Project 001

Sources of Farm Pollution and Impact on River Quality

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Souces of farm pollution and impact on river quality

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This document is intended as a comprehensive record of the field investigations and literature reviews undertaken within the project. It is intended for circulation to members of the NRA's Rural Land Use Group only.

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EXECUTIVE SUMMARY

Reductions in river quality due to organic pollution from livestock farming are a concern in the UK and elsewhere. Effective control is hampered by the sheer number and widespread nature of potential polluting sources. There is a need for practical tools that will allow pollution control resources to be targeted at the worst affected areas and, within these, the problem farms. This project commenced in September 1989, with the main objective of using biological techniques to rapidly detect and pin-point farm pollution problems and to assess the benefits of farm visits.

Extensive fieldwork in West Wales has led to the production of simple biological indicator keys that allow the rapid screening of large areas for polluting farms. These keys rely on a few readily identifiable macroinvertebrate groups and sewage fungus, and can therefore be used by biologists and non-biologists alike. A national database of farm pollution risk has been developed using Geographical Information System software, based upon datasets of livestock numbers and physical factors contributing to farm pollution risk. This allows the targeting of pollution control effort at those area at highest risk. Intensive integrated monitoring of study catchments has identified links between specific farming practices and impacts upon river quality. Studies into the recovery of fish and macroinvertebrate populations following real and simulated farm pollution episodes have also been undertaken. These indicate that recovery of macroinvertebrate populations can be rapid (within 2-3 months), but salmonid populations may take considerably longer depending upon local circumstance.

This Project Record and associated Map Annex is a comprehensive account of the work undertaken during this study, incorporating all information previously reported in interim reports.

KEY WORDS

Livestock, farms, river quality, macroinvertebrates, indicator key, pollution risk.

1. INTRODUCTION

The WRc farm pollution research programme was initiated in 1987 in response to widespread concern over the increasing number of reported farm pollution incidents nation-wide. In collaboration with the water authorities, a catchment study was set up in tributaries of the Eastern Cleddau, West Wales, to investigate the impact of intensive dairy farming activities on the chemical and biological quality of rivers. Findings of this study have been reported in WRc progress reports PRS 2016-M and PRS 2347-M.

The funding of farm pollution research was taken over in 1989 by the National Rivers Authority (NRA) (with a contribution from SNIFFER, the Scotland and Northern Ireland Forum for Environmental Research), who have extended the scope of the work to provide nationally applicable tools for farm pollution assessment and control. The research is primarily targeted at the problems arising from organic farm wastes originating from livestock farming. Since many of the water quality problems associated with livestock farming are intermittent in nature, traditional discrete chemical monitoring is not suited to its assessment. Benthic macroinvertebrate communities, however, are affected by such intermittent pollutions and the impact is observable for some time after the pollution has ceased. Macroinvertebrate monitoring is therefore an effective means of farm pollution assessment and has great potential in future control strategies.

This Project Record, together with the associated Map Annex, is a comprehensive source of information on all work undertaken during the contract period, from 1 April 1990 to 30 September 1992. It is essentially a synthesis of all four interim reports produced during the study, but in addition it is an archive of raw data that is readily accessible for reference purposes and future studies. The majority of raw data is held in digital form as ASCII files on a single TapeXchange tapestreamer cassette (attached to the report). Data can be downloaded onto floppy discs using a TapeXchange tapestreaming system (Version 4.1).

2. PROJECT DESCRIPTION

2.1 Objectives

The overall objective of the agreed work was:

To develop biological methods for the detection of organic pollution from farms and for assessing the effectiveness of remedial measures. Such techniques could be used to quickly assess the pollution status of streams over a wide area in order to target pollution control efforts, and to subsequently assess the benefits of programmes of farm visits.

This overall objective was broken down into the following specific objectives:

1. To identify through biological assessment the principal sources of organic pollution from farms and quantify their impact on water/biological quality and fishery status of receiving streams.

2. To identify farming practices causing pollution and assess the effectiveness of remedial measures using biological techniques.

3. To develop within the NRA a strategy, using biological techniques, for the future monitoring and control of the farm pollution problem.

2.2 Work programme

A programme of work was agreed that would fulfil the objectives listed above.

2.2.1 Review of farm pollution and its control

Information on the impact and control of farm pollution in the U.K. and abroad would be collated and reviewed. This would involve a search of the published scientific literature and liaison with regulatory authorities in other countries.

