seen after one year.

Chironomidae

Chironomidae show great variability in numbers at both impacted and reference sites. High abundance is evident at Site C after one year. This may be a response to the oil pollution through increased availability of nutrients or lack of predators.

Crangonyx pseudogracilis

A pattern similar to *A. aquaticus*, this species is found in the reference site (Site A) and at Site E after one year.

Oligochaeta

Immediately following the impact, large numbers of dead Lumbricidae were seen on the substratum of the Mythe tributary. The Oligochaeta recorded during sampling consist largely of Tubificidae. Sites C and D show an increase in numbers after 187 and 369 days. This may be a response to an increase in available nutrients following breakdown of the oil or other changes e.g. lack of predators.

Sphaeriidae

Sphaeriidae are only found at Sites B (reference) and Site D but no clear evidence of impact is seen. Few individuals were found at Site C, just downstream of the pollution incident, throughout the study although many empty shells were found. This would indicate that this site has, in the past, supported abundant populations of Sphaeriidae, suggesting that the recovery of numbers may require more than the one year of the study.

Table 5.8.6: Summary of Impact of Pollution Incident on abundance of selected taxa

Taxon	Impact of pollution
Asellus aquaticus	Absence from site E until 369 days
Chironomidae	Initial absence, then high abundance at Site C
Crangonyx pseudogracilis	Absence from site E until 369 days
Oligochaeta	Initial low numbers then increase in abundance at Sites C and D
Sphaeriidae	Possible initial mortality due to pollution incident. Evidence of recovery not clear.

5.8.5.3. Univariate Measures

Changes in univariate measures at each site over time are shown in Fig. 5.8.3 and summarised in Table 5.8.9.

Number of Taxa

The number of taxa found in each replicate ranged from 0 to 14. The reference Sites A and B show relatively even numbers of taxa throughout the study. The impacted Sites C and D show low numbers of taxa throughout the study with increases one year after the impact. Four days following the impact, the number of taxa at Site D are not significantly different to Site B because of single individuals of two beetle taxa. Site E shows low numbers of taxa for 187 days following the incident and then an increase after one year. (See Table 5.8.7 for significance tests. Site E was compared to Control Site A because of similar environmental conditions; Sites C and D were compared to reference Site B which, although differing in environmental conditions, was the only unimpacted upstream site).

Table 5.8.7: Significance Table-Mann-Whitney U test applied to Number of Taxa in replicates from Reference and Impacted Sites (ns = not significant, s = significant, $p < 0.05$)

	Days following Pollution Incident				
Comparison between sites	4	34	100	187	369
Reference A (all times) & Site E	s	s	s	s	ns
Reference B (all times) & Site C	s	s	s	s	ns
Reference B (all times) & Site D	ns	s	s	s	ns

The impact has apparently been to reduce the number of taxa at Sites C, D and E. The number of taxa is seen to return to reference levels by 369 days following the incident.

Diversity Index

The Diversity index shows a more variable picture than the number of taxa. Table 5.8.8 summarises the significance tests applied to the diversity indices comparing impacted and reference sites at different times following the pollution incident. Diversity at the reference Site A remains high through the year. Site E shows a diversity approaching this after 369 days. Sites C and D when compared with reference site B, show no significant difference 4 days following the incident but then show a decrease in diversity which is significantly different to that of the reference for 100 - 187 days following the incident. All sites show similar diversity to reference diversities after 369 days.

Figure: 5.8.3: Mythe Tributary - Changes in univariate indices following pollution incident

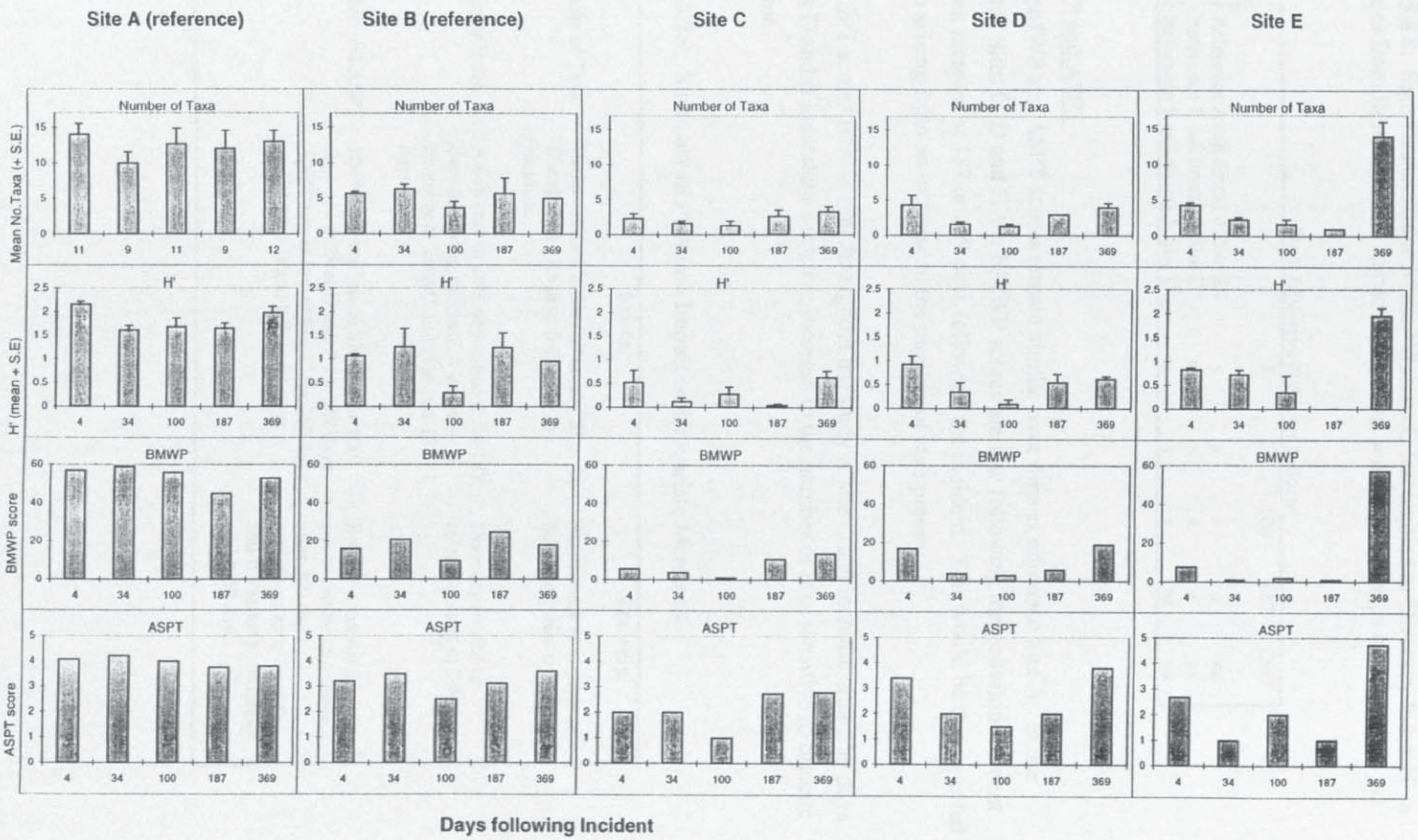


Table 5.8.8: Significance Table Mann-Whitney U test applied to Diversity Index H' in replicates from Reference and Impacted Sites (ns = not significant, s = significant, $p < .05$)

	Days following Pollution Incident				
	4	34	100	187	369
Reference A (all times) & Site E	s	s	s	s	ns
Reference B (all times) & Site C	ns	s	s	s	ns
Reference B (all times) & Site D	ns	s	s	ns	ns

BMWP and ASPT

Both BMWP and ASPT scores remain similar over time at reference Site A . In the impacted Sites C, D and E, the BMWP scores are low following the pollution incident and then increase at 187 or 369 days following the incident. This would be the expected pattern arising from an increase in the number of taxa present.

The ASPT scores remain low throughout the study at site C and increase after 369 days at sites D and E indicating a relative increase in the number of taxa sensitive to organic pollution.

Table 5.8.9: Summary of Pollution Impact on Univariate Measures

	Impact	Recovery
Number of Taxa	Reduction in numbers of taxa at Sites C, D and E for 187 days following the incident.	An increase in number of taxa at all sites at 369 days
Diversity Index	A decrease in diversity at Site E for 187 days following incident. A decrease in diversity at Sites C and D at 100-187 days	Diversity similar to reference sites at 369 days
BMWP and ASPT	BMWP scores low at Sites C for 100 days and at Sites D and E for 187 days. ASPT scores low at Site C for 369 days and at Sites D and E for 187 days	BMWP scores at Sites D and E similar to reference after 369 days. ASPT scores at Sites D and E similar to reference after 369 days.

5.8.5.4. Multivariate Measures

The R Statistic was computed for each sampling time as described in Chapter 4. This showed that the replicates within sites are more similar to each other than replicates between sites. Similarity matrices were therefore produced from data from averaged replicates and used to produce Multidimensional Scaling configurations (MDS plots) comparing each impacted site with the relevant reference site. (See Figs 5.8.4 a-c).

Site A (reference) was compared to Site E. Communities at each sampling time at Site A were similar to each other. Those from Site E showed greater variability with E5 (one year after the incident) showing the greatest similarity to the reference sites (Fig. 5.8.4a). ANOSIM analyses show a significant difference between the communities found at Site E from 4-187 days following the incident and all the communities at the reference Site A (Table 5.8.10). However the community at Site E, 369 days after the incident shows no significant difference to the reference communities when the raw data only is considered but not when the data is transformed to reduce the importance of the abundant taxa and increase the importance of the rarer taxa (Table 5.8.10). This indicates that, in terms of the more abundant taxa, recovery has occurred by 369 days.

Sites C and D were compared with Site B. There was no clear evidence of any progressive similarity to the reference site over time.

Table 5.8.10: Summary of ANOSIM analyses to compare reference site A with each sampling occasion at Site E (s = Significance $p<0.01$ unless otherwise indicated, ns = no significant difference)

level of transformation of taxon abundance	E1 compared with A1-A5	E2 compared with A1-A5	E3 compared with A1-A5	E4 compared with A1-A5	E5 compared with A1-A5
No transform	s	s	s	s	ns
Square root	s	s	s	s	s ($p<0.04$)
Root root	s	s	s	s	s ($p<0.17$)
Presence/absence	s	s	s	s	s ($p<0.007$)

Figure 5.8.4a: Mythe Tributary - MDS configuration of Site A (reference) and Site E

Root/root transformation. Stress = .07

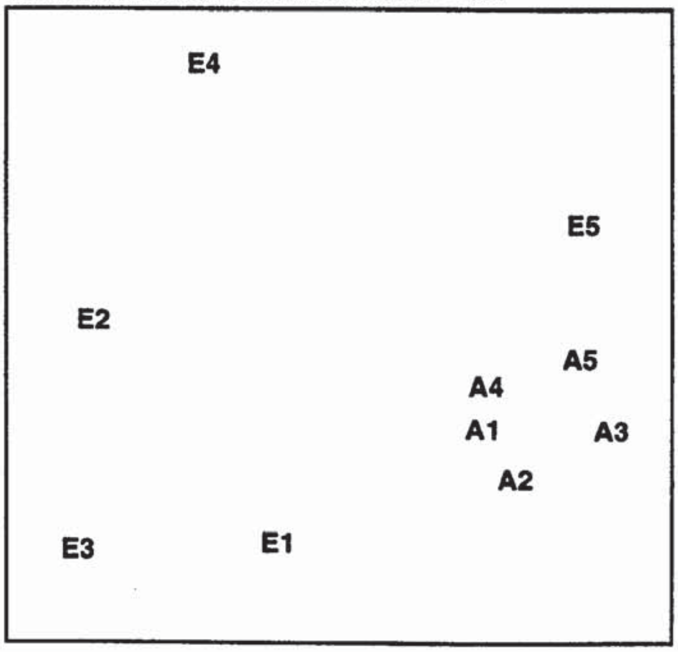


Figure 5.8.4b: Mythe Tributary - MDS configuration of Site B (reference) and Site C

Root/root transformation. Stress = 0.13

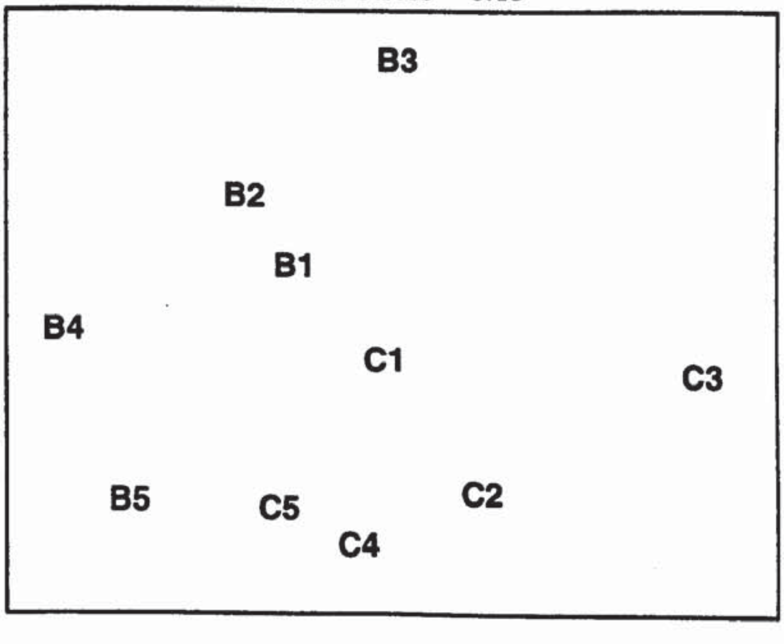
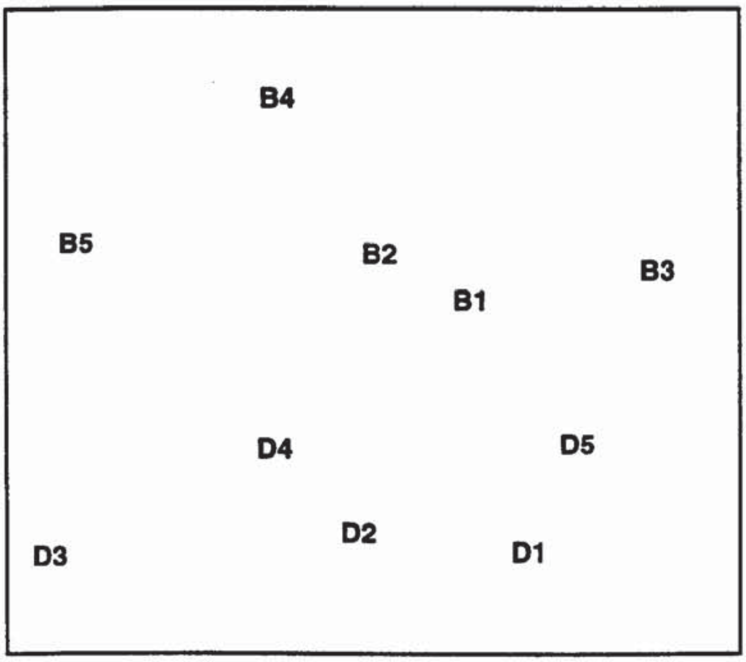


Figure 5.8.4c: Mythe Tributary - MDS configuration of Site B (reference) and Site D

Root/root transformation. Stress = 0.14



5.8.6. Summary

Table 5.8.11 summarises the impact of the pollution incident on the biotic measures used. The impact of the pollution incident upon the macroinvertebrates of this small tributary was great, with few taxa remaining after the incident. Sites C and D show some signs of recovery during the period of study, however lack of comparable upstream sites, necessitated comparison with a pool habitat. It is possible that further patterns of recovery may have been seen if sampling had continued for a longer time period.

Site E was compared to a similar reference site; another tributary of the Mythe. The impact on Site E was great as the depositional nature of the site served as a repository for some of the oil. The emergent vegetation left on site showed no re-growth at the beginning of the following season and the macroinvertebrate community remained sparse. However, one year following the incident, re-growth was evident and certain measures showed recovery: the numbers of certain taxa; *Asellus aquaticus* and *Crangonyx pseudogracilis*; all univariate indices (the total number of taxa, BMWP and ASPT scores, Diversity Index) and a multivariate measure of community similarity using raw data, thus giving relatively greater emphasis to the more abundant taxa.

However certain measures did not show recovery by 369 days: oil was still being released when the substratum was disturbed, certain taxa were absent or low in abundance; notably *Gammarus pulex* which, although abundant at the reference site, was absent from Site E throughout the study. In addition, multivariate measures of community similarity transformed to give greater weight to less abundant taxa did not show recovery at this time. Thus rarer taxa had not colonised in numbers comparable to the reference site.

As the incident had occurred just downstream of the tributary source, the main sources of colonisation are downstream and aerial sources. This would result in the downstream sites being colonised before upstream sites. This was found in the study, with Site E showing greater recovery than the other impacted sites in spite of the fact that oil was still being released during disturbance of the substratum.

Table 5.8.11: Summary of pollution impact on all measures

Measure	Type of impact	Spatial Impact	Temporal Impact
Individual Taxa	Initial mortality followed by an increase in abundance, either to reference or greater than reference levels	All sites (C, D, E)	369 days or more
Number of Taxa	Reduction in numbers of taxa	All sites (C, D, E)	187 days
Diversity (H')	Decrease in diversity	All sites (C, D, E)	100-186 days
BMWP, ASPT	Decrease in BMWP and ASPT scores	All sites (C, D, E)	187-369 days
Multivariate analysis	Presumed decrease in similarity of community when compared with reference	Site E	369 days (greatest similarity to reference but not necessarily complete)
Visual assessment of macrophytes	Emergent vegetation coated with oil and subsequent lack of regrowth	Site E	187 days
Visual assessment of substratum	Oil released when substratum is disturbed	All sites (C, D, E)	>369 days

5.9 Preston Bagot Brook

5.9.1. Description of Incident, Remedial Action

On 25/8/94 a lorry carrying a tanker of Hydrochloric acid was involved in an accident on the M40 motorway in Warwickshire. The tanker was fractured during the accident resulting in a spill of 1500 litres of 30% Hydrochloric Acid. Fire services sprayed down highway with water after the accident, washing the contents of the tanker into the Preston Bagot Brook, the pH of this discharge being 1.9. The stream had a pH of 7.6 prior to the discharge. After the acid had entered the water, the pH dropped to pH 3.2 (100 m downstream) and pH 2.9 (200 m downstream).

Remedial action involved sandbagging of the stream, pumping 4500 litres of the water away and adding 1kg soda lime to the stream in an attempt to restore the pH to normal. The weather at time of the incident, and for 7 days previously, had been dry.

Table 5.9.1 shows the chemical composition of the water at two sites downstream of the incident taken on the day of the incident as well as the minimum levels of the determinands found to cause severe damage or total loss of macroinvertebrate taxa (refer to Fig. 5.9.1 for location). This shows that at a point 2 km downstream of the incident, the pH value had returned to normal within 5.5 hours. The chemical measures also show that the acid had caused the mobilisation and resultant increased concentration of all the metals measured. Most of the determinands shown in Table 5.9.1 were initially higher than the minimum levels known to cause severe damage to macroinvertebrate taxa but had returned to non toxic levels within 5.5 hours.

The NRA fisheries officer recorded dead Leeches, Crustacea, Molluscs and Oligochaeta on the day following the incident but no quantitative information was obtained and the downstream extent of invertebrate mortality was not assessed. Fish counted dead were 1138 Stickleback, 1047 Stone Loach (total 2185). The Fisheries Officer assessed that the extent of damage to the fish was not greater than 1 km.

On 14/9/94, 20 days after the incident, routine standard sampling at the Pettiford Bridge site (see Fig. 5.9.1), 11 km from the incident, indicated a fall in biological water quality from Class 1A to a very low Class 2 and only 9 invertebrate groups were found (Table 5.9.2). A BMWP score of 37 was recorded. Previous scores at the same time of year varied between 56 and 127.

Table 5.9.1: Chemical Analysis after the incident (from NRA data) and minimum levels of determinands to cause damage to invertebrates (Jeffries, 1988)

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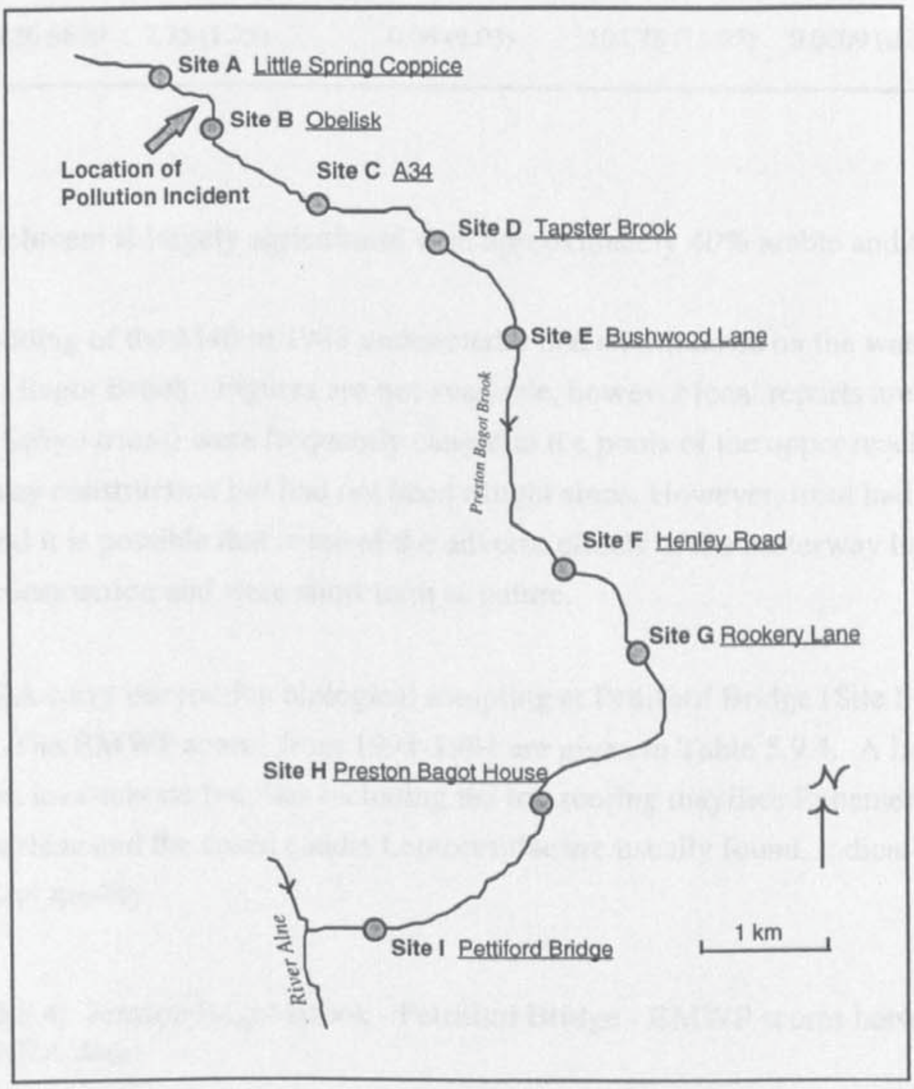
Table 5.9.2: Preston Bagot Brook - Taxa present at Pettiford Bridge, September 1994 (NRA monitoring data)

Taxon	Abundance Category
Potamopyrgus jenkinsi	Common
Sphaeriidae	Few
Tubificidae	Few
Gammarus pulex	Very Abundant
Baetidae	Common
Elmidae	Common
Hydropsychidae	Few
Tipulidae	Few
Simuliidae	Few
Cottus gobio	Present

5.9.2. Site Description and History

Preston Bagot Brook is a first order stream about 12 km in length before its confluence with the River Alne which forms part of the River Avon catchment in Warwickshire (Fig. 5.9.1).

Figure 5.9.1: Preston Bagot Brook: Map to show location of Pollution Incident and Sampling Points



The brook is described as a 'Non-Designated Brown Trout Water' in the River Avon Catchment Management Plan (NRA, 1994). In terms of water quality, it is classified as Band A (NRA, Water Quality in the West Midlands, 1994/1995). The River Quality Objective set by the Environment Agency is RE2. NRA records of water quality from 1992-1994 for the 8 km section of the Brook from Bushwood Lane (see Fig. 5.9.1) to the confluence with the River Alne is given in Table 5.9.3.

Table 5.9.3: Water Quality Data in Preston Bagot Brook 1992-94 (means and standard deviations of 27 samples (NRA data)

Location	BOD (5 day BOD ATU inhibited, mg/l)	NH3 (Total ammonia as N mg/l)	DO (% saturation mg/l)	NH3 (unionised ammonia as NH3, mg/l)
SP 0836 6820	2.35 (1.35)	0.04 (0.03)	104.78 (15.97)	0.0009 (0.0006)

The catchment is largely agricultural with approximately 40% arable and 60% pasture.

The building of the M40 in 1988 undoubtedly had an influence on the water quality in Preston Bagot Brook. Figures are not available, however local reports are that Brown Trout (*Salmo trutta*) were frequently caught in the pools of the upper reaches prior to motorway construction but had not been caught since. However, trout had been seen in 1996 and it is possible that some of the adverse effects of the motorway had been caused by its construction and were short term in nature.

The NRA carry out routine biological sampling at Pettiford Bridge (Site I - see Fig. 5.9.1). The BMWP scores from 1991-1994 are given in Table 5.9.4. A large number of different invertebrate families including the top scoring mayflies Ephemerellidae and Ephemeridae and the cased caddis Leptoceridae are usually found, indicating very good biological quality.

Table 5.9.4: Preston Bagot Brook - Pettiford Bridge - BMWP scores between 1991 and 1994 (NRA data)

Date	BMWP score
Autumn 1991	84
Spring 1992	61
Summer 1992	92
Autumn 1992	127
Autumn 1993	56
Spring 1994	95
Summer 1994	105
Autumn 1994	37

5.9.3. Sampling Procedure

Three replicate Surber samples were taken from 1 site upstream and 8 sites downstream of the source of the incident at the locations indicated on Fig 5.9.1. Table 5.9.5 presents a summary of the environmental parameters at each sampling point. Due to the location of the pollution incident close to the stream source, only one 'reference' sampling site was found. The substrate at this site comprised a high proportion of silt which was not found in the other sampling sites.

All the sites (apart from Site E which was not sampled on the first occasion) were sampled on 8 occasions over a period of 552 days distributed as shown in Table 5.9.6.

Table 5.9.5: Preston Bagot Brook - Summary of Environmental Parameters at each sampling point

Site Name	Location (G R)	Distance from pollution incident	Substrate	Depth (cm)	Width (cm)	Flow m/sec 20/7/95,27/2/95	Discharge (cu. m/s) 20/7/95,27/2/95
Site A - Little Spring Coppice	SP 142 720	600m upstream	60% silt, 40% gravel	5-20	30	0.2-0.285	.006-.026
Site B - Obelisk	SP 144 715	50m downstream	50% pebble, 30% gravel,20% silt	5-15	50	0.2-0.429	.012-.038
Site C - A34	SP 157 704	1900m downstream	60% gravel, 40% silt	7-30	50-100	0.1-0.333	.007-.071
Site D - Tapster Brook	SP 168 701	2900m downstream	40% cobble,20% pebble,40% silt	5-40	100	0.143-0.27	.014-.043
Site E - Bushwood Lane	SP 175 693	4300 m downstream	60% gravel,40% silt	5-14	200	0.36-0.6	.028-.041
Site F - Henley Road	SP 174 672	6700 m downstream	40% cobble, 30% gravel, 20% sand,10% boulder	7-30	100	0.24-0.6	.017-.128
Site G - Rookery Lane	SP 180 665	7800m downstream	40% cobble, 40%pebble/gravel, 10% sand	5-30	250	0.257-0.5	.018-.127
Site H - Preston Bagot House	SP 173 653	9700m downstream	100% gravel/pebble	5-25	400	0.272-0.6	.019-.128
Site I - Pettiford Bridge	SP 162 639	11500m downstream	80% gravel/pebble, 20% brick	10-50	300	0.2-0.375	.029-.177

Table 5.9.6: Preston Bagot Brook - Sampling Times

Sampling time code	Date		Days following Incident
PB1	12 October	1994	48
PB2	15 November	1994	78
PB3	15 December	1994	112
PB4	6 February	1995	165
PB5	28 March	1995	215
PB6	22 May	1995	270
PB7	20 July	1995	329
PB8	28 February	1996	552

5.9.4. Predicted Impact of Pollution

Table 5.9.1 shows the multifactorial nature of the pollution incident, making prediction of the impact on the macroinvertebrate fauna complex. Table 5.9.7 is a summary of the minimum levels of the determinands measured following the pollution incident that are known to cause severe damage or loss to a range of macroinvertebrate taxa. It can be seen that, although sensitivities of different taxa to each determinand vary, the levels were greater than the minimum damage levels such that a differential impact on each taxon would be difficult to predict. It is presumed,therefore, that a total loss of taxa would occur close to the incident.

5.9.5. Results and Analysis

The data is organised as described in Chapter 4. Both impact and recovery have to be inferred by comparison with the unimpacted upstream sites and with NRA data taken before the impact (Table 5.9.4).

5.9.5.1.Water Quality

Table 5.9.9 gives the physicochemical measurements taken at each sampling time. Temperature varied according to season. Values of pH were between pH 7-8.5. Conductivity is variable throughout the stream, and is sometimes high, possibly as a result of agricultural run-off.

Table 5.9.7: Levels of Determinands measured by the NRA following the pollution incident and minimum levels of each determinand known to cause severe damage or loss to a range of taxonomic groups (information from Jeffries, 1988)

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Table 5.9.8: Predicted Impact of Pollution Impact on Biotic Measures

Measure	Predicted effect
Individual taxa	Initial toxicity causes loss of taxa. This is followed by an initial increase in those taxa that are rapid colonisers due to high mobility, multivoltine life cycles and can benefit from the changed conditions. These taxa may then show a decline and a gradual increase in other taxa as conditions resemble those prior to the incident. A downstream decrease in the impact will be evident.
Number of Taxa	Initial low numbers followed by a gradual increase. A downstream decrease in the impact will be evident.
Diversity	Initial low diversity followed by a gradual increase. A downstream decrease in the impact will be evident.
BMWP and ASPT	Initial low BMWP score and ASPT score as taxa are lost followed by an increase over time. An increase in the ASPT score over time would be the result of increased numbers of taxa rather than changes to a community that is less sensitive to organic pollution. A downstream decrease in the impact will be evident.
Multivariate Analysis of Community Similarity	1. Upstream 'reference' sites will show less variability over time than impacted sites (impact). 2. Impacted sites will show a progressively greater similarity to the 'reference' sites over time (temporal recovery). This information can be used to determine a time of recovery within a site. 3. There will be progressively less variability and less increase in similarity of samples over time the further downstream of the impact the sampling point is located (Spatial recovery). 4. Progressive increase in similarity within impacted site i.e. impacted sites will show a decrease in variability between samples the greater the time following the pollution incident.

Table 5.9.9: Physicochemical Conditions in Preston Bagot Brook (Conductivity: µS cm-1 at 25°C)

		Site A (reference)			Site B			Site C		
Date	Days following Disturbance	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond.
12-Oct-94	48	n/a	n/a	n/a	n/a	8.4	269	11.3	8	460
06-Feb-95	165	7.7	7.2	339	7.2	7.6	323	7.5	7.8	350
28-Mar-95	215	6.4	7.3	669	5.1	8.1	590	4.6	8.6	483
20-Jul-95	329	16.5	7.2	531	17.9	8.1	341	17.8	8	639
28-Feb-96	552	5	n/a	558	4.5	8.2	136	3.3	8.4	436

		Site D			Site E			Site F		
Date	Days following Disturbance	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond.
12-Oct-94	48	10.9	8.1	495	n/a	n/a	n/a	10.3	8.1	504
06-Feb-95	165	7.7	7.7	320	7.6	7.8	375	7.7	7.9	399
28-Mar-95	215	4.6	8.3	514	4.8	8.3	793	5.1	8.4	827
20-Jul-95	329	17.4	8	679	18.5	7.8	654	18	8	649
28-Feb-96	552	3.2	8.3	446	3.3	8.2	485	3.2	8.1	469

		Site G			Site H			Site I		
Date	Days following Disturbance	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond.
12-Oct-94	48	10.1	8.1	348	10.7	7.9	729	10.7	7.8	864
06-Feb-95	165	7.2	8	404	n/a	n/a	n/a	n/a	n/a	n/a
28-Mar-95	215	5.8	8.4	830	5.7	8	734	6	8.2	768
20-Jul-95	329	19.5	8.1	689	18.6	8.1	860	18.7	8.2	211?
28-Feb-96	552	3.7	8.4	35	4.6	8.3	535	4.8	8.4	580

5.9.5.2. Selected Taxa

Table 5.9.10 is a summary of the taxa found, and their total abundance. 80 taxa were found throughout the study. With the 3 species of Elmidae found, both larval and adult forms were recorded as separate 'taxa' bringing the total number of taxa to 83.

The five most abundant taxa are listed below and changes in their presence and abundance are summarised in Fig. 5.9.2 (sites with insufficient numbers are not shown).

Chironomidae
Gammarus pulex
Hydropsyche siltalai
Oligochaeta
Potamopyrgus jenkinsi

Table 5.9.10: Preston Bagot Brook - Summary of Taxa found in study

Phylum, Order, Family, Genus, Species	number found in study
TURBELLARIA:TRICLADIDA	
Planariidae	
Dugesia sp.	22
Polycelis tenuis (Ijima)/*nigra* (Muller)	297
Polycelis felina (Dalyell)	3
Dendrocoelidae	
Dendrocoelum lacteum (Muller)	32
MOLLUSCA	
GASTROPODA:PROSOBRANCHIA	
Valvata piscinalis (Muller)	117
Hydrobiidae	
Potamopyrgus jenkinsii (Smith)	19924
GASTROPODA:PULMONATA	
Lymneidae	
Lymnaea peregra (Muller)	314
Lymnaea palustris (Muller)	4
Lymnaea stagnalis (L.)	3
Planorbidae	
Planorbis carinatus (L.)	23
Planorbis contortus (L.)	255
Planorbis crista (L.)	1
Anisus vortex (L.)	84
Gyraulus albus (Muller)	19
Ancylidae	
Ancylus fluviatilis Muller	1043
BIVALVIA	
Sphaeriidae	1785
OLIGOCHAETA	6397
HIRUDINEA	
Piscicolidae	
Piscicola geometra (L.)	33
Glossiphoniidae	
Glossiphonia complanata (L.)	171
Glossiphonia heteroclita (L.)	1
Helobdella stagnalis (L.)	4
Hemiclepsis marginata (Muller)	1
Erpobdellidae	
Erpobdella octoculata (L.)	437
ACARINA:HYDRACARINA	220
CRUSTACEA: ISOPODA	
Asellidae	
Asellus aquaticus (L.)	2044
CRUSTACEA:AMPHIPODA	
Gammaridae	
Crangonyx pseudogracilis Bousfield	5
Gammarus pulex (L.)	11395
INSECTA : MEGALOPTERA	
Sialis lutaria (L.)	30
INSECTA : EPHEMEROPTERA	
Baetidae	
Baetis rhodani Pictet	1932
Heptagenidae	
Ecdyonurus sp.	11
Leptophlebiidae	
Habrophlebia fusca (Curtis)	16
Ephemeridae	
Ephemera danica (Muller)	12
Ephemerellidae	
Ephemerella ignita (Poda)	2303
Caenidae	
Caenis sp.	42

Phylum, Order, Family, Genus, Species	number found in study
INSECTA : PLECOPTERA	
Leuctra hippopus (Kempny)	4
INSECTA : COLEOPTERA	
Dytiscidae	
Agabus biguttatus (Olivier)	2
Agabus sturmii (Gyllenhal)	1
Hydroporinae	1
Dytiscidae larva	22
Hydrophilidae	1
Helophorus sp.	29
Gyrinidae larva	1
Haliplidae larva	1
Helodidae larva	3
Elmidae	
Elmis aenea Muller	217
Elmis aenea larva	219
Limnius volckmari (Panzer)	11
Limnius volckmari larva	12
(*Normandia nitens* (Muller))	2
Oulimnius tuberculatus Muller	44
Oulimnius tuberculatus larva	4
INSECTA : TRICHOPTERA	
Hydropsychidae	
Hydropsyche angustipennis (Curtis)	113
Hydropsyche contubernalis Mclachlan	3
Hydropsyche pellucidula (Curtis)	22
Hydropsyche siltalai Dohler	2557
Polycentropidae	
Holocentropus dubius (Rambur)	2
Rhyacophilidae	
Rhyacophila dorsalis (Curtis)	5
Glossosomatidae	
Agapetus fuscipes Curtis	15
Hydroptilidae	
Hydroptila sp.	14
Lepidostomatidae	
Lepidostoma hirtum (Fabricius)	1157
Limnephilidae	26
Chaetopteryx villosa (Fabricius)	1
Drusus annulatus (Stephens)	12
Halesus digitatus (Schrank)	13
Halesus radiatus (Curtis)	41
Glyphotaelius pellucidus (Retzius)	1
Limnephilus extricatus McLachlan	86
Limnephilus lunatus Curtis	25
Micropterna lateralis (Stephens)	1
Micropterna sequax McLachlan	20
Potamophylax sp.	1
Goeridae: *Goera pilosa* (Fabricius)	56
Sericostomatidae	
Sericostoma personatum (Spence)	199
Leptoceridae	10
Athripsodes sp.	247
Mystacides sp.	3
INSECTA: HEMIPTERA,	
Velia sp.	6
INSECTA : DIPTERA	
Ceratopogonidae	99
Chironomidae	2344
Simuliidae	1455
Tipulidae	146
Dipteran larva	116

Chironomidae

Changes in abundance at the reference site and impacted sites appear to be similar with low numbers initially followed by an increase in abundance in the spring. Thus, no impact of the pollution incident is apparent.

Gammarus pulex

The pattern of abundance at the most impacted sites matches that of the reference site with low numbers initially followed by an increase in abundance at 329 days.

Hydropsyche siltalai

H. siltalai is not present at the reference site, presumably because of a lack of suitable substratum for this rheophilic species (Edington and Hildrew, 1995). At most sites, *H.siltalai* is found in low numbers during the first site visit (48 days following the incident), however the extent to which this may be related to the incident is unclear. During July, 369 days following the incident, numbers are low or absent at all sites which is probably due to emergence of adults (Edington and Hildrew, 1995).

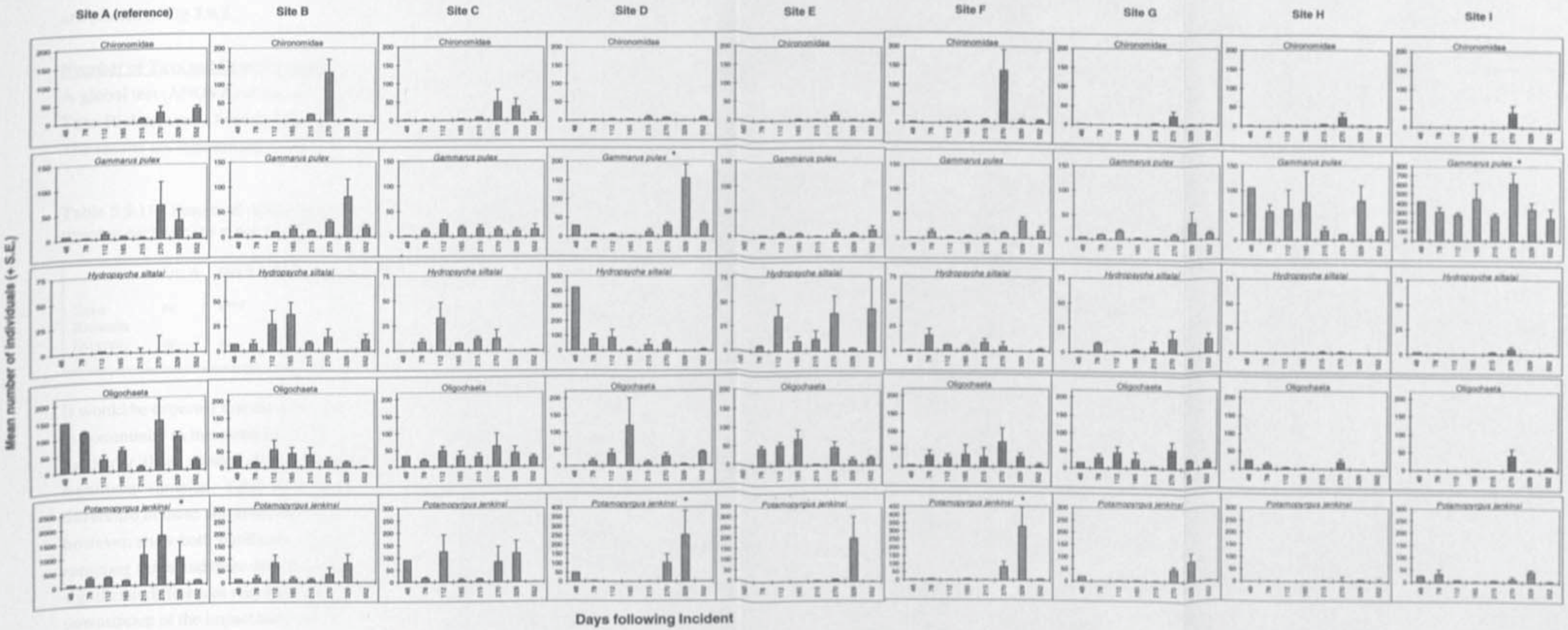
Oligochaeta

Impacted sites B,C,D,E show an increase in abundance 112 - 165 days after the pollution which is not apparent at the reference site. The numbers of Oligochaeta at 165 days following the pollution (February, 1995) was compared with the numbers found at 552 days (Feb 1996) and was shown to be significantly different (Mann-Whitney U test, $p<0.05$). This pattern was not seen in the control site.

Potamopyrgus jenkinsi

Sites B-G show an increase at 215-329 days which matches reference site changes but sites B and C also show another peak at 48-112 days following the incident.

Figure 5.9.2: Preston Bagot Brook - Changes in abundance of selected taxa following pollution incident



*note change of scale
NS = not sampled)

5.9.5.3. Univariate Indices

Univariate indices (Number of Taxa, Diversity Index, and BMWP and ASPT scores) are plotted in Fig 5.9.3.

Number of Taxa and Diversity Index

A global test (ANOVA) of the null hypothesis that there are no significant differences in Taxa Richness and Diversity (H') between sampling times, was performed for each site The results are summarised in Table 5.9.11.

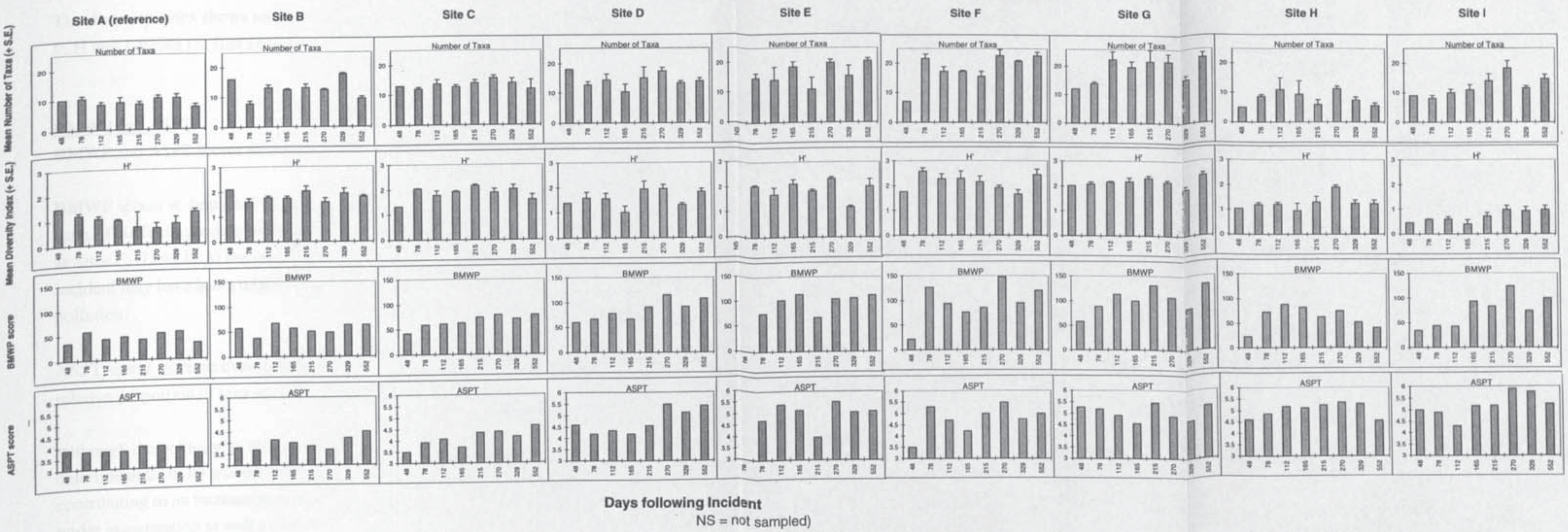
Table 5.9.11: Results of ANOVA to test for significant differences between sampling times at each site (ns = not significant; s = significant: **p*<0.05,***p*<0.01, ****p*<0.001)

	Site A	Site B	Site C	Site D	Site E	Site F	Site G	Site H	Site I
Taxa Richness	ns	s***	ns	ns	ns	s***	s*	ns	s**
Diversity (H')	ns	ns	s**	s*	s***	s*	s*	s**	s*

It would be expected that the reference site (Site A) would show less temporal variation in community as measured by the univariate indices and that the impacted sites, in particular, those situated close to the pollution impact, are more likely to show significant variation. Table 5.9.11 shows that Site A does in fact show no significant difference in these univariate measures between sampling times. The impacted sites, however, show both significant differences and non significant differences between sampling times and there does not appear to be a spatial pattern. Although the significant differences may have been caused by the pollution incident, sites downstream of the impact have differing environmental characteristics to the upstream reference site. They support higher numbers of taxa which may therefore include higher numbers of seasonally occurring taxa thus resulting in greater variation between sampling times.

The reference Site A shows similar numbers of taxa at all times. Sites B, C, D, E and H do not show the expected initial decrease in the number of taxa following the incident although this is shown by Sites F,G and I. When 'total numbers of taxa' over 3 replicates are considered, Site C does show reduced numbers of taxa during the first two sampling visits The taxa initially absent at the latter sites include those known to be sensitive to organic pollution: Leptoceridae, Ephemerellidae, Elmidae. It may be that recolonisation by these taxa occurred at the downstream sites first, as the major source

Figure 5.9.3: Preston Bagot Brook - Univariate measures - changes following pollution incident



of colonisation was from downstream. An alternative explanation is that chronic background pollution from the motorway would preclude the inclusion of these taxa within the community at sites closer to the motorway.

The diversity index shows variability at all sites although shows lower figures at Sites C, F, H and I during the first sampling visit.

<u>BMWP and ASPT</u>

BMWP and ASPT scores show consistency over time at reference Site A.

BMWP scores at impacted sites C, D, F,G, H, I are initially depressed and increase over time. This finding would result from an increase in the number of taxa which is the case at Sites F, G and I. At the other impacted sites, this finding indicates that the pollution incident may have had a relatively greater impact on those taxa sensitive to organic pollution.

ASPT scores increase over time at Sites B, C, D, F and I, showing an increase in the relative proportion of taxa sensitive to organic pollution.

Although the pollution in this case was complex and included toxic levels of a variety of determinands, BOD was not affected. This indicates that some of the families contributing to an increase in scores were relatively more sensitive to the pollution under investigation as well as being sensitive to organic pollution.

Site I is a standard sampling point utilised by the NRA (Table 5.9.4) and BMWP scores taken from 1991-1994 range from 56 to 127, with a mean of 89. However 20 days following the pollution incident, NRA data show a score of 37. The present study, showed a similarly low score of 35, 48 days after the incident, with numbers remaining low until 112 days following the incident, after which the scores range from 75 to 124.

5.9.5.4. Multivariate Analysis

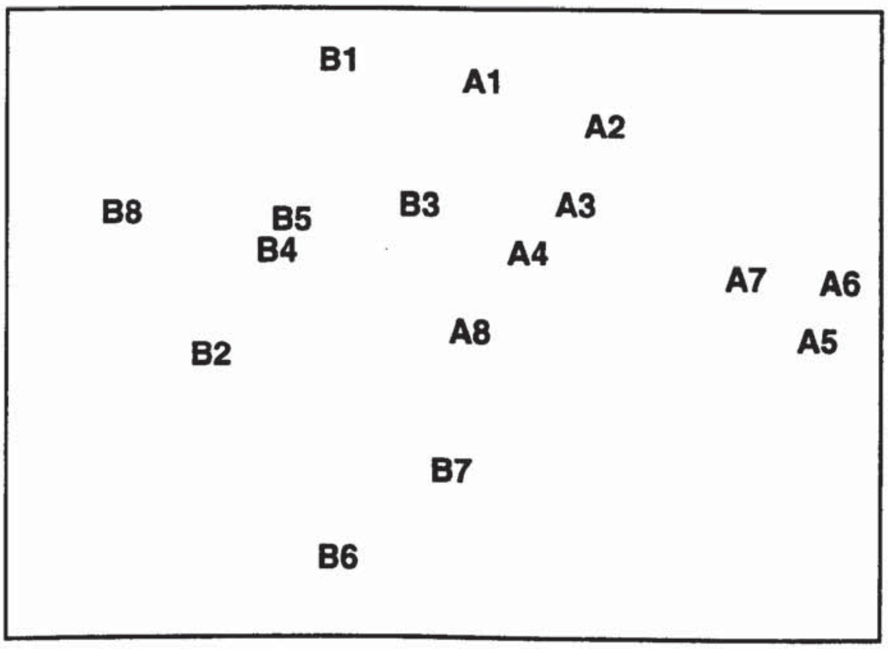
The R Statistic was computed for each sampling time as described in Chapter 4. This showed that the replicates within sites are more similar to each other than replicates between sites. The replicates were therefore averaged and an MDS analysis performed to compare each impacted site with the reference site (A) in turn. The resulting configurations are given in Figs 5.9.4 a-h and have been carried out on both untransformed and root/root transformed data. Table 5.9.12 represents a summary of the findings.

The configurations show that the upstream reference site shows less variability over time than the impacted sites when root/root transformed data is used although not when untransformed data is compared. In other words, greater variability is seen in the impacted sites (apart from Site I) than the reference site if relatively less emphasis is given to the abundant taxa. When abundant taxa dominate the analysis, as is the case with untransformed data, the variability over time in the reference site and impacted sites is similar.