Regional pollution control practices in the UK would also be reviewed in order to provide information for the development of future strategies.

2.2.2 National assessment of farm pollution risk

The nature, extent and distribution of farm pollution problems in the U.K. would be assessed, considering the distribution of livestock farming activity, farm management practices and reported farm pollution problems.

The information collated would be computerised and analysed using a Geographical Information System (GIS) in order to identify catchments within the U.K. at greatest risk from farm pollution. This analysis would indicate areas within which the biological methods developed during this study could be used. Further, it would act as a general aid to the targeting of both resources and attention at farm pollution problems.

2.2.3 Development of rapid biological appraisal techniques

Field investigations would initially be undertaken in West Wales in order to develop a protocol for the development of simple biological indicator keys for the rapid assessment of farm pollution. Such keys would permit use by a wide variety of staff with a minimum of training, so that a large number of sites could be visited with relatively little invested effort. The applicability of the approach to other regions would then be tested by undertaking additional work in another NRA region. The principles established would then be applied elsewhere in the U.K. according to the results of the national assessment exercise.

2.2.4 Detailed investigations of pollution processes

From initial surveys using a provisional indicator key, a number of impacted sites would be identified in order to investigate the specific farming practices causing pollution and the nature of the impact. Continuous water quality monitors linked to a telemetry system (METEORBURST) would be used at selected sites to enable comprehensive monitoring of pollution events, in addition to biological monitoring techniques.

Investigations of the transport and fate of pollutants following slurry spreading was initiated during the Eastern Cleddau catchment study. Further work was considered necessary to obtain suitable baseline data for a computer model being developed by WRc (under NRA Reference No. 007). The 'FARMS' model simulates run-off quality and assesses the risk of slurry application to land. The output from the model is a catchment map of field-scale resolution showing low risk application zones, which can be used to guide slurry disposal on farms.

2.2.5 Salmonid recolonization studies

A study of the recolonization by fish populations of river stretches impacted by gross episodic farm pollution was initiated during the Eastern Cleddau catchment study, when there was a major fish kill in the Deepford Brook following a discharge of slurry. This work would be continued under the new work programme, since further monitoring was necessary. Such studies are important since they allow a more complete picture of the timescale of impact on aquatic communities caused by such catastrophic one-off problems.

2.2.6 Biological recovery of streams

An investigation of the impact of episodic farm pollution on benthic invertebrate communities and the timescale of subsequent recolonization would be undertaken under subcontract by the University of Wales. This would act as a complementary study to the fishery recovery investigations outlined in Section 2.2.5.

2.2.7 Development of a national control strategy

On the basis of the tools developed during the study, a cost-effective strategy would be developed for the assessment and control of farm pollution.

3. LITERATURE REVIEW OF THE IMPACT AND CONTROL OF FARM POLLUTION

3.1 Introduction

A broad assessment of the adverse effects of farm wastes on water quality has been previously presented to the NRA in Bascombe *et al.* (1990). This subsequent review follows directly from these earlier findings, concentrating on the adverse effects of farm wastes from animal husbandry operations. **Since this review was conducted in the early part of 1990, information relating to legislation and pollution control activities are somewhat out of date (some additions were made in 1991). Where new developments are known, notes have been added in square brackets. One major change is the publications of a new code and Good Agricultural Practice (MAFF 1991), which supercedes all previously used advisory booklets mentioned in this review.**

Agricultural non-point source pollution has been identified as a major cause of the failure to achieve water quality objectives (Sherwood 1986), and pollution from farm waste has been considered to exhibit one of the largest and most unwanted expansions (Freeman 1986). The number of farm pollution incidents has steadily increased over the last ten years, as illustrated in Figure 3.1 (Payne 1989). However, 1989 and 1990 produced markedly lower numbers of incidents; since this has coincided with exceptionally dry weather it is uncertain whether this is due to improved farm waste management or climate.

With the increasing intensity of many agricultural activities, greater concern is being expressed over the impacts of these practices on water quality (Porter 1975). The effects of nitrate-based fertilisers, pesticides (e.g. Egboka *et al* 1989), silage (e.g. Beck 1989a) and animal waste disposal practices (e.g. Humenik *et al* 1987, Ritter 1988), are now being addressed as pollution problems. The contribution of agriculture to groundwater contamination is also receiving increasing attention (e.g. Bjerke 1989, Kovacs 1977).

The intensity of agricultural activities in England and Wales has already been described on a region-by-region basis in WRc Report No. PRS 2475-M. Overall trends of note are a general decline in cattle stocks, with dairy farming concentrated in the west and south west of the country. Pig farming predominates in the Anglian and Yorkshire regions, and the steady increase (23%) of this activity in the latter region since 1980 is also of note. There is an overall increase in numbers of sheep throughout all regions, which has resulted in a nation-wide increase of 31% since 1980. A steady rise is also evident since 1984 in the number of agricultural holdings, although total area of agricultural land has increased less rapidly over this same period. Despite these overall changes, indicating a greater extensification of farming activity (through an increase in agricultural area), more concentrated stock densities within specific regions (for example, pigs in Yorkshire) have led to a greater intensification on a much more localised scale.

Variations between regions in the importance of different types of livestock farming have meant that farm waste pollution problems receive different levels of priority. Water quality issues associated with farm waste production in different regions of the U.K. are considered in Section 5.

3.2 Nature of farm wastes

3.2.1 General

Farm wastes are of a variety of types dependent upon their source. Each type of waste poses a number of problems to the aquatic environment through different chemico-physical, biological and bacteriological impacts. The major issues concerned have been generally addressed for the benefit of the farmer by MAFF (1985a), and pollution of surface waters is broadly considered. The general toxicity of pesticides and herbicides is addressed, whilst the impact of organic wastes is considered with reference only to their BOD levels. Different types of agricultural wastes are frequently described in this context (e.g. MAFF 1986a).

The major causes of pollution resulting from livestock wastes which are considered to pose a risk to surface waters are summarised below:

1. BOD, which reduces dissolved oxygen levels in receiving waters, thereby killing aquatic life.

2. Suspended solids, which can smother the riverbed and reduce light penetration.

3. Nutrients, which may stimulate excessive plant growth (particularly phosphate, since this is normally growth-limiting in freshwaters).

4. Ammonia, which is directly toxic to aquatic life (especially in the un-ionized form).

5. Metals, notably copper and zinc, which are toxic to aquatic life.

6. Organic compounds (pesticides and oil-derived hydrocarbons), which are highly toxic to aquatic organisms.

The major sources of these contaminants are as follows:

- slurry;
- parlour, yard washings and dirty water generally;
- silage;
- pesticides (including spent sheep dip residues);
- oil.

These sources are considered in greater detail below.

3.2.2 Slurry

Slurries and manures can provide a valuable resource to the farmer for the fertilisation of agricultural land, despite the tendency to perceive them more as waste materials, with ever greater quantities being produced (Neeteson 1989). This value mainly stems from the high quantities of nitrogen and phosphorus which are present within the slurry and which become available to plants after application to the land. The influence of slurry application on nutrient leaching losses has been investigated by numerous workers (e.g. Furrer and Stauffer 1986). The lowest nitrate losses occur from slurries applied to permanent grassland. However, it should be emphasised that leaching of nitrogen is dependent upon soil structure, moisture and the cropping system adopted.

The mean chemical content of different cattle slurries is illustrated in Table 3.1 (Suss 1989). Typical BOD values for slurry are of the order of 10 000-36 000 mg l^{-1} (Mann 1975). This represents a more concentrated waste product than human sewage, but one which is normally disposed of by application to the land without treatment. Often, large quantities of slurry are stored for extended periods prior to disposal.

Table 3.1 - Mean chemical content of cattle slurry (Suss 1989)

	Dairy	Mean value mixed farms	Bull-farms
Dry matter (%)	7.5	7.7	8.8
Org. matter (%)	74.9	74.9	78.2
NH_4-N	2.4	2.9	2.6
Total N	4.3	5.1	4.6
P_2O_5	2.0	2.1	2.1
K_2O	6.3	6.4	5.3
CaO	2.4	2.1	2.0
MgO	1.1	1.1	1.3

Slurries are also typically rich in ammonia, which is present at concentrations posing a significant toxicity threat to aquatic life. The solids content of slurries varies greatly according to the amount of dilution it receives, but solids inputs to rivers can cause blanketing of the stream bed and loss of microhabitat (Beck 1989a).