The prediction of impact being demonstrated by impacted sites showing progressively greater similarity over time to reference sites (Table 5.9.8) is not found at any site. Given the different environmental parameters of the Reference Site A to the impacted site, the macroinvertebrate community supported by the sites is likely to be different even in the absence of pollution impact. Progressive similarity to upstream communities may therefore not be an appropriate measure of impact and recovery in this case. However, within each site, following the pollution incident, recovery could be demonstrated by communities becoming progressively more similar to each other and more similar to the sampling occasion furthest in time from the pollution incident. In other words, greater evidence of clustering should be evident over time. This is seen to varying extents in Sites F, G and I (Fig. 5.9.4e, f, h).

Figure 5.9.4a: Preston Bagot Brook - MDS configuration of Site A(reference) and Site B

No transformation. Stress value = 0.07



Root/root transformation. Stress value = 0.15

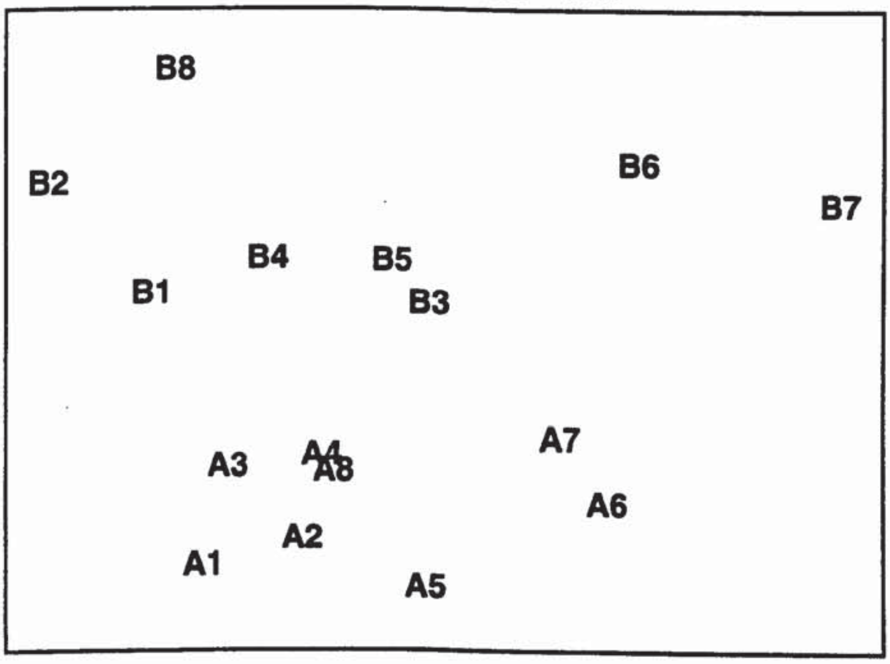
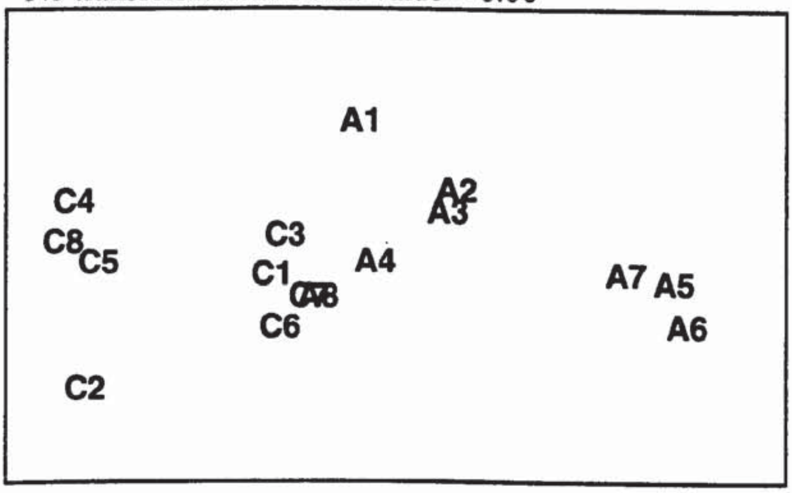


Figure 5.9.4b: Preston Bagot Brook - MDS configuration of Site A(reference) and Site C

No transformation. Stress value = 0.06



Root/root transformation. Stress value = 0.14

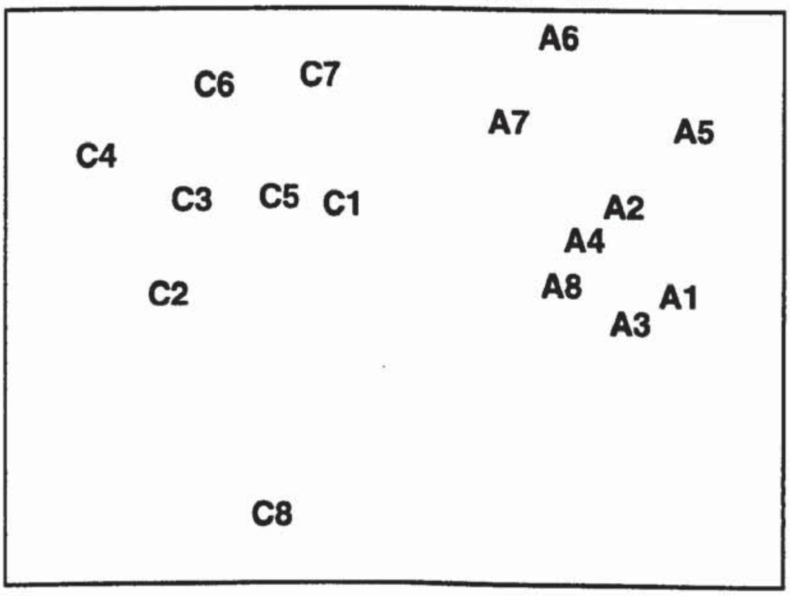
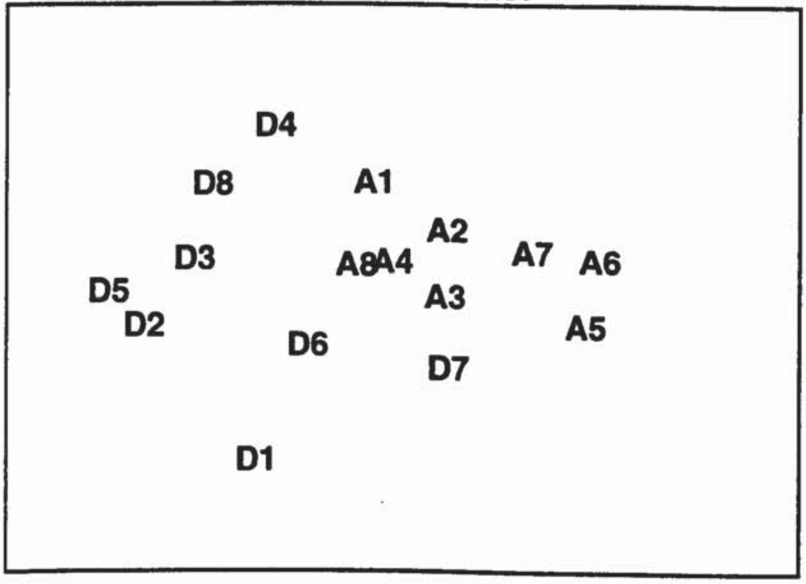


Figure 5.9.4c: Preston Bagot Brook - MDS configuration of Site A(reference) and Site D

No transformation. Stress value = 0.10



Root/root transformation. Stress value = 0.12

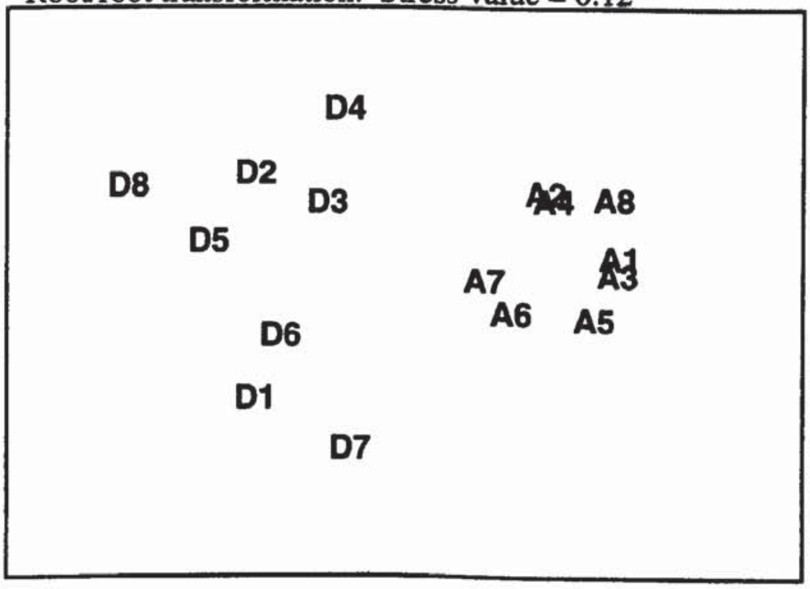
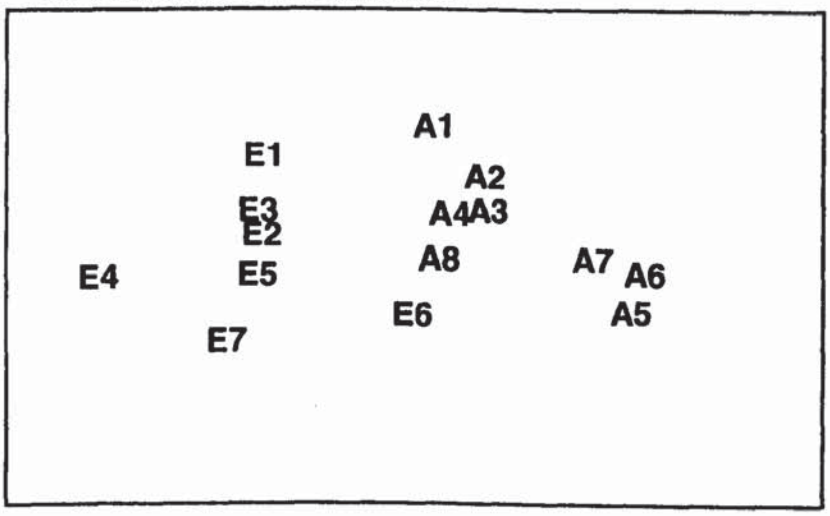


Figure 5.9.4d: Preston Bagot Brook - MDS configuration of Site A(reference) and Site E

No transformation. Stress value =



Root/root transformation. Stress value =

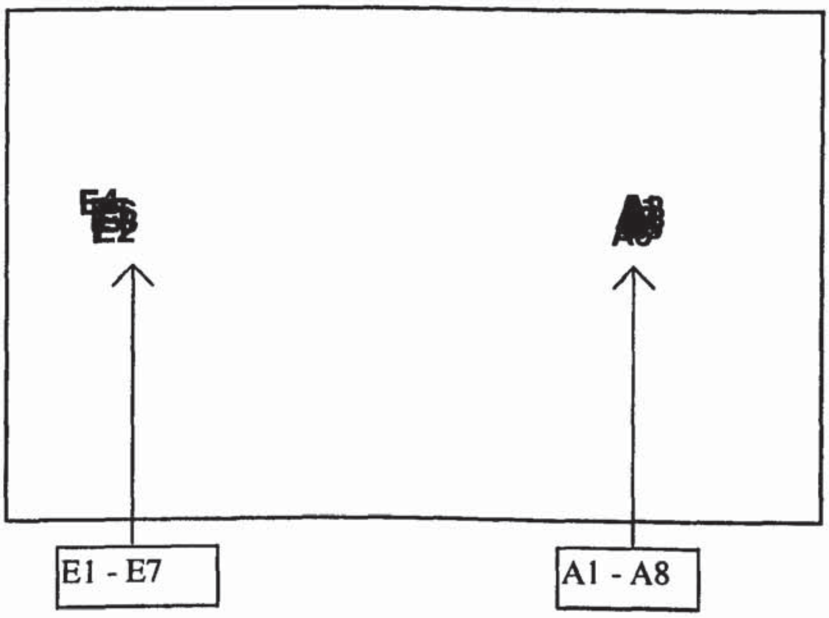
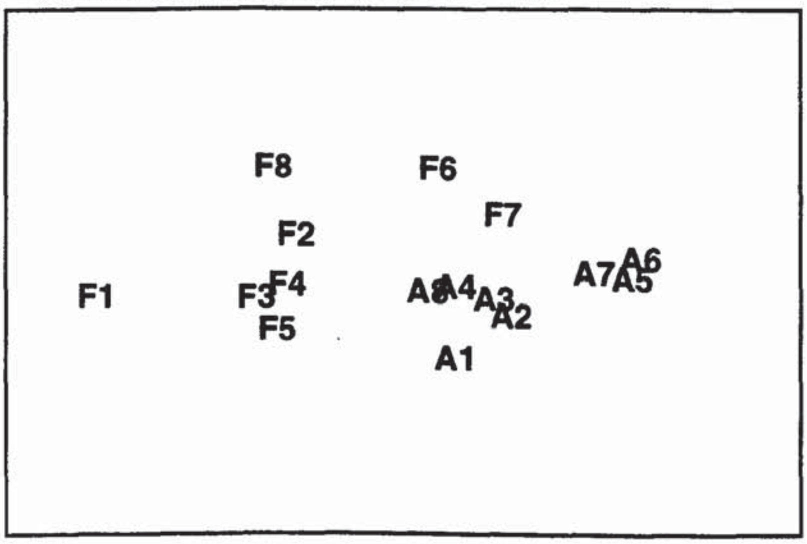


Figure 5.9.4e: Preston Bagot Brook - MDS configuration of Site A(reference) and Site F

No transformation. Stress value = 0.05



Root/root transformation. Stress value = 0.07

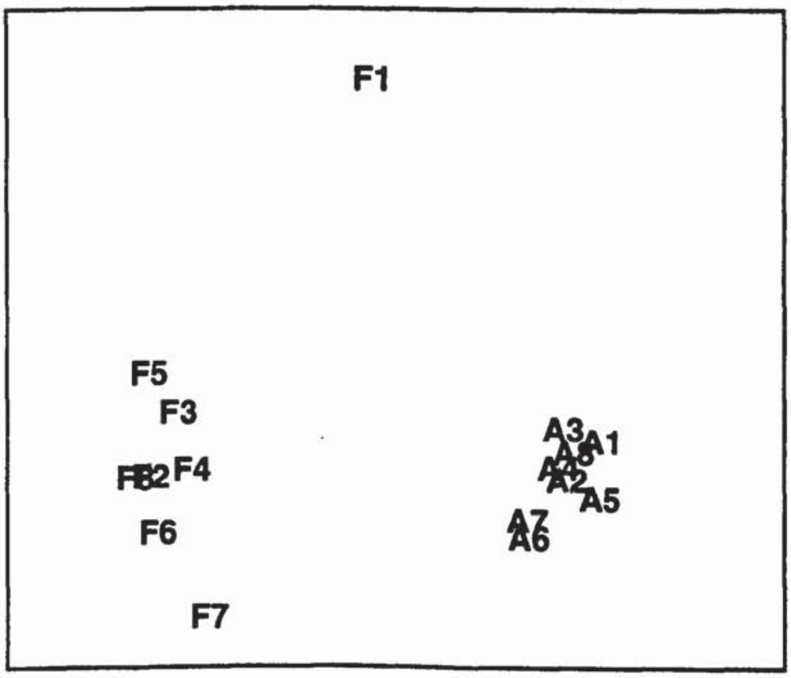
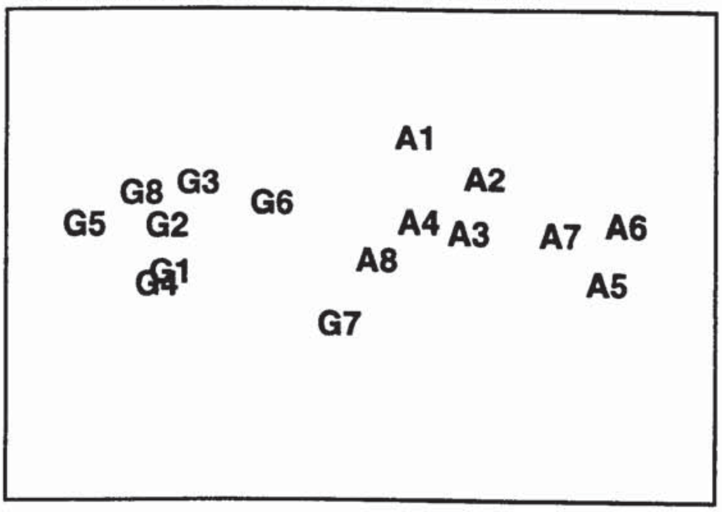


Figure 5.9.4f: Preston Bagot Brook - MDS configuration of Site A(reference) and Site G

No transformation Stress value = 0.06



Root/root transformation Stress value = 0.05

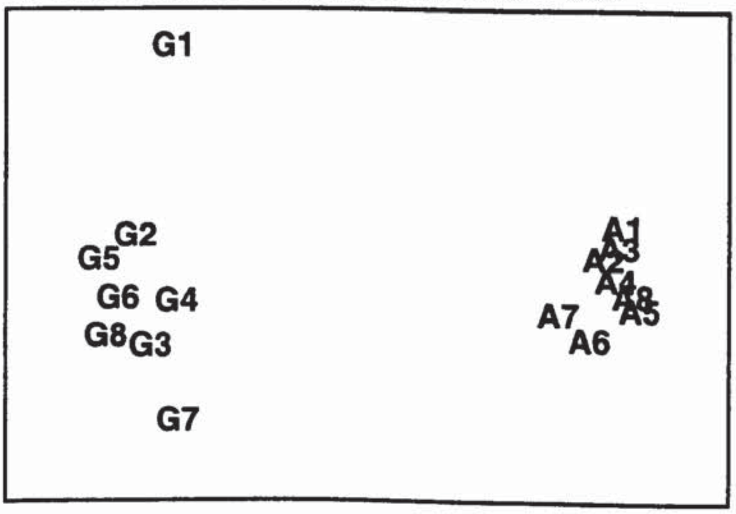
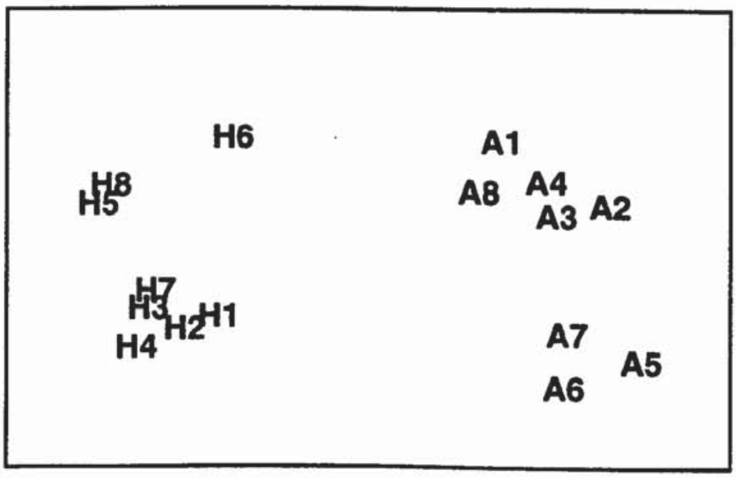


Figure 5.9.4g: Preston Bagot Brook - MDS configuration of Site A(reference) and Site H

No transformation. Stress value = 0.06



Root/root transformation. Stress value = 0.10

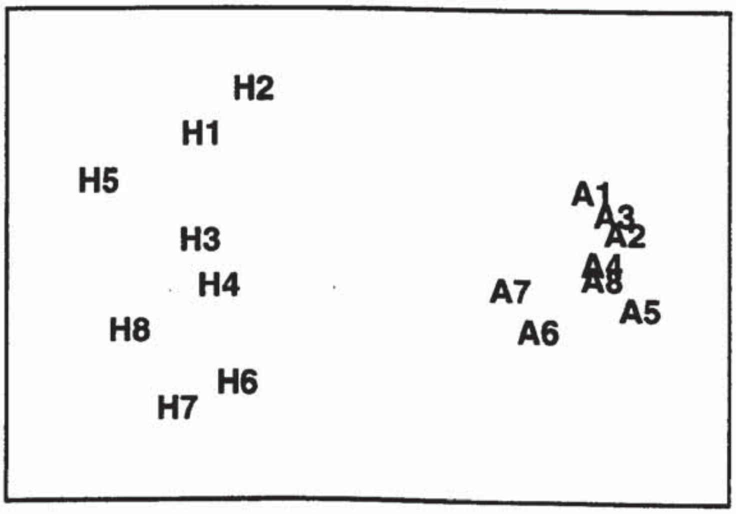
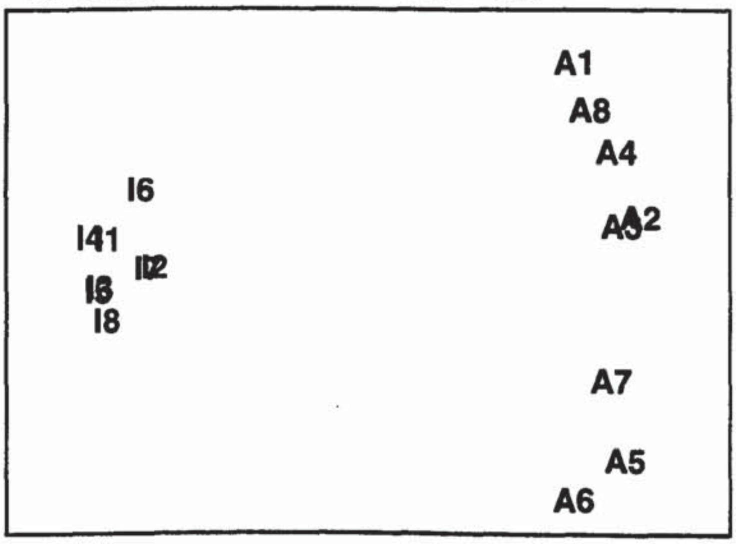
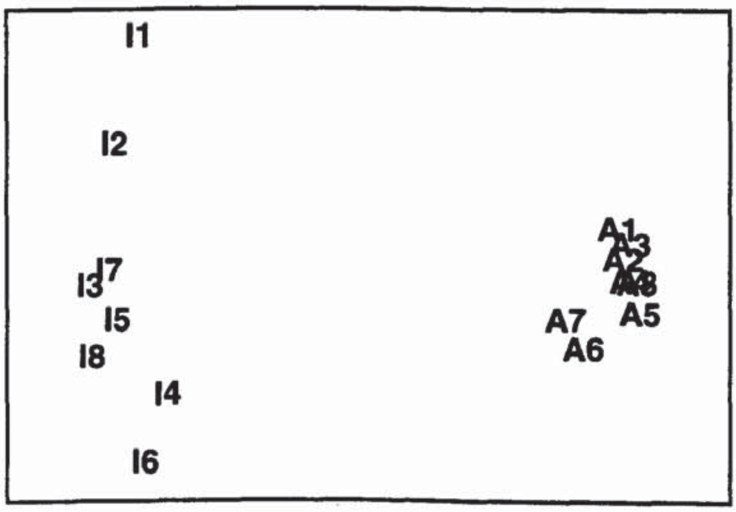


Figure 5.9.4h: Preston Bagot Brook - MDS configuration of Site A(reference) and Site I

No transformation. Stress value = 0.04



Root/root transformation. Stress value = 0.06



5.9.6. Summary

Table 5.9.12 summarises the measures that showed some impact. The acid spill had an immediate effect on fauna within Preston Bagot Brook causing visible fish and macroinvertebrate mortality for at least 1 km downstream of the incident. 20 days after the incident, and 11 km downstream, a decrease in BMWP scores was recorded. The present study commenced 48 days after the incident. It is likely that a great deal of macroinvertebrate recolonisation had already occurred by this time, however the investigation of longer term effects by continuing to assess changes in the fauna for more than 500 days following the incident, provided the rationale for the investigation.

Certain features are worthy of note:

1) Impact on the most abundant taxa was not obvious although certain patterns of change could tentatively be assumed.

2) Impact was found in terms of taxa richness as well as BMWP scores. These were the result of changes in rarer taxa. For example, the cased caddisfly larva; *Goera pilosa* was present in all replicates at sites F and I, 552 days after the incident but showed only sporadic occurrence before this.

3) Multivariate analysis showed a progressive increase in similarity within sites F, G and I as well as greater variability in the communities of the most impacted sites when less abundant taxa are given greater emphasis.

4) Variability of impact on different sites. The prediction that the impact of a pollution incident would be greatest close to the source of the impact and would decrease with distance downstream is not necessarily the case in a stream that has varying environmental parameters and associated fauna. The prediction of impact may require the assessment of each site independently. In this study impact was evident up to 11.5 km downstream of the incident.

Table 5.9.12: Summary of Pollution Impact on All Measures

Measure	Type of impact
Individual Taxa	Chironomidae - no apparent impact *Gammarus pulex, Hydropsyche siltalai*- some evidence of initial low numbers followed by an increase. Oligochaeta, *Potamopyrgus jenkinsi* some evidence of an increase in numbers after the study followed by a decrease.
Number of Taxa	Initially low numbers of taxa at certain sites
Diversity (H')	No evidence of impact
BMWP, ASPT	Low BMWP scores at certain sites Low ASPT scores at certain sites
Multivariate analysis	Greater variability in the communities of impacted sites when less abundant taxa are given greater emphasis. Progressive clustering within impacted sites further downstream

5.10 Sketchley Brook

5.10.1. Description of Incident, Remedial Action

On 10 July 1995, a burst pipe caused a leak of 72,000 litres of fuel oil into the ground floor of a factory. Some of the oil found its way into the surface water system and discharged into the Sketchley Brook in Hinkley, Leicestershire. Within 24 hours, straw bales and absorbent booms were placed up and downstream of the outfall and oil was pumped out of the stream. Three days after the incident, oil mixed with streamwater was still being pumped out, and an area of substrate within 30 m downstream of the outfall, was covered with black oil. The NRA did not keep records of the amount of oil tankered from the site and the quantity of oil entering the brook is not known. The area 30-100 m downstream of the outfall was coated with lesser amounts of oil and a film of oil extended for at least 1 km downstream. NRA staff reported seeing dead *Asellus aquaticus* and Chironomidae within 24 hours of the incident being notified.

5.10.2. Site Description and History

Sketchley Brook is a first order stream, about 5 km in length before its confluence with Harrow Brook which discharges into the River Anker; a tributary of the River Tame (Fig. 5.10.1). The Brook initially drains an area of marshland within a residential area of Hinkley and flows through residential and industrial estates before flowing through arable land. Its volume is greatly enhanced by the outfall from Hinkley Sewage Treatment Works.

Water quality data from 1992-1994 is shown in Table 5.10.1. The General Quality Assessment based upon chemical data was 'F' (Bad quality) in 1990, and 'E' (Poor quality) in 1994 according to NRA data. The Brook receives surface drainage from the industrial estates (between Sites B and D on Fig. 5.10.1) and, in the past, has been subject to highly visible pollution incidents in the form of dark coloured dyes from the local dye works.

Aston University

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5.10.3. Sampling Procedure

Three replicate Surber samples were taken from 3 sites upstream and 3 sites downstream of the pollution incident. The location of the sampling points A-F is illustrated in Fig. 5.10.1 and their environmental parameters are summarised in Table 5.10.2. Two of the 'reference' sampling sites are situated on the Sketchley Brook and one is on a small tributary. The pollution incident issued from a storm sewer overflow discharging into the brook 1.1 km downstream of the source of the brook. Site B (reference) is within a silty, vegetation-filled channel, but the other sites are located on riffles.

Table 5.10.2: Sketchley Brook - Summary of Environmental Parameters at each sampling point

Site Name	Location (G R)	Distance from pollution incident (m)	Substrate	Depth (cm)	Width (cm)	Flow m/sec	Q (cu.m/sec) 3/10/95
Site A - Brookside	SP 430 931	850 m upstream	20%silt/ 50%gravel/ 30%pebble	2.0-7.0	100	0.024-0.22	0.00077
Site B - Nature Reserve	SP 431 932	800 m upstream	100%silt	2.0-10.0	750	0.14-0.19	0.00152
Site C - 5m US Outfall	SP 422 928	5 m upstream	20%sand/ 30%gravel/ 50%pebble	2.0-10.0	150	0.05-0.17	0.0039
Site D - 50m DS Outfall	SP 421 928	50 m downstream	50%pebble/ 50%gravel	2.0-5.0	100	0.125-0.2	0.0064
Site E - Hinkley STW	SP 419 927	300 m downstream	10%silt/ 10%gravel/ 30%pebble/ 50%sand	3.0-10.0	60	0.12-0.25	0.00576
Site F - Nutts Lane	SP 409 924	1100 m downstream	50%gravel/ 50%pebble	10.0-30.0	300	0.2-0.5	0.0864

Seven sampling times were distributed over a period of 365 days following the incident (Table 5.10.3.)

Table 5.10.3: Summary of sampling times

Sampling time Code	Date		Days following incident
Sk1	13 July	1995	3
Sk2	2 August	1995	23
Sk3	31 August	1995	52
Sk4	3 October	1995	85
Sk5	23 November	1995	136
Sk6	19 March	1996	253
Sk7	9 July	1996	365

5.10.4. Predicted Impact of Pollution

Table 5.10.4: Predicted Impact of Pollution Incident on Biotic Measures

Measure	Predicted effect
Individual taxa	Initial toxicity causes loss of sensitive taxa and decrease in abundance of less sensitive taxa. This is followed by a rapid increase in those taxa able to benefit from the changed conditions and then showing a decline, and a more gradual increase in other taxa as conditions resemble those prior to the incident.
Number of Taxa	Initial low numbers followed by a gradual increase.
Diversity	Initial low diversity followed by a gradual increase.
BMWP and ASPT	Initial low BMWP score and ASPT score as taxa are lost and any taxa are dominated by tolerant taxa, followed by an increase over time.
Multivariate Analysis of Community Similarity	1. Upstream 'control' sites will show less variability over time than impacted sites (impact). 2. Impacted sites will show a progressively greater similarity to the 'control' sites over time (temporal recovery). This information can be used to determine a time of recovery within a site. 3. There will be progressively less variability and less movement towards similarity between impacted and control sites, the further downstream of the impact the sampling point is located (Spatial recovery). 4. Progressive increase in similarity within impacted site i.e. within an impacted site those samples closer in time to the pollution incident will show less similarity to each other than those samples further in time.

The oil involved in this pollution is residual fuel oil, used for industrial heating purposes. Residual fuel oils are paraffinic, naphthenic and aromatic hydrocarbons mainly from blends of residues from crude oil distillation. The predicted impact of general oil pollution on the measures used in this study is discussed in Chapter 5.5 (Hewell Brook) and summarised in Table 5.10.4.

5.10.5. Results and Analysis

The data is organised as described in Chapter 4. As no pre-impact measures exist, both impact and recovery have to be inferred by comparison with the three unimpacted upstream sites.

5.10.5.1. Water Quality

Table 5.10.5 gives the physicochemical measurements taken at each sampling time. Conductivity levels are consistently lower at Sites A and B than at Sites C, D, E and F. This may be the result of increased levels of dissolved ions coming from surface water sewers draining industrial and residential estates.

Table 5.10.5: Physicochemical Conditions in Sketchley Brook (Dissolved Oxygen in mg/l; Conductivity: µS cm-1 at 25°C)

		Site A - Brookside				Site B - Nature Reserve				Site C - 5m US Outfall			
Date	Days following Disturbance	Temp (oC)	pH	DO	Cond.	Temp (oC)	pH	DO	Cond.	Temp (oC)	pH	DO	Cond.
13-Jul-95	3	17.7	7.9	n/a	645	17.30	7.4	n/a	n/a	18.5	7	n/a	997
02-Aug-95	23	21	8.1	n/a	694	19.6	7.7	n/a	746	20.7	8	n/a	1219
31-Aug-95	52	15.9	7.4	n/a	648	15.1	6.9	n/a	744	16.6	7.5	n/a	1397
03-Oct-95	85	13.8	7.7	3.4	648	13.1	7.2	7.5	731	14.8	8	6.6	888
23-Nov-95	136	9.6	8.1	12.3	508	8.7	7.9	10.6	511	11.3	7.9	5.6	466
19-Mar-96	253	5.9	8.5	13.4	551	5.3	8.6	13.3	693	10	n/a	10.8	1317
09-Jul-96	365	14.8	n/a	9.2	677	15.6	8.3	5.4	722	17.2	6.4	4.5	1077

		Site D - 50m DS Outfall				Site E - Hinkley STW				Site F - Nutts Lane			
Date	Days following Disturbance	Temp (oC)	pH	DO	Cond.	Temp (oC)	pH	DO	Cond.	Temp (oC)	pH	DO	Cond.
13-Jul-95	3	18.1	8.4	n/a	1138	18.3	8	n/a	n/a	21.9	7.7	n/a	n/a
02-Aug-95	23	19.5	8.8	n/a	1378	19.4	8	n/a	1687	22.6	7.8	n/a	1231
31-Aug-95	52	17.5	8	n/a	1241	15.1	7.3	n/a	n/a	19.7	6.9	n/a	1441
03-Oct-95	85	16.1	8	8.5	1850	13.1	7.4	6.8	1625	15.9	6.9	9.3	1274
23-Nov-95	136	11.3	8	7	164	9.2	7.9	6.9	1503	12.4	7.6	8.4	1142
19-Mar-96	253	9.6	n/a	12.3	1239	6.4	n/a	16.8	1159	8.3	8.1	7.6	1155
09-Jul-96	365	16.4	n/a	5.6	1336	14.5	n/a	6.5	1482	17.8	8.3	10.5	1289

5.10.5.2. Selected Taxa

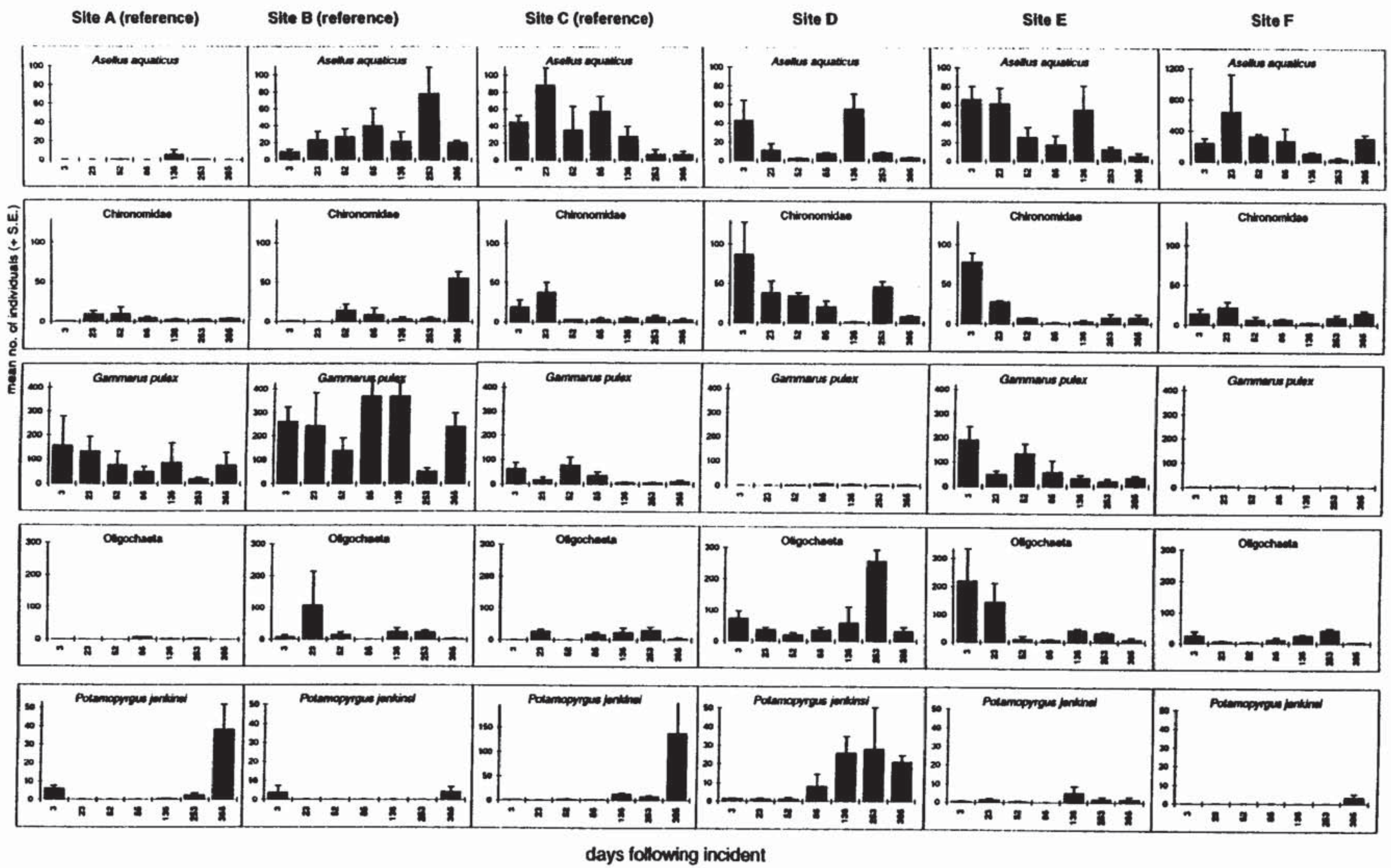
Table 5.10.6 is a summary of the taxa found and their total abundance. Forty taxa were found throughout the study. The five most abundant taxa are listed below and changes in their presence and abundance throughout the study are shown in Fig. 5.10.2 and summarised in Table 5.10.7.

Asellus aquaticus	Oligochaeta
Chironomidae	*Potamopyrgus jenkinsi*
Gammarus pulex	

Table 5.10.6: Sketchley Brook - Summary of Taxa found

Phylum, Order, Family, Genus, Species	Number found in study
PLATYHELMINTHES TURBELLARIA:TRICLADIDA	
Planariidae	
Dugesia sp.	5
Polycelis felina (Dalyell)	130
Polycelis tenuis (Ijima)/*nigra* (Muller)	180
Dendrocoelidae	
Dendrocoelum lacteum (Muller)	2
MOLLUSCA	
GASTROPODA:PROSOBRANCHIA	
Hydrobiidae	
Potamopyrgus jenkinsi (Smith)	943
GASTROPODA:PULMONATA	
Lymnaeidae	
Lymnaea palustris (Muller)	9
Lymnaea peregra (Muller)	10
Lymnaea stagnalis (L.)	1
Lymnaea truncatula (Muller)	6
BIVALVIA	
Sphaeriidae	30
ANNELIDA	
OLIGOCHAETA	4102
HIRUDINEA	
Glossiphoniidae	
Glossiphonia complanata (L.)	14
Glossiphonia heteroclita (L.)	2
Helobdella stagnalis (L.)	1
Erpobdellidae	42
Erpobdella octoculata (L.)	8
Trocheta bykowskii Gedroyc	4
ACARINA:HYDRACARINA	8
CRUSTACEA: MALACOSTRACA: ISOPODA	
Asellidae	
Asellus aquaticus (L.)	8383
CRUSTACEA:MALACOSTRACA :	
AMPHIPODA	
Gammaridae	
Gammarus pulex (L.)	7275
Gammarus tiginus Sexton	21
INSECTA: EPHEMEROPTERA	
Baetidae	
Baetis rhodani Pictet	405
INSECTA : COLEOPTERA	
Dytiscidae	
Agabus bipustulatus (L.)	1
Agabus paludosus (Fab.)	1
Dytiscidae larva	3
Hydrophilidae	
Helophorus grandis Illiger	2
Helophorus sp.	29
INSECTA : TRICHOPTERA	
Polycentropidae	
Plectornemia conspersa (Curtis)	1
Limnephilidae	3
Glyphotaelius pellucidus (Retzius)	2
Micropterna sequax McLachlan	5
INSECTA: HEMIPTERA, HETEROPTERA	
Corixidae	3
Gerridae	
Gerris lacustris (L.)	1
Notonectidae	
Notonecta glauca (L.)	1
Veliidae	3
Velia sp.	
INSECTA : DIPTERA	
Dipteran larvae	8
Ceratopogonidae	2
Chironomidae	1910
Simuliidae	67
Tipulidae	16

Figure 5.10.2: Sketchley Brook - Changes in abundance of selected taxa following pollution incident



Asellus aquaticus

High numbers of *Asellus aquaticus* are found at Site F, just downstream of the outflow from the Sewage Treatment Works, a finding to be expected from this area of high organic matter. At impacted Sites D and E, numbers of *A.aquaticus* are higher immediately after the incident than one year later (Mann-Whitney U test, $p<0.05$) whereas no significant difference is found at the upstream Sites B and C. At Site E, the substrate was coated with oil immediately after the incident and the presence of *A.aquaticus* indicates a tolerance to the changed conditions.

Chironomidae

At impacted Sites D and E, higher numbers of Chironomidae are found immediately after the incident than one year later (Mann-Whitney U test, $p<0.05$) whereas no significant difference is found at the upstream Sites B and C.

Gammarus pulex

G. pulex is a standard component of the upstream fauna, with particularly high abundance at Site B. At impacted Sites D and E, dead *G. pulex* were collected 3 days after the incident. At impacted Site D, *G. pulex* is absent 3 days after the impact and is subsequently present in low numbers. At Site E, numbers of *G. pulex* are higher 3 days after the incident than one year later. This may have been the result of adverse conditions closer to the pollution incident causing drift to occur.

Oligochaeta

This group comprises largely Tubificidae worms. Numbers at reference sites are not high. Numbers at impacted Site D remain even throughout the year with a peak in abundance 253 days following the incident (as this was not seen at the control sites, this may have been a response to the breakdown of oil increasing the availability of food). Site E on the other hand showed high numbers of Oligochaeta 3 days after the incident, and a decrease thereafter.

Potamopyrgus jenkinsi

Numbers at the impacted Site D increase towards the latter end of the study however the same pattern is seen at the reference Sites A and C so no effect of the oil pollution can be deduced.

Table 5.10.7: Summary of impact of pollution incident on selected taxa

Taxon	Impact of pollution
Asellus aquaticus	Increase in abundance 3 days after the pollution as ISites D and E and subsequent decrease
Chironomidae	Increase in abundance 3 days after the pollution at Sites D and E
Gammarus pulex	At Site D initial absence then low numbers. At Site E, high initial numbers then a decrease
Oligochaeta	Site D peak in abundance after 253 days. Site E high numbers 3 days after then decrease.
Potamopyrgus jenkinsi	No apparent impact

5.10.5.3. Univariate Indices

Univariate indices (Number of Taxa, Diversity index, BMWP and ASPT scores) are plotted in Fig. 5.10.3.

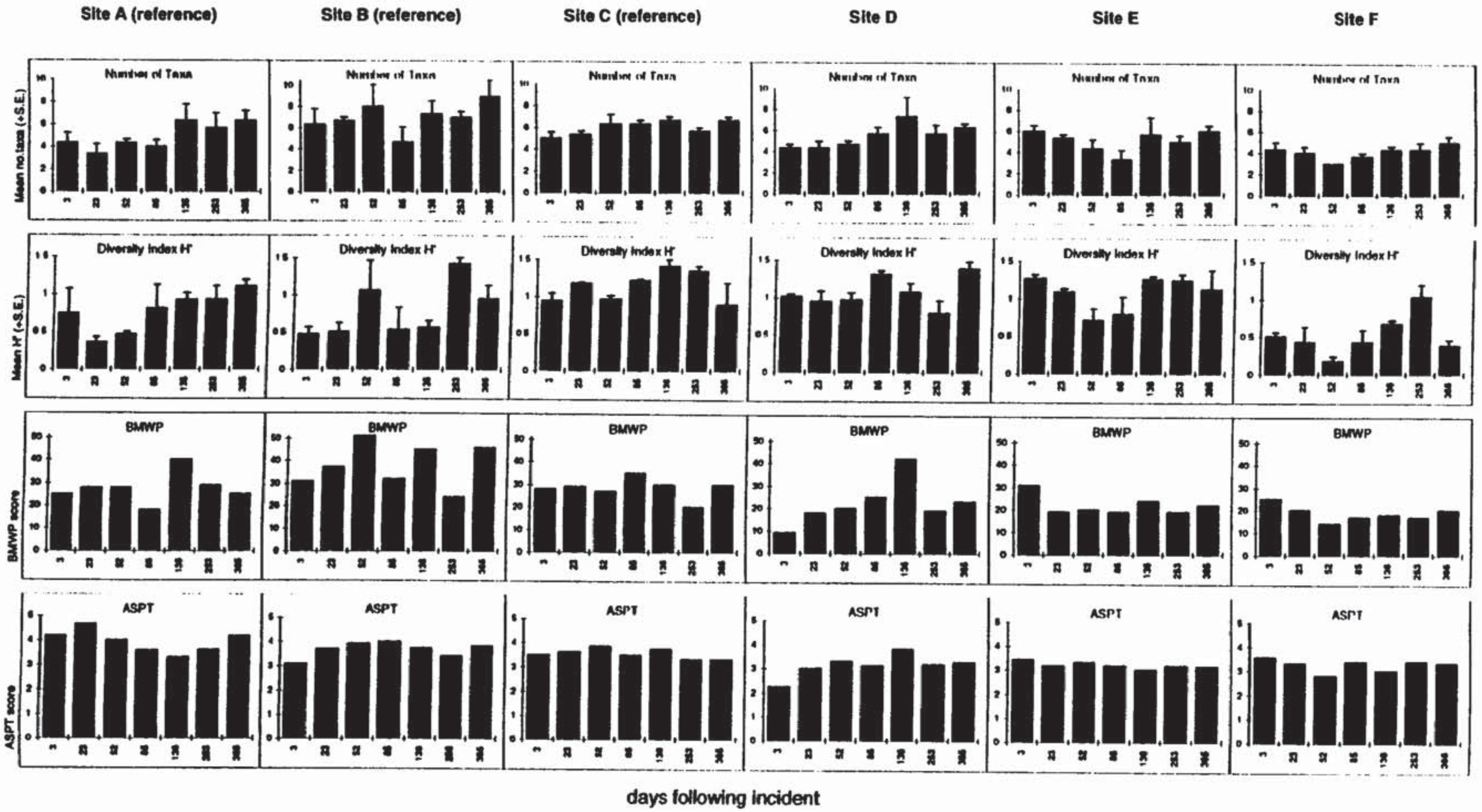
Number of Taxa

No clear pattern in the number of taxa is seen in reference or the impacted sites. When the number of taxa 3 days after the incident is compared with the number of taxa 365 days after the incident, at both reference sites (A,B,C) and impacted sites (D,E,F), no significant difference is found in either the reference or impacted sites (Mann-Whitney U test). At each site, the first two sampling times were compared with the last two sampling times. A significant difference was seen at Site D. However a significant difference was also found at reference Site C (refer to Table 5.10.8).

Table 5.10.8: Number of Taxa. Summary of significance tests (ns = not significant; s = significant: $p<0.05$: Mann-Whitney U test).

SITES	COMPARISONS	SIGNIFICANCE
ABC	3 days vs 365 days after incident	ns
EFG	3 days vs 365 days after incident	ns
A	A1 and A2 compared with A6 and A7	ns
B	B1 and B2 compared with B6 and B7	ns
C	C1 and C2 compared with C6 and C7	s
D	D1 and D2 compared with D6 and D7	s
E	E1 and E2 compared with E6 and E7	ns
F	F1 and F2 compared with F6 and F7	ns

Figure 5.10.3: Sketchley Brook - Changes in univariate indices following pollution incident



Diversity Index

At each site, the first two sampling times were compared with the last two sampling times. A significant difference was seen at the 2 upstream reference Sites B and D. No effect of the pollution on the diversity index is therefore evident (refer to Table 5.10.9)

Table 5.10.9: Diversity Index. Summary of significance tests (ns = not significant; s = significant: $p<0.05$: Mann-Whitney U test).

COMPARISONS	SIGNIFICANCE
A1 and A2 compared with A6 and A7	s
B1 and B2 compared with B6 and B7	s
C1 and C2 compared with C6 and C7	ns
D1 and D2 compared with D6 and D7	ns
E1 and E2 compared with E6 and E7	ns
F1 and F2 compared with F6 and F7	ns

BMWP and ASPT

BMWP scores throughout the study are greater at the upstream reference sites than at the impacted sites. Low BMWP and ASPT scores are seen at Site D, 3 days after the incident but 23 days after the incident, they have increased.

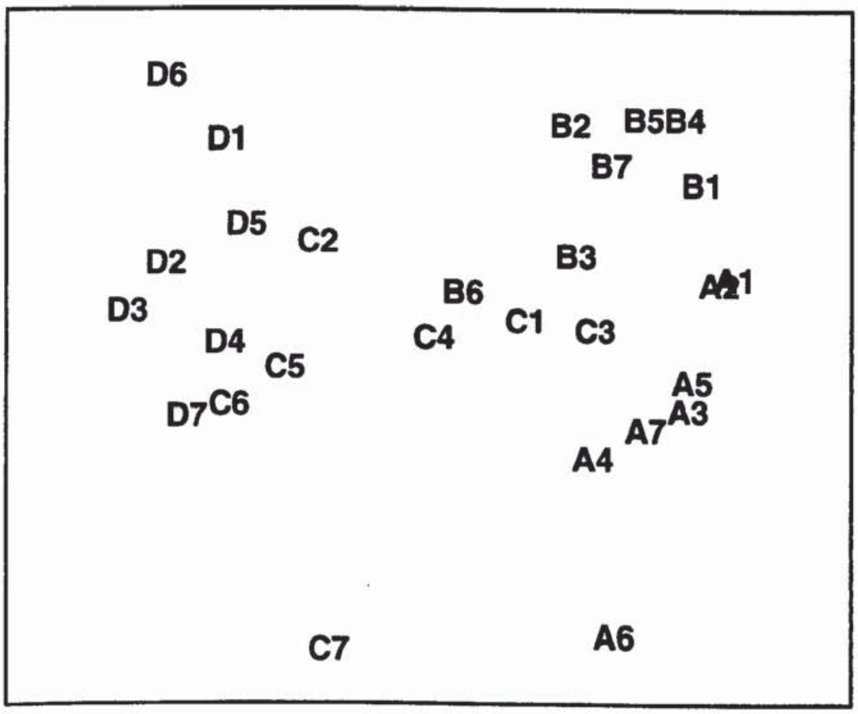
5.10.5.4. Multivariate Measures

The R Statistic was computed for each sampling time as described in Chapter 4. This showed that the replicates within sites are more similar to each other than replicates between sites. Similarity matrices were therefore produced from data from averaged replicates. The similarity matrices were used to construct Multidimensional Scaling configurations (MDS plots) comparing each impacted site with the three upstream reference sites; A, B, C (Figs 5.10.4a-d).

Fig 5.10.4a (MDS plot of Sites A, B, C and D) shows samples from Site D to be more similar to reference Site C than the other two reference sites. Furthermore D1 is least similar to Site C communities and subsequent communities are closer to Site C in similarity. Fig 5.10.4b is an MDS plot of Sites C and D which illustrates the relationships more clearly. An ANOSIM analysis carried out to compare D1 with all the samples found at Site C, using a range of transformations, found no significant difference (Table 5.10.10).

Figure 5.10.4a: Sketchley Brook - MDS configuration of Sites A, B, C (reference) and Site D (impacted)

No transformation. Stress = .13



Root/root transformation. Stress = .16

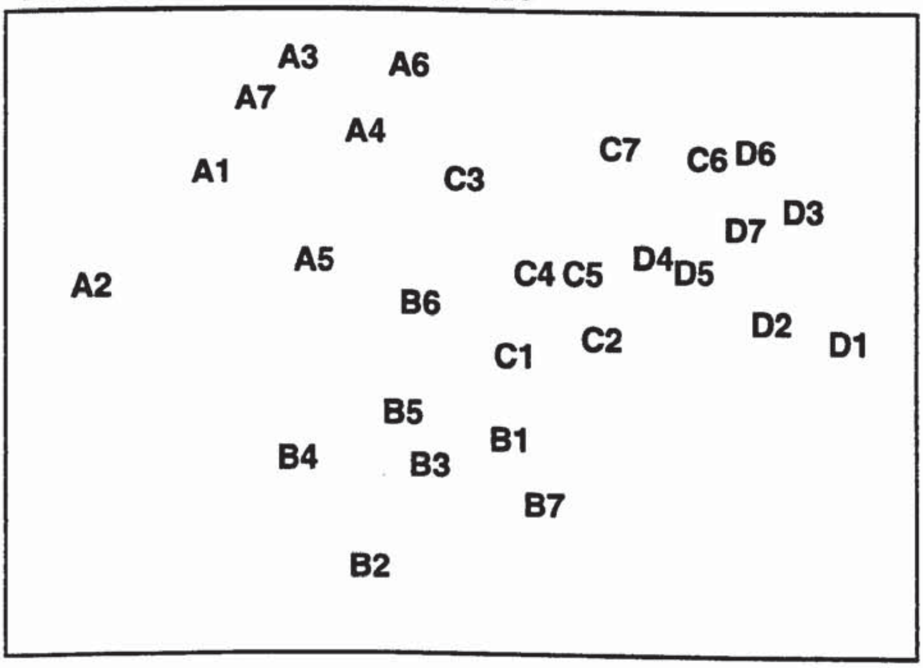
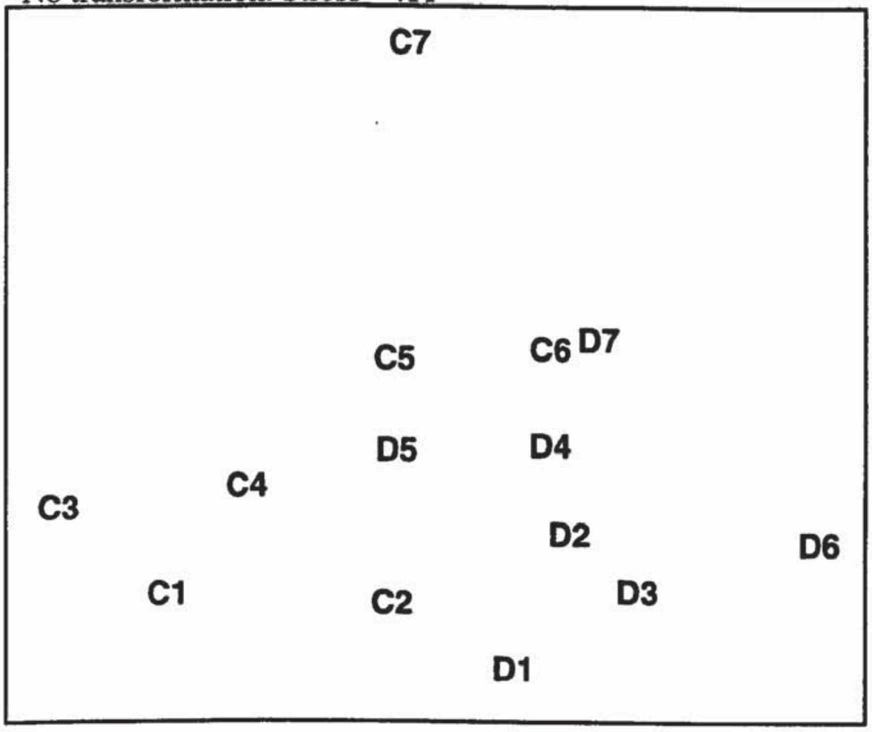


Figure 5.10.4b: Sketchley Brook - MDS configuration of Site C (reference) and Site D (impacted)

No transformation. Stress = .11



Root/root transformatioan. Stress = .14

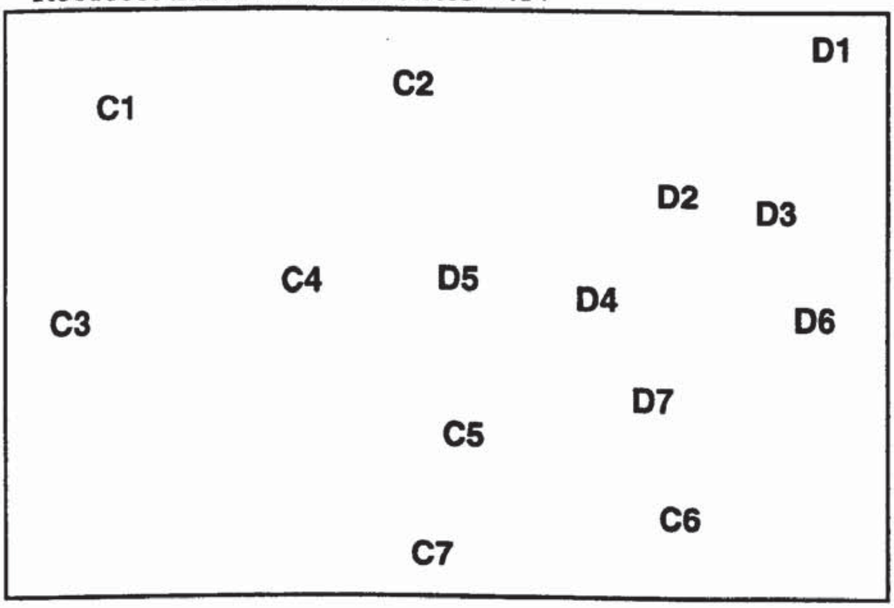
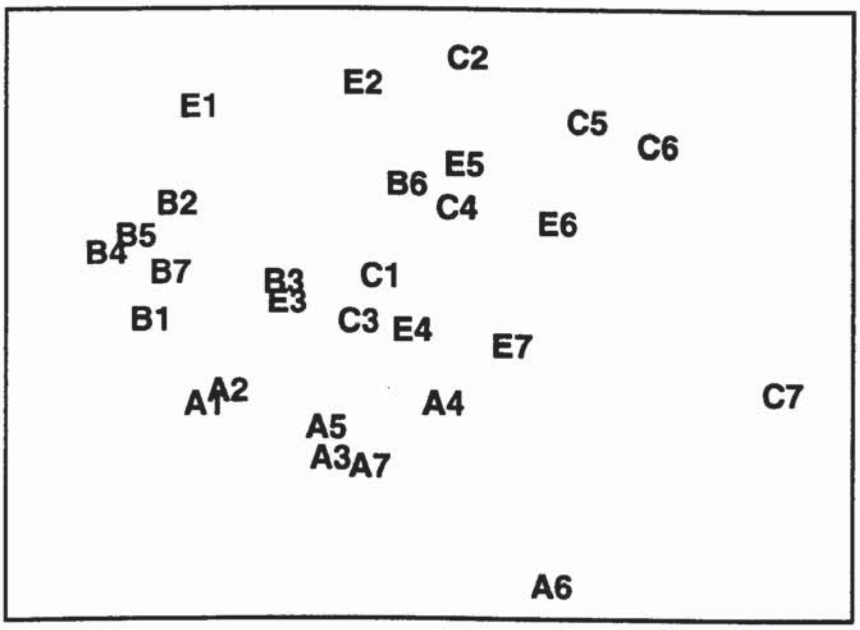


Figure 5.10.4c: Sketchley Brook - MDS configuration of Sites A, B, C (reference) and Site E (impacted)

No transformation. Stress = .11



Root/root transformation. Stress = .18

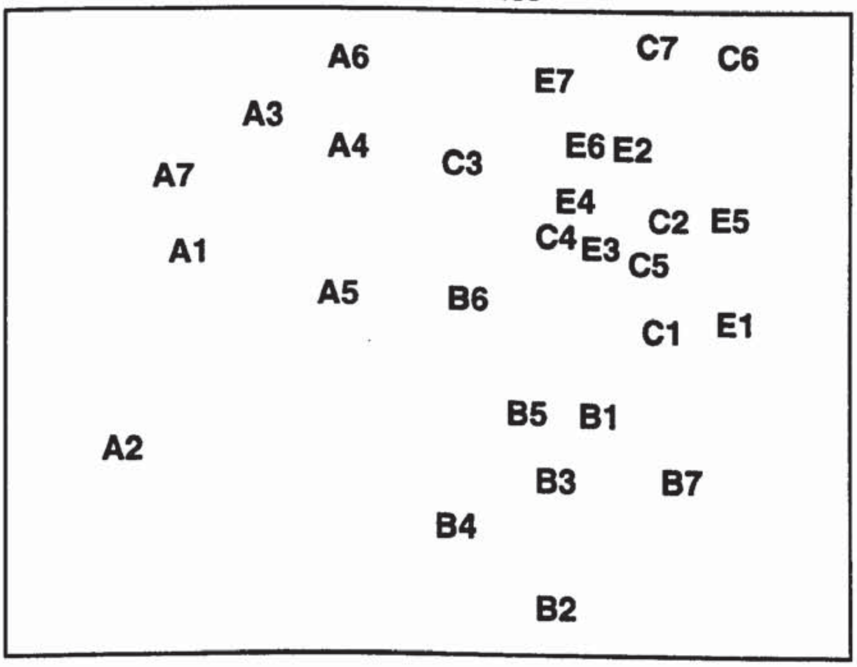
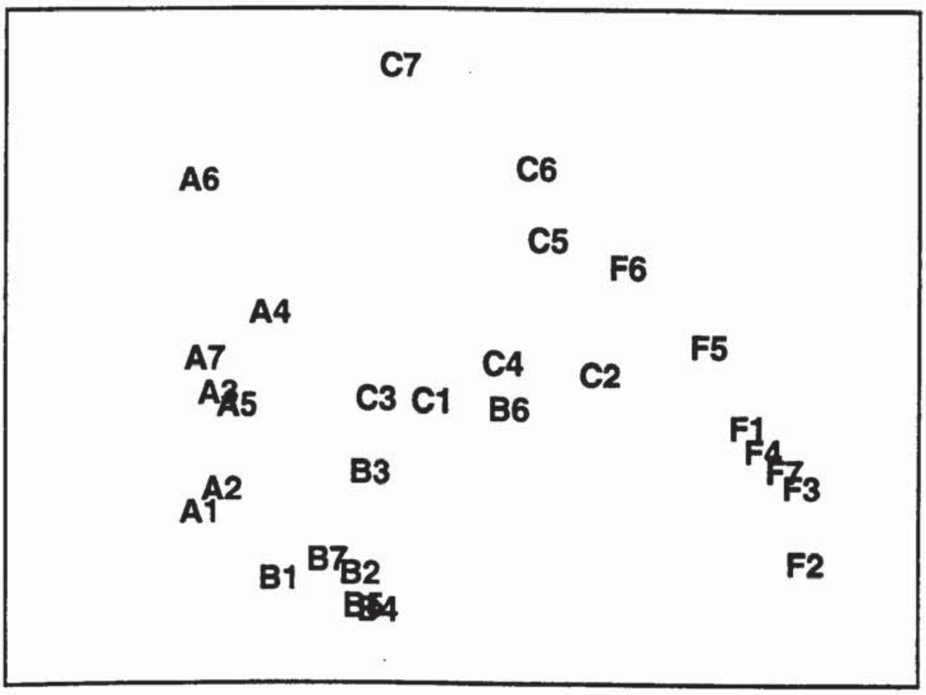
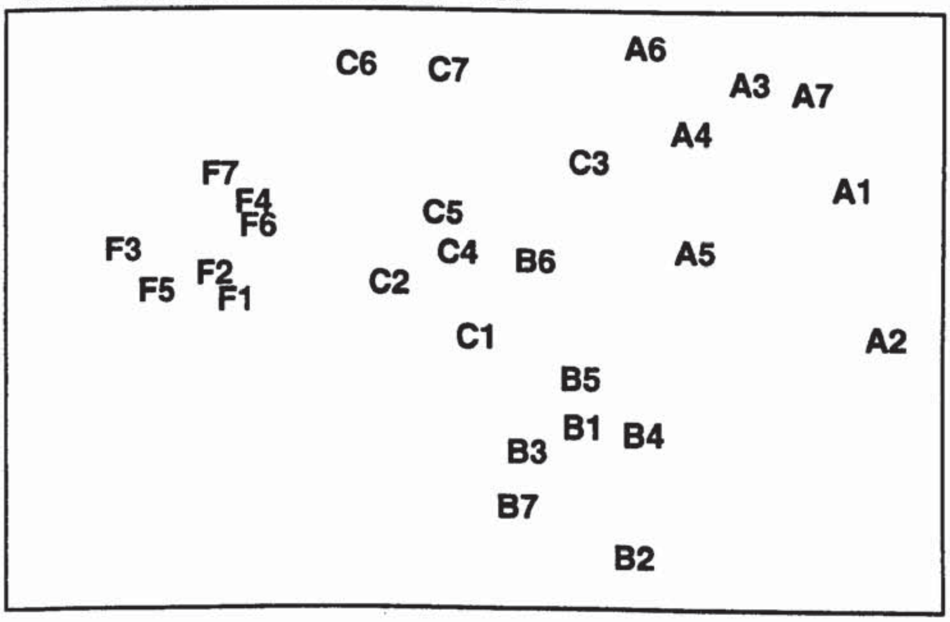


Figure 5.10.4d: Sketchley Brook - MDS configuration of Sites A, B, C (reference) and Site F (impacted)

No transformation. Stress = .11



Root/root transformation. Stress = .16



Closer examination of the significance levels shows that greatest similarity between D1 and C is seen when presence/absence data is used. This implies that any differences that do exist are due more to the abundance of taxa than their presence or absence.

Table 5.10.10 Summary of ANOSIM analyses comparing reference Site C with D1 (ns = not significant p<0.05)

Transformation of Taxon Abundance	D1 compared with C1-C7
No transformation	ns (p< 0.14)
Square root	ns (p< 0.09)
Root root	ns (p< 0.14)
Presence/absence	ns (p< 0.28)

Fig. 5.10.4c (MDS plot comparing Sites A,B,C and E) shows Sites C and E to be similar in their communities throughout the study.

Fig. 5.10.4d (MDS plot comparing Sites A, B, C and F) similarly shows no evidence of impact.

5.10.6. Summary

Table 5.10.11 summarises the pollution impact on all the biotic measures used. Sketchley Brook is a stream of poor chemical and biological quality and subject to intermittent minor pollution discharges. The upper reaches and tributary were found to support large populations of *Gammarus pulex* and moderate populations of the mayfly *Baetis rhodani,* thus acting as a colonisation source for downstream reaches.

Within the sampling programme, *Gammarus pulex* is the only taxon to have shown an initial effect of the oil. Dead individuals of this species were found at Sites D and E 3 days after the incident, suggesting mortality due the toxic fractions within the oil. In addition, higher numbers of *G. pulex* were found at Site E 3 days after the incident than 365 days, indicating possible increase in drift.

Asellus aquaticus and Chironomidae also suffered mortality one day after the pollution incident, as dead individuals were seen by NRA staff. However, this was not reflected in the samples taken 3 days after the incident. Indeed, the samples contained a greater abundance of these two taxa than samples taken one year later. It is possible that after 3 days, the volatilisation of the highly toxic low molecular weight fractions of the oil had occurred (Foght and Westlake, 1987), and that these taxa had been attracted by a potential food source resulting from biodegradation of the oil. Foght and Westlake (1987) suggest that among the most important factors affecting the bacterial degradation of oil in freshwater is the nutrient status of the water and the previous history of the watercourse. The conductivity measurements downstream and just upstream of the pollution outfall are high (Sites C, D, E and F see Table 5.10.5), suggesting a high nutrient status. Although no information exists on previous oil spills into the stream, oil is a common component in urban surface water run-off, therefore oil degrading bacteria are likely to be present within the stream. These factors may therefore provide conditions conducive to the rapid biodegradation of oil.

BMWP scores and multivariate measures provide some evidence of impact at Site D after 3 days. Although not quantified, physical evidence of oil within the substratum of Site D was found after 136 days.

To summarise, the background pollution in Sketchley Brook is high and the pollution incident has not had a great measurable impact on the benthic macroinvertebrates.

Table 5.10.11: Summary of Pollution Impact on All Measures

Measure	Type of impact	Spatial Impact	Temporal Impact
Individual Taxa	Initial decrease in abundance followed by an increase (*Gammarus pulex*).	D,E	3 days
	Initial increase in abundance followed by a subsequent decrease (*Asellus aquaticus, Chironomidae,* Oligochaeta)		
Number of Taxa	No evidence of impact		
Diversity (H')	No evidence of impact		
BMWP, ASPT	Low BMWP and ASPT score	Site D	3 days
Multivariate analysis	Some evidence of initial low similarity to control site and to subsequent samples within site	Site D	3 days
Visual assessment of substratum	Oil released from substratum when disturbed	Site D	136 days

5.11 Spark Brook

5.11.1. Description of Incident

On 14 July, 1994 a member of the public reported a noxious smell and discoloured water in the Spark Brook, Birmingham as it emerged from a culvert. Investigation by pollution control officers revealed that, due to a blockage, foul sewage was entering the Spark Brook. The foul sewage in this area was known to contain a high percentage of industrial effluent. The blockage was cleared within 2 days. The site was visited 7 days after the incident and dead leeches (Erpobdellidae) were found on the substratum.

5.11.2. Site Description and History

The Spark Brook is a small tributary of the River Cole in Birmingham. The NRA do not hold water quality information on the Spark Brook itself, however, the water quality is known to be poor due to unregulated effluents discharging into the largely culverted nature of the brook (Environment Agency pollution control officer, personal communication). The River Cole, into which the Spark Brook flows, has a River Quality Objective of Class 2 for the lower reaches. The Biological Water Quality of the River Cole is moderate with typical BMWP scores of 36-65 (NRA, 1993).

5.11.3. Sampling Procedure

Three replicate Surber samples were taken from 6 sites (see Fig. 5.11.1). Given the upstream location of the pollution incident, it was not possible to have upstream reference sites. One reference site was chosen on the River Cole upstream of the Spark Brook/River Cole confluence (Site A). Three sites were located on the Spark Brook itself (Sites B, C, D) and two further sites on the River Cole downstream of the Spark Brook/River Cole confluence (Sites E and F).

The environmental parameters of the sampling sites are summarised in Table 5.11.1. All sampling sites were situated in riffle areas. The reference site on the River Cole differed from the other sites in its greater flow and average depth.

Figure 5.11.1: Spark Brook - Location of Pollution Incident and Sampling Points

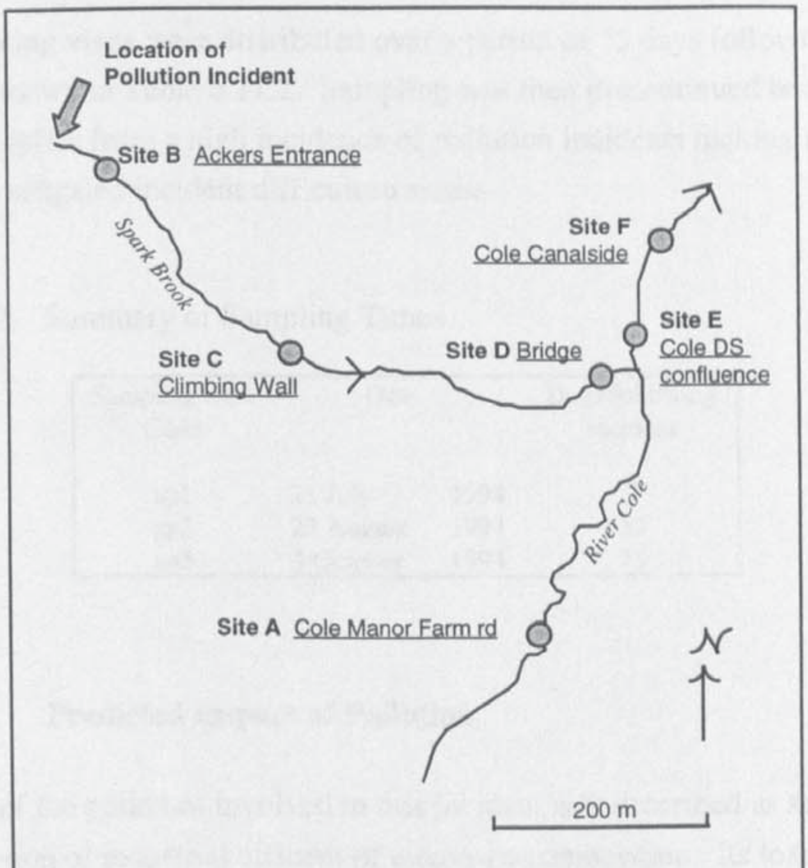


Table 5.11.1 - Spark Brook - Summary of Environmental Parameters at each sampling point

Site Name	Location (G R)	Distance from pollution incident (m)	Substrate	Average Depth (cm)	Width (cm)	Flow m/sec (23/8/94)	Q (cu.m./sec) (23/8/94)
Site A - Cole, Manor Farm (reference)	SP 101 842	River Cole 500m upstream Cole/Spark Brook confluence	50%pebble/ 30%brick/ 20% sand/gravel	10,20	300	0.12	0.1566
Site B - Ackers Entrance	SP 096 846	100 downstream	50%pebble/ 50%gravel/ sand/ silt	2,5	80	0.048	0.0086
Site C - Climbing Wall	SP 098 844	450 downstream	50% pebble/ 50%gravel	2,6	150	0.028	0.007
Site D - Bridge	SP 103 844	820 downstream	50% pebble/ 50%gravel	5,10	110	0.08	0.0134
Site E - Cole, DS confluence	SP 103 845	River Cole 10m downstream Cole/Spark Brook confluence	60%pebble/ 40%gravel	2,5	300	0.048	0.0288
Site F - Cole, canalside	SP 104 846	River Cole 100m downstream Cole/Spark Brook confluence	50%pebble/ 50%gravel	2,10	250	0.06	0.03

Three sampling visits were distributed over a period of 75 days following the pollution incident as shown in Table 5.11.2. Sampling was then discontinued because the brook appeared to suffer from a high incidence of pollution incidents making further recovery from the investigated incident difficult to assess.

Table 5.11.2: Summary of Sampling Times

Sampling time Code	Date		Days following incident
sp1	21 July	1994	7
sp2	23 August	1994	33
sp3	4 October	1994	75

5.11.4. Predicted Impact of Pollution

The nature of the pollutant involved in this incident was described as sewage with a high proportion of industrial effluent of unknown composition. Its toxic nature was apparent from the dead leeches and snails seen close to the culvert. The predicted impact of the incident would therefore be a reduction on numbers of taxa within the Spark Brook and downstream of the Spark Brook/River Cole confluence. Table 5.11.3 outlines the predicted impact of the pollution on the biological measures investigated.

5.11.5. Results and Analysis

The data is organised as described in Chapter 4. No pre- impact measures exist and there are no upstream reference sites. Impact of the pollution incident must therefore be assessed by temporal changes within Spark Brook itself and when compared with reference Site A on the River Cole.

Table 5.11.3: Predicted Impact of Pollution Incident on Biotic Measures

Measure	Predicted effect
Individual taxa	Initial toxicity causes loss of sensitive taxa and decrease in abundance of less sensitive taxa. This is followed by a rapid increase in those taxa able to benefit from the changed conditions and then showing a decline, and a more gradual increase in other taxa as conditions resemble those prior to the incident.
Number of Taxa	Initial low numbers followed by a gradual increase.
Diversity	Initial low diversity followed by a gradual increase.
BMWP and ASPT	Initial low BMWP score and ASPT score as taxa are lost and any taxa are dominated by tolerant taxa, followed by an increase over time.
Multivariate Analysis of Community Similarity	1. Reference sites will show less variability over time than impacted sites (impact). 2. Impacted sites will show a progressively greater similarity to the reference sites over time (temporal recovery). This information can be used to determine a time of recovery within a site. 3. There will be progressively less variability and less movement towards similarity between impacted and control sites, the further downstream of the impact the sampling point is located (Spatial recovery). 4. Progressive increase in similarity within impacted site i.e. within an impacted site those samples closer in time to the pollution incident will show less similarity to each other than those samples further in time. Hypotheses 2-4 assume that the sampling period is long enough for a certain degree of movement of impacted communities towards reference communities to occur.

5.11.5.1. Water Quality

Table 5.11.4 gives the physicochemical measurements taken at the sampling times. Conductivity is generally high at all sites, although is lowest during the last sampling time. The temperature of the water is higher at Site B on the Spark Brook, where it emerges from the culvert.

Table 5.11.4: Physicochemical Conditions in Spark Brook (Conductivity: µS cm-1 at 25°)

		Site A (reference) Cole Manor Farm			Site B - Ackers Entrance			Site C - Climbing Wall		
Date	Days following Disturbance	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond.
21-Jul-94	7	20.5	7.5	995	19.5	7.3	437	20	6.7	473
23-Aug-94	33	17.2	7.6	488	17.7	7.3	354	17.3	7	465
04-Oct-94	75	9.9	7.8	392	12.9	7.5	172	9.2	7.5	240

		Site D - Bridge			Site E - Cole, DS confluence			Site F - Cole, canalside		
Date	Days following Disturbance	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond.	Temp (°C)	pH	Cond.
21-Jul-94	7	19.7	7.5	568	21.4	7.4	480	21.5	7.4	737
23-Aug-94	33	16.9	7.2	461	16.7	7.2	457	16.4	7.3	693
04-Oct-94	75	9	7.5	396	9.1	7.7	278	8.6	7.5	260

5.11.5.2. Selected Taxa

Table 5.11.5 is a summary of the taxa found and their total abundance. 24 taxa were found throughout the study. The six most abundant taxa are listed below and changes in their presence and abundance throughout the study are shown in Fig. 5.11.2 and summarised in Table 5.11.6.

Asellus aquaticus	Oligochaeta
Chironomidae	Physa
Lymnaea peregra	Sphaeriidae

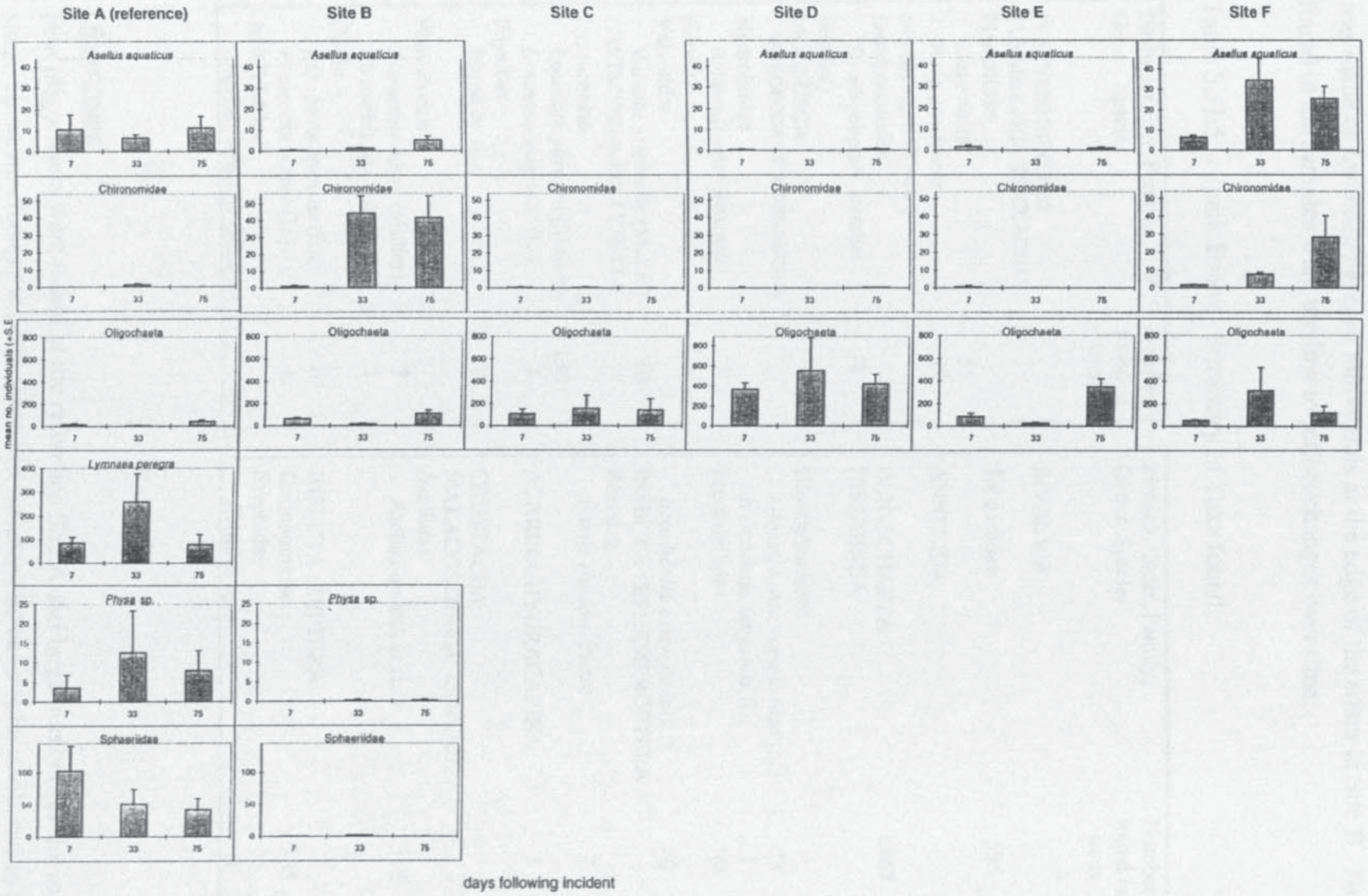
Asellus aquaticus

Numbers of *A.aquaticus* show little variation over time at the reference site (A). Sites B and F show low numbers initially and an increase in numbers over time. The other sites are almost devoid of this species.

Chironomidae

Few Chironomidae were found at the reference site. Most individuals were found at impacted Sites B and F with low numbers at the first sampling visit and a subsequent increase in abundance.

Figure 5.11.2: Spark Brook - Changes in Abundance of Selected Taxa following Pollution Incident



Lymnaea peregra

Most of the individuals of this species were found at Site A(reference) with very few individuals found at the impacted sites. However during the first sampling visit, note was made of *Lymnaea peregra* individuals at the edge of the water at Site B. Numbers found in the samples were too low to reflect changes over time.

Table 5.11.5 - Spark Brook - Summary of Taxa found

Phylum, Order, Family, Genus, Species	Number found in study
PLATYHELMINTHES	
TURBELLARIA:TRICLADIDA	
Planariidae	
Dugesia sp.	51
Polycelis tenuis (Ijima)/*nigra* (Muller)	17
Dendrocoelidae	
Dendrocoelum lacteum (Muller)	8
MOLLUSCA	
GASTROPODA:PROSOBRANCHIA	
Hydrobiidae	
Potamopyrgus jenkinsii (Smith)	5
Valvatidae	
Valvata piscinalis (Muller)	10
GASTROPODA:PULMONATA	
Lymnaeidae	
Lymnaea peregra (Muller)	1307
Lymnaea stagnalis (L.)	2
Physidae	
Physa sp.	75
Planorbidae	
Planorbis albus (Muller)	3
Planorbis carinatus (Muller)	1
Planorbis corneus (L.)	6
Planorbis vortex (L.)	1
ANCYLIDAE	
Ancylus fluviatilis (Muller)	1

Phylum, Order, Family, Genus, Species	Number found in study
BIVALVIA	
Sphaeriidae	595
ANNELIDA	
OLIGOCHAETA	8867
HIRUDINEA	
Glossiphoniidae	
Glossiphonia complanata (L.)	23
Helobdella stagnalis (L.)	1
Erpobdellidae	90
Erpobdella octoculata (L.)	62
INSECTA : EPHEMEROPTERA	
Baetidae	
Baetis rhodani Pictet	2
ACARINA:HYDRACARINA	5
CRUSTACEA: MALACOSTRACA: ISOPODA	
Asellidae	
Asellus aquaticus (L.)	318
INSECTA : DIPTERA	
Chironomidae	385
Syrphidae	
Eristalis sp.	1

Oligochaeta

Few oligochaeta were found at the reference Site A and large numbers at the impacted sites. However changes in abundance are variable and can not be attributed to the pollution impact.

Physa
Most of the individuals of this genus were found at Site A(reference) with very few individuals found at the impacted sites.

Sphaeriidae
Most individuals were found at Site A(reference) with few found at the impacted sites.

5.11.5.2. Univariate Indices

Univariate indices are plotted in Fig. 5.11.3 and changes are descibed below.

Number of Taxa
Site A(reference) shows little variation in the number of taxa over time and impacted Sites B and F show a progressive increase in the number of taxa.

Diversity Index
Site A(reference) shows little variation in the diversity index. Diversity is very low within the Spark Brook. Site B shows an increase in diversity after 33 days.

BMWP and ASPT scores
BMWP and ASPT scores within the impacted sites are very low. Scores at the reference Site A show little variation. Both BMWP and ASPT scores at Sites B and F show increases after 33 days.

5.11.5.3 Multivariate analysis

Similarity matrices were used to construct Multidimensional Scaling configurations (MDS plots) comparing each impacted site with the reference site (Figs 5.11.4a-e)

Sites C and D, when compared with Site A(reference) show no progressive movement over time. Sites B, E and F similarly show no increase in similarity to reference Site A over time, although, when the data is transformed to give less emphasis to the more abundant taxa, there is more variation between times in the communities found at the impacted sites than the reference site, a potential impact predicted in Table 5.11.3. However, the chronic pollution within the Spark Brook is probably greater than in the reference site on the River Cole, and this finding would have been predicted from background pollution levels.

Figure 5.11.3: Spark Brook - Changes in Univariate Measures following Pollution Incident

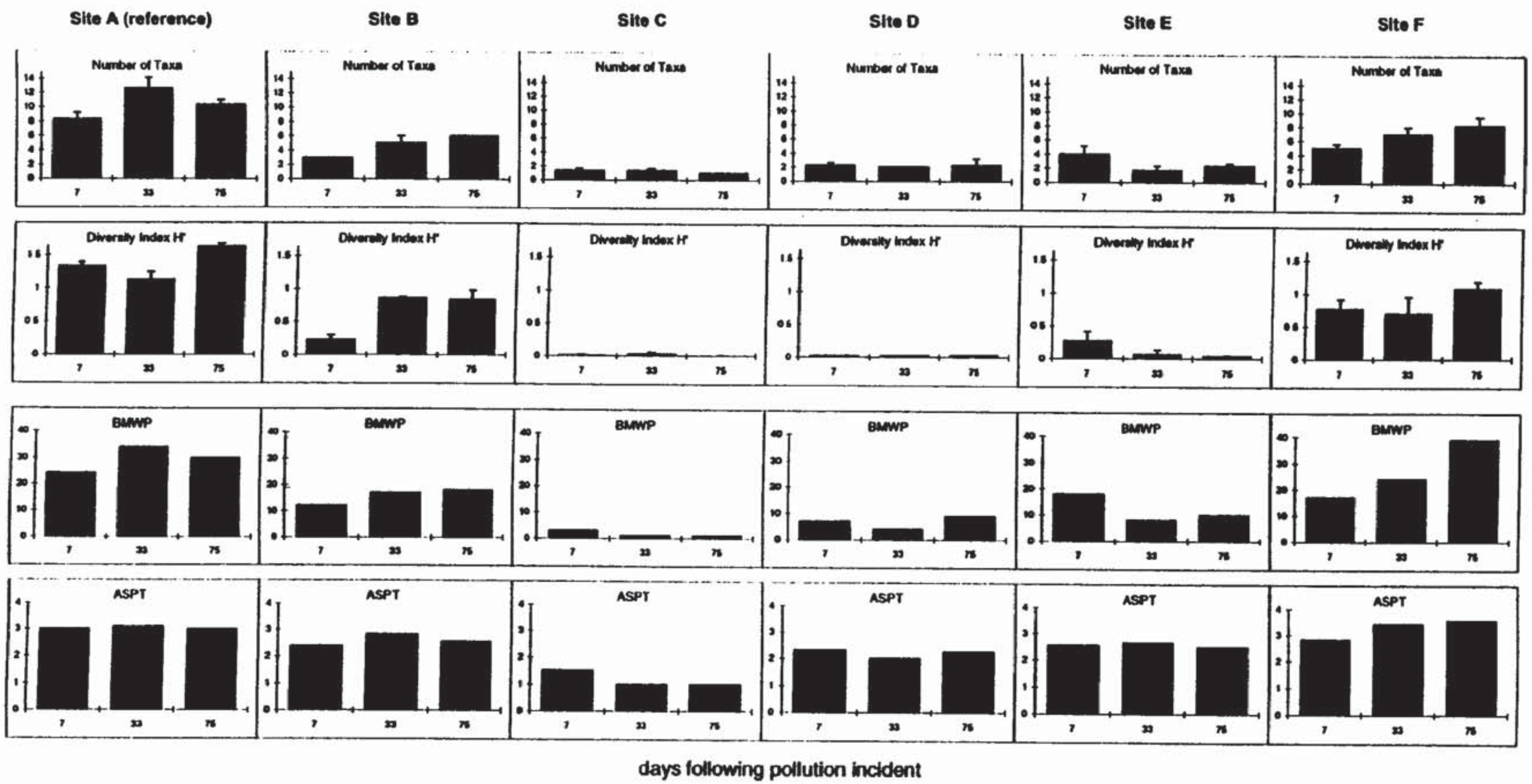
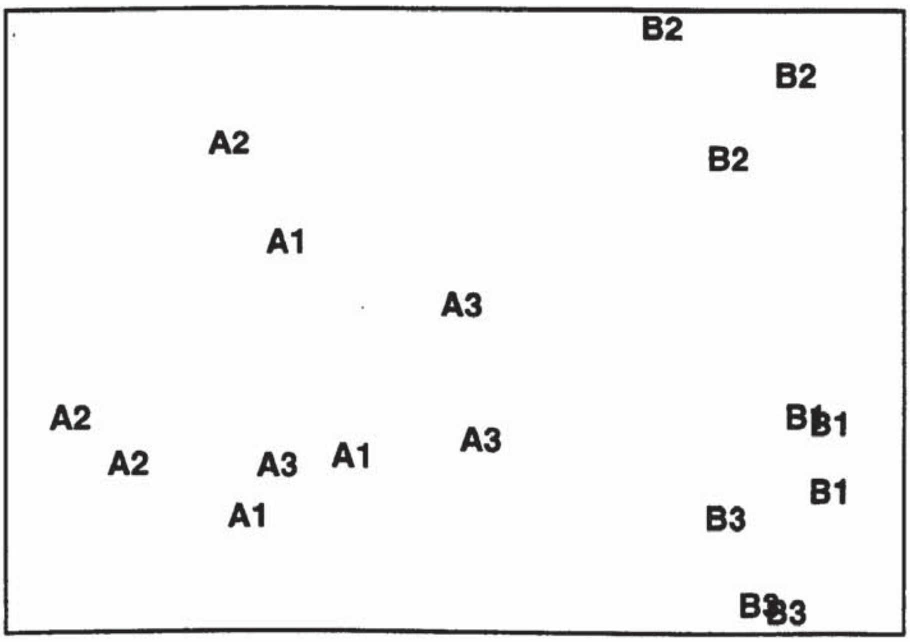


Figure 5.11.4a: Spark Brook - MDS configuration of Site A(reference) and Site B

No transformation. Stress = .09



Root/root transformation. Stress = .07

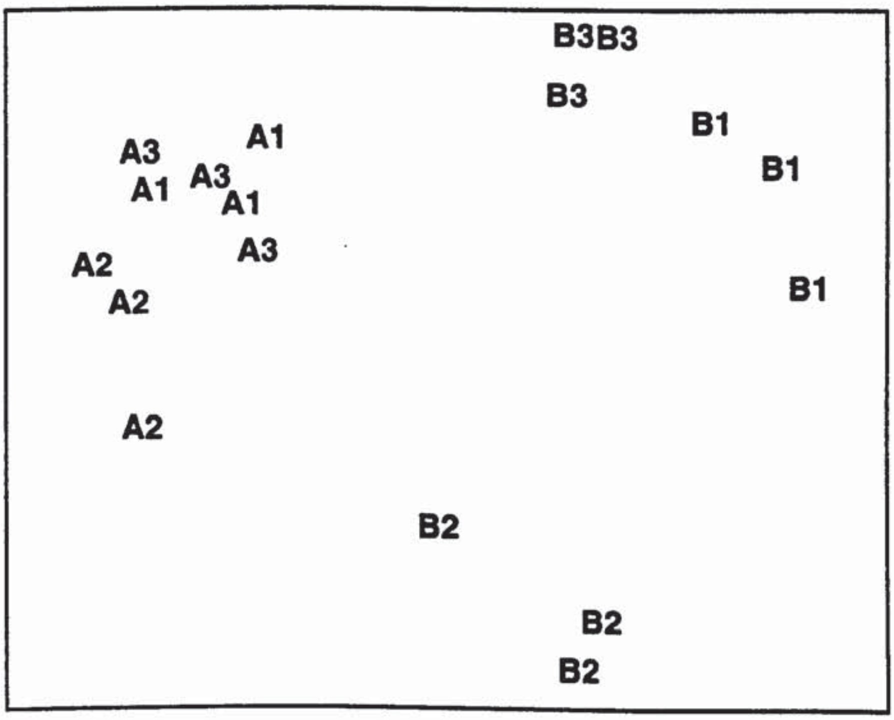
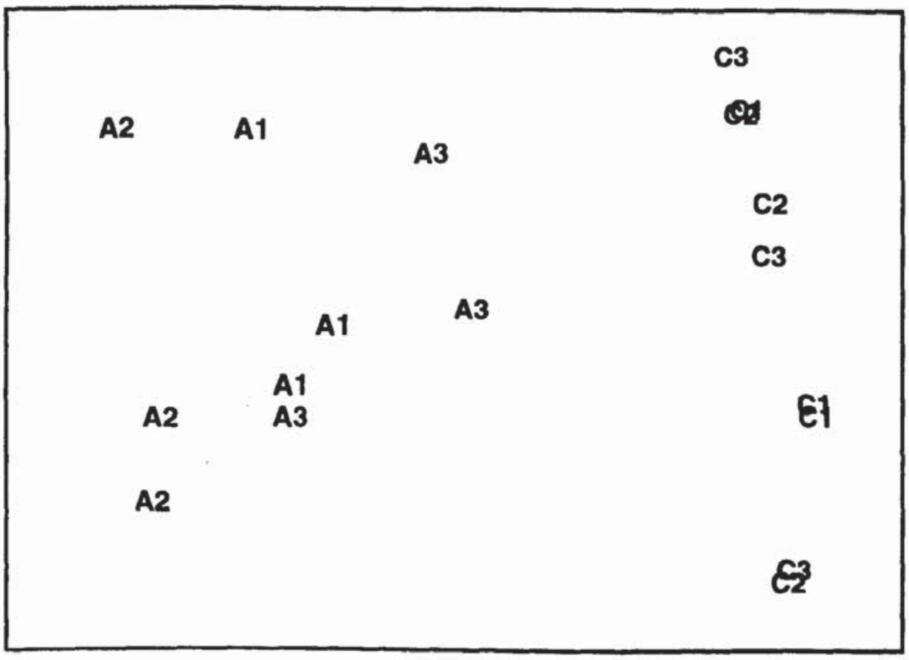


Figure 5.11.4b: Spark Brook - MDS configuration of Site A(reference) and Site C

No transformation. Stress = .06



Root/root transformation. Stress = .01

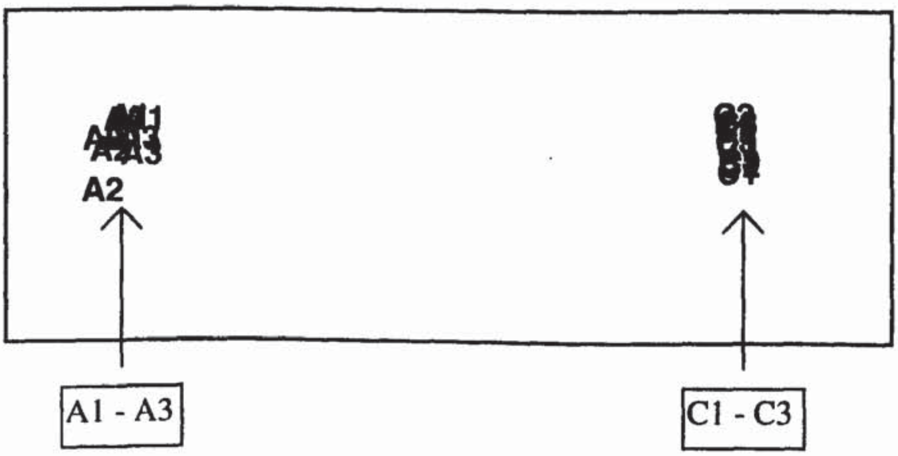
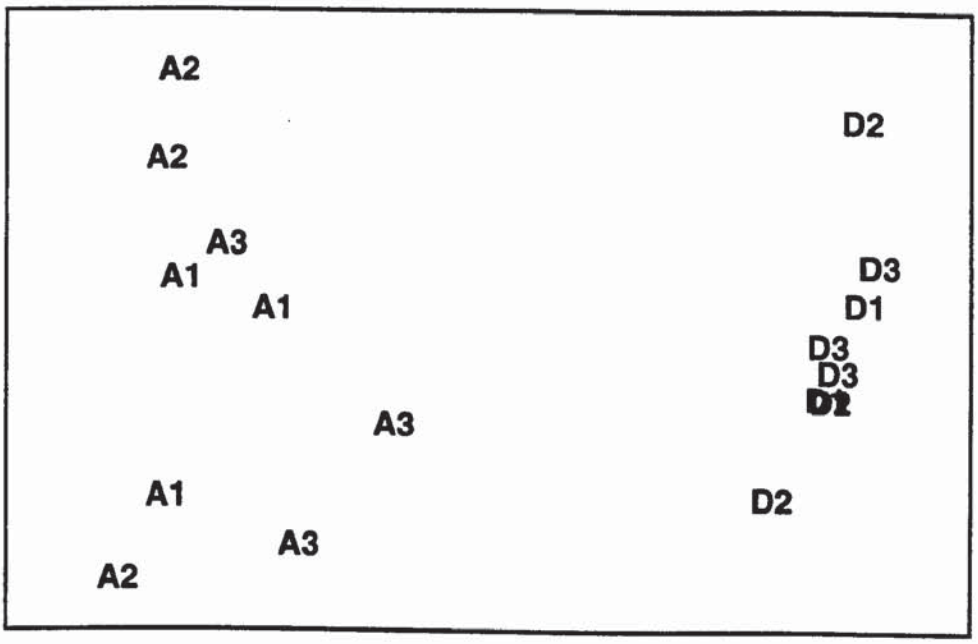


Figure 5.11.4c: Spark Brook - MDS configuration of Site A(reference) and Site D

No transformation. Stress = .04



Root/root transformation. Stress = .01

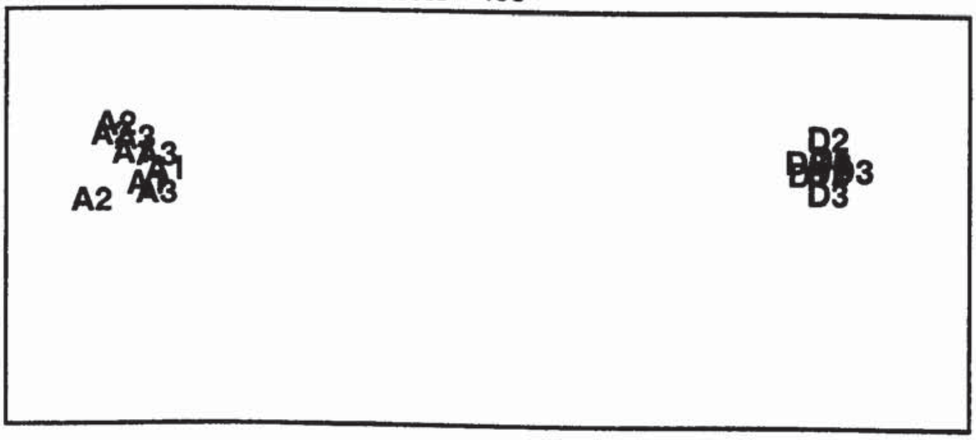
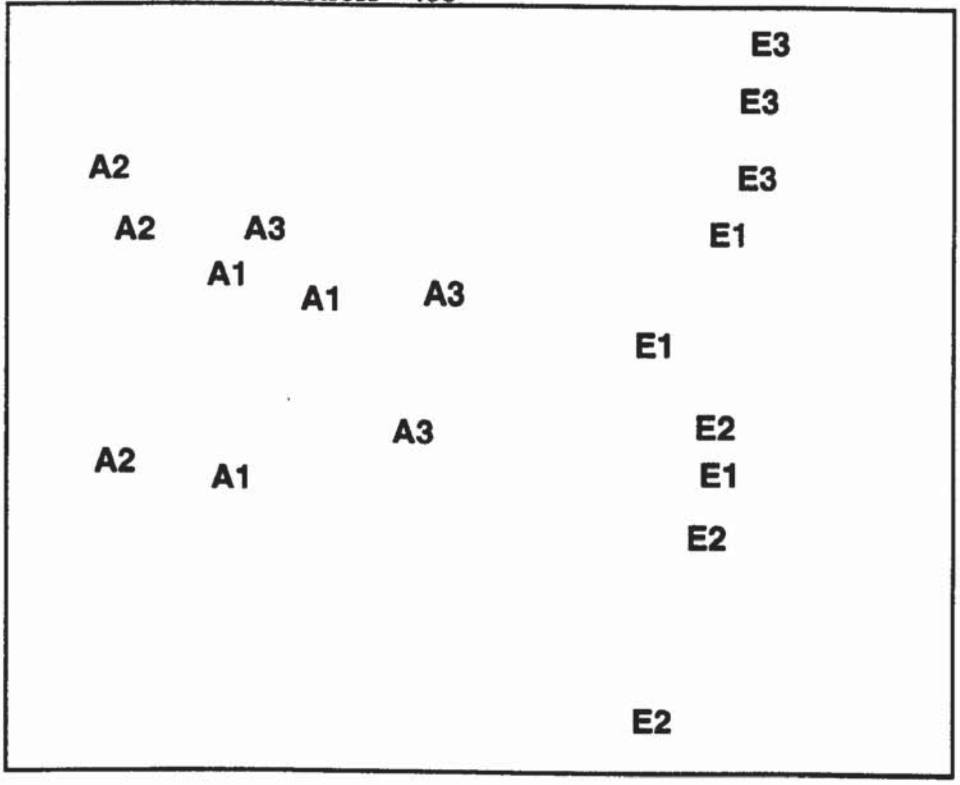


Figure 5.11.4d: Spark Brook - MDS configuration of Site A(reference) and Site E

No transformation. Stress = .08



Root/root transformation. Stress = .05

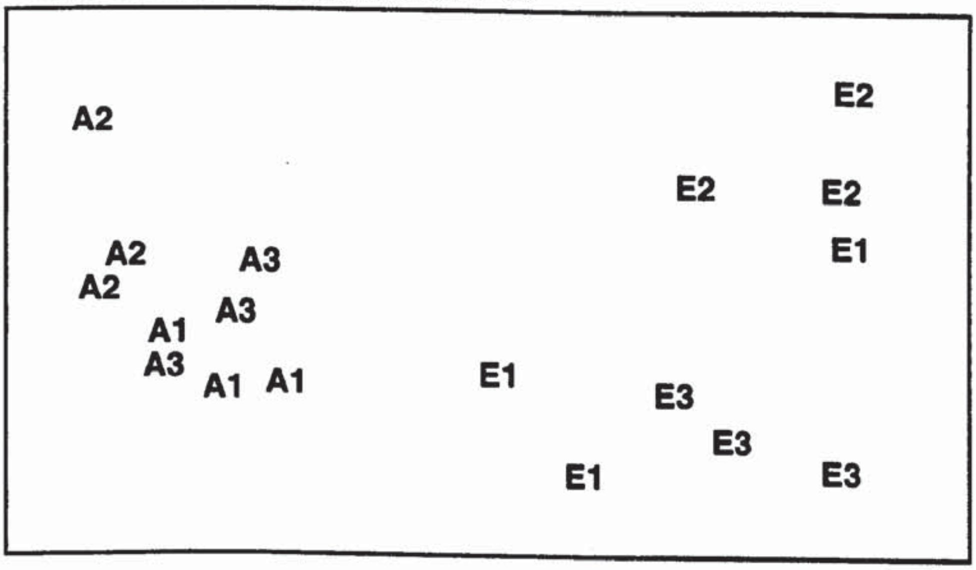
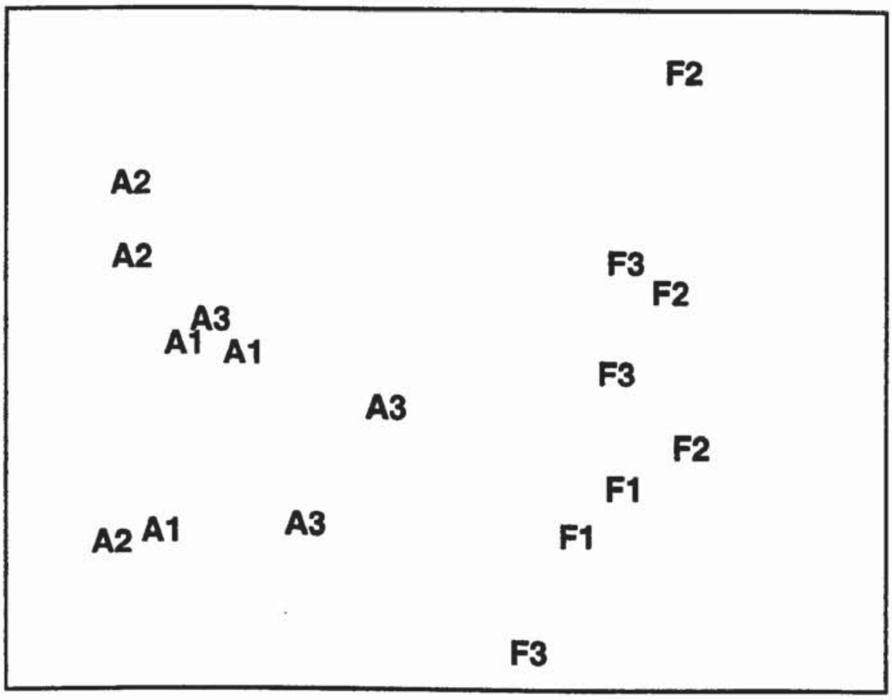
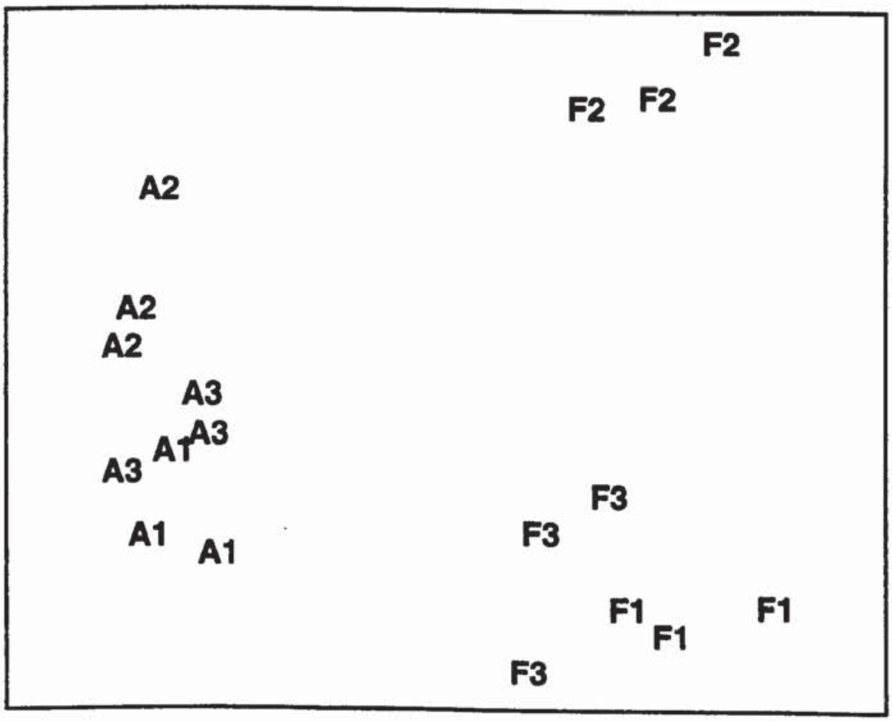


Figure 5.11.4e: Spark Brook - MDS configuration of Site A(reference) and Site F

No transformation. Stress = .10



Root/root transformation. Stress = .08



5.11.6 Summary

Table 5.11.6 summarises the pollution impact on all the biotic measures used.