Slurry derived from intensive pig farming operations can pose a greater risk to water resources owing to the intensive nature of modern pig farming (producing large quantities in localised areas) and the high organic content of the slurry produced. Frequently, pig slurry has a lower solids content than cattle slurry owing to the foodstuffs used. It has also been shown to be rich in heavy metals as a result of their addition to foodstuffs as a bacterial regulator. Poultry slurries can be of an even more liquid consistency and are likely to require drying before application to the land owing to the large volumes produced by intensive operations.

3.2.3 Yard and parlour washings

Large quantities of water are used in processing and cleaning on farms, and this is particularly so in the case of the dairy industry. These activities inevitably result in large quantities of waste water being produced which can have a high organic content. This problem is further compounded by the increasing tendency towards ever more intensive agricultural operations. An increased economy of water use has often been cited as an immediate measure to reduce risk to receiving waters, as well as the implementation of biological treatment measures where risks remain high.

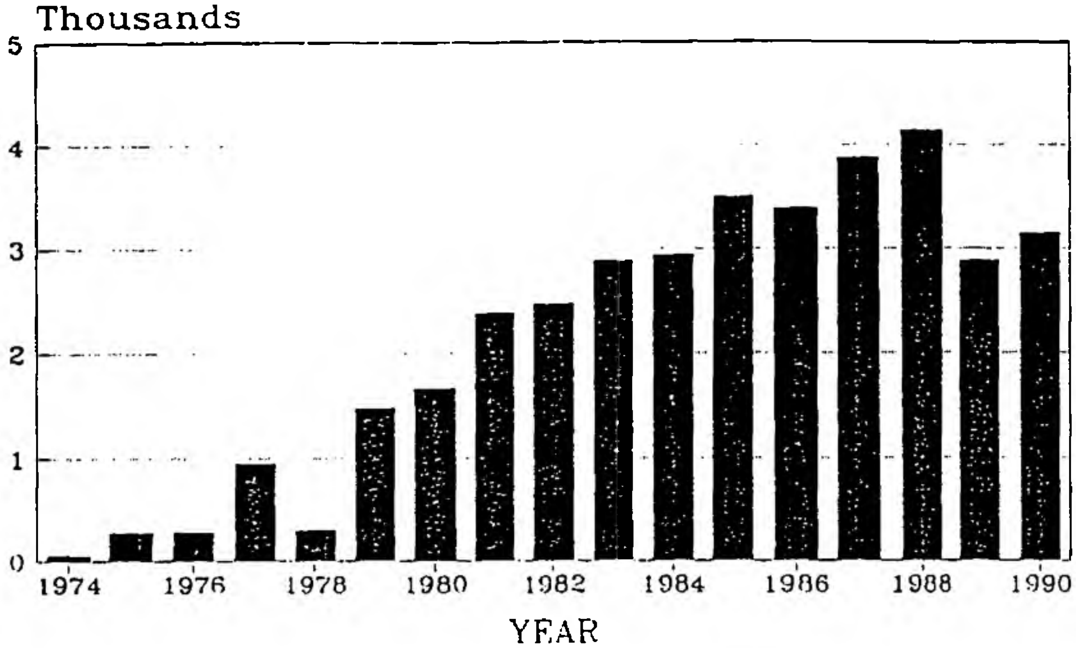
In addition to dirty water arising from farming activities, rainfall gives rise to run-off from farm yard areas; this can pose a particular problem since it is unplanned. The separation of 'clean' waters from roof run-off and 'dirty' waters from contaminated yard and parlour areas is therefore an essential step in the design of any farm management strategy.

3.2.4 Silage

Silage liquor is an acidic organic liquid waste with typical BOD values of the order of 30 000-80 000 mg l^{-1} (MAFF 1987). The highly concentrated nature of this waste poses a severe threat to the aquatic environment as only a small trickle entering a watercourse can result in severe pollution. The adverse effects of silage effluent upon receiving waters are of major concern and farmers have been made aware of the associated risks through numerous guidelines and recommendations (e.g. MAFF 1984).

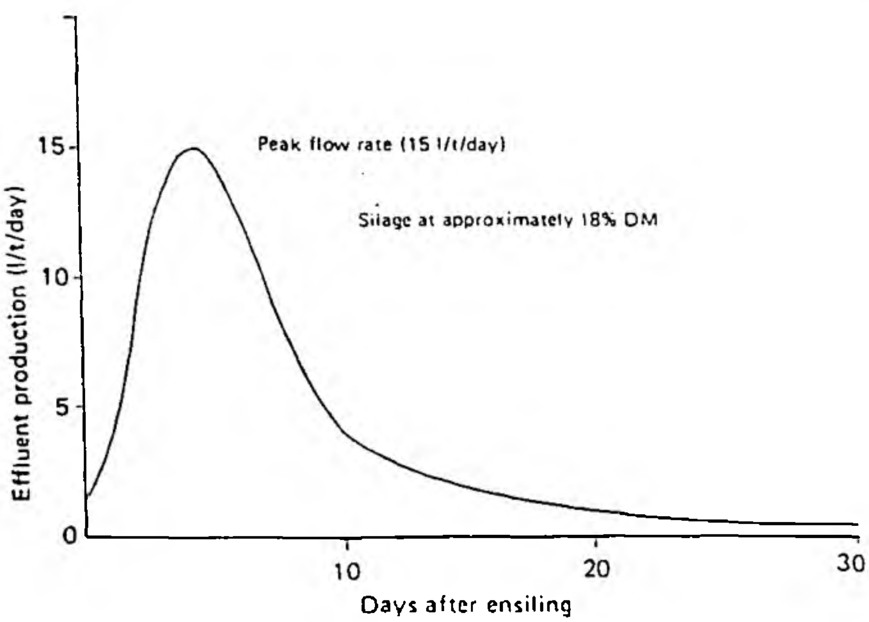
The production of silage effluent from a typical silage clamp is illustrated in Figure 3.2. Peak flows of around 15 litres per tonne silage day^{-1} occur 3-5 days after ensiling. However, the magnitude of liquor production varies with the moisture content of the grass at the time of ensiling. Wilting, which results in a drier silage, therefore reduces liquor production and consequently pollution risk, but the prevailing weather conditions in the U.K. and the need for several early silage cuts means that cut silage will never be completely dry and a liquor will always be produced. Wilting to 25% dry matter has

Figure 3.1 Total numbers of pollution incidents caused by farm wastes in England & Wales, 1974–90.



Data from Payne (1989) and annual farm waste reports (eg NRA/MAFF 1990)

Figure 3.2 Typical production of liquor from a silage clamp (from MAFF 1984).



therefore been recommended (MAFF 1984). Protection from rain-water must also be an important consideration, as additional dilution after ensiling will directly increase the quantity of liquor that can be produced, and therefore increase the degree of pollution risk. Clear definitions are required for the controlled disposal of silage liquor. The discharge of silage effluent can only be undertaken with the written consent of the NRA, and the usual requirement is for BOD to be less than 20 mg l^{-1}. Whilst discharge requirements cannot be relaxed to prevent the storage of large quantities of silage, the quantity of liquor produced can be reduced by minimising the dry matter content of cut silage by cutting during suitable weather, and ensuring its isolation from rain-water during storage.

The introduction of milk quotas and MAFF advisory leaflet advice have encouraged a recent increase in silage production to reduce costs. Whilst it is known that efficient wilting can reduce (and virtually eliminate) silage liquor, and is often a practice favoured by farmers, this is not always possible due to unfavourable weather conditions. Silage pollution is most commonly caused by the inadequate design and failure of storage facilities. Allcock (1989) suggests the adoption of additional external peripheral drains for silage clamps. Prevention of silo floor leakage by the use of hot-rolled asphalt rather than concrete has also been suggested as an improvement (Beck 1989b).

Schua (1970) has considered the effects of discharges from fermentation silos and the threat posed to water quality from increasing volumes of silage liquor, outlining the danger to receiving waters and sewers. These problems were considered to be most serious during peak production periods in summer.

Littlejohn and Melvin (1987) have considered the pollution threat of phenols originating from silage on surface water quality. These effects are more often encountered from sheep dip sources but are also associated with grass fermentation processes.

3.2.5 Pesticides

Pesticides are directly toxic to organisms, and therefore pose a significant threat to aquatic organisms by their entry into surface waters through spillage, surface run-off or unsafe disposal. The general impact of pesticides on water quality has been addressed in a separate NRA-funded WRc study (NRA Project Reference A10(90)2), while the specific impacts associated with the use of sheep dipping materials have been considered in a further study (NRA Project Reference A19.53).

A major problem encountered when investigating the adverse impacts of pesticides on water quality is that effects can often occur while the pesticides themselves are present at concentrations lower than the accepted limit of detection. Furthermore, information on the toxicity and environmental hazard in surface waters of pesticides is often incomplete, owing to the large number of compounds widely used, and the rapid rate of development of new compounds by the industry.