It is clear from the taxa found, that the Spark Brook itself is grossly polluted, supporting very few taxa. Assessing impact and recovery of macroinvertebrates after the pollution incident is therefore difficult in this case, given the high degree of chronic and intermittent pollution events. In addition, due to the upstream nature of the incident, the reference site used is at an unimpacted downstream location and is dissimilar in terms of environmental parameters to the impacted sites.

However, some information can be gleaned from this study. The number of taxa at two of the sites increased over time and this was connected with an increase in the BMWP scores, ASPT scores and Diversity Index. In addition, the impacted site closest to the outfall (Site B) showed many snails; *Lymnaea peregra* at the edge of the brook, out of the water. Although the abundance of *Lymnaea peregra* within the samples at Site B was low throughout the study, there was a progressive increase in abundance over time. It is possible that the behavioural response of this species has enabled it to survive in conditions of intermittent unfavourable water quality, and a focus on the population dynamics and behavioural responses of the snail in a stream of this quality may provide an indication of changing conditions.

Table 5.11.6: Summary of pollution impact on all measures

Measure	Type of impact	Spatial Impact	Temporal Impact
Initial impact of incident	Dead leeches	Sites B, D	7 days after incident
Individual Taxa	Chironomidae, *Asellus aquaticus* - initial low numbers followed by an increase in abundance	Sites B and F	7 days
Number of Taxa	Initial low numbers followed by an increase	Sites B and F	7, 33 days
Diversity (H')	Initial low diversity	Site B	7 days
Multivariate analysis	Greater variability in community composition over time	Sites B, E and F	75 days

5.12 River Wye

5.12.1 Description of Incident, Remedial Action

On 2/12/94 a plating firm discharged excessive levels of cyanide from their works to the foul sewerage system. The cyanide caused a catastrophic failure in the local sewage treatment works and a discharge of toxic levels of cyanide (0.75 mg/l) and untreated sewage into the River Wye at High Wycombe. This was classified by the NRA as a Category 1 incident. The River Wye was affected to its confluence with the river Thames, a distance of 8 km. 3000 fish were killed and the metal plating firm responsible for the discharge was subsequently fined. (NRA, 1996a).

A biologist from Thames region, NRA collected 3 minute kick samples immediately upstream and downstream of the outfall on the day of the incident. On 6/12/94, 4 days after the incident, further samples were taken, one upstream and three downstream. It was found that, even immediately after the incident was reported, there was no obvious difference in BMWP scores between the upstream and downstream sites. No dead organisms were observed in the field. However when the samples were sorted in the laboratory, it was observed that 50% of the *Gammarus* in the downstream sample had died compared with less than 1% in the upstream sample.

5.12.2 Site Description and History

The River Wye rises from chalk springs to the north west of High Wycombe and flows in a south easterly direction for approximately 17 km, to join the River Thames at Bourne End. Major tributaries of the River Wye include the Hughenden Stream and the Wycombe Marsh Brook. The upper reaches of the River Wye are rural. Further downstream the river enters High Wycombe and is culverted under the town centre before emerging to pass through residential and commercial areas and areas of public open space.

The river supports moderate to poor fish populations of mixed coarse and salmonid species. Of the tributaries, the Hughenden Stream supports excellent brown trout populations and the Wycombe Marsh Brook supports poor fish populations of mixed species (NRA, 1996b).

The General Quality Assessment chemical quality of the River Wye is 'good' (Grade B, 1992-94). However BMWP scores are consistently below scores predicted by the RIVPACS (River Invertebrate Prediction and Classification Scheme) and lower than for other streams in the area (see Table 5.12.1). The River is Biological Class C (BMWP score 51-100) in its upper and lower reaches and Class D (BMWP score 16-50) for the stretch below High Wycombe STW. Regular pollution caused by urban run-off from High Wycombe is considered to be the cause of the relatively poor fauna in this river (NRA,1996b).

Table 5.12.1: BMWP scores and achievement of RIVPACS predictions in 1995 (NRA data)

Location of sampling site	BMWP score	Achievement of *RIVPACS score
Bassetbury Lane	62	no
Above Wycombe Marsh Mill	45	no
At Gauging Station, Hedsor	88	yes

*Achieved if the observed BMWP score exceeds the lower confidence limit of the RIVPACS predicted BMWP score for the site.

5.12.3. Sampling Procedure

Three replicate Surber samples were taken from nine sites on the River Wye (See Fig. 5.12.1). Four reference sites were chosen: two were situated on the River Wye upstream of the pollution incident but downstream of High Wycombe town centre; the two other were located on a channel flowing adjacent to the River Wye (the 'dyke') which joins the River downstream of the pollution outfall. Five sampling sites were located downstream of the pollution incident covering a distance of 4.7 km.

The environmental parameters of the sampling sites are summarised in Table 5.12.2. The majority of the sampling sites were located at riffles, although this was not possible at sites D, E and G which had higher proportions of sand and silt.

Table 5.12.2: River Wye - Summary of environmental parameters at each sampling point.

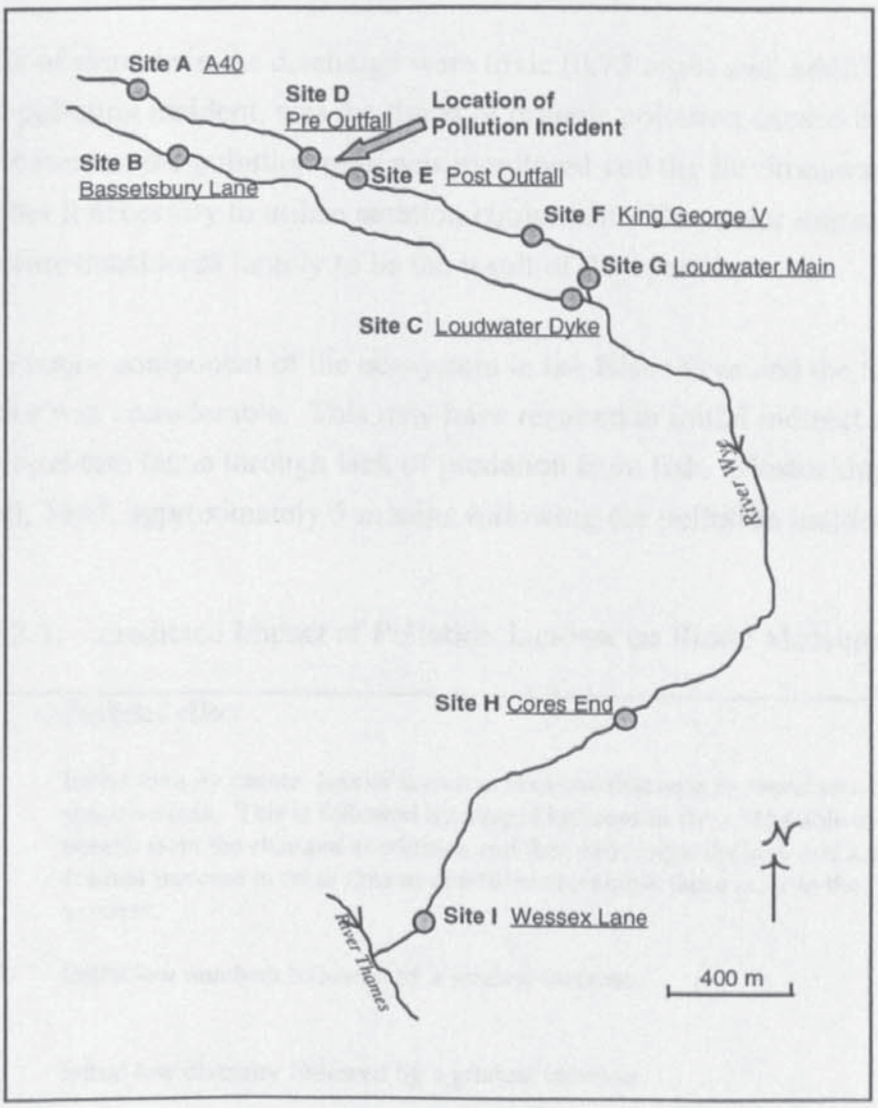
Site Name	Location (G R)	Distance from pollution incident	Substrate	Depth (cm)	Width (cm)	Flow (m/sec) (5/3/96)	Discharge (cu.m/sec) (5/3/96)
Site A (reference) - A40	SU 873926	740m upstream	50% gravel/50%pebble	15-50	400	0.4	0.31
Site B (reference)- Bassettsbury Lane	SU 880922	different channel	70% pebble/20% gravel/10% boulder/rock	5,20	400	0.4	0.11
Site C (reference) - Loudwater Dyke	SU 901905	different channel	50% pebble/40% fine gravel/10% sand	5,20	350	0.63	0.15
Site D (reference)- Pre-outfall	SU 885 921	30m upstream	40% silt/30% sand/30% pebble/debris	15-60	500	0.5	0.45
Site E - Post outfall	SU 887920	30m downstream	40% silt/30% pebble/30% sand	15-60	500	0.75	0.53
Site F - King George	SU 897 922	800m downstream	70% pebble/30% sand	25-40	550	0.43	0.57
Site G - Loudwater Main	SU 902906	1180m downstream	50% sand/30% silt/debris/10% gravel/10%boulder	30-60	600	0.46	0.72
Site H - Cores End	SU 908878	4120m downstream	40% sand/30% pebble/30% gravel	20-50	700	0.71	1.29
Site I - Wessex Lane	SU 896866	4700m downstream	80% cobble/20% pebble	20-50	700	0.71	1.1

Four sampling times were distributed over a period of 459 days following the incident (Table 5.12.3). The first three samples were taken during the four months following the incident. The last sample was taken about one year after the third sample so that sample numbers 3 and 4 are seasonally comparable.

Table 5.12.3: Summary of sampling times

Sampling time Code	Date		Days following incident
Wy1	13 December	1994	11
Wy2	25 January	1995	54
Wy3	6 April	1995	125
Wy4	5 March	1995	459

Figure 5.12.1: River Wye - Map to show location of Pollution Incident and Sampling Points



5.12.4. Predicted Impact of the Pollution Incident

Cyanide is used in iron and steel manufacture, gas production, plating, case hardening, non-ferrous metal production and metal cleaning. The source of the cyanide in this pollution incident was a metal plating firm. When cyanide salts dissociate in aqueous solution, the toxic hydrocyanic acid is formed which is regarded as the primary toxic component. Poisoning by cyanide is effectively by inhibition of oxygen metabolism. Its toxicity is modified by several factors, particularly low oxygen, low pH and high temperature which serve to increase toxicity (Hynes, 1960; Hellawell 1986). The impact of cyanide on the benthic macroinvertebrate fauna is difficult to predict. Hynes (1960) describes an industrial accident in which a large amount of copper cyanide entered the river through a sewage treatment works. The impact of this incident on the fauna was selective with *Asellus*, *Gammarus*, *Baetis* and *Lymnaea* affected to varying degrees and no evidence of impact on *Haliplus* beetles, caddis flies and Chironomidae.

The levels of cyanide in the discharge were toxic (0.75 mg/l) and, additional to the chemical pollution incident, was the threat of organic pollution caused by sewage works failure. However, the pollution plug was monitored and the Environment Agency did not consider it necessary to utilise aeration equipment. The major impact on the river was therefore considered largely to be the result of the cyanide.

Fish are a major component of the ecosystem in the River Wye and the impact on all the fish species was considerable. This may have resulted in initial indirect effects on macroinvertebrate fauna through lack of predation from fish. Restocking occurred on 20th April, 1995, approximately 5 months following the pollution incident.

Table 5.12.4: Predicted Impact of Pollution Incident on Biotic Measures

Measure	Predicted effect
Individual taxa	Initial toxicity causes loss of sensitive taxa and decrease in abundance of less sensitive taxa. This is followed by a rapid increase in those taxa able to benefit from the changed conditions and then showing a decline, and a more gradual increase in other taxa as conditions resemble those prior to the incident.
Number of Taxa	Initial low numbers followed by a gradual increase.
Diversity	Initial low diversity followed by a gradual increase.
BMWP and ASPT	Initial low BMWP score and ASPT score as taxa are lost and any taxa are dominated by tolerant taxa, followed by an increase over time.
Multivariate Analysis of Community Similarity	1. Reference sites will show less variability over time than impacted sites (impact). 2. Impacted sites will show a progressively greater similarity to the reference sites over time (temporal recovery). This information can be used to determine a time of recovery within a site. 3. There will be progressively less variability and less movement towards similarity between impacted and control sites, the further downstream of the impact the sampling point is located (Spatial recovery). 4. Progressive increase in similarity within impacted site i.e. within an impacted site those samples closer in time to the pollution incident will show less similarity to each other than those samples further in time. Hypotheses 2-4 assume that the sampling period is long enough for a certain degree of movement of impacted communities towards reference communities to occur.

5.12.5 Results and Analysis

The data is organised as described in Chapter 4. Some pre-impact information is available in the form of BMWP scores and the families found at NRA monitoring points both up and downstream of High Wycombe STW. With other measures, impact and recovery have to be inferred by comparison with the four reference upstream sites.

5.12.5.1 Water Quality

Table 5.12.5 gives the physicochemical measurements taken at each sampling time. The water is alkaline in nature with sampling sites having a mean pH of 7.8. Conductivity measurements are higher at the impacted Sites E, F, G, H and I than Sites A, B, C and D as a result of discharges from High Wycombe STW (t test, $p<0.01$).

Table 5.12.5: Physicochemical Conditions the River Wye (Conductivity: µS cm-1 at 25^{o})

		Site A A40				Site B Bassettsbury Lane				Site C Loudwater Dyke			
Date	Days following Disturbance	Temp (^{o}C)	pH	DO	Cond.	Temp (^{o}C)	pH	DO	Cond.	Temp (^{o}C)	pH	DO	Cond.
25-Jan-95	54	9.5	7.9	n/a	495	6.4	7.6	n/a	351	7.5	7.8	n/a	401
06-Apr-95	125	11.6	7.9	n/a	445	10.8	7.1	n/a	445	12.2	8.2	n/a	433
05-Mar-96	459	7.6	n/a	15.8	358	6.3	n/a	9.7	375	8	n/a	15.8	425
		Site D Pre Outfall				Site E Post Outfall				Site F King George			
Date	Days following Disturbance	Temp (^{o}C)	pH	DO	Cond.	Temp (^{o}C)	pH	DO	Cond.	Temp (^{o}C)	pH	DO	Cond.
25-Jan-95	54	6.9	7.8	n/a	382	9.1	7.3	n/a	524	8.4	7.5	n/a	476
06-Apr-95	125	12.2	8.2	n/a	423	12.3	7.5	n/a	532	12.5	7.8	n/a	453
05-Mar-96	459	7	n/a	15.3	393	8.5	n/a	10.7	629	8.5	n/a	12.4	582
		Site G Loudwater Main				Site H Cores End				Site I Wessex Lane			
Date	Days following Disturbance	Temp (^{o}C)	pH	DO	Cond.	Temp (^{o}C)	pH	DO	Cond.	Temp (^{o}C)	pH	DO	Cond.
25-Jan-95	54	8.2	7.5	n/a	468	7.7	7.7	n/a	440	6.3	n/a	n/a	501
06-Apr-95	125	12.3	7.9	n/a	476	12.5	8	n/a	460	13.2	8.2	n/a	451
05-Mar-96	459	7.9	n/a	12.3	580	7.5	n/a	13.8	487	7.7	n/a	15	479

5.12.5.2 Selected Taxa

Table 5.12.6 is a summary of the taxa found and their total abundance. 51 taxa were found throughout the study. The most abundant taxa in the study and in the reference sites are listed below and changes in their presence and abundance through the study are shown in Figure 5.12.2 and summarised in Table 5.12.7.

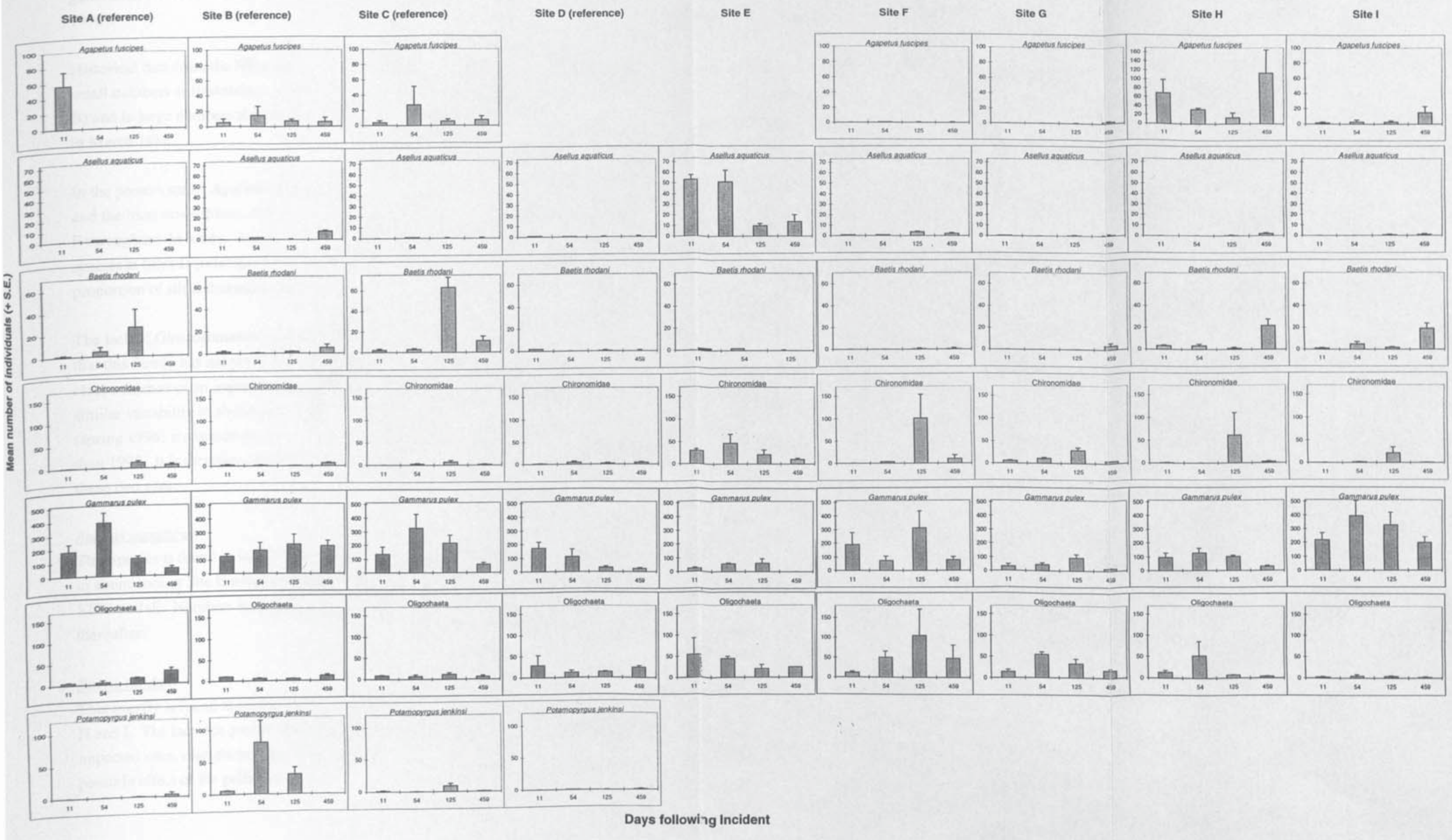
Agapetus fuscipes	*Gammarus pulex*
Asellus aquaticus	Oligochaeta
Baetis rhodani	*Potamopyrgus jenkinsi*
Chironomidae	

Table 5.12.6: River Wye - Summary of Taxa found in study

Phylum, Order, Family, Genus, Species	number found in study
PLATYHELMINTHES	
TURBELLARIA:TRICLADIDA	
Planariidae	
Dugesia sp.	21
Dugesia tigrina (Girard)	1
Polycelis tenuis (Ijima)/*nigra* (Muller)	203
Polycelis felina (Dalyell)	136
Dendrocoelidae	
Dendrocoelum lacteum (Muller)	39
MOLLUSCA	
GASTROPODA:PROSOBRANCHIA	
Valvatidae	
Valvata piscinalis (Muller)	3
Hydrobiidae	
Potamopyrgus jenkinsi (Smith)	398
GASTROPODA:PULMONATA	
Lymneidae	
Lymnaea peregra (Muller)	107
Lymnaea palustris (Muller)	3
Physidae	
Physa sp.	1
Planorbidae	
Anisus vortex (L.)	30
Ancylidae	
Ancylus fluviatilis Muller	250
BIVALVIA	
Sphaeriidae	64
ANNELIDA	
OLIGOCHAETA	2267
HIRUDINEA	
Piscicolidae	
Piscicola geometra (L.)	25
Glossiphoniidae	
Glossiphonia complanata (L.)	29
Glossiphonia heteroclita (L.)	6
Helobdella stagnalis (L.)	3
Erpobdellidae	15
Erpobdella octoculata (L.)	241
ARTHROPODA	
ACARINA:HYDRACARINA	281
CRUSTACEA: MALACOSTRACA: ISOPODA	
Asellidae	
Asellus aquaticus (L.)	458
CRUSTACEA:MALACOSTRACA: AMPHIPODA	
Gammaridae	
Crangonyx pseudogracilis Bousfield	16
Gammarus pulex (L.)	14975

Phylum, Order, Family, Genus, Species	number found in study
INSECTA : EPHEMEROPTERA	
Baetidae	
Baetis rhodani Pictet	542
Leptophlebiidae	
Habrophlebia fusca (Curtis)	
Ephemeridae	
Ephemera danica (Muller)	4
Ephemerellidae	
Ephemerella ignita (Poda)	1
INSECTA : COLEOPTERA	
Elmidae	
Elmis aenea Muller	47
Elmis aenea larva	244
Limnius volckmari (Panzer) larva	6
Riolus sp.	1
INSECTA : ODONATA	
Calopteryx sp.	1
INSECTA : TRICHOPTERA	
Hydropsychidae	
Hydropsyche pellucidula (Curtis)	14
Hydropsyche siltalai Dohler	168
Rhyacophilidae	
Rhyacophila dorsalis (Curtis)	11
Glossosomatidae	
Agapetus fuscipes Curtis	1095
Hydroptilidae	
Hydroptila sp.	1
Limnephilidae	46
Drusus annulatus (Stephens)	38
Halesus radiatus (Curtis)	4
Limnephilus lunatus Curtis	36
Limnephilus rhombicus (L.)	1
Potamophylax latipennis (Curtis)	12
Sericostomatidae	
Sericostoma personatum (Spence)	6
Leptoceridae	3
Athripsodes sp.	11
Mystacides sp.	1
INSECTA : DIPTERA	
Ceratopogonidae	8
Chironomidae	1231
Simuliidae	149
Tipulidae	1
Dipteran larva	2

Figure 5.12.2: River Wye - Changes in abundance of selected taxa following pollution incident



<u>*Agapetus fuscipes*</u>

Agapetus fuscipes is a glossosomatid cased caddis larva. A related genus: *Glossosoma* sp. is described as being found in areas of high oxygen concentration (Friedrich, 1990) and pH values above 5.5 (Fjellheim & Raddum, 1990). There may be more than one generation a year and adults have been collected from spring to autumn (Wallace *et al*, 1990).

Historical data from the NRA show that Glossosomatidae are occasionally found in small numbers at Bassetsbury Lane monitoring point (equivalent to reference study Site B) and in large numbers at King George V monitoring point (equivalent to study Site F) in March 1988.

In the present study, *Agapetus* populations were found at the reference Sites A, B and C, and the most downstream impacted Sites H and I. Only one individual was found at Site F throughout the study. *A. fuscipes* is a rheophilic species and the absence of this species at Sites D (reference), E and G (impacted) would be expected, given the higher proportion of silt and sand at these sites.

The lack of Glossosomatidae at Site F, 469 days after the incident may be an indication that this family has not yet recovered. Further downstream at Sites H and I, there is no clear evidence of an impact on *Agapetus* abundance as upstream reference sites show similar variability in abundance. However, sampling times 3 (spring 1995) and 4 (spring 1996) are seasonally comparable and numbers of *Agapetus* are higher in 1996 than 1995. It is therefore possible that the incident had depressed population numbers at these two sites.

<u>*Asellus aquaticus*</u>

This species is found in very low numbers throughout the sampling area and only occurs in abundance at Site E which is just downstream of the pollution incident and also of the STW outfall. Numbers are high for 54 days following the pollution and decrease thereafter.

<u>*Baetis rhodani*</u>

This mayfly is found at control Sites A and C and the most downstream impacted Sites H and I. The fact that greater abundance is seen 459 days after the incident at the impacted sites, than during the previous season, 125 days after the incident, indicates a possible effect of the pollution.

Chironomidae

Chironomidae are found in low numbers at the four reference sites. Site E, downstream of the outfall shows consistently higher abundance and although Chironomidae were not identified to species, this site supported large numbers of the genus *Chironomus*. Sites F, G, H and I show generally low numbers of Chironomidae, but there is a population increase at 125 days. This population peak may be a consequence of the pollution incident rather than seasonal factors as a similar peak is not seen at the same season one year later.

Gammarus pulex

This species is found in variable numbers at all sites up and downstream of the pollution incident and no evidence of impact could be determined.

Oligochaeta

Oligochaeta are found in low numbers at the upstream control sites and higher numbers downstream of the outfall. Numbers are variable at all the sites and peaks in abundance were seen at 54 and 125 days after the pollution incident at Sites F, G and H.

Potamopyrgus jenkinsi

Few snails of this species were found downstream of the incident.

Table 5.12.7: Summary of impact of pollution incident on selected taxa

Taxon	Impact of pollution (tentative)
Agapetus fuscipes	Absence at Site F for 459 days. Decrease in abundance at Sites H and I for 125 days
Asellus aquaticus	Increase in abundance at Site E for 54 days.
Baetis rhodani	Decrease in abundance at Sites H and I for 125 days
Chironomidae	Increase in abundance at Sites F,G,H and I at 125 days
Gammarus pulex	No evidence of impact
Oligochaeta	Increase in abundance at Sites F, G and H at 54 or 125 days.
Potamopyrgus jenkinsi	No evidence of impact

5.12.5.3. NRA Data

A standard monitoring point of the NRA is at Wycombe Marsh Mill which is close to Site E, just downstream of the pollution outfall. Examination of family data from 1988 -1996 shows no obvious effect of the incident in terms of BMWP and ASPT scores which are variable. The only family to be consistently present in samples, but absent at the time of the incident (samples were taken on the day of the incident) is Baetidae. At another monitoring point, G.S. Hedsor, which is close to Site I just upstream from the Thames/Wye confluence, the families Planariidae and Hydroptilidae are present in samples May1990 -Oct 1994. However, at a standard monitoring time 6 months after the incident, they are absent, and for the following two sampling times covering a period of 23 months after the incident, they are present in low numbers or absent. Similar changes are not seen over the same period at Bassetsbury Lane: another NRA monitoring point situated upstream of High Wycombe STW.

The NRA King George V monitoring point is close to study Site F and the macroinvertebrate families found at both are shown in Table 5.12.8. The samples taken by the NRA are not strictly comparable with the study samples as, although both entail sampling of the substratum for a total of three minutes, the NRA kick sampling method covers a far larger area than 3 Surber samples and is therefore more likely to include rarer taxa. However, comparisons can be made, particularly with the more abundant taxa.

Only one sample is available for this site prior to the pollution incident in March 1988. The clearest difference in the NRA data is that Glossosomatidae and Rhyacophilidae are abundant in 1988 and absent after the pollution incident. The family Rhyacophilidae remains absent according to the study data and only one individual belonging to the family Glossosomatidae is found 125 days following the disturbance. It must be noted that the NRA abundance information for Rhyacophilidae is likely to be an overestimate as this family is not generally found in such large numbers.

5.12.5.4. Univariate Indices

Univariate indices (no. of taxa, Diversity index, BMWP and ASPT scores) are plotted in Figs 5.12.3.

Number of Taxa

The predicted changes are not evident at most of the impacted sites, although Site F shows a lower number of taxa 11 days following the pollution incident than in subsequent samples.

Diversity Index

Site F and I show lower diversity indices following the pollution incident. However similar changes are seen at the upstream Sites A, C and D.

Table 5.12.8: Taxa found at King George V sample site: NRA data and study data

	NRA sample 7/3/88	NRA sample 6/12/94	Study sample 13/11/94	Study sample 25/1/95	Study sample 6/4/95	Study sample 5/3/96
Leptoceridae	*					
Limnephilidae			*	*		*
Rhyacophilidae	*****					
Gammaridae	****	***	***	***	***	***
Haliplidae	*					
Elmidae	*					
Simuliidae	*					
Planariidae	**		*	*	**	*
Dendrocoelidae	*			*		*
Baetidae	**	**			*	*
Ancylidae		**	**	**	**	**
Lymnaeidae	*					
Planorbidae	*	*				
Valvatidae					*	
Sphaeriidae	*	**				*
Glossiphoniidae	**	**				
Erpobdelliodae	**	*	*	*	*	*
Piscicolidae			*			
Asellidae	*	*	*	*	**	*
Oligochaeta	***	**	**	***	***	***
Ceratopogonidae	*					*
Glossosomatidae	*****			*		
Stratiomyidae	*					
Chironomidae		*		**	***	**
Hydracarina				*	*	*

* Logarithmic abundance scale used by the NRA(* 1-9;**10-99;***100-999; ****1000-9999;*****10000+)

BMWP and ASPT

The study data shows no evidence of pollution impact on these two scores in terms of lower scores following the incident and higher scores in subsequent samples.

Analysis of preimpact and postimpact data at the NRA monitoring points show no evidence of impact apart from King George V monitoring point (equivalent to Site F).

Pre impact NRA data for Site F is a BMWP score of 71 and an ASPT score of 4.44. This compares with study data of similar ASPT scores following the pollution incident but much lower BMWP scores ranging from 33-45 for 469 days.

Figure 5.12.3: River Wye - Univariate measures - changes following pollution incident

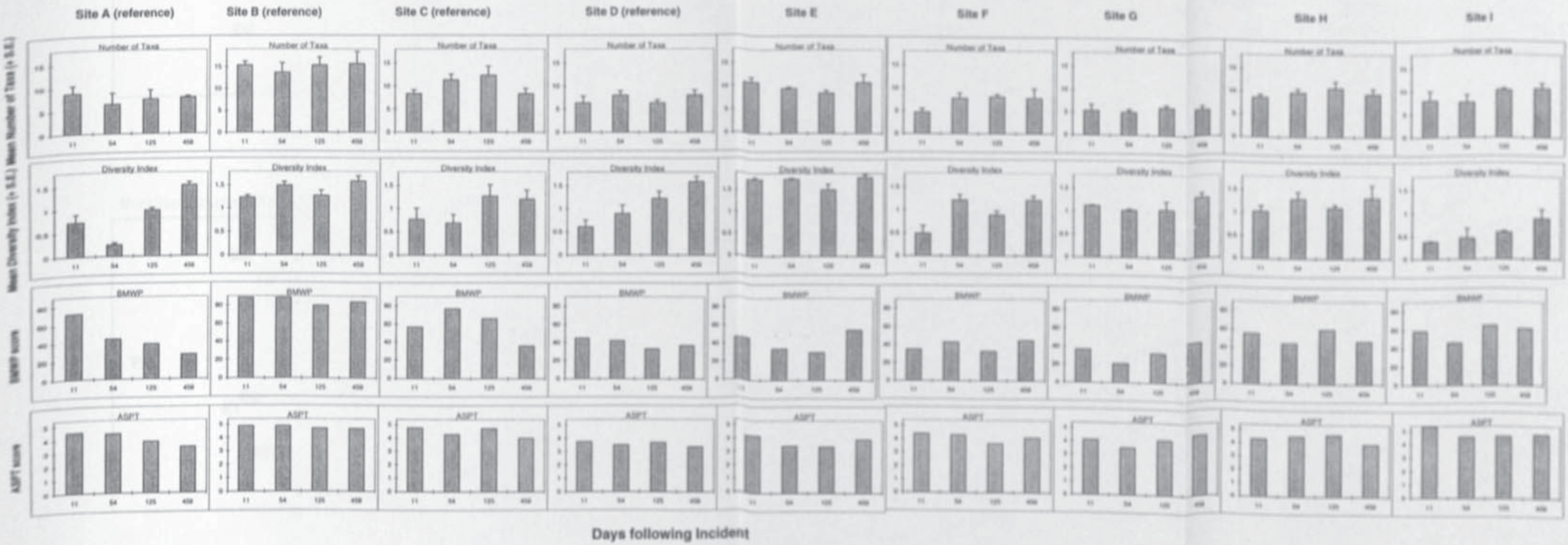
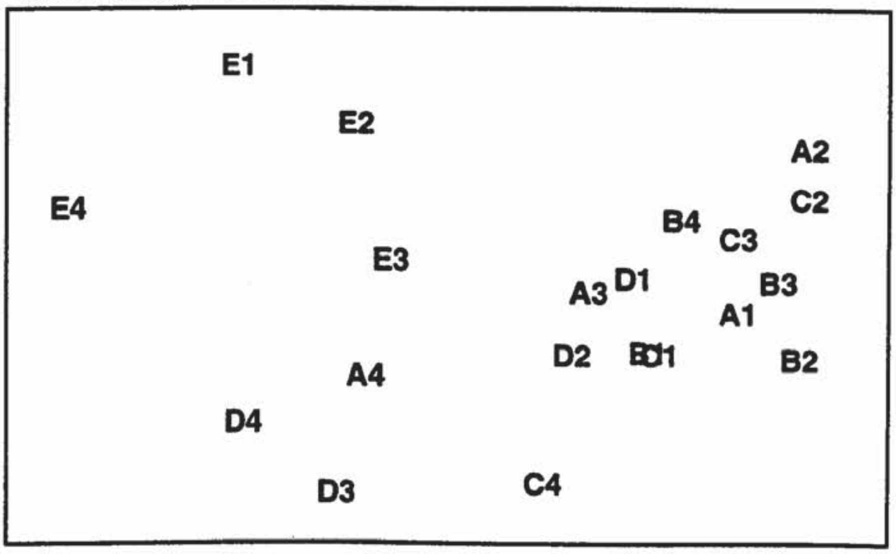


Figure 5.12.4a: River Wye - MDS configuration of Site A-D (reference) and Site E

No transformation. Stress = .07



Root/root transformation. Stress = .15

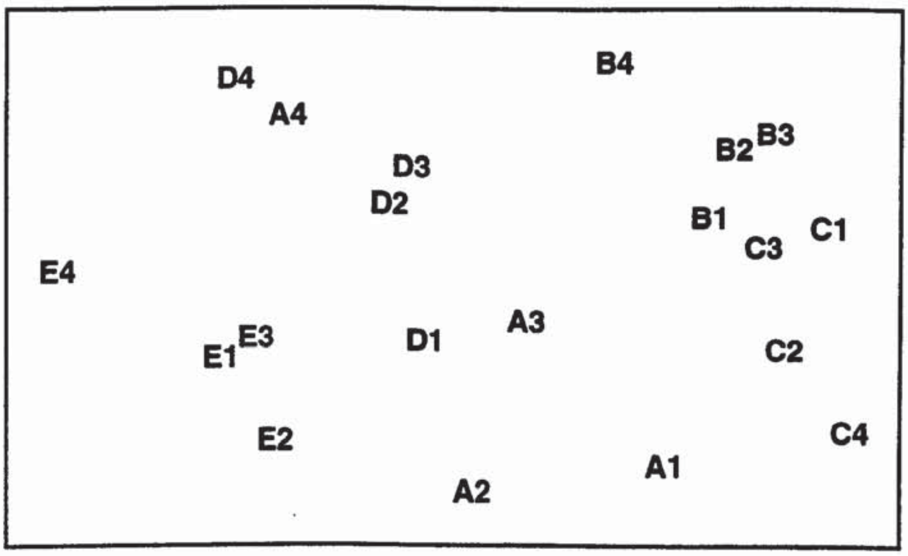
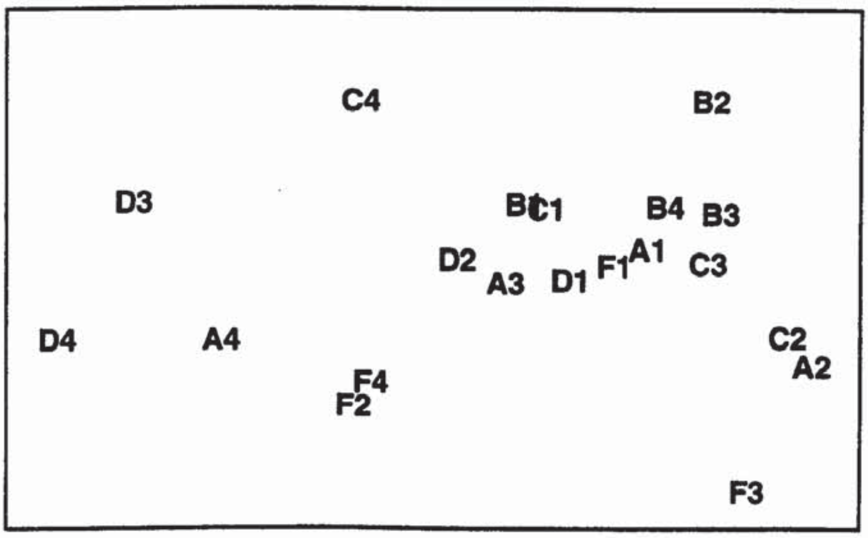


Figure 5.12.4b: River Wye - MDS configuration of Site A-D (reference) and Site F

No transformation. Stress = .07



Root/root transformation. Stress = .18

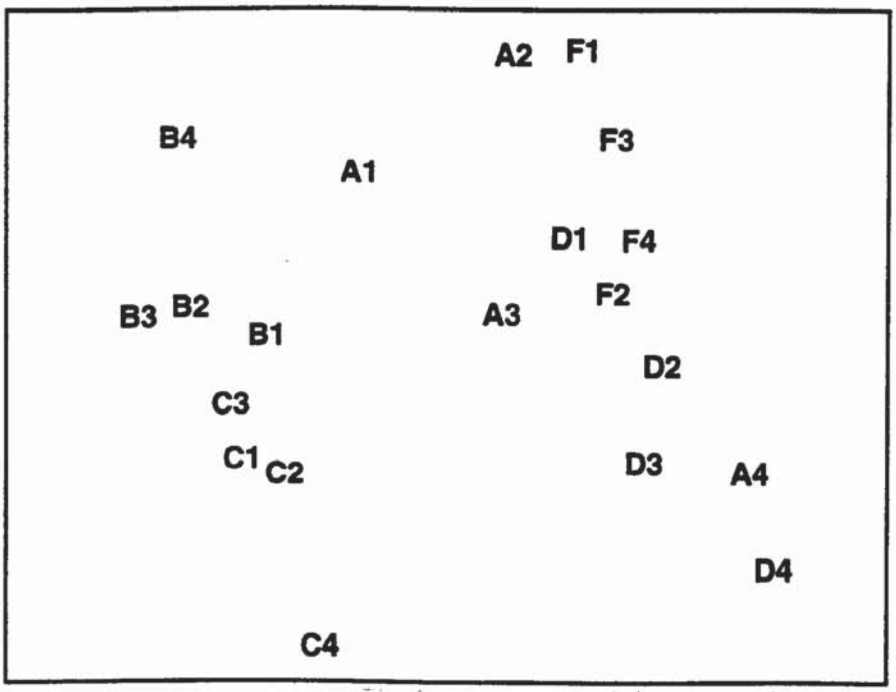
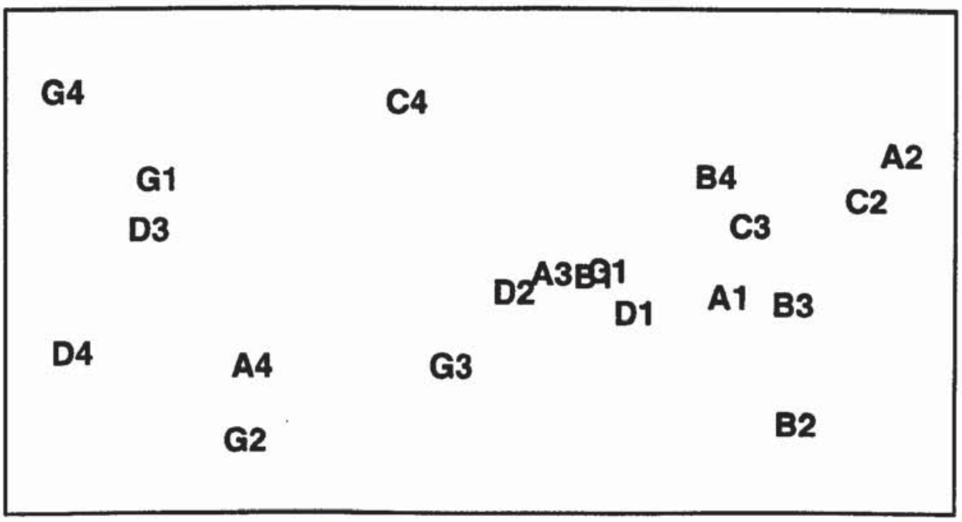


Figure 5.12.4c: River Wye - MDS configuration of Site A-D (reference) and Site G

No transformation. Stress = .07



Root/root transformation. Stress = .18

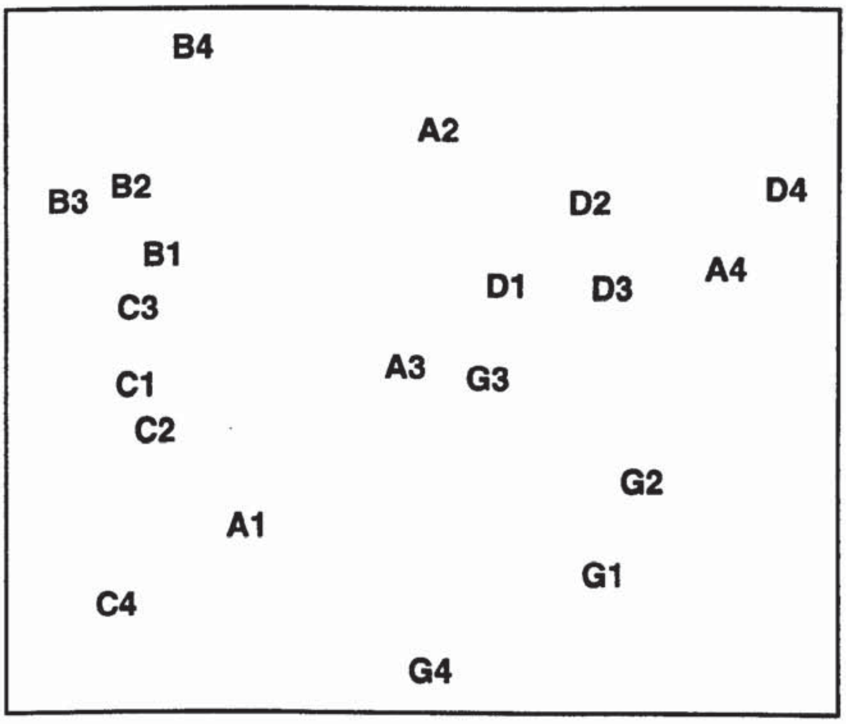
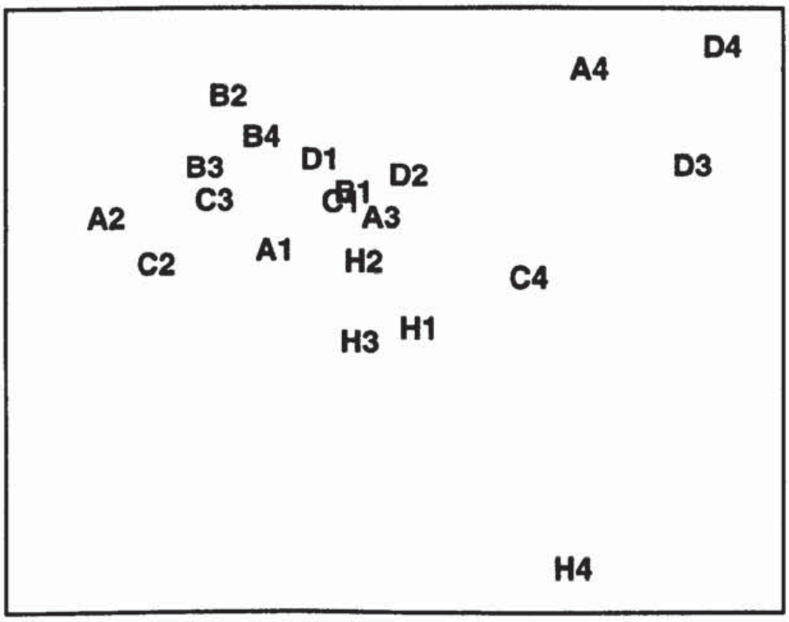


Figure 5.12.4d: River Wye - MDS configuration of Site A-D (reference) and Site H

No transformation. Stress = .10



Root/root transformation. Stress = .19

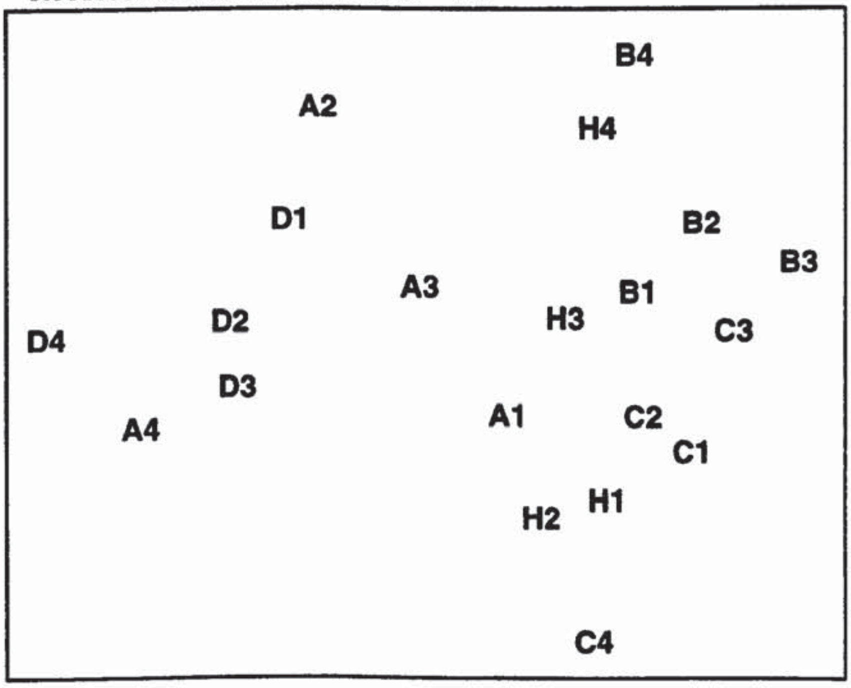
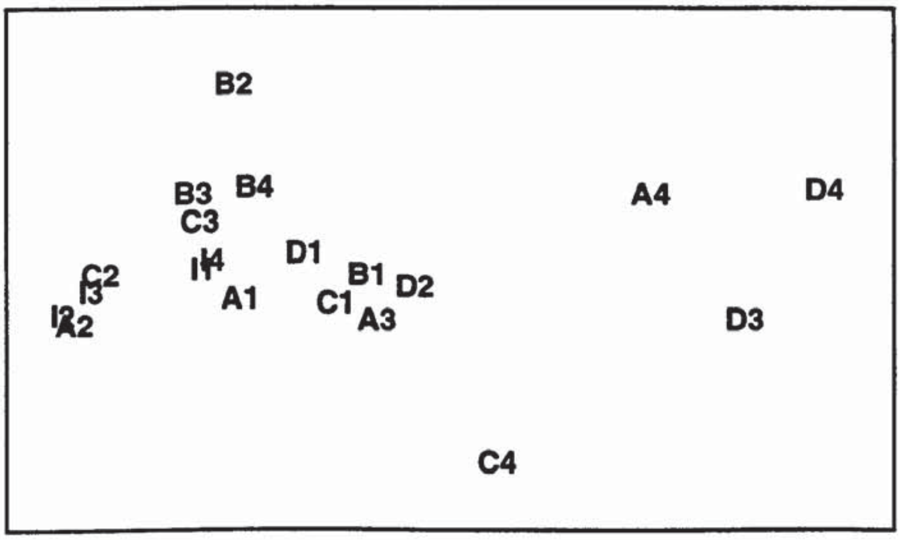
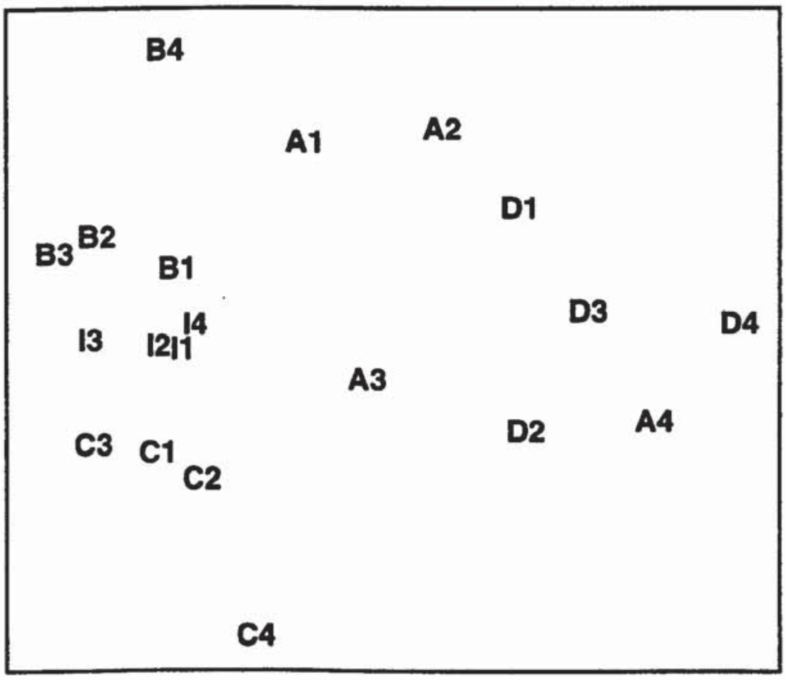


Figure 5.12.4e: River Wye - MDS configuration of Site A-D (reference) and Site I

No transformation. Stress = .06



Root/root transformation. Stress = .18



5.12.5.5. Multivariate Analysis

The R Statistic was computed for each sampling time as described in Chapter 4. This showed that the replicates within sites are more similar to each other than replicates between sites. Similarity matrices were therefore produced from data from averaged replicates. The similarity matrices were then used to construct Multidimensional Scaling configurations (MDS plots) comparing each impacted site with the three upstream control sites using both untransformed data and root/root transformed data (Figs 5.12.4a-e)

None of the hypotheses in Table 5.12.4 were evident in the MDS configurations, thus no effect of the pollution impact was evident.

5.12.6. Summary

Table 5.12.9 summarises the pollution impact on all the biotic measures used.

The River Wye is a largely urban stream with a variety of chronic pollution inputs. In spite of this, the river supports coarse and salmonid fish and, at certain points, a macroinvertebrate fauna characteristic of good quality streams.

The pollution incident had a dramatic effect on the fish. However, the effect on the macroinvertebrate fauna was less obvious and comparison of samples taken immediately up and downstream of the pollution discharge on the same day, revealed no evidence of impact, although *Gammarus pulex* showed evidence of stress downstream of the incident outfall.

Analysis of impact on selected taxa in the study showed 3 patterns; no impact (*Potamopyrgus jenkinsi*, *Gammarus pulex*) low abundance (*Baetis rhodani*, *Agapetus fuscipes*) and an increase in abundance (*Asellus aquaticus*, Chironomidae, Oligochaeta).

Historical data supplied by the NRA indicated a loss of two macroinvertebrate families at Site F. Study data showed no evidence of recovery of these families during the period of study.

Univariate analyses similarly only showed an impact at Site F with low numbers of taxa for 11 days and low BMWP scores throughout the study period.

Multivariate analyses showed no impact of the pollution incident.

Table 5.12.9: Summary of pollution impact on all measures

Measure	Type of impact	Spatial Impact	Temporal Impact
Individual Taxa	*Agapetus fuscipes*- absence	F	>459 days
	A. fuscipes- low abundance	H, I	125 days
	Asellus aquaticus - high abundance	E	54 days
	Baetis rhodani - low abundance	H,I	125 days
	Chironomidae - high abundance	F,G,H,	at 125 days
	Oligochaeta - high abundance	I	at 54 or 125 days
		F,G,H	
NRA family data	Loss of Rhyacophilidae and Glossosomatidae	Site F	>459 days
Number of Taxa	Lower numbers of taxa	Site F	11 days
Diversity (H')	No evidence of pollution impact		
BMWP, ASPT	low BMWP scores	Site F	>459 days
Multivariate analysis	No evidence of pollution impact		

Chapter 6

Pollution incident investigations - Discussion

6.1 Introduction

This chapter brings together all the findings from the investigations into the pollution incidents presented in Sections 5.2-5.12. The first sections, 6.2 and 6.3, discuss the parameters associated with the pollution incidents and the streams that were impacted as well as the methodology used in the assessment of biotic change. In section 6.4 the biotic measures are discussed and their relative utility in the investigation of impact and recovery following pollution incidents is assessed.

Table 6.1 is a summary of the pollution incidents investigated. Both the impacted watercourses and the nature of the pollutant varied between incidents and are considered below.

6.2 Discussion of Watercourse and Pollutant Parameters

6.2.1 Size of Impacted Watercourse

Most of the pollution incidents investigated had impacted first order streams. A large number of incidents are reported by members of the public and, due to the lack of dilution, an incident impacting on a small watercourse is more likely to be visible and indeed the relative impact of an incident on a small watercourse is likely to be greater. The validity of the choice of small watercourses was therefore felt to be high in the investigation of actual pollution incidents, although the findings may also apply to larger watercourses.

Table 6.1: Summary of pollution incidents investigated

Incident	Stream Order	Prior Quality of Stream (NRA - biological water quality)	Pollutant (Type and amount)	Category of Pollution Incident	Immediate impact of pollution incident	Location of Reference Sites
Callow Brook	First Order	poor to moderate	Heating Oil	3	No impact visible	Upstream (2 sites)
Canley Brook	First Order	poor to moderate	Lubricating (Suds) Oil	1	Dead fish and invertebrates	Upstream (1 site), Tributary (2 sites), pre impact data.
Chinn Brook	First order	not known	Unknown toxicant associated with Fluoroscene dye	2	Dead invertebrates	'Tributary' (2 sites)
Hewell Brook	First order	moderate	Kerosene	2	Coating of substratum and emergent vegetation	Upstream (2 sites)
Mousesweet Brook	First order	poor	Hydrochloric Acid	2	Dead fish	'Tributary' (1 site)
Mythe Tributary	First order	unknown	Heating Oil	2	Dead invertebrates	Upstream (1 site) Tributary (1 site)
Preston Bagot Brook	First order	moderate to good	Hydrochloric Acid	1	Dead fish and invertebrates	Upstream (1 site)
River Lostock	Second order	moderate	Pesticide	2	Low biotic scores	Upstream (2 sites)
River Wye	Third order	moderate to good	Cyanide	1	Dead fish	Upstream (1 site) Tributary (1 site)
Sketchley Brook	First order	moderate to poor	Fuel Oil	2	Dead invertebrates	Upstream (2 sites) Tributary (1 site)
Spark Brook	First order	poor	Sewage (domestic and industrial effluent)	2	Dead invertebrates	'Tributary' (1 site)

6.2.2 Water quality of Impacted Watercourse

The water quality of a number of the streams prior to the pollution impact was known from NRA data. Although this information was often general, it gave some indication of the relative biological quality of the streams. Most of the watercourses investigated fell into the moderate to poor range of quality with only two water courses, the River Wye and Preston Bagot Brook, showing moderate to good quality. The study had aimed initially to focus on pollution incidents in streams of relatively good quality that would not be subjected to influences from other anthropogenic activities, such that changes in the biota could be more easily attributable to the pollution incident. However, the majority of incidents appeared to impact streams that were already subjected to other pollution impacts, particularly those associated with urban surface water drainage, thus necessitating the choices made.

6.2.3 Nature of Pollutant

There is wide variation in the type of pollutant impacting the watercourses, with 5 being oil-related, 1 cyanide, 2 acid, 1 sewage, 1 pesticide and 1 unknown toxicant associated with fluoroscene dye. The variety of the pollution incidents investigated provided the study with incidents that were representative of the diversity of actual pollution incidents (EA, 1996). Pollution incidents involving oil predominate within the study reflecting the importance of oil-related incidents reported to the Environment Agency (see Section 2.2.1 and EA, 1996). Although in most of the incidents, the initial nature of the pollutant was known, its subsequent impact on the watercourse was complicated by other substances. For example, in Hewell Brook, the kerosene may have been mixed with other unknown substances on its passage to the brook (Chapter 5.5). Similarly, in Preston Bagot Brook and Mousesweet Brook, the pollutant was known to be hydrochloric acid. The acid caused mobilisation of heavy metals within the sediment to levels that were potentially toxic (Chapters 5.9 and 5.7). The multifactorial nature of pollutant release is typical of actual pollution incidents and reduces the validity of impact prediction based upon laboratory derived toxicological information. Within the present study therefore, predictions of the impact on the biota based on toxicological data were kept general.

6.2.4. Category of pollution incident

The severity of the pollution incident was assessed according to the NRA Categories of Pollution Incidents (Environment Agency, 1996, see Appendix A). These are broad categories used to assess the severity of an incident according to the immediate impact on the ecology and/or the use to which the watercourse is put, as well as remedial measures required. The categorisation of the incidents investigated does not necessarily coincide with the NRA categorisation of the same incident largely due to the assessment of a 'readily observable effect on invertebrate life'. The present study involved a greater length of time assessing obvious impacts on the macroinvertebrates than is available to investigating pollution control officers and was therefore more likely to find an effect on the invertebrate life.

Two of the incidents were described as Category 1 in that there was an extensive fish kill (River Wye and Preston Bagot Brook). The other incidents were described as category 2 in that there was a significant fish kill, a readily observable effect on invertebrate life or the bed of the watercourse was contaminated. Only one incident was in Category 3, a minor incident with no readily observable effect on invertebrate life (Callow Brook).

6.2.5. Immediate impact of pollution

The pollution incidents investigated were chosen because they were seen to be or were considered likely to have an impact on the ecology of the watercourse. One incident investigated as part of the pilot project (Callow Brook) had no readily observable impact on the brook but all other incidents resulted in fish or invertebrate mortality or coating of the substratum and associated vegetation with the pollutant. In one case the incident was located by low biotic scores at a standard monitoring point used by the NRA (River Lostock).

6.3. Discussion of Sampling Parameters

6.3.1. Sampling Methodology

The sampling methodology described in Chapter 3 was used for the majority of the incidents. The pilot studies at Callow Brook and Mousesweet Brook differed in their sampling methodologies as did the study of the River Lostock. At Callow Brook, five rather than three Surber samples were taken at each sampling point and at Mousesweet Brook, a 3-minute kick sample was taken at each sampling point. Although useful in the assessment of variability and rarer taxa, five samples were rejected in favour of three samples due to the logistics of transport and processing. Stratified sampling was used which reduced the variability between samples (see Chapter 3). The single kick sample as used by the NRA, gave no account of the variability within a site thus limiting its utility as a measure of change between sites and between times. The biological information following the incident on the River Lostock was obtained from the Environment Agency (formerly the NRA). They utilised a 3 minute kick sample with field sorting and no replication.

6.3.2 Reference sites

The choice of relevant reference sites is crucial to the understanding of changes at the impacted sites. In most of the incidents studied, the reference sites were situated on the same watercourse as the impact and located upstream of the point of impact. The advantage of this is that environmental conditions are likely to be similar to those of the impacted sites and it also gives some indication of the value of the upstream sites as colonisation sources. This site selection was possible in Callow Brook, River Lostock, River Wye and Sketchley Brook. In some incidents, the upstream sites, although giving some indication of a source of colonists, showed different environmental parameters to the impacted sites, largely because of the upstream location of the pollution incidents. This served as a constraint upon their utility as reference sites (Mythe Tributary, Preston Bagot Brook, Canley Brook). Three of the pollution incidents were situated near the source of the watercourse so that there were no upstream reference sites (Chinn Brook, Spark Brook, Mousesweet Brook). In these cases, a reference site was chosen upstream of the affected tributary's confluence with a larger watercourse which, again was limited in terms of providing an adequate comparison site.

Reference sites may also be subjected to a variety of inputs that are not initially apparent but affect their utility as a comparison or standard against which community change can be compared in the impacted sites. Within Hewell Brook, two reference sites were chosen upstream of the impacted sites. These differed significantly in their community composition with some evidence of a chronic pollution source affecting one of the sites. Thus, the community change at the impacted sites in this situation was compared with the reference site considered most appropriate as a recovery end-point to each impacted site, as well as using a combination of the two reference sites.

To summarise, the choice of suitable reference sites was of crucial importance in the assessment of recovery following a pollution incident but was often difficult to achieve.

6.4 Discussion of Pollution Impact as assessed by the biotic measures

6.4.1 Selected Taxa

The most abundant taxa within each stream investigated were selected for analysis of abundance changes following the pollution incidents (See Chapter 4 for selection criteria). Initial attempts were made to focus on those taxa that were most abundant at the reference sites. At the first site investigated, Hewell Brook, it was found, however, that the most abundant taxa at the reference sites were not the same as those of the stream as a whole, for reasons that may or may not be related to the pollution incident. One of the reference sites, for example, included a high proportion of *Hydropsyche angustipennis* which was rare at the impacted sites and at the other reference site. Conversely, the most abundant taxa when the stream was considered as a whole, included *Asellus aquaticus* and Chironomidae, which were not common at one of the reference sites.

It was decided, therefore, to investigate the five most abundant taxa within each stream and in addition, the five most abundant taxa at the reference sites if these are not the same as the former. In practice it was found that there was a high degree of overlap between these two groups.

Each taxon is considered in the following ways:

1) its response to the incident. (presence/absence, abundance, behaviour, source of colonising individuals),
2) its response to the incident in relation to the response of other dominant taxa within the stream,
3) the relationship between its response and its known response to conditions of pollution from a variety of sources,
4) the relationship between its response and its ecology, i.e. position in food web, feeding mechanism, favoured habitat within stream, life history, behavioural and physiological response to disturbance, mode of colonisation/dispersal.

Table 6.2: Taxa considered for analysis following each pollution incident

Incident	Number of taxa analysed	The most abundant taxa during study	The most abundant taxa at reference sites	Taxa selected for analysis
Callow Brook	6	*Asellus aquaticus* *Baetis rhodani* Chironomidae Oligochaeta *P. jenkinsi*	*Baetis rhodani* Chironomidae *Gammarus pulex* Oligochaeta *P. jenkinsi*	*Asellus aquaticus* *Baetis rhodani* Chironomidae *Gammarus pulex* Oligochaeta *P. jenkinsi*
Canley Brook	6	*Asellus aquaticus* Chironomidae *Gammarus pulex* Oligochaeta *P. jenkinsi*	Sphaeriidae Chironomidae *Gammarus pulex* Oligochaeta *P. jenkinsi*	*Asellus aquaticus* Chironomidae *Gammarus pulex* Oligochaeta *P. jenkinsi* Sphaeriidae
Chinn Brook	5	*Gammarus pulex* Baetidae Oligochaeta *Ancylus fluviatilis* Chironomidae	*Gammarus pulex* Baetidae Oligochaeta *Ancylus fluviatilis* Chironomidae	*Gammarus pulex* Baetidae Oligochaeta *Ancylus fluviatilis* Chironomidae
Hewell Brook	6	Chironomidae *Gammarus pulex* Oligochaeta *P. jenkinsi* Sphaeriidae	*Gammarus pulex* *Hydropsyche siltalai* Oligochaeta *P. jenkinsi* Sphaeriidae	Chironomidae *Gammarus pulex* *Hydropsyche siltalai* Oligochaeta *P. jenkinsi* Sphaeriidae
River Lostock	6	*Asellus aquaticus* *Baetis rhodani* Chironomidae Oligochaeta Sphaeriidae	*Baetis rhodani* Chironomidae *Lymnaea peregra* Oligochaeta Sphaeriidae	*Asellus aquaticus* *Baetis rhodani* Chironomidae *Lymnaea peregra* Oligochaeta Sphaeriidae
Mousesweet Brook	5	*Asellus aquaticus* Chironomidae *Lymnaea peregra* Oligochaeta *P. jenkinsi*	*Asellus aquaticus* *Baetis rhodani* Chironomidae *Lymnaea peregra* Oligochaeta	*Asellus aquaticus* *Baetis rhodani* Chironomidae *Lymnaea peregra* Oligochaeta *P. jenkinsi*
Mythe Tributary	5	*Asellus aquaticus* Chironomidae *C..pseudogracilis* Oligochaeta Sphaeriidae	*Asellus aquaticus* Chironomidae *C. pseudogracilis* Oligochaeta Sphaeriidae	*Asellus aquaticus* Chironomidae *C. pseudogracilis* Oligochaeta Sphaeriidae
Preston Bagot Brook	6	*Hydropsyche siltalai* Chironomidae *Gammarus pulex* Oligochaeta *Potamopyrgus jenkinsi*	*Asellus aquaticus* Chironomidae *Gammarus pulex* Oligochaeta *Potamopyrgus jenkinsi*	*Asellus aquaticus* *Hydropsyche siltalai* Chironomidae *Gammarus pulex* Oligochaeta *Potamopyrgus jenkinsi*
River Wye	7	*Agapetus fuscipes* *Asellus aquaticus* Chironomidae *Gammarus pulex* Oligochaeta	*Gammarus pulex* Oligochaeta *P. jenkinsi* *Baetis rhodani* *Agapetus fuscipes*	*Agapetus fuscipes* *Asellus aquaticus* Chironomidae *Gammarus pulex* Oligochaeta *P. jenkinsi* *Baetis rhodani*
Sketchley Brook	5	*Asellus aquaticus* Chironomidae *Gammarus pulex* Oligochaeta *P. jenkinsi*	*Asellus aquaticus* Chironomidae *Gammarus pulex* Oligochaeta *P. jenkinsi*	*Asellus aquaticus* Chironomidae *Gammarus pulex* Oligochaeta *P. jenkinsi*
Spark Brook	5	*Asellus aquaticus* Chironomidae *Lymnaea peregra* Oligochaeta Sphaeriidae	*Asellus aquaticus* *Lymnaea peregra* Oligochaeta *Physa* sp. Sphaeriidae	*Asellus aquaticus* Chironomidae *Lymnaea peregra* Oligochaeta *Physa* sp. Sphaeriidae

6.4.1.1. *Agapetus fuscipes*

A. fuscipes is a glossosomatid cased caddis larvae commonly found in the stony substratum of flowing waters and lake shores (Wallace *et al,* 1990). This species can form one of the most important components of the macroinvertebrate fauna of streams of high pH. An ecological study of the fauna of a chalk stream in Yorkshire found that within the stony bed, *Agapetus* was a major part of the fauna, forming more than 25% in numerical terms and co-dominant with Oligochaeta and *Gammarus* (Whitehead, 1935).

Glossosomatidae are categorised as 'mineral scrapers', feeding on algae and associated microflora (Cummins and Klug, 1979). The method of feeding involves fixing one end of the case to the stone surface, and grazing algae within reach of the case.

Its mode of existence (a case of stones fixed to the upper surface of substratum) precludes rapid behavioural avoidance of adverse conditions. Although the larva is quick to leave its stone case when disturbed (I.D.Wallace, personal communication, 1997), its movements within the substratum are not extensive and population establishment of a particular area appears to be through oviposition of the adults, the females of which enter the water to lay eggs on the surface of stones.

Agapetus with its fixed case, does not form a major part of the drift fauna. Hynes (1960) found that 7 months following the cessation of gross organic pollution of a stream, the fauna was diverse and varied including *Gammarus*, *Polycentropus* and *Baetis* which had evidently colonised through immigration from tributaries rather than reproduction. The species within the tributaries that had not colonised the main stream were those with good hold-fast mechanisms (flat mayflies, *Agapetus* and limpets), or cryptic habits (stoneflies).

Investigations into the relationship between the colonisation of stones by *Glossosoma* sp. (a related genus) and by periphyton in a creek in Montana, USA, involved the creation of an artificial barrier of jelly around the stones (McAuliffe, 1983). This prevented lateral colonisation by *Glossosoma* but did not prevent colonisation by drift. After 56 days, no colonisation was apparent suggesting that larval colonisation is largely lateral in this genus and not through drift.

There may be more than one generation of *A. fuscipes* a year and adults have been collected from spring to autumn (Wallace *et al*, 1990).

The related genus, *Glossosoma* sp. is described as being found in areas of high oxygen concentration and shows a low tolerance to organic pollution (Friedrich, 1990). Fjellheim and Raddum (1990) in summarising the acid-sensitive invertebrates used in the Norwegian monitoring programme found that the related species, *Glossosoma intermedium* was the most sensitive trichopteran species, being found only at values above pH 5.5 (Fjellheim & Raddum, 1990). This caddis larva is therefore sensitive to chronic conditions of low oxygen and low pH but nothing is known of its sensitivity to intermittent pollution events.

Within the present study *Agapetus fuscipes* was only found in the River Wye (cyanide pollution incident). Historical data from the NRA indicate its numerical importance as a component of the benthic fauna prior to the pollution incident. For the period of study (469 days) following the incident, it was absent within 0.8 km of the pollution outfall. In addition, two sites further downstream (4.1 and 4.7 km from the incident) showed a seasonal reduction in abundance. The only measurable impact on the selected taxa following the cyanide pollution was on these glossosomatid caddis larvae. The major means of colonisation mechanism of this species; aerial colonisation through egg-laying by adults which are relatively weak fliers, may explain the slow recovery of this taxon. After 469 days, numbers of *A.fuscipes* at impacted sites were still not close to numbers prior to the incident or to numbers further downstream. Therefore in spite of no measured impact on other invertebrate taxa following the pollution incident, consideration of this taxon, with its feeding position making it susceptible to changes in water quality, its inability to show behavioural avoidance other than emergence from the case and drift, and its slow recolonisation methods show that full recovery to preimpact conditions may take longer than the time period of the study (1.5 years).

6.4.1.2. *Asellus aquaticus*

A. aquaticus is widely distributed in freshwater in the British Isles, being found in habitats ranging from clear streams to stagnant, polluted ponds (Gledhill *et al*, 1993). Food consists primarily of decomposing plant material and the associated fungi and bacteria. *A. aquaticus* ranges through the substratum as it feeds (personal observation).

In Britain, *A.aquaticus* has two generations each year with life spans ranging from 3-12 months (Gledhill *et al,* 1993). Large females that have over-wintered breed early in the year (February-March) followed by smaller females. The resultant generation breeds at a small size in June-October. Thus breeding occurs during February-October and there are numerous overlapping cohorts with some sexually mature adults present throughout the year.

A. aquaticus is often tolerant of conditions which exclude other crustaceans. In the revised Saprobic system used in Germany (Friedrich, 1990), its saprobic index is 2.7 ('kritisch belastet' to 'stark verschmutzt': extremely damaged to strongly polluted). In Norwegian streams, it is found at pH levels down to pH 5.0 (Fjellheim and Raddum, 1990). *A. aquaticus* is recognised as a characteristic species of the zone of recovery following organic pollution and is tolerant of severe organic pollution (Hellawell, 1986). In streams polluted by organic effluents it frequently replaces *Gammarus pulex* as the dominant detritivore (Hawkes and Davies, 1971).

Little information is available on the movements of *A.aquaticus* within the watercourse. It avoids light and is more likely than *Gammarus pulex*, the other common crustacean in the streams investigated, to be found within the substratum than on the surface (personal observation). This may be related to its poor swimming abilities in comparison with *G.pulex* which can rapidly re-establish its position within the substratum if dislodged by the water current (Townsend and Hildrew, 1976). It is presumed therefore that the most important means of movement within the stream would be by up and downstream movements through the substratum rather than by drift associated with re-attachment to the substratum.

This species has been found to show relatively fast recolonisation following elimination of populations by pollution incidents. For example *A. aquaticus* and Chironomidae were among the first to recolonise a stream bed after an insecticide spill primarily through downstream movement from unaffected tributaries (Raven and George, 1989). Hynes (1974) found a varied fauna in a small, stony stream in southern England within 6 months of cessation of gross sewage pollution which included *A.aquaticus.* Individuals could only have colonised from unaffected tributaries. Another example of recovery from a pollution incident of copper cyanide in the River Lee, showed recolonisation of *A.aquaticus* within a year from up and downstream sources as no tributaries were present, even though this species had been eliminated for 3 miles (Hynes, 1974).

A.aquaticus was among the most abundant taxa in the majority of the pollution incidents investigated in this study and populations showed a range of responses following the incidents. Numbers of *A.aquaticus* were initially reduced at sites downstream of the incident outfall after two incidents studied (Mousesweet Brook and Spark Brook) and after three incidents were absent for a period of time (Hewell Brook, Mythe Tributary and River Lostock). In Canley Brook there was no measured evidence of impact throughout the study and Callow Brook showed no impact initially and then increased abundance close to the outfall and absence at a distance from the outfall. Two incidents showed higher numbers at the first sampling periods following the incident (Preston Bagot Brook and River Wye). At Preston Bagot Brook, the first sampling time was already 48 days following the pollution incident and so the response measured did not occur immediately after the pollution incident. In the River Wye, *A. aquaticus* was only abundant at the sampling point just downstream of the outfall from a sewage treatment works and the increased abundance for 54 days following the incident may have been due to the lack of predation by fish which were removed by the cyanide and are important predators of *A.aquaticus* (Maitland and Campbell, 1992). Reduced predation by fish may also have been a factor in the increase in abundance of *A.aquaticus* following an insecticide pollution (Raven and George, 1989).

Numbers of *A.aquaticus* at Hewell Brook were low immediately after the incident but were similar to that of reference sites 101-150 days following the pollution. As this is in the period January to March, the increase could be the result of recruitment from overwintering females. The initial increase is greatest at the upstream impacted sites indicating that recruitment may have been the result of drift from unimpacted reference sites situated further upstream.

At one incident showing an initial decrease in this species, there is subsequently an increase in abundance which is greater than that expected when compared with reference sites (Hewell Brook). This occurs 394-610 days following the incident and may be the result of increased recruitment resulting from the increased availability of detritus following the breakdown of the oil coating on the substratum. A similar increase is seen at the sampling site closest to the outfall in Callow Brook, 7 days after an oil spill and may have occurred in Preston Bagot Brook. A possible explanation is the lack of competition/predation following a toxic incident enabling those organisms with rapid colonisation characteristics to take advantage of the changed conditions. Another possibility is the ability of *A. aquaticus* to take advantage of changing conditions e.g. a new food source provided by degradation of oil, resulting in a relative increase in numbers.

To summarise, *A. aquaticus* was affected by a range of pollutants in the study resulting in complete absence or reduced abundance. Subsequent colonisation was rapid resulting either from recruitment through reproduction or colonisation from upstream sources. The subsequent increase in numbers was, in some cases, greater than that of reference sites.

6.4.1.3. Baetidae

All the Baetidae collected that were identified to species were found to be *Baetis rhodani* and the following discourse assumes that all the Baetidae found belong to this species.

Baetis rhodani is classified as a scraper and collector-gatherer (Elliott *et al.*, 1988) and feeds primarily on benthic algae (Bohle, 1978). A long period when adults are present and the multivoltine capacity of the life cycle make this one of the most abundant Ephemeroptera species occurring in running water in the British Isles. The time from hatching to emergence can be as little as one month at 14-16^{o}C (Bohle, 1978). Distribution occurs through nocturnal drift and upstream and downstream movements of both larvae and adults (Elliott *et al.*, 1988). Ephemeropteran larvae comprise a major proportion of drifting animals in a wide variety of streams (Bernard *et al.*, 1990; Elliott, 1971; Gammeter and Frutiger, 1989). The drift of mayflies can be affected by a number of factors including food availability (Bohle, 1978), unfavourable water conditions (Coutant, 1964; Wallace and Hynes, 1975; Gammeter and Frutiger, 1989), and increased predation (Peckarsky, 1980). Elliott (1971) suggests that short excursions into the drift are part of the normal behavioural activity of this species and being strong swimmers, they are well able to return rapidly to the substratum.

Ephemeroptera are generally sensitive to organic pollution and their presence and abundance forms a component of many measures of water quality (Hellawell, 1986; Barbour *et al.*, 1995). *Baetis rhodani* differs from other mayfly species in being tolerant of a moderate degree of organic pollution and is a named species in biotic score systems in the United Kingdom, for example; the Trent biotic index and the BMWP score (Woodiwiss, 1964; Chesters, 1980). The response of this genus to a wide variety of pollutant impacts has been investigated; oil (Barton and Wallace, 1979; Crunkilton and Duchrow, 1990) acid (Bernard *et al.*, 1990), ammonia (Gammeter and Frutiger, 1988). In a number of studies monitoring recovery following pollution incidents, Baetidae have been found to be among the first taxa to recolonise affected areas after Chironomidae and oligochaeta (Gore, 1982; Pontasch and Brusven, 1988; Hoiland *et al.*, 1990; Sibley *et al.*, 1991; Turner, 1992).

From the above information, it would be expected that *Baetis rhodani* would be removed or decrease in abundance due to the toxic effects of a pollution incident, but that, given suitable substrate for the growth of benthic algae and a suitable colonisation source, recovery could be rapid.

At Preston Bagot Brook, numbers are low throughout the study except at one of the impacted sites furthest downstream when a peak in abundance is seen 48 days after the pollution incident. At the impacted sites closest to the pollution outfall, abundance of *B.rhodani* is low for 78 - 215 days following the pollution. However, given the lack of an adequate reference site to enable estimates of the reference populations to be made, it is assumed that individuals of *B. rhodani* would be found throughout the year and that peaks in abundance would be apparent in at certain times according to local factors. Thus no impact of the pollution incident can be ascertained in this case.

B.rhodani shows recovery of numbers following the pesticide impact in the River Lostock with a single individual found after 42 days and abundance similar to that of the nearby reference site after 423 days.

Following the cyanide pollution incident on the River Wye, the two impacted sites where this species is present show lower abundance during March, 125 days following the incident than one year later in April. Although this may be due to natural variation in abundance, the numbers during the first season were lower than those of the reference sites at the same time indicating a possible impact of the population.

At Chinn Brook, *B. rhodani* had not colonised the affected tributary at the end of the study, 287 days after the incident. The tributary, not having upstream sources of *B. rhodani* available for colonisation through drift, would therefore be expected to show slower re-establishment through upstream migration and aerial colonisation of adults. However populations were present in the Chinn Brook, within a short distance of the tributary so it may be that this species does not form a standard component of the tributary fauna.

In summary, *Baetis rhodani* has been found to be impacted by a number of the pollution incidents investigated and in spite of predicted rapid recolonisation due to high propensity to drift and multivoltine reproductive capacity, this has not necessarily been the case. In the Chinn Brook *Gammarus pulex* showed more rapid recolonisation of the affected tributary and this may be due to this species' greater propensity to upstream migration (Townsend and Hildrew, 1986), although, as previously mentioned, other factors may have been responsible for the continued absence of this taxon. In the River Lostock, abundance levels of *B.rhodani* did not match upstream reference sites until 423 days following the incident. This finding is longer than would have been predicted given the close proximity of sources available for recolonisation and the known rapid breakdown of the pollutant (cypermethrin) (Crossland, 1982). It is possible that, although the cypermethrin leak had been halted, there was some continued pollution from a sump which was subsequently cleaned out.

6.4.1.4. Chironomidae

Larval Chironomidae often dominate the invertebrate fauna in fresh water in terms of abundance and sometimes biomass (Cranston, 1982; Williams and Feltmate, 1992). They are small particle feeders, grazing from surfaces or filtering from the water and a few are plant feeders or predators. In Britain, the family is represented by over 450 species in 120 genera, with keys available for the identification of the larval forms of most genera and a large proportion of species (Smith, 1989).

There is great variation in life history patterns, some species being uni- or bivoltine and other species, particularly Orthocladiinae, being multivoltine.

Williams and Hynes (1976) showed that in a study of the colonisation methods employed by different taxa, Chironomidae demonstrated colonisation primarily through drift and aerial oviposition by adults although upstream migration and movement from within the substrate was also significant.

Chironomidae are said to be able to survive in habitats characterised by high adversity and low stability (Williams and Feltmate, 1992). Chironomidae are often a dominant part of the macroinvertebrate fauna found in situations polluted by a variety of substances; sewage, oil, heavy metals etc. (Brinkhurst and Kennedy, 1965). In addition, many moderately polluted streams support more species of Chironomidae than unpolluted streams (Lenat, 1983). Chironomidae have been shown to be tolerant to many pollutants and some taxa may show a positive response to oil (e.g. Woodward and Riley, 1983). Rosenberg and Wiens (1976) in a study of 29 Chironomidae species on oiled substrates showed an equal number of species responding positively, negatively or not responding at all.

Morphological, physiological and behavioural mechanisms that aid some species of Chironomidae in withstanding the effects of organic pollution include a higher oxygen storage capacity because of the presence of haemoglobin which enables survival in conditions of low oxygen (Williams and Feltmate, 1992). In addition, Chironomidae have been shown to avoid pollutants and increased water flow by burrowing into the sediment, thus utilising the hyporheic zone as refugial space (Wentsel *et al.*, 1977; Palmer *et al.*, 1992).

The wide range of tolerance found among the different species of Chironomidae is an indication of their value for biomonitoring purposes (Wilson, 1987; 1994). However few monitoring programmes identify the Chironomidae to species level. The known increase in the relative proportion of Chironomidae in the macroinvertebrate community in situations of greater pollution has been used as a metric in the Rapid Assessment Protocol developed in the USA (Plafkin *et al.*, 1989). This metric comprises the ratio of EPT (Ephemeroptera/ Plecoptera/ Trichoptera) abundance to Chironomidae abundance.

The survival and increase in abundance of Chironomidae after a variety of anthropogenic disturbances is well documented ((Ladle *et al.*, 1980; Milner, 1994; Smith and Distler, 1981; Yasuno *et al.*, 1982, 1985; Victor and Ogbeibu, 1986; see review by Hellawell, 1986). This may be a consequence of the fact that the Chironomidae comprise a large number of species adapted to a variety of environmental conditions such that most aquatic habitats are likely to support at least one species of this family. Although the majority of pollution studies do not identify Chironomidae beyond family level, exceptions to this give some indication of the underlying changes within the group following impact. For example, after a simulated pollution incident of the pesticide Temephos on a tributary of the Yamaguchi River in Japan, there was a great increase in the abundance of Chironomidae. This was found to be due to the following taxa; *Orthocladius* spp. (probably *O.kanii* and *O.glabripennies*) and *Diamesa* sp. These genera were not, however, found at the reference sites (Yasuno *et al.*, 1982).

After Oligochaeta, Chironomidae larvae have been the first to colonise substrates polluted by oil (Harrel, 1985; Woodward and Riley, 1983; Siron, 1981; Pontasch and Brusven, 1988). This is possibly due to a tolerance of the chemical nature of the degrading oil and high dispersal abilities enabling rapid exploitation of a potential food source provided by stimulation of algal growth (Siron, 1981; Lock *et al*, 1981). The reduced density of both invertebrate competitors (e.g. Ephemeroptera, Trichoptera, Gastropoda) and predators (e.g. Plecoptera and fish) may also be a factor in the creation of conditions for population increases following pollution incidents.

Following some of the pollution incidents investigated within the present study, Chironomidae were seen to show changes in abundance which were similar to changes at the reference sites and could not therefore be attributed to the pollution incident (Canley Brook, Hewell Brook, Preston Bagot Brook). For example, the pollution incident in Hewell Brook apparently occurred during a period of seasonally low abundance for this family (late autumn/winter), and numbers increased in the summer following the incident at both the upstream reference sites and the impacted sites.

However a number of the pollution incidents investigated showed evidence of a peak in abundance of Chironomidae 3-125 days after the impact which was not apparent at the reference sites or at the same sites during the same season one year later (Chinn Brook, Mousesweet Brook, River Wye, Sketchley Brook, Spark Brook). In the Mythe Tributary, following oil pollution, numbers of Chironomidae increased from scattered individuals during the year following the pollution incident to high abundance after one year, a change not reflected in the reference sites.

As different species of Chironomidae tolerate different environmental conditions, a more defined response to the pollution incidents may have been apparent if the Chironomidae had been identified to genus or species level. For instance, Yoder and Rankin (1995), found that identification of Chironomidae to genus as part of the multiple metric approach to biological assessment, led to the discovery that the use of the metric '% composition of the genus *Cricotopus* spp.' was the best differentiator of complex organic and toxic wastes from other impacts on two rivers in Ohio, USA.

However, despite numerous studies indicating the value of this approach in water monitoring, (Cranston, 1982; Wilson, 1987; Rosenberg and Snow, 1975; Lenat, 1983), generally this has not been adopted by monitoring agencies, presumably because of the greater time required for identification with correspondingly greater cost implications.

The present study has shown that, in common with other studies following pollution incidents, Chironomidae can show increased abundance followed by a decrease and this may be a feature of the 'pioneer' stage of recovery (Heckman, 1983). The reasons for the increase in abundance seen following a number of the pollution incidents are probably due to a combination of factors e.g. increase in food source, tolerance of, or avoidance of toxicant, lack of predators or competitors, but at the level of the present investigation the relative contribution of each can not be determined.

In summary, the greatest value of Chironomidae in assessment of pollution incident impact and recovery may rely on the identification to species. Although this was not carried out in the present study, it can be seen that temporary abundance of Chironomidae following some of the pollution incidents, is an important finding which has implications for the use of reference sites and ascertaining patterns of recovery.

6.4.1.5. *Crangonyx pseudogracilis*

C. pseudogracilis are small freshwater shrimps originating in North America but now well established in ponds, lakes, canals and slow-flowing part of rivers throughout much of England (Sutcliffe and Carrick, 1985). This species can reproduce throughout the year, with spring and early summer being the peak breeding time. Its mode of colonisation is probably similar to that of the amphipod *Gammarus pulex*, comprising movements through the substrate with occasional excursions into the drift.

C. pseudogracilis only formed a major part of the benthic fauna at one of the watercourses investigated; Mythe Tributary. It was absent in the impacted tributary for 187 days following the incident and was found after 369 days but at an abundance lower than in the reference tributary. Its response to the oil pollution matched that of *A.aquaticus.*

6.4.1.6. *Gammarus pulex*

The amphipod *Gammarus pulex* occurs in a wide variety of lotic and lentic freshwater habitats in Britain. *G. pulex* is omnivorous but can be classified as a shredder; the food of preference being coarse particulate organic matter (CPOM) which has been colonised by microorganisms (Cummins and Klug 1979).

Breeding occurs throughout the year with a peak of activity in Spring and early Summer. Females may produce more than 6 broods and in mid summer, 3 generations may be breeding at the same time (Gledhill *et al*, 1993). Individuals commonly drift downstream at night and also move upstream (McCahon *et al*, 1991). Individuals are often abundant in drift but are also active within the water column, drifting for short distances before attaching to substrate (Elliott, 1971). In the study of a stream in the Lake District in England, Elliott (1971) found that along with *B.rhodani*, *G.pulex* was found to be faster than other taxa at returning to the substrate from the drift. *G. pulex* is active within the substrate and can show considerable upstream migration (Soderstrom, 1987). Williams and Williams (1993) found that, at certain times of the year, the numbers migrating upstream approached or exceeded the numbers drifting downstream. Williams and Hynes (1976) investigating the relative recolonisation methods of invertebrate taxa in a Canadian stream, found that Amphipoda only showed evidence of upstream migration through the substratum, although numbers were small. Turner (1992) however, found colonisation through drift to be the most important method, followed by within-substrate and then upstream movements.

Gammarus pulex is sensitive to the deoxygenation that is often associated with organic pollution. *Gammarus* is killed within five hours at oxygen concentrations in the water of 1 mg/litre (Hawkes and Davies, 1971). A major factor affecting its distribution in rivers is the number of hours during the night that the oxygen levels fall below this concentration (Hawkes and Davies, 1971). Higher levels of oxygen (e.g. 7.5 mg/litre at $15^{o}C$), although not low enough to result in mortality, may cause physiological stress (Maltby *et al*, 1990).

G.pulex is relatively simple to keep and rear in laboratory conditions, leading to its widespread use as a test organism in ecotoxicology (McCahon and Pascoe, 1988a; 1988b; Taylor *et al.*, 1991). The sublethal responses of *G.pulex* to a variety of toxicants has been investigated e.g. the 'scope for growth' method (Maltby *et al*, 1990; 1992), behavioural changes (Costa, 1966; Borlakoglu and Kickuth, 1990), changes in feeding rate (McCahon *et al.*, 1989), disruption of precopulatory behaviour (McCahon *et al.*, 1991; Turner, 1992).

G.pulex has been shown to drift in response to adverse water conditions, for example Mitchell *et al.*, (1993) found an increase in drift during experimental addition of the pesticide Lindane. Increased drift has also been found following a simulated pollution incident of reduction of oxygen (Turner, 1992). This consisted largely of live drift such that organisms were subsequently able to resume normal behaviour within the substratum.

McCauley (1966) found that, downstream of an oil pollution, *Gammarus* populations were absent for at least two years. Following the installation of treatment works for industrial and sewage effluent on the River Derwent, *G.pulex* was among the first taxa to recolonise along with Chironomidae and *Asellus aquaticus* (Brinkhurst, 1965). Similar findings were obtained 7 months following improved sewage disposal facilities in a stream in Southern England (Hynes, 1960). In this case, colonisation was apparently through drift from unaffected tributaries and it was suggested that *G.pulex* was one of the early colonisers being more mobile and easily swept away by the current than species with cryptic habits and good holdfast mechanisms e.g. Stoneflies, *Agapetus* and *Ancylastrum*.

Following a spill of the pesticide chlorpyrifos in the River Roding in Southern England, *Gammarus pulex* did not return to sites on the affected reach 0.35-18.6km downstream of the spill, until 48-93 weeks following the incident. This recolonisation was slower than that of *Asellus aquaticus* and Chironomidae but more rapid than *Caenis moesta* and *Oulimnius tuberculatus* and appeared to be related to the location of unimpacted tributaries (Raven and George, 1989).

Numbers of *G. pulex* at the impacted sites at Hewell Brook: sites C, D and E are low for 53 days following the incident and then increase in numbers; an increase which is maintained apart from a decrease in July, 1995. This decrease occurred throughout the stream and may have been related to seasonal factors (high temperature conditions resulting in low oxygen levels).

Within the Chinn Brook study, *G. pulex* started to colonise the impacted tributary 139 days after the incident when it was found in low numbers with an increase in numbers seen during subsequent sampling visits. Colonisation could only have been from downstream sources and the distance upstream for the colonisers to move was only 200 metres.

At Canley Brook, no impact of pollution on the abundance of *G. pulex* was found although dead individuals were seen following the incident.

The situation in Sketchley Brook is complex. Dead individuals were seen following the pollution incident and *G.pulex* was absent at the most impacted site 50m downstream of the outfall 3 days after the incident. However at the next sampling point, 300 m downstream of the outfall, *G. pulex* were higher than one year later; this may simply be the result of natural variation or may have been the result of behavioural avoidance by *G.pulex* leading to drift and subsequent abundance increases downstream.

Within the Mythe tributary, *G. pulex* is a consistent component of the fauna at the reference site, an adjacent tributary, but had not colonised the impacted tributary during the period of the study, 369 days following the incident. There were no upstream sources of this species, and numbers within the Mythe Brook itself were not high, possibly due to fish predation. Colonisation by *G.pulex* in this situation would be expected to be slower than by taxa that have aerial means of dispersal.

G.pulex is shown to have been affected by a number of the pollution incidents studied showing initial absence or a decrease in abundance. This is followed by a steady increase in abundance which is predictable in nature and timescale, given the relevant sources of recolonisation.

6.4.1.7. *Lymnaea peregra*

The snail *Lymnaea peregra* is the commonest and most widespread freshwater snail in Britain and probably the commonest in Europe, tolerating a wide range of ecological conditions (Macan, 1977; Fitter and Manuel, 1986). It belongs to the subclass Pulmonata; snails which breathe atmospheric air from an internal 'lung'. Snails are scrapers, having adaptations for grazing upon substances, particularly periphyton, that adhere to surfaces (Cummins and Klug, 1979).

In temperate regions, Gastropoda tend to have annual life cycles often with a number of overlapping generations (Harman, 1974). *Lymnaea peregra*, being a pulmonate snail and able to utilise atmospheric air for respiration, can exist in anaerobic waters or the marginal areas of the watercourse for extended periods of time. However, for development of their eggs to occur, they must be in contact with dissolved oxygen (Richard, 1965, quoted in Harman, 1974). Dispersal is largely through crawling, and pulmonate snails are often found in ephemeral aquatic environments e.g. roadside ditches and marshes (Harman, 1974).

Snails are known to be particularly sensitive to heavy metals (Harman, 1974). In addition, snails are not found in conditions of low pH which may be related to their requirements for calcium (Fjellheim and Radum, 1990). *Lymnaea peregra* is tolerant of a moderate amount of organic pollution (Hellawell, 1986; Friedrich, 1990)

The recolonisation habits of *L. peregra* do not involve dispersal over a wide area and this mollusc has been shown to be the last taxon to recolonise an area following a disturbance. A incident of copper cyanide pollution in the River Lee in England described by Hynes (1960) resulted in the elimination of *Lymnaea peregra* for 16 miles. Few sources of recolonisation existed between the outfall and 10 miles downstream. One year after the incident, this species was found within half a mile downstream of the outfall having presumably colonised from upstream and was present to within 8.5 miles downstream, the recolonisation sources being tributaries and movement upstream. This contrasted with the other affected taxa: *Gammarus*, *Baetis* and *Tanytarsus* which had regained their former status within a year. However a study of the River Cole in Birmingham, England, found rapid recolonisation by *Lymnaea peregra* after cessation of sewage effluent (B.Hawkes, personal communication). This had apparently occurred through the distribution through flood events of egg masses which had been deposited on debris.