3.2.6 Heavy metals

Heavy metals pose significant risks to water quality when present in high concentrations. Sources of heavy metals include slurries, notably those of intensive pig and poultry operations where metals (especially copper and zinc) are incorporated into animal food supplies as bactericides (MAFF 1986b). Van Erp and Smilde (1989) have considered the risk of contamination of waters from excessive application of slurries containing copper, zinc and cadmium, and concentrations are presented in Table 3.2. Fleming (1989) has discussed the problems associated with the application of pig slurry to grassland. At regular rates of application, copper and zinc in particular were found to accumulate at the soil surface. This has been considered in some areas to pose a secondary danger to grazing sheep (e.g. MAFF 1985a) as well as a potential problem for humans where application is to arable crops or where waters within the catchment are abstracted (Joseph and Clark 1982). McGrath *et al* (1983) have considered seasonal variations in the metal content of sheep livers, concluding that where slurry is spread there is a risk of contamination to grazing animals.

Table 3.2 - Copper, zinc and cadmium levels in slurry (from Van Erp and Smilde 1989)

	Cd	Zn (g/100 kg manure)	Cu
Cattle slurry	0.03	5	3
Pig slurry	0.07	38	22
Chicken slurry	0.11	64	18
Chicken manure	0.35	205	57
Broiler manure	0.41	218	68

3.2.7 Hydrocarbons

Hydrocarbons can cause serious problems in surface waters by preventing the exchange of oxygen across the water surface and through direct toxic effects. Such effects can cause catastrophic local impact upon aquatic communities, and necessitate expensive remedial operations if large quantities enter a watercourse. Fuels and oils are often stored in large quantities on farms, and pose a particularly high pollution risk where storage facilities are unbunded, or where spillage or leakage occurs during filling operations.

3.3 Sources of water pollution

General risks to surface water quality posed by farming activities can arise from a wide range of sources. Poor quality farm building work, and the inadequate supervision of farm workers have been considered to be factors of importance in increasing the potential for water pollution. These risks are often exacerbated by the occurrence of unexpected weather conditions, for which inadequate planning has been made.

Agricultural activities involving wastes which pose a risk of water pollution can be broadly divided into three phases: storage, treatment and disposal. These three phases are discussed separately below.

3.3.1 Storage

Major risks posed through the storage of agricultural wastes arise as a result of the following factors:

- o lack of separation of clean and dirty waters in mixed drainage systems, leading to the production of larger volumes of liquid wastes;
- o inadequate storage capacity;
- o deliberate breaching of the embankments of full slurry lagoons in an attempt to allow the supernatant to escape;
- o building overflow pipes on effluent tanks;
- o the sudden collapse of lagoon embankments, manure storage walls or silage clamp walls;
- o cracks in silage clamp floors;
- o clamps with rubble floors sited over land drains;
- o silage stored outside of silage clamps or bales.

With increasing agricultural intensity, there is an increasing trend to provide capacity on the farm to store ever greater quantities of silage and agricultural wastes. In the case of silage, the increase in pollution risk due to larger quantities is being partly offset by the trend towards big-bale storage. However, in general the standard of construction and maintenance required of storage facilities will be more important in the future if larger quantities of farm wastes are to be contained. The new regulations on the storage of slurry, silage and fuel oil (DoE 1991) set minimum standards for storage facilities and are a useful development.

Ensuring an adequate period between silage cuts enables the percentage dry matter to be maintained within silage, and should be encouraged as a lower volume of liquor is produced. The selection of a suitable site for storage, and the efficient design, maintenance and management of clamps and silos are vital for the prevention of accidental spillage. The practice of bankside storage should be actively discouraged, as the proximity to waters considerably increases the likelihood of accidental spillage.

With the increasing quantities of materials stored in the U.K., there is clearly a greater need for tightened control over the prevention of spillage. A greater emphasis should be placed on the prevention, rather than the amelioration after the event, of the adverse effects of stored wastes on water quality.

3.3.2 Treatment

The treatment of farm wastes is not a widespread practice in the U.K. farming industry. Treatment to produce a less hazardous end-product may allow a greater initial quantity of a waste to be handled at the farm, provide a smaller quantity of a less hazardous waste for disposal to the land, or produce an end product which can be profitable in itself (e.g. composting). In certain cases, such as for agrochemical wastes, treatment may be a necessary measure before safe disposal is permissible.