Cairns and Dickson, (1977) describing a spill of ethyl benzene and creosote into the Roanoke River, Virginia, USA found an impact on the benthic community for up to 7 miles below the spill, however some snails were observed to have survived the spill.

A similar instance of survival of polluted conditions was found in the present study. At the two most impacted sites on the Hewell Brook following the pollution incident, individuals were seen at the edge of the water channel, having crawled out of the water column. It is possible that *L.peregra* was displaying active avoidance of the unfavourable conditions following the incident. However, as numbers of *L. peregra* were low in Hewell Brook, it is difficult to ascertain any initial impact of the pollution incident through analysis of the numbers found. It has been observed that a related species, *Lymnaea palustris,* frequently leaves the water in the littoral zone and this behaviour enabled individuals to escape the acute toxicity of a chlorine spill (Heckman, 1983). During the study of a drought on the River Roding (Extence, 1981), *L. peregra* showed active avoidance and subsequent survival of drought conditions.

Following the pollution incident in Spark Brook, *L.peregra* individuals were observed at the edge of the channel, above the water level. Although the abundance of *L.peregra* within the samples at this site was low throughout the study, there was a progressive increase in abundance over time. It is possible that the behavioural response of this species has enabled it to survive in conditions of intermittent unfavourable water quality.

At Hewell Brook, increased abundance of *L. peregra* was seen at one site upstream and downstream of the outfall during May and July 1995 (222 and 288 days following the incident). The increase was particularly noticeable at the most impacted site and may have been due to an increase in the food resource following bacterial breakdown of the oil. This can result in subsequent stimulation of algal growth (Mason, 1996; Rosenberg and Snow, 1975). Similar effects have been found in other studies. Lock *et al* (1981) reported significant increases in chlorophyll a (a measure of algal biomass), diatoms and bluegreen algae on experimentally oiled substrates that had not previously been colonised with periphyton. Raven and George (1989), investigating the impact of an insecticide spill, found little evidence of mollusc mortality caused by the spill but a temporary increase in *L. peregra* abundance during the 6 months following the spill. They suggest that this may have been due to reduced fish predation, an increase in food substances (decaying macroinvertebrates) or the removal of arthropod competitors e.g. mayflies. Skoog (1978) (quoted in Gregory, 1983) found that *L. peregra* grew best on a diet of mixed blue-green algae and that the juveniles attained their highest growth on diatoms. The abundance increase at the most impacted site within Hewell Brook was indeed largely composed of juveniles.

The spill of Hydrochloric acid in the Mousesweet Brook had an impact on the population of *L. peregra* attached to the submerged aquatic vegetation in the River Stour 3.5 km downstream of the outfall. Colonisation in this case was rapid and by 83 days after the incident, numbers were similar to the reference site. In this instance, suitable colonisation sources were close; within 100m upstream. Within the Mousesweet Brook itself, no upstream sources of colonisation existed and few individuals were found within a year of the incident. Recruitment through reproduction of surviving individuals and from downstream sources on the River Stour were the methods of colonisation available, which would result in a relatively slow re-establishment of populations of this taxon within the Brook.

The cypermethrin spill on the River Lostock did not have any apparent effect on *Lymnaea peregra* populations, a finding to be expected as this pesticide affects largely arthropods.

In summary, *Lymnaea peregra* is catholic in its choice of habitats and has behavioural means of avoiding temporary reductions in water quality. Following mortality, however, recolonisation of affected areas can be slow in comparison with other taxa involving crawling through the substratum up or downstream. If, however, upstream sources of colonisation are present, the distribution of egg masses may result in the rapid recruitment of juveniles. There is some evidence of a potential stimulatory effect on populations following one of the pollution incidents.

6.4.1.8. Oligochaeta

Oligochaeta form a numerically important part of the benthic fauna of rivers, with Tubificidae predominating in depositing substrata and Naididae preferring eroding substrata (Learner *et al.*, 1978).

In general, they graze on algae and bacteria although some species of Naididae are predatory or parasitic (Learner *et al.*, 1978).

Naidids reproduce both sexually and asexually. Asexual reproduction occurs throughout the year and sexual reproduction is infrequent and may only take place once a year (Learner *et al.*, 1978). Some Tubificidae are able to reproduce asexually as well as sexually, and species found in abundance in polluted waters show this feature as well as showing sexual reproduction during most times of the year (Brinkhurst, 1965). In general, Oligochaeta are most abundant during the summer months when rates of growth and reproduction are stimulated by higher temperatures and plentiful food although in some species e.g. *Nais elinguis* Muller, Spring may be the time of highest recruitment (Learner *et al.*, 1978; Eyres *et al.*, 1978; Brinkhurst and Kennedy, 1965).

Ladle and Ladle (1992) suggest passive redistribution is important in some Tubificidae such as *Tubifex tubifex* (Muller) and *Limnodrilus hoffmeisteri* Claparede. These taxa are unable to swim actively and are swept along with eroding sediments in Autumn and Winter. Breeding and cocoon production can take place rapidly following the high flows. Hynes (1970) found Oligochaeta to be relatively uncommon in drift. Thus a distinction must be drawn between catastrophic drift and behavioural drift with catastrophic drift being the main means of dispersal of Oligochaeta. Williams and Hynes (1976) found that Oligochaeta colonised small areas of substratum by moving from both upstream and downstream sources.

The ability of Oligochaeta to reproduce rapidly in favourable conditions has led to the suggestion that local peaks in the populations of Tubificidae may be related to the time that has elapsed since the last catastrophic event affected that particular group of individuals (Brinkhurst, 1965).

Tubificidae and Naididae respond to organic enrichment by an increase in abundance (Learner *et al.*, 1978; Hawkes and Davies, 1971, Brinkhurst and Kennedy, 1965; Brinkhurst, 1965) and Tubificidae have also been found to dominate the macroinvertebrate fauna in rivers polluted by industrial waste (Eyres *et al.* 1978).

Yoder and Rankin, (1995) found that the metric '% Oligochaeta', when applied to a macroinvertebrate communities in Ohio rivers, was indicative of complex impacts comprising effluents from municipal Waste Water Treatment Plants and toxic chemical plants.

A number of studies of macroinvertebrate changes following pollution incidents have found that Oligochaeta are the first to colonise the impacted substratum (Cairns and Dickson, 1977; Douglas and McCreanor, 1990; Harrel, 1975; Raven and George, 1989; Crunkilton and Duchrow, 1990). In some instances this is due to survival during the pollution incident, e.g. the pesticide pollution investigated by Raven and George, (1989).

Smith and Distler (1981) suggest that, following the spill of an unknown chemical, vertical movement from within the substrate is a possible source of colonisation for Oligochaeta. Victor and Ogbeibu (1986), in assessing the impact of an application of insecticide (Lindane-based) to a tropical stream, found large population increases in Oligochaeta and Chironomidae and suggested that individuals may have escaped the effects of the incident through passive or active avoidance by being located deeper within the substratum and subsequently responded to a decrease in predation or competition by rapid reproduction.

Harrel (1985) investigating the effects of a crude oil spill into a small stream found that Oligochaeta were the most tolerant and the first taxon to be found in large numbers. Some Oligochaeta sampled were found to have oil in their gut. Rosenberg and Snow (1975) investigating the colonisation of oiled artificial substrates, found increased numbers of Oligochaeta in oiled vs control substrata in one of the rivers studied.

The majority of the Oligochaeta found in the present study consisted of Tubificidae and Naididae.

In most of pollution incidents investigated within the present study, a peak in Oligochaeta populations was seen during the year following the incident; a peak that was not reflected in the reference sites or in some cases in the same site during the following year (Chinn Brook, Hewell Brook, Mythe Tributary, Preston Bagot Brook, Mousesweet Brook, River Wye and Sketchley Brook). No Oligochaeta were found at the most impacted site 15 days after the pollution incident in the Hewell Brook, but 29 days later, numbers were high and remained higher than reference sites for 288 days. The oil incident may therefore have had a stimulatory effect on the abundance of Oligochaeta at this site.

A number of reasons can be suggested for this increase, which as previously mentioned, has been found by other investigators: survival of initial pollution incident; tolerance of changed conditions; rapid reproductive capacity, thus an ability to take advantage of favourable conditions e.g. lack of predators or competitors, increase in food resource.

Oligochaeta present difficulties in identification, such that identification to species often requires the presence of sexually mature adults. There are therefore problems with the use of this category - different species and families of Oligochaeta may have different sensitivities to different kinds of pollutant. However two features are found in the Tubificidae and Naididae, the most frequently found families of Oligochaeta in the streams investigated: firstly, they respond to an increase in organic enrichment with an increase in numbers, and secondly; the increase in abundance can be rapid over a short period of time if conditions are favourable. In this way, Oligochaeta are similar to Chironomidae in that they show rapid recolonisation following disturbed situations and can be described as 'pioneer' taxa (Heckman, 1983).

6.4.1.9. *Potamopyrgus jenkinsi*

A hydrobiid snail native to New Zealand which has spread throughout Britain and Europe since its introduction in 1859 (Ponder, 1988), *P. jenkinsi* demonstrates apomictic parthenogenetic reproduction and can reproduce rapidly given the right conditions (Forbes *et al*, 1995). Due to apparently low levels of genetic diversity of these obligate parthenogenetic snails, it has been suggested that associated slow adaptation to environmental change combined with low competitive ability, restricts its association to man-made or disturbed habitats (Hauser *et al*, 1992; Ponder, 1988).

Distance chemoreceptors enable *P. jenkinsi* to locate food and it has been found to eat a wide range of substances including green algae, leaf litter from aquatic macrophytes and allochthonous sources, dead and decaying animal material (Haynes and Taylor, 1984).

In Hewell Brook, this species is found in quantity only at sites B (reference), C, D and E but is absent from the furthest downstream sites F and G and in low numbers at one of the upstream control sites; A. At sites C, D and E, *P. jenkinsi* is not present immediately after the pollution. The incident may have caused an initial impact on *P. jenkinsi*, eliminating numbers just downstream of the outfall for 15-29 days. At the reference site B, *P.jenkinsi* is present throughout the study with an abundance pattern showing 2 peaks at 53 and 222 days following the incident. Population peaks of *P. jenkinsi* were also found at sites D and E 53-222 days after the pollution incident. The rapid increase of this species at these impacted sites may have been associated with reduced competition from other species combined with a relatively greater ability to take advantage of favourable habitats in terms of reproductive ability, mobility and eclectic food preferences (Haynes and Taylor, 1984; Haynes *et al.*, 1985).

A peak in numbers is also seen following the incidents at the Mousesweet Brook and Preston Bagot Brook, which in both cases is not matched by similar increases at the reference sites.

Given the efficient dispersal and reproductive abilities of this taxon, and the finding of peaks in population after pollution incidents, *P. jenkinsi* can be regarded as characteristic of the pioneer stage of recovery along with Chironomidae and Oligochaeta.

6.4.1.10. Sphaeriidae

Sphaeriidae, small bivalve molluscs, are hermaphroditic or self fertile and only a single individual may be required to extend the ecological and geographic range of a species. Reproduction involves the brooding of a small number of larvae in the shell cavity which are released as fully formed young mussels. Mechanical distribution is therefore a key element in their dispersal (Fuller, 1974). There is some evidence that Sphaeriidae will burrow deep into the substrate given unfavourable conditions e.g. drought (Ingram, 1941; quoted in Fuller, 1974). It is not known whether this avoidance mechanism is also found during pollution-induced deterioration in water or sediment quality.

Servos and Mackie (1993) found Pisidiid clams to be tolerant of short term depressions of pH (down to 3.5) during spring snowmelt with no subsequent effect on survival or long term effect on reproduction.

As indicated with Oligochaeta and Chironomidae, there may be problems with the level of taxonomic identification used in that different species of *Sphaerium* may have varying responses to pollution (e.g.Friedrich, 1990). Two species found in large numbers in organically polluted waters in Britain are *Sphaerium corneum* and *S. lacustre* (Hynes, 1970).

In Hewell Brook, Sphaeriidae were only frequent at reference site A and found in consistent but small numbers at a site located some distance from the pollution outfall (Site G). Sphaeriidae were initially absent at the most impacted site (Site C) and then showed an increase in numbers 150 days following the incident. This may have been due to an improvement in environmental conditions. This sampling point had a high proportion of silt and deposited organic matter. The incident resulted in the elimination of any Sphaeriidae present within the substrate and gradual recolonisation must have occurred from upstream sources.

The relative paucity and variability of populations found at the reference and impacted sites following the pollution incidents in this study make any hypotheses regarding impact and patterns of recovery impossible to make.

6.4.2. Summary of abundance changes in selected taxa following pollution incident

The problems associated with the selection of the most abundant taxa for analysis have already been mentioned (Section 4.3.2). However with these caveats in mind, analysis of the response of the individual components of the community is essential to the understanding of changes in other measures used and indeed the validity of those measures in reflecting total community response.

Oligochaeta were consistently found to be the first taxon to colonise a site following the pollution incidents investigated with individuals present within a few days. Following some incidents, similarly rapid colonisation was also shown by Chironomidae (Chinn Brook, Sketchley Brook) and by *Potamopyrgus jenkinsi*. *Asellus aquaticus* can also show rapid recolonisation (Sketchley Brook), but generally this species was found to colonise sites after Chironomidae and Oligochaeta. Subsequent colonisers were usually *Gammarus pulex* and *Baetis rhodani*.

Taxa showing slower colonisation within the study were *Goera pilosa* and *Agapetus fuscipes* (cased caddis larvae)and the snail *Lymnaea peregra*.

The differential colonisation patterns of the above mentioned taxa can be due to a number of factors.

- Dispersal Mechanisms Macroinvertebrates show a variety of dispersal mechanisms within each generation. These comprise movement through the substratum, active entry into and exit from drift, passive dislodgement into drift and subsequent attachment to substratum as well as more complex means of dispersal e.g. the attachment of bivalve larvae to fish and the ingestion of *P.jenkinsi* by fish and their subsequent evacuation in a living state (Haynes *et al.*, 1985). The detailed behaviour of individual taxa in terms of their dispersal through the stream habitat is relatively poorly known (Peckarsky, 1983). Drift was once thought to be the most important mechanism leading to a great deal of research on the phenomenon of drift and the compensatory mechanisms required to prevent continuous downstream displacement of populations (Elliott, 1971; Muller, 1982; Allan and Russek, 1985; Otto and Sjostrom, 1986; Soderstrom, 1987; Brittain and Eikeland, 1988; Bernard *et al.*, 1990; Williams and Williams, 1993). However the importance of drift as a distribution mechanism may have been overstated and it is clear that organisms are in a constant state of flux, with a variety of biotic and abiotic factors contributing to their movements (Townsend and Hildrew, 1976; Wiley and Kohler, 1984; Brittain and Eikeland, 1988; Palmer *et al.*, 1992).

- Life history characteristics: Taxa with a multivoltine life-cycle would show more rapid recolonisation of areas than taxa with a univoltine life cycle. Parthenogenesis and asexual reproduction through fragmentation (as shown by *Potamopyrgus jenkinsi* and Tubificidae respectively) are also conducive to rapid population increases.

- Survival of a proportion of the organisms during the pollution incident: The environmental heterogeneity of the stream system leads to variation in the exposure of organisms to the toxicant. In addition, those taxa e.g. Oligochaeta that have been shown to display a tolerance to adverse water quality would be more likely to survive conditions of sublethal pollution stress and therefore be amongst the initial colonisers of a site. Survival may also be a consequence of active behavioural avoidance of adverse conditions e.g. burrowing into the substratum or crawling out of the water as was seen in the case of *Lymnaea peregra*.

A number of taxa showed peaks in abundance following the pollution incidents which were greater than those found at the reference sites. This was found in Oligochaeta, Chironomidae, *Potamopyrgus jenkinsi, Lymnaea peregra* and *Asellus aquaticus*. The reasons for this increase could be attributed to a variety of factors, the relative importance of which can not be ascertained from the findings of the present study. Factors include tolerance of changed conditions, rapid reproductive capacity and colonisation mechanisms, increased food resource, decreased competition and predation.

The dynamic nature of streams results in a situation where aquatic species are adapted to variable environments. However the degree to which different groups can colonise a disturbed area varies and this finding can be used to provide a measure of the recovery status of a particular point within the stream following a pollution incident. As in other ecosystems, there are opportunistic species which have a high rate of population growth and equilibrium species which are adapted to compete more effectively for resources and have a lower rate of population increase (MacArthur, 1960).

6.4.3. Taxa Richness

Taxa richness, simply measured as the number of taxa in a sample, is one of the simplest ways in which community data can be represented. Its utility depends to a large degree on the level of identification employed within the study and on the predicted characteristics of the community response to a disturbance. The underlying principle is that stable biological communities in streams have high species richness (DeShon, 1995). Reice and Wohlenberg (1993), state that species richness is one of the most important attributes of the benthos to be measured in order to determine ecosystem health although population estimates of individual species should also be considered if resources permit.

The number of species becomes restricted as the severity of disturbance of the natural habitat increases (Brinkhurst, 1965). However, the number of species in a community is also influenced by other factors, for example: geographical location (the fauna of the

British Isles is less diverse than that of Europe which in turn is less diverse than that of America); and local environmental conditions, acid streams support fewer species than chalk streams (Jeffries and Mills, 1990), eroding substrata within streams support more species than depositing substrata (Hynes, 1970; Brown and Brussock, 1991). Lenat (1983) found that taxa richness in the family Chironomidae does not always decline with increasing pollution. Moderate pollution and sedimentation may actually cause increases in taxa richness so they suggest that this parameter be used with caution in environmental assessment.

Within the present study, taxa richness was found to be reduced after a number of pollution incidents and a subsequent increase in the number of taxa was taken to be evidence of temporal recovery. The changes seen in this measure were not always reflected by other measures used. For example, in Callow Brook, initial pollution impact was seen in the presence and abundance of certain taxa and this was not reflected in the taxa richness. At Sketchley Brook, taxa richness showed no impact of the pollution incident, whereas an impact was shown by consideration of the abundance of selected taxa and by multivariate community analysis.

In comparison with other univariate measures of the community, taxa richness was found to provide valuable information relating to recovery. Some sites showed similar estimates of recovery to those assessed by the other univariate measures, for example Hewell Brook, Mythe Tributary, Spark Brook. Following the pollution incident at Chinn Brook, recovery as assessed by taxa richness was shown to be more sensitive to changes than the diversity index used which showed no evidence of impact. This was also found by Sheehan and Winner (1984), who demonstrated that species richness was more strongly and significantly correlated with copper levels than was the diversity index. Resh and Jackson (1993), similarly, found that richness measures were accurate in detecting impact where it occurred and had greater effectiveness than the Shannon-Wiener diversity index.

A number of the pollution incidents have impacted streams that support low numbers of taxa. These taxa tend to be present throughout the year and seasonal extinctions and appearances are not features of these streams. Taxa richness, whilst not taking temporal taxon replacements into account, and therefore being less representative of changes that are taking place in diverse streams, may nevertheless have greater utility as a measure of recovery in streams that do not have a high proportion of seasonally occurring taxa.

This simple measure, although not taking into account the more subtle changes in community composition, nevertheless, in most cases where an impact is seen, gives similar information to that provided by the more computationally demanding diversity index, and in some cases, has been shown to be more sensitive than the diversity index.

Taxa richness continues to play a part in the development of indices and forms a part of the multiple metric approach to community analysis now being developed in different states of the USA (DeShon, 1995).

6.4.4. Diversity index

The Shannon-Wiener Index is not concerned with the biological status of each species within the community but only with the ratio of abundance of organisms to the number of species and can be greatly affected by opportunistic species, such as those of the Tubificidae and Naididae which rapidly increase in abundance over a short period of time. Learner *et al.* (1978) found that an increase in these groups on the River Cynon produced a change in the index value which suggested an increase in pollution. However this was not substantiated by chemical or physical analysis.

Values of H' >3 are considered to indicate unpolluted conditions, values from 1-3, moderate pollution and values <1 heavy pollution (Wilhm and Dorris, 1968). All the reference sites in the streams investigated in the present study had diversity index values less than 3 and most showed values less than 1 at certain points during the study. The changes appeared to be related to the abundance levels of a few taxa which are known to show great variability over time.

Although diversity indices are still widely used in pollution studies (Resh and McElravy,1993; Norris and Georges, 1993; Metcalfe-Smith, 1994; Victor and Ogbeibu, 1986; Woodward and Riley, 1983), they are not generally employed for the purposes of river water quality monitoring, particularly in Europe (De Pauw and Hawkes, 1993).

The predicted impact of the response of the structural aspects of the community as measured by the Diversity Index H' were of a reduction in H' following the incident, with a relatively high abundance of fewer taxa. This was found in many of the incidents investigated although the relation of this index to other measures is variable.

In Sketchley Brook, Chinn Brook, River Wye, Mousesweet Brook and Preston Bagot Brook, no impact on H' could be ascertained as the result of the incidents. Of these incidents, in Preston Bagot Brook and the River Wye, no impact was found on the other measures used, whilst in Sketchley Brook an impact was seen both in the abundance of individual taxa and multivariate community similarity. In Chinn Brook, BMWP and taxa richness showed an impact and within Mousesweet Brook an impact was found on taxa richness and community similarity.

At some sites the diversity index shows similar changes to the other univariate values (richness, BMWP and ASPT scores), with low diversity following the incident and a gradual increase over time (Callow Brook, Hewell Brook, River Lostock, Mythe Tributary). However the index is not as sensitive as other measures in assessing community changes over time. In Hewell Brook, recovery in terms of H' occurs within 53 days, whereas other measures including richness, show longer levels of impact; more than 101 days. At the River Lostock, the cypermethrin leak resulted in decreased H' which subsequently increased to reference levels before the BMWP and richness measures.

Findings that univariate measures other than H' show greater sensitivity to changes in pollution gradients have been demonstrated in other studies. Camargo (1992), investigating the changes in a range of indices along a stream affected by an organic effluent, found H' to be more sensitive to changes in the relative abundance of species and less sensitive to the number of species. This study found BMWP and species richness to be more sensitive indicators of total macroinvertebrate response.

Other workers investigating community responses to pollution have also found that taxa richness, or even total abundance of individuals to be a more sensitive measure of changes in pollution concentration or community recovery following disturbance than the diversity index (e.g. Winner *et al.*, 1995; Sheehan and Winner, 1984; Doeg *et al.*, 1989; Guiney *et al.*, 1987).

Although it has been suggested that a lack of changes in the diversity index following a pollution incident was due to toxicity affecting all taxa equally (Hoehn *et al.*, 1974), this is only one of a variety of interpretations of changes in H'. Statzner (1981), suggested that H' was mainly influenced by factors affecting a single species rather than the whole community. He found that changes in populations of species that are frequently common in streams such as *G.pulex*, *P.jenkinsi*, *B.tentaculata* and *Dreissena polymorpha* accounted for a large proportion of the variation in H'.

At Canley Brook, along with other measures, the incident did not show expected changes in H'. However diversity changes at the *reference* sites were significant over time and depended apparently on the abundance changes of *G. pulex*. In this case, the ratio of abundance of organisms to the number of species is a reflection of only one aspect of the ecological status of the community.

The ecological significance of low diversity in situations other than those that are clearly dominated by post pollution populations of Chironomidae and Oligochaeta is not clear. A measure of % Oligochaeta may be more informative as such communities recover. Maltby *et al.*(1995), find that the use of a combination of measures is useful in the determination of impact e.g. diversity indices to assess the alterations in community structure and biotic indices to assess changes in freshwater quality .

The Diversity Index may indeed be useful in situations where a stability of diversity has been demonstrated to be the normal situation; e.g. areas supporting large numbers of taxa with successive seasonal abundance of different taxa. Although, even in these situations, a variety of other measures are probably more informative and it is the case that the multimetric approach, widely used and undergoing development in the USA, does not incorporate the use of a measure of diversity relating to evenness of abundance distribution of taxa.

In the majority of the streams investigated in this study, pollution inputs were complex and streams were often subjected to other inputs, limiting the abundance and variety of taxa present. The loss or addition of a species to a community can be a more significant ecological event than changes in the abundance of a species which can be influenced by a variety of seasonal and environmental factors. Other measures involving less computational time, for example, the simple measure of the number of taxa gave the same information and in certain cases proved more sensitive than the diversity.

6.4.5. BMWP and ASPT

Following the incidents, the total BMWP and ASPT scores were calculated at each sampling point, and qualitative comparisons between impacted and reference sites were made from the graphical data. Replicates were combined to calculate the BMWP score whereas the other univariate measures (H' and Number of Taxa), utilised the means of the three replicates and could be subjected to statistical analysis. The point at which the

BMWP scores are considered similar to those of the reference site therefore involves a greater degree of subjectivity.

Following certain pollution incidents, it was found that the changes in BMWP and ASPT scores matched changes in the other univariate measures of taxa richness and Shannon-Wiener Diversity. In Canley Brook and the River Wye no impact was demonstrated using any of these measures. In the River Lostock, Mythe Tributary and Spark Brook, the measures showed changes which were similar in spatial and temporal terms.

Two incidents showed impacts on the BMWP and ASPT scores which were more extensive than the impacts on other univariate measures. Sketchley Brook showed an impact on ASPT and BMWP scores and no impact on other univariate measures. This was largely due to the absence of *Gammarus pulex*, a relatively high scoring taxon, immediately after the pollution incident. At Preston Bagot Brook, a spill of hydrochloric acid, which also resulted in elevated levels of heavy metals, had a greater impact on those taxa sensitive to organic pollution at certain impacted sites as demonstrated by changes in ASPT and BMWP scores that were not matched by changes in the taxa richness measure.

At other incidents ASPT was either not impacted when other indices showed an impact (Callow Brook, Mousesweet Brook), or shown to recover more rapidly than other indices (Hewell Brook, River Lostock). The latter incidents showed that a qualitative return to a community tolerant of a particular level of organic pollution was achieved before other measures of recovery.

In Hewell Brook, pollution impact is apparent for a greater length of time when taxa richness is considered than either the BMWP and ASPT scores. Although the BMWP score incorporates a measure of richness in that it increases with the numbers of families present, irrespective of their pollution tolerance, the measure of taxa richness utilises the community information to a higher level of identification; primarily genus and species. This may result in the finding of more subtle changes.

The value of utilising a score that does not incorporate abundance was brought into question on a number of occasions within the study, where a few individuals of a particular taxon caused changes in ASPT scores, leading to interpretations that were not supported by other analyses of the data. In Callow brook, following an oil pollution incident, a single individual of *Gammarus pulex* was found. This increased the ASPT score at this site but in fact significantly lower numbers of this species were found downstream of the pollution incident when compared with the reference sites. Drifting individuals were also thought to have the same effect in Hewell Brook on the first sampling occasion following the incident, thereby raising the univariate measures. Within Canley Brook, great variability was found in BMWP and ASPT scores, largely as a result of the presence or absence of a few individuals of Limnephilidae; a high scoring family.

Although some relationship between taxa richness and BMWP score is to be expected as the latter incorporates an element of richness in the summation of the number of families found, the finding of similar changes in ASPT following pollution incidents involving a variety of pollutants, e.g. oil, acid and insecticide could demonstrate that the scores may have more general applicability.

The taxa showing the greatest tolerance to organic pollution, Oligochaeta and Chironomidae, are also often the taxa to be found following a pollution incident involving a variety of pollutants (See Section 6.4.1.4 and 6.4.1.8). These two groups have the lowest scores on the BMWP scale and when they form a large proportion of the taxa present, reduce the resulting ASPT score.

A more refined assessment of the community and its tolerance to organic pollution would be the allocation of a tolerance score to each taxon recorded, not just the families present. It is conceivable, given adequate toxicological and survey information, that a variety of score systems could be applied to components of a community depending on the pollution input under investigation e.g. acid, organic, pesticide etc. However, the value of such score systems to transient pollution incidents that do not show any residual toxicity is questionable.

The finding that the basic community information; the identity and abundance of the taxa, still convey more information than changes in BMWP scores, only highlights the fact that rather than reliance on univariate measures to understand changes within the community, account needs to be taken of the components contributing to changes in the measures (Fryer, 1987).

6.4.6. Conclusion of Univariate Measures

To demonstrate impact and recovery on a stream utilising a univariate measure, the more sensitive the measure is, the more defined the assessment of recovery can be. This is important because it must be remembered that the benthic macroinvertebrates are being utilised as indicators of ecosystem change, and the loss of information that is necessarily involved when community information is summarised in a single score or index, could result in a false assessment of biotic recovery.

Measures of diversity have been found to vary greatly, even within the reference sites and provide a reflection of the changes in abundance of a few taxa rather than an ecologically relevant representation of the community in response to a disturbance. ASPT scores are also found to be relatively insensitive to changes following disturbance. BMWP scores, although giving greater weight to families least tolerant to organic pollution, have been shown in some instances to be more sensitive than other indices although in other instances, greater sensitivity is found when taxa richness is considered. The former finding may be related to the fact that the taxa to appear first following a disturbance have the lowest BMWP scores, but, as colonisation proceeds beyond a certain stage, further definition of change may not be provided by this biotic score.

In summary, it appears that the simple measure of taxa richness gives as much as or more information than the other measures, whilst making no assumptions about the structure of the community or relative sensitivities of the taxa involved.

6.4.7. Multivariate Analysis

The value of multivariate analysis of community data is that it utilises a larger proportion of the data than the univariate measures that were considered above. Community similarity matrices were constructed which incorporate the taxa present as well as the abundance measures of each taxon.. The assessment of impact and recovery was carried out in 3 ways.:

1. In all the pollution incidents, impact was predicted to have caused changes within the community resulting in a community that differed from the reference. As time elapsed, the communities at the impacted site were predicted to show a progressive increase in similarity to the reference site such that no significant difference in

communities would provide a measure of recovery. In addition, it was predicted that a spatial pattern would be evident in that communities further away from the pollution impact would show progressively less impact and less progression in community similarity over time than communities close to the impact. The strength of this method of analysis relies to a large extent on the choice of appropriate comparison sites which have not received a pollution impact.

2. Impacted communities were predicted to show greater variability over time than the reference sites. This prediction did not presume that the impacted communities would be the same as the reference communities in the absence of pollution. This assumes that communities not impacted by a pollution incident would show greater stability or community similarity over time (Clarke and Warwick, 1994). Increased variability at the impacted site may not, however, be attributable to the pollution incident but may, for example, be the result of other discharges connected with the outfall.

3. The communities within the impacted sites are predicted to show progressively greater similarity to each other, with samples taken within a short time of the pollution incident showing greater dissimilarity than samples taken some time after the pollution incident. This assumption does not require a reference site for comparison, the spatial configuration of samples within each site over a period of time following the incident is all that is required. The rationale for this assumption is based upon the colonisation patterns of streams and other habitats following a disturbance which tends to follow a pattern of rapid initial colonisation of taxa followed by a more steady increase in taxa (Sheldon, 1984).

The disadvantage of the community level approach to the assessment of impact is that detailed patterns of change within the community may be masked e.g. the abundance of individual taxa may increase or decrease.

An advantage is that the spatial configurations allow progressive changes over time to be seen and significance analyses can be carried out to confirm or reject apparent differences. In addition, the taxa within the communities that are primarily responsible for these differences can be extracted and compared with the analyses of abundance changes in the numerically dominant taxa. The degree of transformation of the community abundance and subsequent comparisons gives some indication of the degree to which community similarity is dependent upon the more abundant taxa or the less frequent taxa.

The significance tests carried out between samples are particularly useful in situations where the reference sites are known to be similar to the polluted sites prior to impact. Otherwise, multidimensional scaling is most valuable in the graphical representation of community similarity and the changes over time. This, in conjunction with other measures, provides some indication of the way in which the community has changed following a pollution incident and the important components of that change.

Within the study, following some of the incidents, community similarity of the impacted sites to reference sites is found after a period of time - which can be taken to be an indication of recovery. However, recovery of other measures, for example, abundance of individual taxa may not be apparent. For example in the Chinn Brook, similarity of the impacted sites to the reference sites is found after 287 days when the data are root/root transformed. However, the most frequent taxon within the reference site is *Gammarus pulex* and as numbers remain low within the impacted site they may not have yet have recovered by 287 days following the incident, which was the end of the study.

Multivariate analysis can show an impact not demonstrated by other measures. For example in Hewell Brook, one of the sites situated some distance downstream of incident outfall did not show an impact using the individual taxa or univariate measurements. However the MDS plot showed a progressive increase in similarity over time to the reference sites.

MDS plots can show impact for a greater length of time than the other measures, indicating the value of consideration of the whole community or as much of the community as possible rather than condensation into a single number. For example in Hewell Brook, the most impacted site shows a continued difference to reference sites in untransformed and root/root transformed data after 610 days whereas other measures e.g. diversity and taxa richness show recovery at 101 days and 394 days respectively.

At some pollution incidents, multivariate analysis is not shown to offer any more information than the other measures e.g. River Lostock, Mousesweet Brook, and Sketchley Brook. At these sites BMWP scores and multivariate analysis show a similar temporal impact. In some instances, multivariate analysis can show no evidence of impact although some impact is discernible through other means e.g. individual taxa, BMWP scores (River Wye). Within the Mythe tributary, the last sampling occasion, 369 days following the incident showed a recovery according to certain univariate measures. However, although multivariate analysis showed some recovery when the raw data were considered, when greater weight was given to the less common taxa, recovery was not apparent, a finding which was also clear from consideration of the individual taxa present.

In Preston Bagot Brook, greater variability is seen in the impacted sites (apart from Site I) than the reference site if relatively less emphasis is given to the abundant taxa. When abundant taxa dominate the analysis, as is the case with untransformed data, the variability in the reference site and impacted sites is similar. The prediction of impact being demonstrated by impacted sites showing progressively greater similarity over time to reference sites is not found at any site. Given the different environmental parameters of the reference Site A to the impacted site, the macroinvertebrate community supported by the sites is likely to be different even in the absence of pollution impact. Progressive similarity to upstream communities may therefore not be an appropriate measure of impact and recovery in this case.

Greater variability at the impacted sites than the reference sites is seen at Spark Brook. However this may be related to the respective community compositions at the sites; the impacted sites supporting a relatively impoverished community with taxa that are variable in their abundance over time and space, and the reference site being more diverse with fewer dominant taxa. This may in fact be a consequence of the use of this measure as an assessment of an impacted site; if data are not transformed, it would be largely a reflection of the great variability in abundance that is shown by a few rapidly colonising taxa.

The third prediction of greater clustering of communities over time within impacted sites was only found at one of the pollution incidents investigated: Preston Bagot Brook, where it was seen to varying extents in three of the impacted sites. This measure may be more useful if a greater number of sampling times is used and if the sampling times are more evenly spaced.

In summary, the multivariate approach to assessment of impact and recovery is a valuable tool which is often more appropriate as a summary of the relative differences in community composition between two sites than the univariate indices in that a greater proportion of the information that has been collected is retained in the analysis.

If appropriate reference conditions are available for comparison, then the progressive change in impacted communities over time is a powerful tool in the assessment of temporal recovery.

Attempts to infer impact and recovery in other ways, e.g. differential variability at impacted sites compared with reference sites or progressive increase in similarity between communities within an impacted site were not successful, due possibly to the great within site and between site variability associated with factors not connected with the pollution incident and to the fact that the sampling times following the pollution incident were relatively few (not more than 9) and they were not evenly spaced in time.

Chapter 7

Laboratory Study

7.1. Introduction

The effect of a transient pollution incident depends upon a large number of variables related both to the pollutant (type of pollutant, length of discharge, pattern of discharge), and to the receiving environment (history of pollution discharge, physical and chemical characteristics of the stream, composition of the ecological community etc.).

The results of the pollution incidents investigated showed that limited transient pollution may have a visible impact on various aspects of the ecology within the watercourse, for example, dead fish and invertebrates, but may not have a clear measurable impact on the stream macroinvertebrate fauna in terms of abundance or composition. For example, the suds oil incident in Canley Brook caused visible fish and invertebrate mortality but no measurable impact on the invertebrate community within the sampling points chosen, even though reference conditions included pre-impact data as well as upstream sites and tributaries. This finding may have been a consequence of the sampling methodology chosen, however other possibilities could also be considered. At the two most impacted sites, rapid recolonisation could have occurred before the first sampling occasion following the pollution incident. As a large portion of the watercourse upstream of the source of the pollution incident is culverted, this is unlikely to have been a major source of colonisation. The most likely source of colonisation would therefore be from within the impacted section of the watercourse and given the apparent rapidity of colonisation, this may have resulted from individuals that had survived the passage of the pollutant through active or passive avoidance. Given that there would be spatial variability in the level of impact of a pollutant within a stream, it is likely that the the fauna would be exposed to variable concentrations with the possibility of survival for a proportion of the community. In addition, detection and active avoidance of the pollutant by individuals would also result in increased chances of survival and subsequent colonisation.

It was considered that a laboratory study to investigate the superficial behavioural responses of selected macroinvertabrates to a pollutant would give some insight into those factors that may be important in understandig ecological impact of pollution incidents, but which are not necessarily apparent when measuring only the structural community changes.

Research into the effects of pollution incidents have tended to focus on the presence/absence of macroinvertebrate taxa, with absence being the result of mortality or avoidance (drift) (Jones and Howells, 1975; Smith and Distler, 1981). Steinman (1993) observed that:

> "most studies have treated the disturbance event itself as a black box presumably because they have observed only its after-effects. However disturbances are not instantaneous and it seems likely that interesting ecological processes are taking place during the disturbance".

It has been suggested that behavioural experiments can be utilised to determine the mechanisms for observed phenomena (Peckarsky, 1983; Borlakoglu and Kickuth, 1990).

There is evidence that macroinvertebrates can detect and respond in a measurable way to pollution concentrations within the water column and sediment that are far lower than LD50 concentrations and in certain cases lower than maximum 'no effect concentrations' (e.g. Heinis and Swain, 1986; Gammeter and Frutiger,1989; Crossland *et al.*, 1992).

Behavioural responses to sublethal levels of toxicants can take a variety of forms e.g. reduction in feeding response or increase in undulatory behaviour (Johnson *et al.*, 1990; Heinis *et al.*, 1990). A number of these responses will be adaptive in that they result in the organism being better able to survive in suboptimal conditions. Included amongst the adaptive responses is the repertoire of behaviours that can be subsumed under the heading 'avoidance behaviour' in that they result in reduced exposure to a perceived reduction in optimal conditions. The following behaviours can be described as having an avoidance result.

1) Drift There is a vast literature on the phenomenon of drift in macroinvertebrates. Drift occurs naturally in streams and is considered to be an important mechanism in downstream colonisation of the streambed. Drift has also been shown to be a response to the introduction of toxicants into a stream (Edwards *et al.*, 1991;

Sibley *et al.*, 1991; Crossland *et al.*, 1992; Mitchell *et al.*, 1993). Less well documented is the role played by active avoidance of suboptimal conditions by entry of organisms normally associated with the substratum, into the water column, commonly termed 'behavioural' drift (Wiley and Kohler, 1984). Hopkins *et al.* (1989) found an increase in 'behavioural' drift following an experimental pollution of acid in a Sierra Nevada stream. Similarly, Bernard *et al.*, (1990) found a drift response after the reduction of pH in a stream and the addition of Aluminium. As more than 95% of the animals found in the drift subsequently recovered when placed in circumneutral water, this was also described as behavioural drift.

2) Burrowing into the Substrate Use of the hyporheic zone as a refuge under conditions of high flow has been quoted as a behavioural avoidance mechanism (Marchant, 1988; Palmer *et al.*, 1992). There is little evidence for this behaviour under conditions of anthropogenic pollution (an observational example is provided by Sheehan & Winner, 1984).

3) Climbing out of the water or burrowing into the bank Many species of snail can live in suboptimal physical situations for a limited period of time as a response to a temporary decline in water quality and this has been demonstrated as a response to pollutants (see examples in Section 6.4.1.7; Harman, 1974; Extence, 1976; Heckman, 1983; Edwards *et al.*, 1991).

4) Increase in activity This behavioural response is related to the above three responses which may be the functional result of a simple increase in activity on the part of the individual animal . This would increase the likelihood of encountering more favourable environmental conditions. Scherer and McNicol (1986), investigated the behavioural responses of a predatory stonefly to insecticide pulses in the laboratory and found an increase in locomotor activity and abandonment of preferred microhabitat, suggesting that this is the mechanism whereby increased drift numbers are seen following a pollution incident.

Initially, three widely distributed macroinvertebrate taxa were chosen for study: *Gammarus pulex, Lymnaea peregra* and *Asellus aquaticus*. Two flowing water channels were available, one with perspex sides, enabling extensive viewing of the test animals and another meandering channel comprising riffle and pool areas. Pilot studies involved placing cages containing the test animals with associated substrate and food in the flowing water channels. However, despite numerous adaptations in the construction details and the design of both the cages and the flowing water channels, it was not possible to keep *Gammarus pulex* and *Asellus aquaticus* alive under these conditions. It was considered likely that pollutants associated with the channel were contaminating the water and, although changes were made to rectify this, the time subsequently available necessitated limiting the study to a single taxon under static water conditions.

7.2. Objectives and approach

Aim - To determine the short-term behavioural responses to a toxicant and to discuss the degree to which they may be adaptive in avoidance of sub-optimal or lethal conditions.

The specific objectives were:
1. to determine whether there is a behavioural response to an introduced toxicant;
2. to define the behavioural responses.

The response of *Gammarus pulex* to the introduction of a toxicant (Hydrochloric acid) was assessed in the laboratory. This species is common and widely distributed, forming an important part of the freshwater ecosystem in a large proportion of shallow waters. In freshwater ecosystems with allochthonous material forming a major source of energy, changes in population dynamics of *G. pulex*, one of the most important species within the functional group responsible for shredding leaf material, is likely to have an important impact upon other trophic levels within the stream. A number of factors contribute to the presence of this species in the field including substratum, food, predators and oxygen levels (Rabeni and Minshall, 1977; Adams *et al.*, 1987). To enable the behavioural response to be related to its potential adaptive significance in real-life situations, the laboratory conditions were designed to be close to optimal field conditions experienced by the species tested and included the addition of substratum and food. Previous trials indicated that *Gammarus pulex* survived and formed breeding populations under the conditions provided.

The experimental design enabled the recording of behaviour prior to and following the introduction of the pollutant. Initial trials utilised video equipment to record behaviour. However the definition of the image was not clear enough to enable detailed observations of individuals to be carried out, and direct obervation was employed instead. The pollutant was introduced into one portion of the container enabling spatial location of the shrimps to be related to concentration of the pollutant. The container was visually divided into four sections and the directional movement of the shrimps from one section into the other was recorded. Qualitative and quantitative records of the behaviour of the shrimps were also made

Figure 7.1a: Photograph to show test trays used in study

Figure 7.1b: Photograph to show a *Gammarus pulex* test specimen and substratum

7.3. Methods

7.3.1. Test Apparatus

PVC test containers measuring 10 cm by 30cm were filled to a depth of 2 cm with a layer of washed commercially available gravel (10 mm irregular crushed quartzite blended with natural gravel from the Upper Trent river terrace, Weeford pit). Water was added to a depth of 5 cm from the base of the container. The water used was aerated, dechlorinated Birmingham tap water (temperature 14°C, pH 7.5+-0.2, conductivity 250 ± 5 $\mu S.cm^{-1}$).

The container was divided into four even sections by means of dividers above the surface of the water enabling visual location assessment to be carried out (Figure 7.1).

Food was added in the form of conditioned hazel leaves (*Coryllus avellana*), with a 1 sq. cm of leaf placed in each of the four sections and weighted with a piece of gravel.

Test animals were obtained from a breeding population maintained in the laboratory. The original source of this population was a stream near Hinkley, Leicestershire. Males and non-brooding females 6-7 mm in length were used for the test. One individual was placed in the experimental container and one in the control container for each replicate test session.

The pollutant; 3 mls Hydrochloric acid pH 1.7, was introduced by syringe into one end of the container at a point in time when all *G.pulex* within the containers were situated at the opposite end. Previous studies had shown that if the toxicant was introduced into the section containing an animal, immediate mortality ensued. The concentration used had been shown to reduce the pH in the container to pH 4.6 over 12 hours resulting in mortality to *G.pulex* within the container. Preliminary studies showed that exposure to the container with the pollutant added for one hour followed by transfer to clean water caused no subsequent mortality of individuals. Water lacking the toxicant was introduced by syringe into the control containers.

All experiments were conducted at room temperature ($14^{o}C \pm 2^{o}C$) between 10.00 and 14.00 over a 2 week period.

7.3.2. Test procedure

Preliminary studies had been carried out to assess the behavioural repertoire of *G.pulex* under the laboratory conditions provided. Animals were fairly active, moving within and on the surface of the substratum and occasionally swimming through the water column. A variable amount of time was spent feeding on the leaves provided. The following definable behaviour categories were recorded:

1. <u>'Burrowing' behaviour</u>. The animal moves into the substratum for a period of 2 seconds or longer. The frequency and the length of time spent in the 'burrowed' position was recorded;
2. <u>'Nosing' behaviour</u>. This involved the animal inserting the front portion of its body in crevices within the substratum for a period of not more than one second. The frequency of this behaviour was recorded;
3. <u>Location.</u> In addition to the behavioural categories, a record was kept of the location of the animals within the test containers. This enabled assessment of the location in relation to the pollution gradient as well as providing some measure of the activity level by recording the number of boundary crossings within the sampling period.

The behavioural responses of individual *G.pulex* according to the above categories were recorded before and after the introduction of the toxicant.

Test animals were acclimatised in the test trays for one hour before the study. This was followed by a sampling period of 40 minutes before the pollutant was added, followed by another sampling period of 40 minutes, after which the animals were returned to clean water and not used again. Within the sampling period, each shrimp was observed alternately for a sampling interval of 2 minutes and location within the tray, frequency of nosing behaviour, length of time of burrowed behaviour and feeding were recorded.

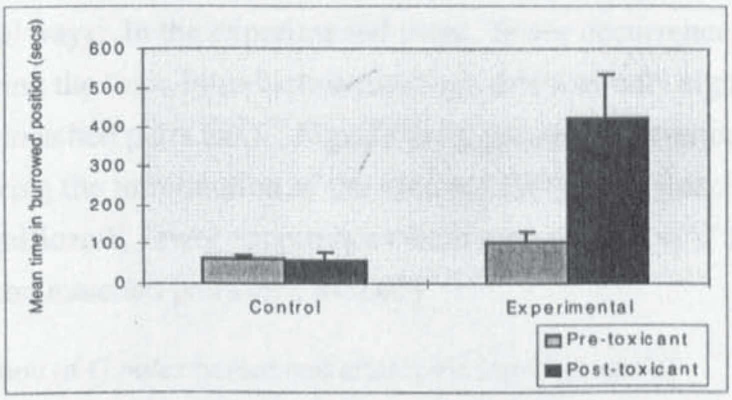
Recording of behaviour was initially carried out using a video recorder. This had the advantage of recording a number of replicate experimental and control conditions at the same time. However the resulting definition on the screen was not sufficient to enable the detailed assessment of the behavioural response to be made. This necessitated sequential replication and direct observation of behavioural changes. Replication was sequential in time as only 2 shrimps (one impacted, one control) could be observed during the experiment. The experiment was repeated 12 times. The location of the tray dosed with toxicant was randomised between replicates.

7.4. Results

7.4.1. Burrowed Position

Figure 7.2 shows the amount of time in the burrowed position by *G.pulex* before and after a toxicant was introduced. The control group shows no difference in time spent in burrowed position before and after introduction of 'toxicant' (water). The experimental group shows a significant increase in the amount of time spent in the burrowed position following introduction of toxicant (Wilcoxon matched pairs test, $p<0.02$).

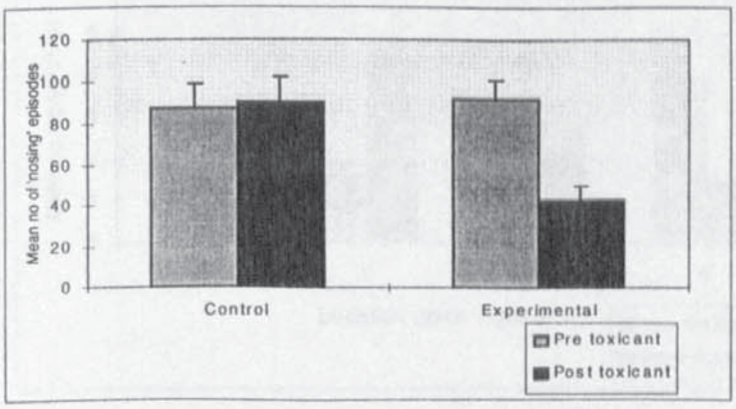
Figure 7.2: Burrowing behaviour of *G.pulex* before and after indroduction of a toxicant (means ± S.E.)



7.4.2. Nosing episodes

Figure 7.3 shows the number of nosing episodes before and after a toxicant was introduced. The null hypothesis is that there is no difference in the frequency of nosing behaviour before and after the introduction of the toxicant. This is found in the control trays. In the experimental trays, there is a significant reduction in the amount of nosing behaviour following the introduction of the toxicant (Wilcoxon matched pairs test, $p<0.01$ level).

Figure 7.3: Nosing behaviour of *G.pulex* before and after introduction of a toxicant (means ± S.E.)



7.4.3. Location

The location of *G.pulex* at the beginning of each 2 minute sampling interval was recorded. The mean number of times *G.pulex* individuals are located in each section of the test tray before and after introduction of the toxicant is shown in Figure 7.4a (controls) and Figure 7.4b (experimental). The null hypothesis is that *G.pulex* in control and experimental conditions would show similar location changes following introduction of a toxicant. It was expected that, if avoidance behaviour had occurred, there would be a greater number of occurrences in the tray sections furthest removed from the section into which the toxicant had been introduced (Section 'a'). No difference in the location of shrimps before and after the introduction of a toxicant was seen in the control trays. In the experimental trays, fewer occurrences were seen in section 'a' following the toxic introduction although this was only significant at the 10% level (Wilcoxon matched pairs test). Significantly greater occurrences were evident in Section 'b' following the introduction of the toxicant (Wilcoxon matched pairs test, $p<0.01$), and significantly fewer occurrences were seen in section 'd' following the toxicant (Wilcoxon matched pairs test, $p<0.05$)

Figure 7.4a: Location of *G.pulex* before and after toxic input (controls)

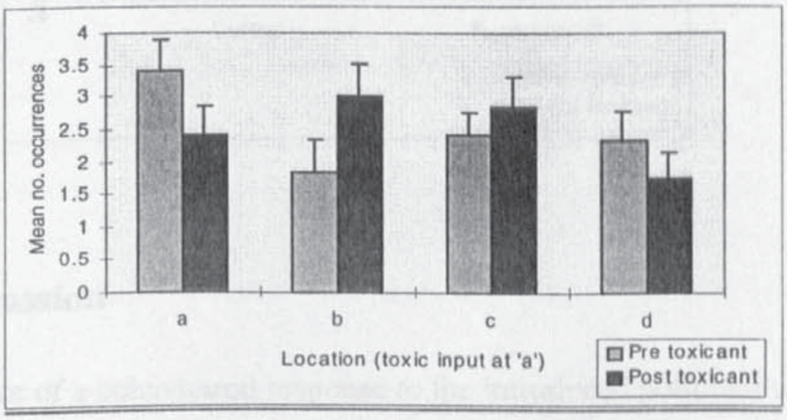
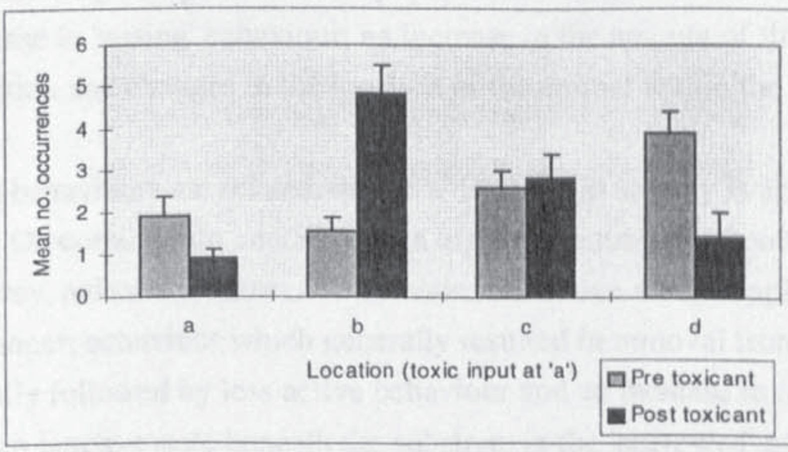


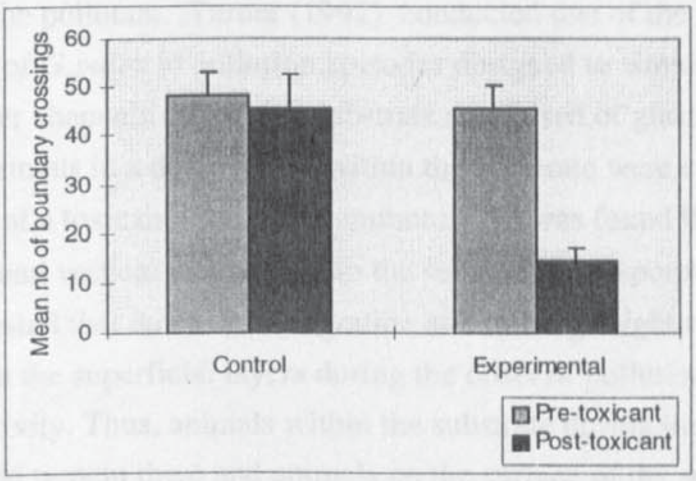
Figure 7.4b: Location of *G.pulex* before and after toxic input (experimental)



7.4.4. Activity levels

An estimate of the activity levels of *G.pulex* can be obtained by counting the number of times an individual crosses from one section of the tray to another within a sampling period. The null hypothesis of no significant difference in activity levels before and after the introduction of the toxicant is seen in the control trays. In the experimental trays there is a significant reduction in the number of boundary crossings after the introduction of a toxicant (Wilcoxon matched pairs test $p<0.01$).

Figure 7.5: Number of boundary crossings by *G.pulex* before and after introduction of a toxicant (means ± S.E.)



7.5. Discussion

Clear evidence of a behavioural response to the introduced pollutant was seen in *G.pulex*.

This comprised a range of quantifiable aspects of behaviour: a decrease in activity levels; a decrease in 'nosing' behaviour; an increase in the amount of time spent in the burrowed position and changes in the location of the animal within the container.

The first three behaviours are related, in that a decrease in activity is apparent following the pollution. On coming into contact with a high concentration of pollutant in section 'a' within the tray, animals were seen to convulse and move around rapidly in a non-directional manner, behaviour which generally resulted in removal from this section. This was usually followed by less active behaviour and an increase in the amount of time spent in an inactive state beneath the substrate in the 'burrowed' position. The increase in occurrence of individuals in section 'b' may have been related to the fact that the increase in the burrowed position was most likely to occur once initial reactions had resulted in removal of the individual from an area of high toxicity (in the experimental group, 60% of the burrowing episodes occurred in section 'b' following the pollution, compared with 26% prior to the pollution).

The category of behaviour described as 'nosing' appeared to be exploratory in nature and was associated with periods of activity. Following the introduction of the pollutant, a significant decrease in this behaviour was seen which was related to the increased periods of inactivity.

The entry of the individuals into the substratum was apparently an active response to the presence of the pollutant. Turner (1992) conducted one of the few studies to investigate the response of *G.pulex* to pollution episodes designed to simulate field conditions. Flowing water channels containing substrate composed of glass beads were used and the number of animals in a drift net and within the substrate were counted following introduction of a toxicant (unionised ammonia). It was found that there was an increase in both drift and vertical migration into the substrate in response to the toxicant. Turner (1992) suggested that downward migration and drifting might result from the location of individuals in the superficial layers during the onset of pollution episodes coupled with increased activity. Thus, animals within the substrate during the onset of the pollution episode would remain there and animals on the surface of the substrate would drift. The present study was able to focus on the reaction of single individuals to a polluted situation and found that the response consisted of active entry into the substrate rather than passively remaining within the substratum during the pollution episode.

G.pulex is found in a wide variety of both still and running waters (Gledhill *et al.*, 1993). Although the present study utilised static conditions, given the range of microhabitats and flow regimes within running waters, the findings are likely to be more generally applicable.

The initial convulsive activity shown on contact with the pollutant, may have resulted in entry of the animal into the drift if flowing water conditions had been used in the study. It is not possible from this study to assess the relative importance of drift and migration into the substrate in avoidance of suboptimal conditions. It is likely that a number of factors would contribute towards the particular avoidance reaction taken by an individual, for example current speed and substratum type.

This study demonstrates the importance of providing the test individual with conditions that are conducive to survival as far as possible. The simplicity of the test methodology enables the assessment of the behaviour patterns of a variety of test species to be carried out. In addition a wider range of pollutants and substrate characteristics could be investigated to gain further understanding of the ecologically relevant factors contributing to the repertoire of behavioural responses to adverse conditions.

It has been demonstrated that movement into the substrate, a behaviour that has been observed to occur during situations of high and low flow, forms part of the repertoire of avoidance behaviour of *G.pulex* to adverse water conditions. Consideration of this finding in relation to the pollution incident studies suggests that a transient pollutant, whilst resulting in the mortality of a proportion of the benthic macrofauna, may also elicit behavioural responses which may be adaptive in that they result in decreased exposure to the toxicant and subsequent survival. This differential survival must therefore be considered when measuring structural changes in the macroinvertebrate community following short term pollution although at present, the degree to which this occurs is not known.

7.6. Conclusions

The major findings from this part of the study are that:

1. *Gammarus pulex* shows a behavioural response to adverse water conditions by migration into the substratum.

2. Encounters with toxic conditions resulted in a decrease in activity as measured by movement through a container and exploratory movements within the substratum.

3. Some evidence of movement away from high concentrations of toxicant was demonstrated and the behavioural means by which this was achieved appeared to be a rapid increase in convulsive activity in areas of high concentration followed by a decrease in activity when more conducive conditions were encountered.

Chapter 8

Discussion

8.1. Introduction

This chapter brings together the findings presented in Chapters 6 and 7, considers their practical application and indicates possible directions for future research.

The assessment of impact and recovery following discrete pollution events, through measurement of the structural aspects of the benthic macroinvertebrates, is discussed in Sections 8.2 . In Section 8.3, the concept of recovery is considered and a continuum of recovery is described, with different biotic measurements, giving different time scales of recovery.

A new biotic score is introduced in Section 8.4.: the Recovery Index, which gives different weights to species according to their relative ability to survive and colonise an area following a discrete pollution incident. This was designed to give a conservative estimate of recovery according to the composition of macroinvertebrate species present. Section 8.5 deals with the CIMAH regulations and gives suggestions for background survey information required in an area of high risk, as well as using the existing biological community to indicate the likely impact and recovery following a pollution incident. The chapter concludes with Section 8.6, in which suggestions are made for future work in this area and ways in which future developments would be most profitable.

8.2. Biological response to pollution incidents

"There is a tendency on the part of science to suppose that nothing happens until it perceives it " (Andrews, 1992).

All the pollution incidents investigated (except Callow Brook) showed an initial impact on the biota, in that fish and/or macroinvertebrate mortality was evident. Following certain pollution incidents, subsequent measurement of the biotic response revealed no clear impact on the biota. This may have been due to a variety of factors (discussed in Section 6.4.), and although this study can not attribute importance to these factors, their consideration and discussion may contribute to an understanding of the conditions under which biological monitoring of pollution incidents is appropriate.

8.2.1. Sampling methodology

Numerous authors have emphasised the importance of taking an adequate number of samples in the quantitative assessment of macroinvertebrate community response (e.g. Elliott, 1977). However, much biomonitoring is carried out under cost and time constraints, therefore limiting the amount of sampling that can be carried out (Resh and McElravy, 1993). The present study utilised stratified sampling, whereby 3 samples were collected from a defined habitat type within the watercourse (riffles). The number of samples taken represents the minimum of effort required for quantitative sampling. The finding that differences between replicates taken within a sampling point were generally less than the differences between replicates taken at different sites or times, indicates that this approach can still give a valuable assessment of the community at the site. However, subtleties in changes of the abundance of a particular taxon may not be apparent.

Two other points need to be highlighted in association with a sampling programme that comprises a relatively small number of replicates: firstly, the abundance levels of the less common taxa may not be high enough for statistical comparisons to be carried out, and secondly, the sampling may miss the presence of rarer taxa. These two points may be particularly pertinent to streams that are rich in taxa. In these instances, an increase in the number of replicates would give greater information on changes in the abundance of the less common taxa, thus providing greater sensitivity to environmental changes.

8.2.2. Level of identification

The present study utilised identification to species level for most taxonomic groups. However, the number of taxa in streams suffering from chronic pollutant input are often dominated by Oligochaeta and Chironomidae, which were not identified further than subclass and family level. A number of studies have shown that Chironomidae, in particular, comprise species showing a range of environmental requirements (see Section 6.4.1.4), such that a more defined response may have been possible if identification to species level had been carried out. The Oligochaeta found in the study belonged primarily to the family Tubificidae. However, even within this family species may differ in their requirements (Section 6.4.1.8).

In summary, although identification to species level is the ideal situation (Resh and Unziger, 1975), identification of certain groups can be difficult and time-consuming. For example, in spite of their demonstrated value in the assessment of pollution (Wilson, 1987; 1994), species level identification of Chironomidae has not been widely adopted in biomonitoring. However, the present study has demonstrated the limited biological impact of a number of pollution incidents in streams dominated by a pollution tolerant fauna. In these situations identification of the Oligochaeta and Chironomidae to a higher level may be the only appropriate approach towards biological assessment of impact in spite of the greater cost and time involved.

8.2.3. Attenuation of pollutant /Avoidance response

The heterogeneity of habitat and hydrological patterns within a watercourse can result in a situation where patches are exposed to differential impact from a transient pollution event. This would result in differential exposure of invertebrates to the toxicant according to their location within these patches, such that some individuals may survive to provide sources of colonists for impacted patches. In addition, it has been shown (Chapter 7) that active avoidance of a toxic pollutant can occur.

In the field situation, it is difficult to determine the relative importance of active and passive avoidance of a pollution incident, however, situations where this might occur would result in rapid recovery as the community would suffer reduced mortality. This may have been the case in one of the incidents, Canley Brook (Section 5.3), where some invertebrate mortality following a pollution incident was evident. However, no apparent impact was found on the macroinvertebrate community when the benthos was sampled. In this instance, rapid colonisation from upstream areas was not likely as a large proportion of the upstream channel was culverted. Colonisation from tributaries to the most impacted sites was also not a possibility, thus suggesting that some survival of the community had occurred.

8.2.4. Limited/Adapted fauna

A number of the pollution incidents investigated in the study took place in watercourses that were subject to a high degree of chronic and intermittent pollution impacts and these streams showed little impact or rapid recovery of the macroinvertebrate community. The benthic fauna within these streams may have been previously exposed to a high degree of disturbance in terms of intermittent pollution events which would thus result in a limited fauna (Payne, 1989). In addition, there is some evidence that invertebrate populations may show a genetic adaptation in their tolerance of and response to pollutants (Klerks and Weis, 1987). In the incident investigated in Canley Brook (Section 5.3), the pollutant was suds oil, which had also been responsible for most of the previous pollution incidents within the brook. It is therefore conceivable that some genetic adaptation may have occurred.

Most taxa living in stream communities show adaptive responses to disturbance at various levels (see Section 2.4.1), and although it is possible to consider that a community can be 'adapted' to a situation of high disturbance, this may simply be the manifestation of selective forces operating to favour those taxa showing appropriate characteristics: e.g. high fecundity, multivoltine capacity etc. (Poff, 1992).

8.3. Biological Assessment of Recovery

8.3.1. Relative utility of Biological measures

Recovery can be defined as:

> "the return to an ecosystem which closely resembles unstressed surrounding areas"(Gore, 1985), or "pre-disturbance levels" (Gore, 1990) (See Section 2.4.1).

The biotic measures used in this study can be compared, to ascertain which are the most sensitive in the assessment of recovery: i.e. which show impact for the longest period of time.

The data did not permit the strict use of statistical similarity in all cases thus necessitating subjective judgements. However, there is a pattern in the relative sensitivities of different measures that emerges from a number of the pollution incidents studied.

It was predicted that the pollution incidents would result in initial removal of taxa from the most impacted sites and this would then be followed by the return of these taxa over a period of time. Thus, following an incident, there would be an absence or low abundance of selected taxa, low taxa richness, low diversity index and low biotic scores. This would be followed by an increase in the abundance of the taxa selected for measurement, in taxa richness, diversity index and biotic scores (BMWP and ASPT scores), until a reference level was reached which would indicate the recovery of the impacted sampling point.

Although, in general, this pattern was found to occur following the pollution incidents, the different measures used indicated a more complex and less predictable process. Firstly, a number of the incidents showed no measurable impact on the biological measures used. This finding is considered in Sections 8.2.3 and 8.2.4. Secondly, although an absence or lower abundance of taxa was found following a number of the pollution incidents, the subsequent changes in the taxa varied in terms of timing of the reappearance and abundance changes. In addition, some taxa showed a stimulatory response following certain pollution incidents.

In some cases, consideration of selected taxa revealed impacts that extended for some distance downstream of the incident and were apparent for a longer period of time than other assessments of impact; the cased caddis larvae *Agapetus fuscipes* in the River Wye (Section 5.12) and *Goera pilosa* in the Preston Bagot Brook (Section 5.9).

The variation in response between taxa was not necessarily reflected in the univariate measures. The Shannon Wiener diversity index was found to be relatively insensitive to changes following the pollution incidents (see Section 6.4.4), with changes in the values often being the result of changes in the abundance of a single taxon. The BMWP scores and ASPT scores were frequently found to be influenced by single, possibly drifting, individuals and to reflect the fact that the taxa most tolerant to organic pollution were also the taxa most likely to survive or be present immediately following a pollution incident (see Section 6.4.5). As recovery progressed, these biotic scores were not necessarily the best reflectors of community change, particularly if the changes were largely abundance changes in particular taxa rather than the colonisation of those taxa showing greater sensitivity to organic pollution.

Taxa richness, measured as the number of taxa within a sampling point, was found to be a relatively simple and sensitive measure of the changes following the pollution incidents (Section 6.4.3). Its advantage is that it makes no assumptions about the relative sensitivities of the taxa or the 'ideal' abundance distribution of the taxa. Its disadvantage is that it does not take account of temporal taxon replacements and variations in abundance. It has been suggested that it may have greater utility as a measure of recovery in streams that do not have a high proportion of seasonally occurring taxa (Section 6.4.3).

The multivariate approach to the analysis of the community (see Section 6.4.7) involved three predictions of response of the impacted communities, two of which involved changes in comparison with a reference site:

- as time elapsed, the communities at the impacted sites were predicted to show a progressive increase in similarity to the reference site, such that the criteria of no significant difference in communities would provide a measure of recovery;

- impacted communities were predicted to show greater variability than the reference sites with progressively less variability over time as the sites recovered. This assumes that those communities that are not impacted by a pollution incident would show greater stability or community similarity over time (Clarke and Warwick, 1994);

- the communities within the impacted sites were predicted to show progressively greater similarity to each other, with samples taken within a short time of the pollution incident showing greater dissimilarity than samples taken some time after the pollution incident.

The first prediction proved the most useful in the assessment of changes following pollution impact, with certain incidents showing a clear progression in similarity over time compared with reference conditions. Following some incidents, multivariate consideration of the community changes gave a longer time-span to the assessment of recovery than the univariate measures.

The second prediction, of greater variability in community composition over time at the impacted sites than the reference sites, was seen in some instances, although this measure did not provide more information than univariate measures. However, any greater variability found may be related to the respective community composition at the sites; the impacted sites supporting a relatively impoverished community with taxa that are variable in their abundance over time and space, whilst the reference site is more diverse with fewer dominant taxa. If the data is not transformed (as described in Section 4.3.5) the use of community similarity as an assessment of changes *within* an impacted site would largely be a reflection of the variability in abundance that is shown by a few rapidly colonising taxa. However, the use of the measure of stability and variability in community composition has not been widely applied in stream ecosystems and may simply be a reflection of the variability in abundance of the 'pioneer' taxa, in which case transformation would result in the loss of this information. Another problem with this prediction of post pollution impact in communities is that it relies upon visual examination of the MDS plots and subjective assessments of relative variability.

It is therefore concluded that an assessment of the relative increase or decrease in variability of community composition between sites is not the most appropriate measure for the assessment of impact following pollution incidents.

The third prediction of greater clustering of communities over time within impacted sites was only found at one of the pollution incidents investigated, Preston Bagot Brook, where it was seen to varying extents in three of the eight impacted sites. This measure may be more useful if a greater number of sampling times is used and if the sampling times are more evenly spaced than was generally the case in this study.

In summary, the multivariate approach to assessment of impact and recovery is a valuable tool which is often more appropriate as a summary of the relative differences in community composition between two sites than the univariate indices since a greater proportion of the information that has been collected is retained in the analysis. If appropriate reference conditions are available for comparison, then the progressive change in impacted communities over time relative to the reference conditions was found to be the most powerful method of assessment of temporal recovery. Of the univariate indices, the Number of Taxa appears to be the most appropriate measure of change.

8.3.2. Patterns of Recovery

A pattern of recovery that has been seen following a number of the pollution incidents studied is that of the order of appearance of certain taxa and, in a number of instances, an increase in the abundance of selected taxa.

Oligochaeta were consistently found to be the first taxon to colonise a site following the pollution incidents investigated, with individuals appearing within a few days (Section 6.4.1.8). Similarly rapid colonisation was also shown by Chironomidae and by *Potamopyrgus jenkinsi* following some incidents (Sections 6.4.1.4., 6.4.1.9). *Asellus aquaticus* showed rapid recolonisation, but generally this species was found to colonise sites after Chironomidae and Oligochaeta. Subsequent colonisers also included *Gammarus pulex* and *Baetis rhodani* (Sections 6.4.1.6., 6.4.1.3).

A number of taxa showed peaks in abundance following the pollution incidents which were greater than those found at the reference sites. This was shown by Oligochaeta, Chironomidae, *Potamopyrgus jenkinsi, Lymnaea peregra* and *Asellus aquaticus* (Section 6.4.1).

Taxa showing slower colonisation within the timescale of the study were *Goera pilosa* and *Agapetus fuscipes* (cased caddis larvae) (6.4.1.1), although if samples had been taken over a longer period of time following the incident, it is possible that further colonisation by other taxa would have been seen. The colonisation of the snail *Lymnaea peregra* appeared to be relatively slow if colonisation sources were not close by (6.4.1.7).

In terms of the community measures, the diversity index was the least sensitive to changes following pollution impact, thus the diversity of the community could be described as having recovered first (Section 6.4.4). ASPT was often the next measure to show recovery (Section 6.4.5). Of the univariate measures, BMWP scores and taxa richness both showed the most sensitivity in recovery assessment (Sections 6.4.3., 6.4.5). This similarity is to be expected, given the taxon richness component within the BMWP score.

Multivariate analysis of similarity, particularly if the data were transformed in order to decrease the relative contribution of the most abundant taxa, appeared to be the most sensitive measure of community recovery (Section 6.4.7).

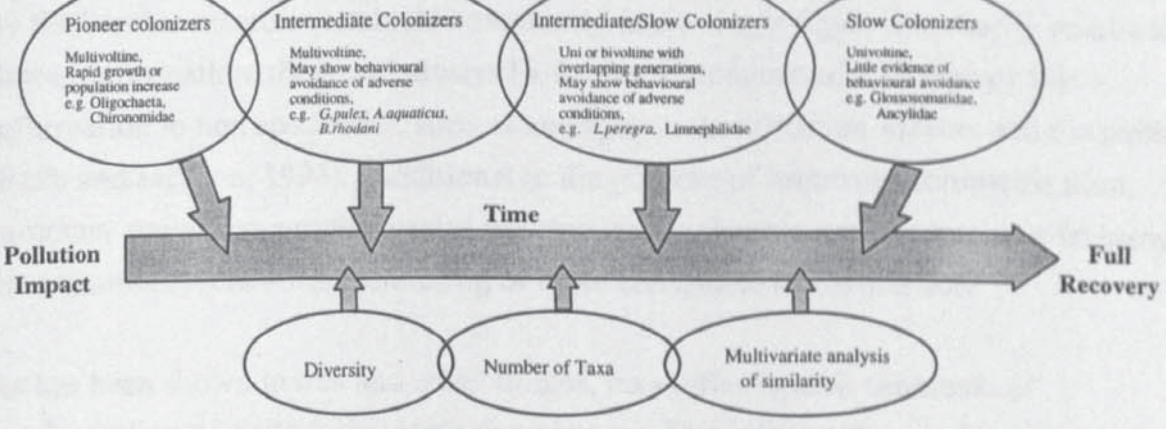
8.3.3. Recovery continuum

The findings of differential recovery times among measures of the biota, can be illustrated as a 'recovery continuum', whereby different aspects of the community are seen to recover at different times (see Fig. 8.1.). The location of the measures refers to relative, rather than absolute times, of recovery.

The individual taxa can be divided into general groups according to their life history characteristics, dispersal abilities and avoidance potential of adverse conditions. The taxa showing rapid growth, multivoltine life history and high dispersal are likely to colonise an impacted area first, and the last to colonise an area are those taxa with low dispersal abilities, uni- or semivoltine life cycles and no ability for behavioural avoidance. Information concerning dispersal abilities and life cycles is available in the literature for many taxa (see summaries in Section 6.4.1). Less information is available on the ways in which different taxa avoid adverse conditions.

The finding of differential temporal recovery rates among the community measures used in the present study can also be illustrated in the continuum. Thus, Diversity shows recovery before Number of Taxa, and the last of the community measures to show recovery is the multivariate analysis of similarity. BMWP and ASPT scores are not included for the reasons mentioned in Section 8.3.1: they do not supplement information provided by the Number of Taxa and may indeed, due to the limitation of the level of identification to family level, provide less information. Secondly, scores following a pollution incident may purely be depressed by the presence of the initial colonisers - the Chironomidae and the Oligochaeta.

Fig. 8.1: Recovery Continuum - to illustrate the relative recovery times of different measures of the macroinvertebrate community



The recovery continuum is a concept that illustrates the *value judgements* involved in the assessment of recovery. Recovery endpoints can be chosen at various points along recovery endpoints, and the most appropriate measure for monitoring this endpoint can be chosen.

It must be acknowledged that macroinvertebrates are being used as *indicators* of ecosystem recovery. For a 'full recovery' to have occurred, i.e.

> "the return to an ecosystem which closely resembles unstressed surrounding areas"(Gore, 1985),

complete understanding of the structure and functioning of an ecosystem is required. There are very few ecosystems in which this situation has been achieved (Fryer, 1987), therefore the most sensitive measures of macroinvertebrate recovery, i.e. those which give the longest timescale to recovery are preferable for practical assessments. Thus, the multivariate analysis of similarity would be the most appropriate approach to community analysis. However, although more sensitive than the univariate measures used, the multivariate approach still does not make full use of the species/abundance data, the collection of which involves a high proportion of the cost of biomonitoring. A method of utilising the macroinvertebrate information is required that would be more sensitive to changes on the recovery continuum.

8.4. The Recovery Index

8.4.1. Introduction

Although the process of summarising the large amount of information that is provided by the benthic macroinvertebrate community into a single figure necessarily results in a loss of information, there will always be the need for summaries to convey this information to non specialists, such as managers, other decision-makers and the public (Resh and Jackson, 1993). Additional to the purpose of improved communication, summary scores can provide useful information on changes within particular features of the community, enabling monitoring of these changes to be carried out.

As has been shown in this and other studies, taxa differ in their timescale of recolonisation of a site following a disturbance. This information can be utilised to produce an index which summarises the extent to which a community has recolonised a particular site following a pollution incident. Thus, taxa with high colonisation abilities (i.e. those described as pioneer taxa on the recovery continuum presented in Section 8.3.3), would have the lowest 'Recovery Score', and taxa with low colonisation abilities would have the highest Recovery Score. The resulting score would give definition to the recolonisation potential of individual taxa. The Recovery Scores of all the taxa within a community could be combined to give a 'Recovery Index' which could be used as a measure of the state of recovery of a particular community relative to a reference community.

8.4.2. Terms used to derive the Recovery Index

The three most important factors that influence the differing recolonisation times of the taxa following a pollution incident were highlighted in Section 6.4.2:

1. the dispersal of a taxon during its lifetime;
2. the reproductive life cycle of the taxon;
3. the survival of the taxon following a discrete pollution episode.

Information about each of these factors can be estimated from the literature and from observations made during the course of the present study and can be used to give an indication of the potential of a taxon to recolonise an area within a stream following a discrete pollution episode.

The following terms are evaluated for each taxon:

Movement within lifetime (M) - This term is designed to represent the dispersal abilities of the organism (i.e. the movement of the organism during its lifetime) on a scale of 1 to 3. Thus taxa showing relatively large amounts of movement, e.g. *Gammarus pulex*, would score 1, and taxa that are largely sedentary e.g. *Ancylus fluviatilis*, the river limpet, would score 3. Where different life stages show different dispersal abilities, for example insects with aerial adult stages, the stage showing the greatest dispersal is used to derive the score. Thus taxa with a high score have a low dispersal ability and would be expected to show slower colonisation of a disturbed area.

M_i = degree of movement of i^{th} taxon during lifetime

- 3: (little movement)
- 2: (some movement),
- 1: (rapid movement).

Life Cycle Strategy (L) - This term refers to the reproductive strategy of the taxon in terms of the number of generations per year, and taxa are again given a score of 1 to 3. Taxa showing a multivoltine capacity and a capacity for asexual reproduction, e.g. Oligochaeta, would score 1 and would be expected to show rapid colonisation of an area. Taxa showing univoltine or semivoltine life history with no overlap of generations score 3 and would be expected to show slower colonisation of an area.

L_i = life cycle strategy of ith taxon

- 3: (univoltine or more than one year to complete life cycle),
- 2: (bi voltine),
- 1 : (multivoltine, or showing asexual reproduction).

Avoidance Capacity (A) - The basis for the inclusion of this term lies in the finding within the present study that a taxon can show active avoidance of adverse conditions, e.g. by burrowing into the substratum. The ecological basis for this behaviour has been described earlier (Section 7.1) and can be summarised as:

> 'a collection of behaviours, shown in response to disturbance, that increase the probability of survival'.

The term refers to the ability of a taxon to display avoidance of a short-term deterioration in water quality, and, as with the other terms, taxa are allocated a score of 1 to 3. For example, animals showing burrowing behaviour, were presumed to have the capacity to burrow in response to adverse conditions. Pulmonate snails, which breathe atmospheric air, were given a score of 1 as they have the capacity to avoid suboptimal conditions by crawling out of the water: planorbid snails do not have the same capacity and are therefore given a score of 3. Due to the general lack of information regarding these adaptive behaviour patterns, scores have been largely subjectively derived with some information provided from the literature (see Appendix E).

A_i = Avoidance capacity of ith taxon

- 3: (low avoidance capacity),
- 2: (some avoidance capacity),
- 1: (high avoidance capacity).

Table 8.1 lists a range of invertebrate taxa with scores for each of the terms described above. These scores are based on literature concerning life history characteristics of the organisms and on recorded responses to disturbances, both anthropogenic and natural in origin, as well as on findings from the present study (see Appendix E for a list of the references used). If the information for a particular term indicates that a taxon falls into two categories, for example if a taxon is described as both univoltine (L score = 3) and bivoltine (L score = 2), then an average score is allocated (L score = 2.5). Where information was not available, a subjective decision was made based upon personal experience. A species that generally has one generation a year may show different life histories according to environmental conditions or geographic location (Elliott *et al.*, 1988). The values of M, L and A assigned to a species must therefore remain representative of general conditions, and most importantly, *relative* to the values given to other species.

Scores have been assigned at the species level for the majority of orders, but generally correspond to the level of identification most widely used. For example, identification of Oligochaeta remains at the 'Class' level. If the re-establishment of the full structural complexity of an impacted site is defined as the desired recovery endpoint, as has been suggested in Section 8.3.1, then identification should be to species level whenever possible.

The scores refer to running waters and assume that there are sources of colonisation within the watercourse (either upstream of the incident or unaffected tributaries). A different score system would have to be adopted for lentic waters, as certain taxa that colonise areas within streams rapidly, (e.g. the amphipod shrimp; *Gammarus pulex*), would show slower colonisation of ponds. In lentic habitats, the relative colonisation abilities of those taxa with aerial life stages, for example mayflies, would be greater.

8.4.3. The Recovery Index

The three terms described above each provide some information regarding the recolonisation abilities of a particular taxon and can be combined to provide a summary score for each taxon, or a 'Recovery Score' (see Table 8.1):

Recovery score (RS) = $M_i \times L_i \times A_i$ *Equation 8.1*

Where i = i^{th} taxon in a sample

To produce a Recovery Index (RI) for a particular community, the scores obtained for each taxon within a sample are added as follows:

$$\text{Recovery Index (RI)} = \sum_{i=1}^{n} (M_i L_i A_i) \qquad \textit{Equation 8.2}$$

Where i = ith taxon in a sample
n = number of taxa in sample

As an illustration of the operation of the Recovery Index, Table 8.2 gives hypothetical lists of taxa from 2 sites (taken from the reference sites within the study data), with associated Recovery Scores and Indices. It can be seen that, although Site A has only two more taxa than Site B, it has a higher Recovery Index than Site B. Site A would therefore be predicted to support a community that will take longer to recover when impacted by a pollution incident than Site B.

Table 8.2: Taxa lists from 2 sites and associated Recovery Scores and Indices

Site A		Site B	
Taxon	Recovery Score	Taxon	Recovery Score
Dendrocoelum lacteum	12	*Polycelis felina*	4
Lymnaea peregra	6	*Polycelis tenuis/nigra*	4
Planorbis albus	27	*Lymnaea peregra*	6
Planorbis vortex	27	*Ancylus fluviatilis*	27
Ancylus fluviatilis	27	Oligochaeta	6
Sphaeriidae	12	*Glossiphonia complanata*	12
Oligochaeta	6	*Glossiphonia heteroclita*	12
Glossiphonia complanata	12	Erpobdellidae	6
Asellus aquaticus	2	*Asellus aquaticus*	2
Gammarus pulex	1	*Gammarus pulex*	1
Baetis rhodani	1	*Baetis rhodani*	18
Hydropsyche angustipennis	27	*Ephemera danica*	18
Athripsodes sp.	9		
Limnephilus extricatus	12		
RECOVERY INDEX	**181**	RECOVERY INDEX	**116**

Table 8.1: Recovery Scores for Selected Macroinvertebrate Taxa

Invertebrate Taxa	M	L	A	Recovery score
PLATYHELMINTHES				
Planariidae				
Dugesia lugubris (Schmidt)/*polychroa* (Schmidt)	2	1	2	4
Polycelis felina	2	1	2	4
Polycelis nigra (Muller)/*tenuis* (Ijima)	2	1	2	4
Dendrocoelidae				
Dendrocoelum lacteum (Muller)	2	3	2	12
MOLLUSCA				
GASTROPODA: PROSOBRANCHIA				
Valvatidae				
Valvata piscinalis (Muller)	2	2	3	12
Hydrobiidae				
Bithynia tentaculata (L.)	2	2	3	12
Potamopyrgus jenkinsi (Smith)	2	1	3	6
GASTROPODA: PULMONATA				
Lymnaeidae				
Lymnaea palustris				
Lymnaea peregra (Muller)	2	3	1	6
Lymnaea stagnalis	2	3	1	6
Physidae				
Physa fontinalis (L.)	2	3	3	18
Planorbidae				
Planorbis albus Muller	3	3	3	27
Planorbis carinatus	3	3	3	27
Planorbis contortus (L.)	3	3	3	27
Planorbis planorbis (L.)	3	3	3	27
Planorbis vortex (L.)	3	3	3	27
Ancylidae				
Ancylus fluviatilis Muller	3	3	3	27
Bivalvia				
Unionidae				
Anodonta sp.	3	3	2	18
Sphaeridae				
Sphaerium sp.	3	2	2	12
Pisidium sp.	3	2	2	12
OLIGOCHAETA	3	1	2	6
HIRUDINEA				
Piscicolidae				
Piscicola geometra (L.)	1	3	2	6
Glossiphoniidae				
Hemclepsis marginata (Muller)	1	3	2	6
Glossiphonia heteroclita (L.)	2	3	2	12
Glossiphonia complanata (L.)	2	3	2	12
Helopbdella stagnalis (L.)	2	3	2	12
Theromyzon tessulatum (Muller)	2	3	2	12
Erpobdellidae				
Erpobdella octoculata (L.)	1	3	2	6
Trocheta bykowskii Gedroyc	1	3	2	6

Table 8.1: Recovery Scores for Selected Macroinvertebrate Taxa (continued)

Invertebrate Taxa	M	L	A	Recovery Score
ARTHROPODA				
ARACHNIDA:HYDRACARINA	2	1.50	2	6
CRUSTACEA: MALACOSTRACA: ISOPODA				
Asellidae				
Asellus aquaticus (L.)	1	1	2	2
Asellus meridianus Racovitza	1	1	2	2
CRUSTACEA: MALACOSTRACA : AMPHIPODA				
Gammaridae				
Crangonyx *pseudogracilis* Bousfield	1	1	1	1
Gammarus pulex (L.)	1	1	1	1
CRUSTACEA : MALACOSTRACA: DECAPODA				
Astacidae				
Austropotamobius pallipes (Lereboullet)	1	3	2	6
INSECTA: EPHEMEROPTERA				
Baetidae				
Baetis buceratus Eaton	1	2	1	2
Baetis muticus (L.)	1	2	1	2
Baetis rhodani Pictet	1	1	1	1
Baetis scambus Eaton	1	2	1	2
Baetis vernus Curtis	1	2	1	2
Centroptilum luteolum (Muller)	1	2	1	2
Cloeon dipterum (L.)	1	2	1	2
Heptageniidae				
Ecdyonurus dispar (Curtis)	2	3	3	18
Ecdyonurus insignis (Eaton)	2	3	3	18
Rithrogena semicolorata (Curtis)	2	3	3	18
Heptagenia sulphurea (Muller)	2	1	3	6
Ephemerellidae				
Ephemerella ignita (Poda)	2	2.5	3	15
Ephemeridae				
Ephemera danica Muller	3	3	2	18
Caenidae				
Caenis horaria (L.)	3	3	2	18
Caenis luctuosa Burmeister	3	2.5	2	15
Caenis robusta Eaton	3	2.5	2	15
Caenis rivulorum Eaton	3	3	2	18
INSECTA: PLECOPTERA				
Nemouridae				
Nemoura sp.	3	3	3	27
Leuctridae				
Leuctra fusca (L.)	3	3	3	27
Perlodidae				
Isoperla grammatica (Poda)	2	3	2	12

Table 8.1: Recovery Scores for Selected Macroinvertebrate Taxa (continued)

Invertebrate Taxa	M	L	A	Recovery score
INSECTA : ODONATA				
Coenagriidae				
Coenagrion puella (L.)	2	3	3	18
Ischnura elegans (van der Linden)	2	3	3	18
Agriidae				
Agrion splendens (Harris)	2	3	3	18
Calopterigidae				
Calopteryx sp.	2	3	3	18
INSECTA : HEMIPTERA, HETEROPTERA				
Nepidae				
Notonectidae				
Notonecta glauca L.	1	3	1	3
Corixidae				
Sigara dorsalis (Leach)	1	3	1	3
Sigara falleni (Fieber)	1	3	1	3
INSECTA : COLEOPTERA				
Haliplidae	2	3	2	12
Brychius elevatus (Panzer)	2	3	2	12
Dytiscidae	2	3	2	12
Deronectes depressus (Fabricius)	1	3	2	6
Hydroporus marginatus (Duftschmid)	1	3	2	6
Gyrinidae	1	3	2	6
Gyrinus sp.	1	3	2	6
Elminthidae	3	3	2	18
Elmis aenea (Muller)	3	3	2	18
Esolus parallelopipedus (Muller)	3	3	2	18
Limnius volkmari (Panzer)	3	3	2	18
Oulimnius tuberculatus (Muller)	3	3	2	18
INSECTA : MEGALOPTERA				
Sialidae				
Sialis lutaria (L.)	2	3	2	12
INSECTA : TRICHOPTERA				
Rhyacophilidae				
Rhyacophila dorsalis (Curtis)	2	3	2	12
Glossosomatidae				
Agapetus fuscipes Curtis	3	3	3	27
Glossossoma boltoni Curtis	3	3	3	27
Polycentropidae				
Plectrocnemia conspersa (Curtis)	3	3	3	27
Polycentropus flavomaculatus (Pictet)	3	3	3	27
Psychomyiidae				
Psychomyia pusilla (Fabricius)	3	3	3	27
Tinodes waeneri (L.)	3	2	3	18

Table 8.1: Recovery Scores for Selected Macroinvertebrate Taxa (continued)

Invertebrate Taxa	M	L	A	Recovery Score
Hydropsychidae				
Hydropsyche angustipennis (Curtis)	3	3	3	27
Hydropsyche conturbernalis McLachlan	3	3	3	27
Hydropsyche instabilis (Curtis)	3	3	3	27
Hydropsyche pellucidula (Curtis)	3	3	3	27
Hydropsyche siltalai Dohler	3	3	3	27
Hydroptilidae				
Hydroptila tineoides Dalman	2	2.5	2	10
Ithytrichia sp.	2	2.5	2	10
Limnephilidae				
Anabolia nervosa Curtis	2	3	2	12
Halesus radiatus (Curtis)	2	3	2	12
Limnephilus decipiens Kolenati	2	3	2	12
Potamophylax cingulatus (Stephens)	2	3	2	12
Stenophylax lateralis (Stephens)	2	3	2	12
Molannidae				
Molanna angustata Curtis	3	3	2	18
Beraeidae				
Beraeodes minutus (L.)	2	3	2	12
Leptoceridae				
Athripsodes bilineatus (L.)	1.5	3	2	9
Athripsodes cinereus (Curtis)	1.5	3	2	9
Mystacides azurea (L.)	1.5	3	2	9
Mystacides nigra (L.)	1.5	3	2	9
Trienodes bicolor (Curtis)	1.5	3	2	9
Goeridae				
Goera pilosa (Fabricius)	3	3	3	27
Silo pallipes (Fabricius)	3	3	3	27
Brachycentridae				
Brachycentrus subnubilis Curtis	2.5	3	3	22.5
Sericostomatidae				
Sericostoma personatum (Spence)	2	3	2	12
Corixidae	1	1.5	1	1.5
INSECTA: DIPTERA				
Tipulidae	1	2	2	4
Psychodidae	2	1	2	4
Culicidae	1	1	2	2
Ceratopogonidae	1	1	2	2
Chironomidae	1	1	2	2
Simuliidae	1	1	1	1
Empididae	1	2	2	4
Rhagionidae	1	2	2	4
Tabanidae	1	2	2	4

8.4.4. Assessment of recovery using Recovery Index

The Recovery Index for an impacted community can be compared with that of a reference community to give an assessment of recovery - the Recovery Coefficient (RC):

$$\text{Recovery Coefficient (RC)} = \frac{\text{Recovery Index for impacted site}}{\text{Recovery Index for reference site}} \times 100 \qquad \textit{Equation 8.3}$$

The closer the Recovery Coefficient is to 100%, the greater the recovery, although, given the inherent variability of the natural world, an RC of 100% would not be expected.

To illustrate the operation of the Recovery Coefficient relative to other community measures, a hypothetical set of data showing a period of recovery over time is presented in Table 8.3. Taxa lists, in the form of presence/absence data for a reference site and four impacted sites have been compiled and the Recovery Index and number of taxa have been calculated. To compare different measures of recovery, the Recovery Coefficient, the number of taxa as a percentage of the reference and the Bray-Curtis Similarity Coefficient comparing each sample with the reference sample have been calculated. These three measures are illustrated in Fig. 8.2. It can be seen that the Recovery Coefficient presents a more conservative estimate of recovery at each impacted site in comparison with the other two measures. For example, impacted Site 4 shows 80% of the number of taxa present in the reference site, whilst the Recovery Coefficient is 69%. This is because a number of high scoring taxa (i.e. those with low colonising potential, for example *Ancylus fluviatilis* and *Polycelis felina*) continue to be absent in comparison with reference conditions.

Due to the extra weighting given to those taxa that show slower colonisation, the Recovery Coefficient would show greater definition towards the latter stages of recovery in discriminating between recovery conditions than other measures.

Table 8.3: Hypothetical taxa lists from a reference site and four impacted sites

Taxon	Presence of Taxon *				
	Reference	Impacted 1	Impacted 2	Impacted 3	Impacted 4
Polycelis felina	*				
Polycelis tenuis/nigra	*		*	*	*
Dendrocoelum lacteum	*			*	*
Potamopyrgus jenkinsi	*	*	*	*	*
Planorbis vortex	*			*	*
Ancylus fluviatilis	*				
Oligochaeta	*	*	*	*	*
Glossiphonia complanata	*			*	*
Erpobdellidae	*	*	*	*	*
Hydracarina	*		*	*	*
Gammarus pulex	*		*	*	*
Baetis rhodani	*		*	*	*
Ephemera danica	*				*
Elmis aenea	*			*	*
Hydropsyche siltalai	*				*
Athripsodes sp.	*			*	*
Drusus annulatus	*				
Agapetus fuscipes	*				
Chironomidae	*	*	*	*	*
Simuliidae	*		*	*	*
NUMBER OF TAXA	20	4	9	14	16
RECOVERY INDEX	226	20	33	111	156
RECOVERY COEFFICIENT		8.85%	14.6%	49%	69%
NUMBER OF TAXA (% OF REFERENCE)		20%	45%	70%	80%
BRAY CURTIS SIMILARITY COEFFICIENT (compared with reference)		33%	62%	82%	89%

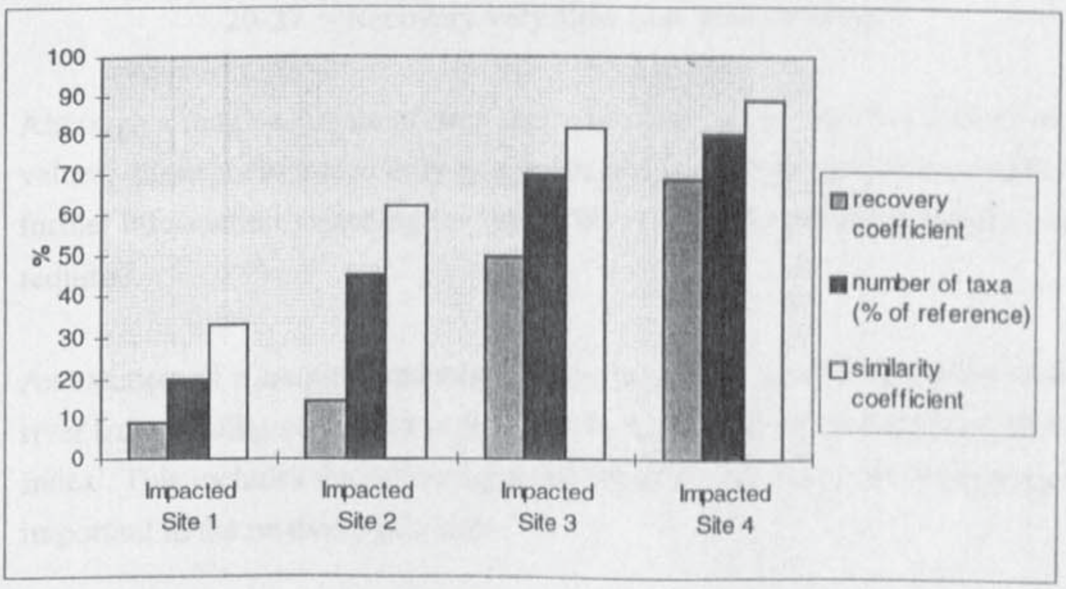


Figure 8.2: Hypothetical data set; Recovery Coefficient and Number of Taxa compared with reference, Bray-Curtis Similarity Coefficient (comparing each sample with reference sample).

8.4.5. Prediction of recovery using the Recovery Index

The Recovery Index of a sampling area can be used to predict the *relative* timing of recovery following a pollution incident. If a high proportion of the taxa in a sample have low Recovery Scores, recovery is likely to be rapid, and conversely, if a high proportion of the taxa show high Recovery Scores, recovery is likely to be slower. To obtain an Average Recovery Score per taxon, the Recovery Index can be divided by the number of taxa. This would give a score that is independent of sampling effort, and gives greatest importance to the Recovery Scores within the sample through elimination of the taxa richness component of the Recovery Index.

$$\text{Average Recovery Score (ARS)} = \frac{\text{Recovery Index of sample}}{\text{Number of Taxa in sample}} \qquad \textit{Equation 8.4}$$

Samples showing a high ARS would therefore show greater time to recolonise a site following a pollution incident than sites showing a low ARS. Preliminary ARS values to predict the relative speed of recovery could be set as follows:

ARS value = ≤ 6 - Recovery rapid (days/weeks)
7-12 - Recovery moderate (weeks/months)
13-19 - Recovery slow (months)
20-27 - Recovery very slow (one year or more)

Although a rough estimate of time scales has been given with the above values of ARS values, these are intended only as a guide and in order to predict time scales accurately, further information regarding the type and extent of the pollution impact would be required.

An example of a simple framework for the predictive assessment of the recovery of a river from a pollution impact is provided by Cairns (1990) in the 'Ecosystem Recovery Index'. This includes the following terms which relate to the factors considered most important in the recovery of a site:

1. existence of nearby epicentres for providing organisms to reinvade a damaged system;
2. transportability or mobility of dissemules;
3. condition of the habitat following pollutional stress ;
4. presence of residual toxicants following pollutional stress ;

5. physical-chemical environmental quality after stress e.g. alteration of the substratum or elimination of certain biota;
6. management or organisational capabilities for immediate and direct control of damaged area.

Each term is given a score on a scale of 1 to 3 (1 - poor; 2 - moderate; 3 - good), and the scores are multiplied to give an estimate of the chances of rapid recovery as follows;

Ecosystem Recovery Index score:	400+	:	chances of rapid recovery excellent
	55-399	:	chances of rapid recovery fair to good
	< 55	:	chances of rapid recovery poor

Each term can be re-examined to discuss its utility within the context of the present study. Term 1 refers to refugia and is an important factor in determining the timing of recolonisation of an area. The importance of refugia upstream of a disturbed area has been indicated in the present study whereby sites lacking upstream colonisation sources show slow recolonisation of macroinvertebrates (see Section 5.5 and 5.7). The location of possible refugia can be defined if the area of potential impact is known. Further definition to this term would be provided by knowledge of the speed of movement of each taxon from the colonisation source, with greater rapidity of colonisation from upstream rather than downstream sources.

The Recovery Index presented in this study builds upon term 2 in Cairns' Ecosystem Recovery Index. The 'M' score in the Recovery Index provides further definition for term 2 relating to the mobility of dissemules. The life history characteristics and avoidance potential ('L' and 'A' scores) used in the Recovery Index are not mentioned in Cairns' Ecosystem Recovery Index, but are particularly important in the timing of recovery.

Terms 3 to 5 relate to the type of pollution impacting the watercourse. Within the present study, only those incidents involving oil have resulted in physical habitat disturbance or the presence of residual toxicants (as assessed by the visual presence of oil during sediment disturbance or by information provided by the regulatory authority). Therefore no further information can be offered by the present study to enable further definition of these general categories.