While the use of treatment facilities seems a potentially valuable method to manage the large amounts of waste produced by intensive livestock farming, in practice such treatment options are not only expensive to install and run, but are frequently overloaded, and in most cases poorly understood and maintained by the farmer. These problems mean that treatment plants, rather than providing a solution to farm waste management, can result in further pollution problems, particularly if the effluent is to be discharged to a watercourse. Treatment plants are also subject to failure, and if large volumes of wastes are being handled the pollution risk is high.

3.3.3 Disposal

In the majority of cases, the most frequently chosen disposal method for agricultural wastes is application to land. A range of equipment can be employed: travelling and static irrigators, conventional tanker systems or soil injection methods for the disposal of slurries and dirty water, and muck spreaders for manures. Animal wastes are a valuable source of nutrients, but there are a number of risks attached to land disposal practices and care is required to prevent water quality impacts.

The principal causes of water pollution from land disposal are as follows:

- over-application of slurry to the land, causing saturation and/or surface run-off;
- the application of slurry to frozen soils;
- application to steeply sloping land, to soils of low permeability, or to land adjacent to water courses;
- application during periods of rainfall, especially when the land is at field capacity;
- disposal guns left unattended for extended periods;
- irrigation pipes with leaks;
- reliance on a neighbour's land for slurry disposal;
- the deliberate emptying of slurry tankers into ditches.

Official guidelines operating until recently gave rather vague recommendations for minimising of the pollution threat posed by disposal methods, for example, the recommendation not to dispose 'close to' a stream (MAFF 1985a). The new Code of Good Agricultural Practice (MAFF 1991) gives improved guidance, but there is a need for more precise advice and the development of waste management plans on individual farms, approved by regulatory authorities.

The disposal of slurry to public sewer has been considered as a safe, although potentially expensive, alternative to land disposal. Guidelines produced for disposal to sewer included the enforcement of control by issuing a 'consent' to discharge upon receipt of a 'trade effluent notice' from the discharger, giving details including the premises, the operation producing the effluent, composition of the effluent and the volumes and times of discharge. However, discussion between the then National Water Council and NFU concluded that the application of farm wastes to land was preferable wherever this was possible and that the minimising of discharges to public sewers was to be favoured. It was considered important that silage liquor, sheep dip wastes, pesticides, herbicides, and petroleum and oil products should not be discharged to sewers.

The disposal to land of 'dirty water', i.e. effluent consisting of water contaminated with manure, urine, cleaning materials, crop seepage and other waste products (MAFF 1989a), has been described with reference to its polluting nature (MAFF 1989a,b,c). Probably the most important single preventative measure that can be taken by the farmer to prevent water pollution is the separation of clean and dirty water at source. The dilution of wastes from yard areas by rain water serves to increase the quantity of waste waters profoundly, causing increased problems of ensuring safe disposal. Remedial action is simple, often only requiring the installation of guttering to farm buildings and the isolation of this water from yard areas. This can clearly be implemented at minimal cost to the farmer, but is a measure which is not eligible for financial aid under the Farm and Conservation Grant Scheme, since it is deemed not to pertain directly to the prevention of pollution (as it relates to the handling of clean water rather than effluent).

The safe transport and handling of slurry prior to disposal to land are described for the benefit of the farmer in advisory booklet 2126 (MAFF 1985b). Considerable information is provided on the effective handling of these wastes; the adverse effects of over-application and accidental spillage are considered separately in booklet 2200 (MAFF 1985a).

Soil injection of slurry offers an alternative which has a number of advantages over surface application. Hall (1986) has considered the status of soil injection research in the U.K. An extensive research programme on soil injection has been carried out at WRc in order to find better spreading methods to prevent possible air and water pollution problems arising from surface spreading. Soil injection has been considered to be a means for meeting environmental quality objectives, as it offers reduced disposal costs, better pasture hygiene, nutrient management and soil loosening. To an extent the quantity that can be injected is governed by land slope and soil moisture condition at the time of application.

Slope: The control of surface run-off demands safe injection on slopes without run-out from the cavity (made by injection) at the base of the slope. To facilitate this, a maximum gradient of 13 degrees has been chosen, this being largely defined by steeper slopes not allowing the safe negotiation of tractor and tanker. Run-off can be controlled by the percentage of dry solids (i.e. the mobility) within the slurry applied.