Term 6 in Cairns' Index, which relates to the remedial action following a pollution incident, is a rather generalised estimate of the likelihood of action being taken and can be broken down into several terms which would relate to the speed with which regulatory agencies can deal with an impact. Given that the regulatory agencies within England and Wales have certain guidelines concerning response time following notification of an incident, and assuming standard access to remedial equipment, the most important and variable factor in determining agency response to an incident is the detection of the pollution incident. Analysis of the pollution incidents investigated in the present study suggest that the following factors may be important in this context:

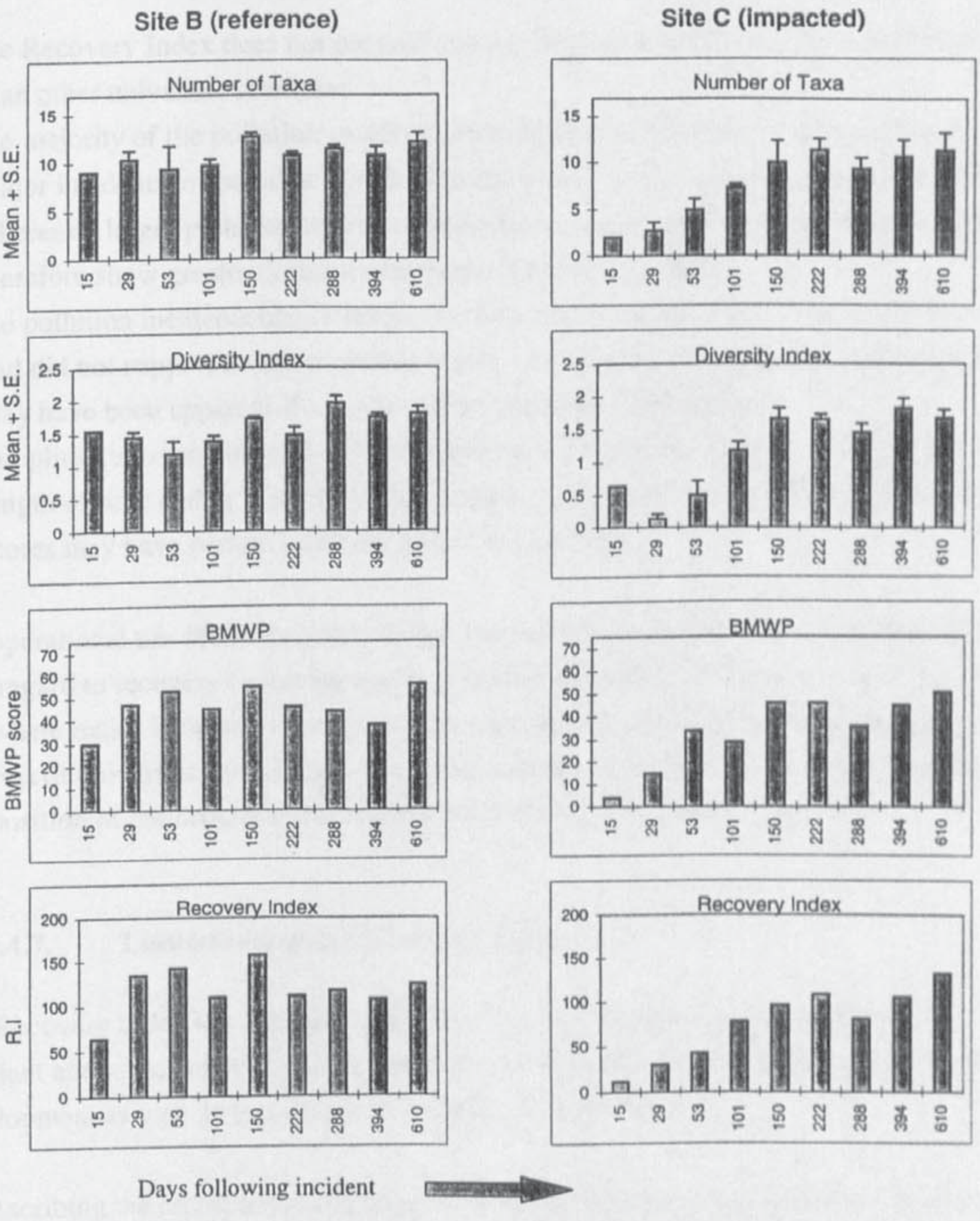
- regular biological monitoring of incident outfall (this led to identification of the pesticide pollution of the River Lostock - see Section 5.6);
- the extent to which a pollution incident would change parameters within the watercourse enabling initial detection, (e.g. changes in colour and smell, or the presence of dead animals);
- visibility of watercourse to members of the public (e.g. a pollution incident within a culverted watercourse or in an isolated watercourse is less likely to be detected than an incident in a watercourse adjacent to a public footpath);
- public awareness of the procedure to follow if an incident is detected.

To summarise, the ARS gives further definition to the colonisation potential of macroinvertebrate taxa within a sampling point and can therefore be used to predict the relative recovery rates of macroinvertebrate communities. This can be used on its own or as part of a predictive model, e.g. Cairns' Ecosystem Recovery Index, involving other parameters relating to the location of refugia, pollution type and to remedial action by agencies.

8.4.6. Application of Recovery Index to Study Data

The Recovery Index was calculated for a number of sites within the pollution incident studies to ascertain its behaviour in relation to the other measures used. It was found that the Recovery Index did not provide more information than the other univariate measures used, thus the predicted greater sensitivity in timescale of recovery was not apparent. This general finding is illustrated with two sites taken from the Hewell Brook pollution incident (see Section 5.5), one reference and one impacted site. Fig. 8.3 shows the Recovery Index, the Number of Taxa, BMWP scores and the Diversity Index over time at the two sites. As with the other univariate measures, the Recovery Index shows variability at the reference site and an increase over time at the impacted site. In this particular case, little further information is added through use of the Recovery Index, although use of this measure shows most clearly the highest values on the last sampling occasion.

Fig. 8.3: Changes in Recovery Index and other Univariate Measures at 2 sites following pollution incident (Hewell Brook)



Thus the predicted greater sensitivity in timescale of recovery of the Recovery Index over other univariate measures was not found when applied to the results following the pollution incident studies. A number of reasons can be postulated for this finding:

- the Recovery Index does not provide more information in the assessment of recovery than other univariate measure;
- the majority of the pollution incidents investigated in the present study were not major incidents in spatial or temporal terms and showed apparently rapid recovery whereas a larger pollution incident would have a longer recovery timescale and may therefore show greater definition in terms of recovery stages;
- the pollution incidents had impacted streams that suffered from chronic pollution and did not support a varied benthic fauna. A more defined response to the incidents may have been apparent if a more varied fauna had been present;
- sampling following the pollution incidents was carried out for a relatively short length of time and it is possible that further colonisation by taxa with high recovery scores may have occured after the period of sampling.

The operational use of the Recovery Index therefore requires further testing particularly with regard to recovery following major pollution incidents. A literature search found that where major incidents were described, information on recovery was generally given in terms of univariate summaries of the macroinvertebrate community rather than the composition of the taxa, making calculation of the recovery index impossible.

8.4.7. Limitations of the Recovery Index

The Recovery Index has been developed for use in situations that have suffered a transient and toxic pollution event. This section describes uncertainties inherent in its development as well as limitations in its potential applicability.

- Ascribing the colonisation potential to different taxa has been carried out in a largely subjective way based on the information gathered in the present study as well information contained in the literature. The accuracy of this information is open to debate and in particular there is ambiguity over the assignment of scores 2 and 3 in terms M and A. However, it was felt that for many taxa, enough information exists on their colonisation potential to enable a distinctions on a scale of 1 to 3 to be made.

- As the index is the product of scores for each taxon recorded, it is related to sample size and effort. Where comparisons are to be made between sites and times, it is important therefore that the sampling procedure is standardised, for example, a defined number of Surber samples within a defined habitat, as used in the present study.

- The utilisation of species data in the construction of the Recovery Index is intended only as an assessment of recovery following a pollution incident. It is not an assessment of water or habitat quality. The 'Avoidance' term applies particularly to a transient pollution event rather than chronic pollution. There is scope, however, for the assessment of recovery following chronic pollution in a similar way, if this term is excluded in the derivation of the score.

- Use of the Recovery Index presumes that a pollution incident has resulted in a certain amount of mortality and that the agents causing the mortality have passed, leaving no residual toxicity.

Because of the greater weighting given to taxa that are slower to recolonise a site, estimates of recovery using the RI may result in a longer timescale than other measures. Application of the RI, however, is less likely than other measures to result in the erroneous assumption of site recovery and could therefore result in greater environmental protection.

8.4.8. Further Development of the Recovery Index

Although the Recovery Index was developed for situations in which the pollutant involved had general toxicity, it may also be of use in situations where a pollutant has specific toxicity to certain taxa only. In this case, the Recovery Coefficient (Equation 8.3) could be calculated so that the unaffected taxa are excluded from the calculation, and recovery is assessed according to the differential colonisation potential of the impacted taxa. For example, the pesticide pollution in the River Lostock, as would be inferred from the toxicological data, had greatest impact on the insects and no apparent impact on snails. In this case the snails would be excluded from the analysis of recovery.

Hellawell (1978) lists the ideal requirements of a pollution index, which can equally be adapted to apply to the Recovery Index where differential recolonisation potential is summarised, rather than differential pollution sensitivity. Hellawell states that the index should reflect the following:

1. the general overall trend of responses of members of the community or populations of species to the environmental stress under consideration;
2. the differing degrees of intensity of the individual responses of faunal components or the indicator value of species;
3. the relative abundance of species or other taxa;
4. some measure of the overall diversity of the community.

In fact, as is the case with most indices, the Recovery Index does not incorporate all the above measures - with points 2 and 3 not being represented. Some indices, e.g. the revised Saprobien Index (Friedrich, 1990), include some measure of the reliability of a particular taxon as an indicator (point 2). An understanding of the performance of the Recovery Index in different situations may result in information about the relative value of taxa as indicators of recovery. For example, an understanding of the variability of populations of taxa between years and between seasons, particularly for those taxa having high colonisation scores, would give further definition to their utility in the assessment of recovery.

Populations of species are variable and subject to great fluctuations from year to year, however presence/absence information attaches as much importance to a drifting individual as to an established population of a species. Therefore a term relating to the abundance of a taxon would be a useful development and could be based upon the predicted abundance pattern of a particular taxon in an unstressed community. The Recovery Score could incorporate a simple abundance weighting for each taxon depending on the predicted abundance of that taxon.

8.5. CIMAH regulations

8.5.1. Introduction

Industrial premises involving dangerous substances are covered by the Control of Industrial Major Accident Hazards (CIMAH) regulations, as described in Section 2.2.3. These regulations require the site manufacturer, as part of the information to be made available to the Health and Safety Executive, to provide baseline data against which an assessment of the effects of a potential accident on persons and the environment can be made. This includes a "thorough survey of the environs of the site, including surface water and groundwater or aquifer catchments", and "assessment of the possible environmental consequences of the accident, with particular reference to ecotoxicity and persistence of effects" (DOE, 1991).

The DOE guidelines concerning the environmental survey suggest that detailed ecological survey work is conducted to provide a set of data for species distribution and abundance. This would serve as a baseline against which the immediate impact of an accident could be evaluated and recovery could be monitored.

Some of the findings from the present study can be used to develop a protocol utilising benthic macroinvertebrates for the biomonitoring of a potential pollution impact on a watercourse.

Standard survey methodology to assess impact from a discharge point requires up and downstream samples, and before and after samples (Underwood, 1992). The main contribution to this methodology from the present study is that the downstream samples should extend for some distance downstream, and in some cases for a number of kilometres. This is because the presumed linear, monotonic response to a pollution incident, with decreasing impact with distance from the pollution outfall has been shown to be too simplistic (Section 8.3.1). Given the variability of habitats and impacts within many stream systems in populated areas, there is great variability in the biota supported at different points. Thus, although the concentration of a pollutant may be highest in the areas closest to the discharge point, the ecosystem may be more sensitive and therefore suffer greater impact at points further from the discharge point.

8.5.2. Suggested protocol for site survey

A framework for the assessment of the impact of a pollution incident on the lotic ecosystem downstream of a potential outfall is presented in Fig. 8.4. (based upon the experience gained during the present study). Whilst each step is described further in the following paragraphs, detailed recommendations resulting from the study refer particularly to the assessment and monitoring of the benthic macroinvertebrate community structure.

8.5.3. Description of procedure

A. Define Pollutant and assess toxicological information
The pollutant that has the potential to have a hazardous impact on the environment needs to be identified. In addition, the potential for a number of pollutants to be involved in a pollution incident and the possible resulting chemical changes requires assessment. The behaviour of a pollutant upon release into the environment needs to be addressed including the possibility that reactions between chemicals may increase the subsequent complexity of the pollutant analysis. For example, the release of Hydrochloric acid into a stream may result in the mobilisation of heavy metals within the substratum (e.g. see Section 5.9.1).

Information regarding the toxicological properties of the pollutants involved can be obtained from a number of databases. This information is largely derived from laboratory data resulting in the loss of a certain amount of certainty when applied to field situations and to components of the ecosystem that have not been used in the toxicity test, although laboratory experiments incorporate a certain degree of caution in that the worst case scenarios are assessed, i.e. maximum concentrations of pollutants that would be found in the environment.

B. Model pollutant behaviour
Given knowledge of the maximum amount of a particular pollutant on a site that could be released into a watercourse accidentally, an estimate of its dilution and dispersion can be calculated based upon knowledge of the hydrological properties of the stream. Several computational packages exist to enable modelling of the fate and behaviour of a pollutant event (Norton *et al.*, 1996). For example, a package titled 'SIRIUS' (Spills In Rivers User Interactive Software) provides concentration/time profiles as the pollutant passes selected points and allows for dilution and degradation through absorption and chemical degradation (Cole and Oakes, 1995).

C. Obtain Ecosystem Information for Potential Area of Impact
Information regarding the characteristics of the receiving ecosystem can be obtained from a variety of sources including published literature, local knowledge and information held by regulatory and non-statutory agencies, for example species and habitat data held by the Wildlife Trusts/English Nature/Scottish Heritage; River Corridor Surveys (Environment Agency - Fisheries and Conservation Dept.) and

biological monitoring data(Environment Agency - Pollution Control Dept.). In addition, field observations and site surveys can be carried out to characterise habitats and associated flora and fauna. This information should be obtained for the maximum distance downstream of potential pollutant outfall that may be affected, as estimated by the concentration/time/distance profiles.

D. Estimate Exposure of Ecosystem to Pollutant

The toxicological information and the ecosystem information can be combined to predict the nature and extent of a potential pollutant impact upon the ecosystem. For example, the primary toxicological effect may be on Arthropoda, but may leave fish unaffected. Secondary effects may then result from the loss of Arthropoda, for example: an increase in the populations of competitors; a decrease in detritus processing; loss of fish food. Few ecosystems are well enough understood to enable more than rough estimates of the potential ecological processes to be made. Nevertheless such predictions are necessary in the derivation of measures as described in the next step.

E. Define Features to be protected.

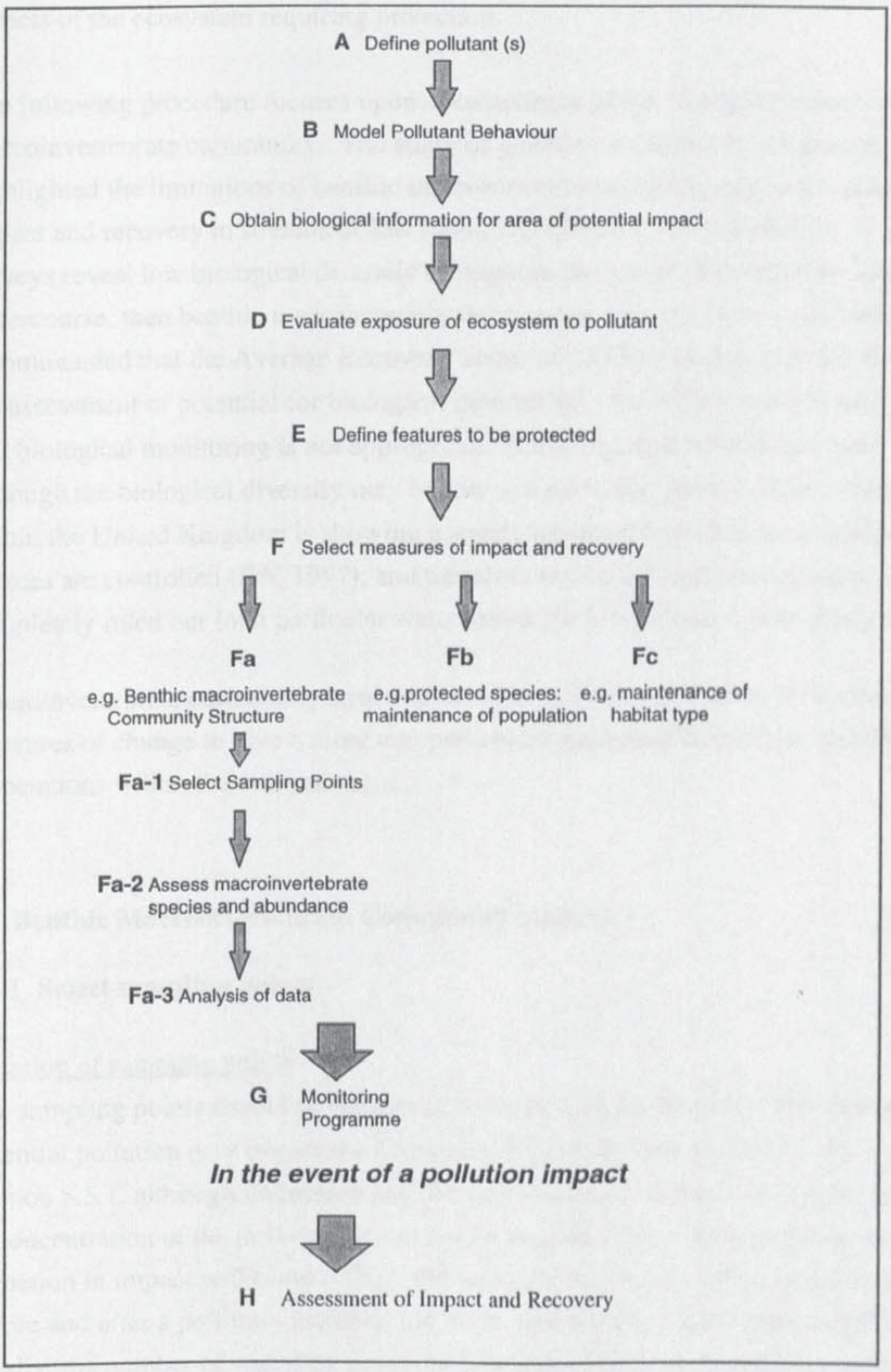
This step aims to define those aspects of an ecosystem that require protection using the knowledge of potential impact gained from the previous step. The features could be general, e.g. 'protection of the flora and fauna within the channel', or more specific, e.g. 'maintenance of the area as a trout spawning habitat'. Identification of the important features within the ecosystem enables the measures to be selected which can be used in the assessment of impact and recovery. The use of benthic macroinvertebrates, as described in Section 2.3.1, can be as an indicator of ecosystem health, as well as being an important component of the lotic ecosystem and therefore a feature to be protected.

F. Select measures of impact and recovery

The selection of measures relates directly to the features of the ecosystem identified as requiring protection. Where particular species have special economic or conservation status, e.g. the Freshwater Crayfish, measures could focus upon the relevant life history stages of that species.

Figure 8.4. suggests three areas of measurement of impact and recovery: Benthic macroinvertebrate community structure, maintenance of populations of a selected

Fig. 8.4: CIMAH regulations - Framework for assessment of potential ecological impact to a watercourse



species and maintenance of habitat types e.g. stands of emergent vegetation. The detailed measures of the latter two require development according to the particular aspects of the ecosystem requiring protection.

The following procedure focuses upon measurement of the structural aspects of the macroinvertebrate community. The study of pollution incidents in the present thesis has highlighted the limitations of benthic macroinvertebrate monitoring in the assessment of impact and recovery in streams of low biological diversity (Section 8.2.7). If initial site surveys reveal low biological diversity throughout the extent of potentially impacted watercourse, then benthic macroinvertebrate structure may not be a useful measure. It is recommended that the Average Recovery Score: ARS (Section 8.4.5) could be used in the assessment of potential for biological monitoring. An ARS score ≤ 6 would suggest that biological monitoring is not appropriate. It must be emphasised however, that although the biological diversity may be low at a particular point in time, water quality within the United Kingdom is showing a steady improvement as chronic pollution sources are controlled (EA, 1997), and therefore biological monitoring can not be completely ruled out for a particular watercourse for longer than a year at any one time.

Macroinvertebrate community structure could be used in association with other measures of change to give a more comprehensive understanding of the variability and associations within the ecosystem.

Fa Benthic Macroinvertebrate Community Structure

Fa-1 Select sampling points

Location of sampling points

The sampling points should be located to cover at least the distance downstream that a potential pollution may impact the ecosystem (Fig. 8.3: Steps B and D). As stated in Section 8.5.1, although dispersion and dilution will result in the downstream reduction in concentration of the pollutant, it can not be assumed that a corresponding downstream reduction in impact will occur. Thus, the most important changes to be monitored before and after a pollution incident, are those which occur within each sampling point. Ideally, the number of sampling points chosen should depend upon the variation within the watercourse along the site of potential impact. For example, a site should be situated downstream of each new tributary to give impact information on the whole range of macroinvertebrate community structures to be found within the sampled habitat.

However cost, time and access constraints may limit this ideal to the selection of a limited number of sampling points within the area of potential impact.

Reference conditions can be provided by sampling points situated upstream of the likely impact and showing environmental parameters that are as closely related to the downstream sampling points as possible. However, the points highlighted in Section 6.3.2 showed that upstream sites similar to the impacted sites in terms of their environmental parameters are not always easily located, so the information they provide should be viewed as additional to the information provided by pre-impact samples.

Similarity of sampling points

Standard practice (e.g. Green, 1979), is to select sampling points that are as similar as possible in terms of environmental parameters such that any changes may, with greater confidence, be attributed to the pollutant. However, the present study has shown that even when superficially similar in terms of environmental parameters, there is great environmental heterogeneity of habitats along streams, particularly in areas receiving a variety of unknown effluents and being influenced by tributaries with varying water qualities. It is therefore recommended that each sampling point be viewed in isolation, with the focus being upon changes within, rather than changes between, sampling points. In the event of a pollution impact, the monitoring information from each site can be compared with information obtained following the impact.

Stratification of Sampling

Stratified sampling, as carried out in the present study, is a practical way of obtaining quantitative samples with less inter-sample variability. It is suggested that riffles are chosen as they are taxonomically diverse, suffer exposure to ambient water conditions, are usually accessible and can easily be quantitatively sampled (see Section 3.4.1).To obtain a more cautious estimate of recovery, biological assessment should involve habitats located within sections of the watercourse that are predicted to show the longest residence time of the potential pollutant: the pools. However this needs to be balanced against the relative difficulties of quantitative sampling and the lower diversity of taxa in pools. It is therefore suggested that although sampling should focus on riffle areas, some pool areas should also be included within the programme. If suitable riffle areas are not available within the watercourse, then consideration needs to be given to the use of colonisation samples (DOE, 1983).

It must be remembered that the form of stratified sampling undertaken within the present study can not be directly compared with other forms of sampling e.g. standard kick samples as carried out by the Environment Agency. The sampling technique adopted in the present study samples only those taxa found in the habitat types selected, and, due to the limited area covered by 3 replicates, may miss the rarer taxa. If a site is known to contain taxa requiring special protection (IEA, 1995), the sampling programme needs to be adapted to take account of this.

Fa-2 Sample macroinvertebrate species and abundance

To assess the macroinvertebrate species and abundance within a site, a minimum of 3 Surber samples should be taken within a riffle. A Surber sampler is recommended as it gives a quantitative dimension to data collection and because of its ease of use within a range of substratum types and depths. If initial background information (Step C in Fig. 8.3) suggests that high numbers of taxa are present in an area, the use of a greater number of replicate samples would increase the number of selected taxa that can be used for abundance analysis, as well as reducing the within-site variability vs between-site variability (Resh and McElravy, 1993). It is suggested that where more than 10 taxa are found during initial assessments, and where resources permit, 5 Surber samples should be taken.

The Institute of Environmental Assessment (1995) suggest that an initial water quality invertebrate sample survey is undertaken and if it achieves any of the following values, then the samples taken should be analysed to species level wherever practical;

- 26 or more families of invertebrates
- BMWP score of 150 or greater
- ASPT score of 6.48 or greater

However, the present study has found measurable impact on the macroinvertebrate community at levels far lower than those suggested above. It is therefore recommended that taxa (apart from Oligochaeta, Chironomidae, Sphaeriidae and Simuliidae) be identified to species.

Abundance levels of taxa should be recorded. The present study recorded the numbers of each taxon within each replicate. This enabled population changes within selected taxa or transformation of abundance to be carried out if required. It is therefore recommended that total numbers within each replicate are recorded.

As part of the monitoring programme, simple environmental measures should be taken. Changes in these may occur independent of a pollution incident, and which may lead to changes in the macroinvertebrate community structure. The eleven physical and chemical variables listed below were found by Moss *et al.* (1987) to explain 65.7% of taxon occurrence. Variables 1-8 were determined from maps, published climatological data and data provided by the water industry. Only variables 9 to 11 were measured in conjunction with each sampling season. It is recommended therefore, that these three variables are measured as part of the monitoring programme and descriptions of the sampling points include information obtained on the other 8 variables.

1. Distance from Source
2. Slope
3. Altitude
4. Total Alkalinity (annual mean)
5. Chloride (annual mean)
6. Total oxidized nitrogen (Nitrate + nitrite)(annual mean)
7. Mean annual air temperature range (July mean-January mean)
8. Mean annual air temperature (mean of January, April, July and October means)
9. Mean Substratum Particle Size
10. Mean water width
11. Mean water depth

Fa-3 Analysis of Community Data

The information obtained from the present study suggests that the most useful measures to be obtained in the analysis of community change are:

- number of taxa;
- nonmetric multidimensional scaling of communities based upon a similarity matrix utilising the Bray-Curtis Similarity Coefficient.

Although the present study has shown that following certain incidents, population changes of selected taxa demonstrates impact and recovery where other summary measures of the community show no impact, it is not necessarily clear which taxa should be selected. It is recommended, therefore that species information is used in the derivation of the Recovery Index as described in Section 8.4. It must be emphasised that these are measures extracted from the raw biological data and all result in a loss of a certain amount of the information available. It is therefore always necessary that the data be obtained and assessed by a competent biologist.

G Establish a monitoring programme

To establish patterns of interannual variability to act as a baseline for comparison to potential post-impact changes, systems need to be sampled for a number of years (Cooper and Barmuta, 1993). In addition, many stream communities are subject to seasonal variations. It is suggested, therefore, that the monitoring programme entails three samples per year: in Spring, Summer and Autumn.

H In the event of a pollution incident

Samples should be taken at the sampling points in the same way as the monitoring programme. Samples should be taken as soon as possible following the incident and at regular intervals thereafter, at periods not less than the monitoring times. The measures used to compare pre and post impact communities are those described in Fa-3.

It is recommended that recovery within each site be assessed using two methods:

1. Multivariate analysis of community similarity

2. Recovery Index (as described in Section 8.4.)

The first method uses nonmetric multidimensional scaling to assess community changes over time in comparison with reference samples derived from both reference sites and pre impact data. This method retains a high proportion of the species/abundance information and has the potential to identify the taxa that are responsible for differences between samples. Recovery endpoints can be established in statistical terms as similarity to reference or to pre-impact samples.

The second method utilises the differential colonisation potentials of the taxa within the community to derive a measure of recovery. Recovery endpoints can be defined in terms of the Recovery Coefficient which, again require the use of reference samples, whilst having regard to the variability within the community as assessed from the pre-impact monitoring data.

8.6. Future Research

This section outlines suggestions for further research that would contribute towards our understanding of the responses of benthic macroinvertebrates to pollution incidents.

- Most of the pollution incidents investigated in the present study were Category 2 incidents (See Section 6.2.4). One of the requirements for further work is the collection of data following major pollution events. Although the impact of major incidents has been reported in the literature, little information is given about subsequent recovery. Study of these incidents would also test the utility of the Recovery Index.

- The present study has, in demonstrating an avoidance response by a selected macroinvertebrate, shown the importance of understanding the detailed responses of individual taxa to disturbances and relating these to perceived structural changes. Further research in this area could involve measurements of the immediate avoidance responses of selected macroinvertebrate taxa to a variety of anthropogenic and natural disturbances, and the influence of environmental parameters on these responses.

- The finding within the present study of slow recolonisation if headwaters are polluted (Section 5.5. and 5.7), and the use of the substratum as a refuge (Chapter 7), indicates the importance of understanding the relative significance of different refugia in recolonisation following disturbance. Further investigations could concentrate on communities of headwaters and their variability over time. In addition, the extent to which the hyporheos functions as a refuge from disturbance and the relative importance of active and passive avoidance requires investigation.

- Recolonisation pathways: the relative importance of each pathway following a short-term pollution incident has not been addressed in the present study but is an important area for future research.

- Further work to develop the Recovery Index would initially involve refinement of the scores allocated to each taxon, with particular emphasis on the measures used to assess avoidance response (A) and movement within lifetime (M).

- Further work is required to investigate mechanisms for the assessment of recovery from a pollution incident impacting a chronically polluted stream. One approach would be to concentrate on the population dynamics of a few, commonly distributed taxa; for example, the Chironomidae. In the present study, the number of Chironomidae in the more polluted waters were often too low in the samples to enable quantitative comparisons to be carried out. It would be necessary, therefore, to adopt different sampling methods aimed directly at this particular group; e.g. emergence traps, sampling in pool as well as riffle areas, collecting a larger number of samples. In addition, species and population changes of other taxa predominating in pool habitats e.g. Oligochaeta and Sphaeriidae could be investigated in greater detail.

- A simple framework for the predictive assessment of the recovery of a river from a pollution impact was discussed in Section 8.4.5 (Cairns, 1990). Requirements for the further development of this model require terms relating to the habitat heterogeneity, size of impact, history of site, detection of a pollution incident as well as terms related to the pollutant. In addition, refinements to the term relating to sources of refugia could be incorporated.

Chapter 9

Conclusions

Biological monitoring of rivers as a means of assessing water quality, and more recently, general environmental quality is extensively used in Europe and North America. Benthic macroinvertebrates are the component of the stream ecosystem most commonly chosen for monitoring and their use in the assessment of ecosystem response to chronic situations involving water quality and habitat alterations is well established. However, the use of macroinvertebrates to study pollution incidents is fragmentary and less well developed. Those studies that have investigated the impact of actual pollution incidents on stream fauna have tended to concentrate on the initial impact of the incident, giving little attention to subsequent changes, i.e. recovery, of the biota. The utility of the macroinvertebrate community in the assessment of impact and recovery following a pollution incident requires further investigation, particularly as the relative importance of pollution incidents in their impact on the environmental quality of rivers is likely to increase, as regulatory controls on chronic discharges become more stringent.

The responses of the macroinvertebrates within the substratum to a number of actual pollution incidents was investigated. Most of the pollution incidents were Category 2 incidents which did not have a major impact on the watercourse. The following measures of the community were used:

- changes in the abundance of selected 'dominant' taxa;
- number of taxa;
- diversity index;
- BMWP and ASPT scores;
- multivariate similarity of communities sampled at different times and sites using nonmetric multidimensional scaling.

Varying degrees of impact and recovery were found within each pollution incident depending upon the measure used. It was found that change following pollution incidents was most adequately represented by changes in the abundance of selected taxa, by the number of taxa and by multivariate analysis of the communities. The importance of adequate reference conditions was highlighted.

It became clear during the study that measures of the community need to take into account the varied responses shown by the comprising taxa. To gain further understanding of the detailed processes that may occur when a taxon is exposed to a pollution incident, a laboratory study was carried out to assess the behaviour of a selected macroinvertebrate, *Gammarus pulex*, in response to sublethal levels of a toxicant. An increase in burrowing behaviour was found. This behaviour is known to occur in response to flood disturbance but may also have the consequence of increased survival and subsequent recolonisation following anthropogenic disturbance in the form of toxic short-term pollution. This finding suggests that an understanding of avoidance behaviour is required when ascertaining the macroinvertebrate community response to pollution incidents, and that further investigations into the extent of avoidance behaviours among different taxa, and the environmental factors controlling them, are required.

The finding of differential colonisation in the taxa following the pollution incident study, as well as observations of behavioural changes in the field and in the laboratory study, led to the development of a novel assessment of recovery within streams - the 'Recovery Index'. A 'Recovery Score' is allocated to each taxon, which is derived from the three parameters considered most pertinent to the colonisation potential of a taxon. The Recovery Scores for each taxon within a sample are combined to produce a Recovery Index which can be used to assess spatial or temporal recovery. By allocating higher scores to taxa that are slower to recolonise sites, utilisation of the 'Recovery Index' in the assessment of recovery will result in longer recovery times following pollution incidents than other measures investigated within the study, thus reducing the likelihood of a false assessment of recovery.

The Control of Industrial Major Accident Hazards (CIMAH) regulations have associated guidelines for environmental survey to provide a set of data against which the effects of an accident could be judged. Suggestions for the development of these guidelines are given in Chapter 8, based upon the findings from the present study and incorporating the use of the 'Recovery Index'.

The overall aims and objectives of the thesis as described in Chapter 1 have been met, although, as with all opportunistic pollution incident studies, a major limitation in impact assessment was the lack of adequate pre-impact data. Notwithstanding this, the study gave definition to the utility of using benthic macroinvertebrates in the assessment of impact and recovery following pollution incidents. The following conclusions and findings have arisen from the study.

1. Pollution incidents can have a measurable impact on the benthic macroinvertebrate communities of impacted streams.

2. Complexities of impact and recovery are such that no single biotic measure was adequate in describing recovery. Some taxa showed absence then an increase, other taxa showed a stimulatory response.

3. Different measures of the macroinvertebrate community differ in their utility in the assessment of recovery following pollution incidents.

4. Of the univariate measures, the number of taxa was found to be the most useful.

5. Multivariate analysis of similarity was found to be useful if adequate reference conditions were available.

6. Active behavioural avoidance by burrowing into the substratum was demonstrated in a laboratory study for a selected macroinvertebrate species.

7. A Recovery Index was developed to reflect the differential colonisation rates shown by the different taxa following the pollution incidents. The index was derived from recovery scores given to taxa according to three factors considered important in determining the recolonisation of a taxon following a pollution incident: their life cycle; the extent of movement within their lifetime and their avoidance response to disturbance. The Recovery Index was predicted to give a more sensitive assessment of recovery time following a pollution incident than the other measures used within the study. The Recovery Index needs to be tested using a wider range of data than that available following the pollution incident studies investigated, particularly data gathered in relation to major pollution incidents.

8. Guidelines for baseline monitoring and impact assessment utilising benthic macroinvertebrates within watercourses covered by CIMAH regulations have been developed.

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Appendix A

Definitions of EA Pollution Incident Categories (Environment Agency, 1997).

Major Incidents - Category 1
These are serious incidents. The category covers incidents which include at least one of the following.

1. the spillage to a watercourse or to groundwater of a chemical which has a persistent effect;
2. an abstraction is closed because of pollution at or approaching the intake;
3. the death of more than 100 fish of any notable species;
4. the major or repeated failure of significant effluent treatment plant which causes gross contravention of consent conditions together with a readily observable impact on the receiving water;
5. the deployment of heavy equipment (i) by the Agency or other body to remedy pollution;
6. recreational activity is curtailed.

Significant Incidents - Category 2
The category covers incidents which include at least one of the following:
downstream abstractors are notified of the risk;
10-100 fish of any notable species, die because of the pollution;
an obvious significant loss of invertebrates (i.e. a visual inspection shows dead snails, dead *Gammarus* etc.);
livestock farmers are warned of the risk;
there is noteworthy contamination of the bed of the watercourse;
a noteworthy reduction of the amenity value by odour or appearance affecting more than 50 metres of the watercourse;

Minor Incident - Category 3
All incidents not qualifying as Category 1 or 2 i.e.
notification of abstractors not necessary;
fish kill of less than 10 fish (of no particular importance to the affected water);
no readily observable effect on invertebrate life;
water not unfit for stock watering;
bed of watercourse only locally contaminated;
minimal environmental impact and amenity value only marginally affected.

Unsubstantiated incident - Category 4
A reported pollution incident which upon investigation proves to be unsubstantiated i.e. no evidence can be found of a pollution incident having occurred.

Appendix B

References used to summarise published literature on biological investigations of pollution incidents

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Appendix C

A sample data set for one pollution incident. Full data sets are available from the School of Engineering, Aston University, *Birmingham*, England

Chinn Brook Site A - Tributary Culvert

	1Aa	1Ab	1Ac	2Aa	2Ab	2Ac	3Aa	3Ab	3Ac	4Aa	4Ab	4Ac	5Aa	5Ab	5Ac	6Aa	6Ab	6Ac
D. lacteum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dugesia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polycelis ten/nigra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
P. jenkinsi	1	0	0	0	0	0	0	0	0	5	5	10	1	3	4	1	2	2
Planorbis vortex	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea peregra	0	0	1	0	0	0	1	1	1	2	3	2	3	4	2	0	1	5
Ancylus fluviatilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sphaeriidae	0	0	0	0	0	0	0	0	0	1	2	0	0	0	0	0	1	0
Physa sp.	0	0	0	0	0	0	1	1	0	21	4	12	1	1	1	1	3	4
Oligochaeta	99	105	38	0	0	0	3	10	12	58	88	85	180	229	150	124	176	220
G. complanata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Erpobdellidae	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
E. octoculata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
H. sanguisuga	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hydracarina	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Asellus aquaticus	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Gammarus pulex	0	0	0	0	0	0	0	0	0	6	1	6	3	0	1	1	0	0
Baetidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Ephemerella ignita	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
H. angustipennis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tinodes waeneri	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Limnephilidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
L. rhombicus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C. villosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dytiscidae larva	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Velia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ceratopogonidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chironomidae	51	12	8	0	0	0	240	290	80	155	144	36	8	15	6	61	14	10
Eristalis	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Simuliidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tipulidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fly larvae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix C (continued)

Chinn Brook Site B - Tributary Bridge	1Ba	1Bb	1Bc	2Ba	2Bb	2Bc	3Ba	3Bb	3Bc	4Ba	4Bb	4Bc	5Ba	5Bb	5Bc	6Ba	6Bb	6Bc
D. lacteum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dugesia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polycelis ten/nigra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
P. jenkinsi	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	1
Planorbis vortex	1	0	2	0	0	0	2	2	4	1	0	0	3	0	0	1	2	1
Lymnaea peregra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ancylus fluviatilis	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Sphaeriidae	0	0	0	0	0	0	4	1	2	2	5	1	0	0	4	2	5	4
Physa sp.	31	62	8	0	0	0	115	80	66	72	42	13	90	194	107	172	160	320
Oligochaeta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
G. complanata	3	0	4	0	0	0	0	0	0	0	1	0	1	1	0	0	0	0
Erpobdellidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
E. octoculata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
H. sanguisuga	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hydracarina	0	0	0	0	0	0	0	0	0	7	9	3	2	1	2	0	0	5
Asellus aquaticus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gammarus pulex	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Baetidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ephemerella ignita	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
H. angustipennis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tinodes waeneri	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Limnephilidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
L. rhombicus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C. villosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dytiscidae larva	17	22	33	0	0	0	67	30	47	3	1	4	6	2	1	18	80	20
Velia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ceratopogonidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chironomidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Eristalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Simuliidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tipulidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fly larvae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix C (continued)

Chinn Brook - Site C Ash Tree

	1Ca	1Cb	1Cc	2Ca	2Cb	2Cc	3Ca	3Cb	3Cc	4Ca	4Cb	4Cc	5Ca	5Cb	5Cc	6Ca	6Cb	6Cc
D. lacteum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0
Dugesia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polycelis ten/nigra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
P. jenkinsi	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis vortex	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea peregra	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ancylus fluviatilis	10	1	3	0	0	0	2	9	7	1	9	2	13	0	0	0	1	6
Sphaeriidae	0	2	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Physa sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oligochaeta	10	0	0	0	0	0	0	2	0	6	0	14	3	8	7	80	13	0
G. complanata	0	2	2	0	0	0	0	0	1	4	3	3	2	1	2	0	0	0
Erpobdellidae	0	1	0	0	0	0	5	2	1	2	0	1	6	2	1	0	0	0
E. octoculata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
H. sanguisuga	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
Hydracarina	1	9	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Asellus aquaticus	1	6	0	0	0	0	0	0	0	2	2	0	1	0	1	0	0	0
Gammarus pulex	180	300	160	0	0	0	260	240	250	370	80	84	404	160	270	122	540	400
Baetidae	5	1	2	0	0	0	4	0	0	35	4	9	88	48	20	2	1	3
Ephemerella ignita	1	5	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
H. angustipennis	0	0	0	0	0	0	1	0	1	0	0	1	1	0	1	0	0	0
Tinodes waeneri	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Limnephilidae	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
L. rhombicus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C. villosa	8	0	4	0	0	0	1	0	0	3	0	0	0	0	0	0	0	2
Dytiscidae larva	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Velia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ceratopogonidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chironomidae	0	55	0	0	0	0	0	0	0	1	0	0	0	0	0	7	0	0
Eristalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Simuliidae	0	0	0	0	0	0	0	0	0	0	0	0	6	2	0	0	0	0
Tipulidae	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Fly larvae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix C (continued)

Chinn Brook Site D 15m US confluence	1Da	1Db	1Dc	2Da	2Db	2Dc	3Da	3Db	3Dc	4Da	4Db	4Dc	5Da	5Db	5Dc	6Da	6Db	6Dc
D. lacteum	0	0	0	0	0	0	1	0	1	1	1	0	0	0	0	0	0	0
Dugesia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polycelis ten/nigra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
P. jenkinsi	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis vortex	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea peregra	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ancylus fluviatilis	2	1	1	0	2	4	0	1	0	0	0	0	1	0	0	0	0	1
Sphaeriidae	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0
Physa sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oligochaeta	17	0	0	0	0	0	0	0	1	10	4	22	2	9	10	2	19	5
G. complanata	0	1	0	0	1	0	0	2	1	1	0	1	0	1	1	0	0	1
Erpobdellidae	0	0	1	0	0	0	2	1	5	1	2	4	3	2	1	4	0	0
E. octoculata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
H. sanguisuga	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hydracarina	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Asellus aquaticus	0	0	0	0	0	0	0	0	0	1	5	3	0	0	0	0	0	0
Gammarus pulex	112	190	220	320	330	220	284	130	230	310	340	85	66	79	50	240	200	140
Baetidae	0	0	1	0	2	2	0	0	0	15	5	6	135	53	84	19	29	66
Ephemerella ignita	14	1	3	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
H. angustipennis	0	0	0	0	0	0	5	3	2	3	0	6	3	0	2	0	1	2
Tinodes waeneri	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Limnephilidae	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1
L. rhombicus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C. villosa	0	0	0	1	0	0	0	1	1	0	0	0	0	0	0	0	0	0
Dytiscidae larva	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Velia sp.	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ceratopogonidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Chironomidae	3	0	1	0	0	0	0	0	5	0	0	0	0	0	0	0	1	1
Eristalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Simuliidae	0	0	0	0	0	2	0	0	0	0	0	0	3	2	1	0	0	0
Tipulidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fly larvae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix C (continued)

Chinn Brook Site E - 15 m DS confluence	1Ea	1Eb	1Ec	2Ea	2Eb	2Ec	3Ea	3Eb	3Ec	4Ea	4Eb	4Ec	5Ea	5Eb	5Ec	6Ea	6Eb	6Ec
D. lacteum	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1	0
Dugesia sp.	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polycelis ten/nigra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
P. jenkinsi	6	10	6	0	1	0	0	0	0	1	0	0	0	0	0	0	0	1
Planorbis vortex	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea peregra	5	2	3	0	2	0	0	0	0	1	0	0	1	0	0	0	0	0
Ancylus fluviatilis	33	27	26	8	21	7	10	7	4	4	8	9	1	4	4	0	0	7
Sphaeriidae	0	2	3	0	0	1	3	0	1	0	0	0	0	0	0	0	0	0
Physa sp.	0	0	0	0	0	0	1	0	0	1	0	0	1	0	0	0	0	1
Oligochaeta	33	12	10	0	0	0	35	8	12	16	7	7	1	3	28	8	11	25
G. complanata	5	12	9	0	14	3	3	1	1	2	1	1	2	2	2	0	0	2
Erpobdellidae	1	4	0	0	0	0	4	5	1	0	0	1	3	1	2	4	1	3
E. octoculata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
H. sanguisuga	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hydracarina	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Asellus aquaticus	0	0	0	0	0	0	0	1	0	1	1	1	1	0	1	0	0	0
Gammarus pulex	160	220	190	195	400	109	200	190	170	135	105	101	215	47	50	200	84	120
Baetidae	14	8	3	34	15	1	3	2	2	11	8	4	27	34	44	7	7	5
Ephemerella ignita	4	2	2	0	4	3	0	0	0	0	0	0	0	0	0	0	0	0
H. angustipennis	0	0	0	0	0	0	3	3	0	0	0	0	0	0	0	0	1	0
Tinodes waeneri	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Limnephilidae	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
L. rhombicus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C. villosa	4	6	7	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0
Dytiscidae larva	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Velia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ceratopogonidae	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Chironomidae	12	20	4	0	0	1	1	0	0	0	0	1	1	0	0	1	0	0
Eristalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Simuliidae	0	0	0	2	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Tipulidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fly larvae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix C (continued)

Chinn Brook Site F - 80m DS confluence	1Fa	1Fb	1Fc	2Fa	2Fb	2Fc	3Fa	3Fb	3Fc	4Fa	4Fb	4Fc	5Fa	5Fb	5Fc	6Fa	6Fb	6Fc
D. lacteum	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0	0
Dugesia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polycelis ten/nigra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
P. jenkinsi	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Planorbis vortex	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea peregra	2	1	0	4	8	0	0	0	0	2	1	3	1	0	0	0	1	5
Ancylus fluviatilis	8	2	6	10	5	8	66	7	16	19	32	22	24	30	7	7	14	15
Sphaeriidae	0	0	1	0	0	0	5	0	3	0	0	0	0	0	0	0	0	1
Physa sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oligochaeta	30	5	2	0	0	0	12	25	29	10	4	34	32	54	26	46	50	58
G. complanata	5	3	2	5	2	2	3	2	1	2	1	1	1	3	2	1	3	2
Erpobdellidae	0	3	0	0	0	0	4	5	1	1	4	1	2	1	1	3	0	1
E. octoculata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
H. sanguisuga	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hydracarina	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Asellus aquaticus	2	0	1	0	0	0	1	0	1	2	0	1	0	0	0	0	0	0
Gammarus pulex	140	58	220	210	270	120	118	15	85	155	64	55	155	49	60	170	116	74
Baetidae	5	68	2	10	23	10	4	2	7	27	15	14	76	18	49	38	20	18
Ephemerella ignita	5	2	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
H. angustipennis	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	1	0	0
Tinodes waeneri	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Limnephilidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
L. rhombicus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
C. villosa	0	0	0	0	0	0	1	1	1	1	0	0	0	0	0	0	0	0
Dytiscidae larva	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Velia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ceratopogonidae	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Chironomidae	8	11	5	0	0	1	0	0	0	2	2	0	0	0	0	5	24	14
Eristalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Simuliidae	0	0	0	0	1	0	0	0	0	0	0	0	2	1	0	0	1	4
Tipulidae	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Fly larvae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix D

The Biological Monitoring Working Party (BMWP) score

	Families	Score
Mayflies	Siphlonuridae, Heptageniidae, Leptophlebiidae, Ephemerellidae, Potamonthidae, Ephemeridae, Taeniopterygidae, Leuctridae, Capniidae,	
Stoneflies	Perlodidae, Perlidae, Chloroperlidae,	
Riverbug	Aphelocheiridae	10
Caddisflies	Phryganeidae, Molannidae, Beraeidae, Odontoceridae, Leptoceridae, Goeridae, Lepidostomatidae, Brachycentridae, Sericostomatidae	
Crayfish	Astacidae	
Dragonflies	Lestidae, Agriidae, Gomphidae, Cordulegasteridae, Aeshnidae, Corduliidae, Libellulidae	8
Caddisflies	Psychomiidae, Philpotamiidae	
Mayflies	Caenidae	
Stoneflies	Nemouridae	7
Caddisflies	Rhyacophilidae, Polycentropidae, Limnephidae	
Snails	Neritidae, Viviparidae, Ancylidae	6
Caddisflies	Hydroptilidae	
Mussels	Unionidae	
Shrimps	Corophiidae, Gammaridae	
Dragonflies	Platycnemidae, Coenagriidae	
Water Bugs	Mesoveliidae, Hydrometridae, Gerridae, Nepidae, Naucoridae, Notonectidae, Pleidae, Corixidae	
Water Beetles	Haliplidae, Hygrobiidae, Dytiscidae, Gyrinidae, Hydrophilidae, Clambidae, Helodidae, Dryopidae, Elminthidae, Chrysomelidae, Curculionidae	5
Caddisflies	Hydropsychidae	
Craneflies	Tipulidae	
Blackflies	Simuliidae	
Flatworms	Planariidae, Dendrocoelidae	
Mayflies	Baetidae	4
Alderflies	Sialidae	
Leeches	Piscicolidae	
Snails	Valvatidae, Hydrobiidae, Lymnaeidae, Physidae, Planorbidae	3
Cockles	Sphaeriidae	
Leeches	Glossiphoniidae, Hirudidae, Erpobdellidae	
Hoglouse	Asellidae	2
Midges	Chironomidae	1
Worms	Oligochaeta	

Appendix E

References used in the Derivation of the Recovery Scores (Table 8.1)

Allan, 1995; Bernard *et al.*, 1990; Bohle, 1978; Borchardt, 1993; Borlakoglu and Kickuth, 1990; Brinkhurst, 1965; Brinkhurst and Kennedy, 1965; Cairns and Dickson, 1977; Costa, 1966; Coutant, 1964; Crisp and Gledhill, 1970; Crunkilton and Duchrow, 1990; Doeg *et al.*, 1989; Douglas and Macreanor, 1990; Edington and Hildrew, 1995; Elliott, 1977; Elliott *et al.*, 1988; Elliott and Mann, 1979; Ellis, 1978; Extence, 1981; Fitter and Manuel, 1986; Forrow, 1995; Friday, 1988; Frost *et al.*, 1976; Fuller, 1974; Gammeter and Frutiger, 1989; Geldiay, 1956; Ghetti and Gorbi, 1985;
Giller *et al.*, 1991; Gledhill *et al.*, 1993; Gore, 1979;1982; Griswold *et al.*, 1982; Guiney *et al.*, 1987; Harman, 1974; Harrel, 1985; Hauser *et al*, 1992;
Haynes *et al*, 1985; Hayner and Taylor, 1984; Heckman, 1983; Hoehn *et al.*, 1974; Hoiland *et al.*, 1994; Hopkins, 1961; Hopkins *et al.*, 1989; Hynes, 1960; 1970; Hynes, 1993; Jeffrey *et al.*, 1986; Kenk, 1974; Ladle and Ladle, 1992;
Ladle *et al*, 1980; Lake *et al.*, 1989; Lake and Schreiber, 1991; Macan, 1977; Mann, 1953; Mann, 1967; Marchant, 1988; McAuliffe, 1983; Minshall *et al.*, 1983; Mitchell *et al.*, 1993; Morrison, 1990; Otto and Sjostrom, 1986; Palmer et al., 1992; Pond Action, 1996; Pontasch and Brusven 1988; Raven and George, 1989;
Reice *et al.*, 1990; Resh and Rosenberg, 1984; Retallack *et al.*, 1981;
Reynoldson, 1978; Rosenberg and Wiens, 1976; Savage, 1989; Sheehan and Winner, 1984; Sheldon, 1984; Turner, 1992; Victor and Ogbeibu, 1986; Wallace *et al.*, 1986; Wallace *et al.*, 1990; Walton, 1980; Wentsel *et al.*, 1987; Williams and Hynes, 1976; Williams and Williams, 1993;Winner *et al.*, 1995; Wood and Petts, 1994; Yasuno *et al.*, 1981; 1982.