Soil moisture: This factor is very much influenced by soil type. Most success has been had by injection in the spring on structured soils when they are in their most plastic condition. The moist surface soil layers and lower evapotranspiration rates of the crop allow satisfactory results for autumn injections.

The combination of slurry with animal litter provides a greater dry matter content, and therefore constitutes a lower pollution risk to receiving waters as pollutant mobility is reduced. Although in the U.K. there has been a tendency not to use litter in most animal housing, this is not the case throughout Europe, and the use of alternative litter types for dairy cattle has been considered (e.g. Peltola 1986).

3.3.4 General

MAFF advice concerning the storage and handling of slurry is considered to be inadequate for farms situated close to waters, on steep slopes, or within high rainfall areas. The absence of appropriate advice for farms in these conditions leads to the poor siting, construction and maintenance of such facilities. Overflowing and swamping of inadequate handling systems is frequently cited as a cause of pollution, resulting in the adoption by farmers of environmentally dangerous practices such as the deliberate breeching of lagoon walls to allow slurry seepage, and large-scale spreading operations being carried out in winter despite contrary MAFF advice. Inadequate knowledge therefore leads to insufficient care being taken and results in increased pollution risk.

3.4 Impacts of wastes on surface water quality

3.4.1 Physico-chemical effects

Middlebrooks (1972) has provided an account of the effects of agricultural practices on water quality, detailing increases in total solids concentrations, nutrient enhancement (notably phosphorus and nitrogen), temperature increases, and the effects of insecticides and herbicides.

Wallace (1971) has considered the inter-related variables determining the effects of land run-off on water quality in receiving streams and designed a computer model to estimate the effects of agricultural run-off on dissolved oxygen (DO) concentration in rivers. The relative significance of this type of pollution compared to discharges of municipal and trade waste waters was also considered. Results showed that existing waste water discharges have negligible effects on the DO concentration, but storm run-off was responsible for serious oxygen depletion. In the Iowa River it was found that concentrations of DO lower than 5 mg l^{-1} could extend for up to 75 miles along the river length, often lasting for periods of over 24 hours and recurring several times each summer.

Placido and Soldatini (1988) have investigated the Cornia river, Tuscany, Italy, a catchment characterised by non-intensive agriculture and an absence of industrial discharges. After studying its hydrology, sampling stations were set up above and below each confluence, and water quality parameters were determined. The results indicated that while dissolved oxygen was generally maintained at around saturation level, intermittent impacts associated with agricultural waste sources could result in a dissolved oxygen concentration as low as 23 per cent.

A number of farm pollution studies in the literature deal with livestock farming as a cause, either singly or in combination with other sources, of lake or reservoir eutrophication. Examples are the studies on a number of Irish loughs (John *et al* 1982), the **Canonsville Reservoir** in New York State (Brown *et al* 1989) and **Lake Okeechobee** in Florida (Flaig and Ritter 1989).

3.4.2 Bacteriological effects

Several investigations have considered the bacteriological effects of farm wastes on water quality. In such studies, the most useful microbiological parameters are considered to be faecal indicator bacteria. The bacteriological content of farm slurries has already been reviewed (WRc Report PRS 2484). Excreta from livestock has been highlighted as the most probable source of cryptosporidium in cases of suspected drinking water contamination (DoE/DoH 1990).

The microbiological monitoring of rivers and estuaries has not until recently been encouraged within the U.K., and management decisions about water quality have only infrequently involved the results of bacteriological analysis (White and Godfree 1985). However, with the development of Environmental Quality Objectives for rivers, research is now being funded by the NRA into the microbiological quality of freshwaters and the sources and mechanisms of contamination.

3.4.3 Biological effects

Little information is available in the scientific literature on the effects of farm pollution on river biota. The WRc study on the Eastern Cleddau catchment provided valuable information on biological impact (Schofield and Bascombe 1990), and this has been continued during the present study (this report, Sections 5, 6, 7, 8 and 9).

In Ireland, phosphorus enrichment of **Loughs Ennell** and **Sheelin** from inadequately treated sewage and pig slurry has caused inhibition of charophyte (stoneworts) growth and stimulation of other algae and macrophytes, leading to a contraction in the extent of important charophyte beds. Champ (1990) reports that nutrient levels in Lough Sheelin and the frequency of pollution incidents around the catchment increased through the 1970s due to a rapid increase in pig numbers (from 9000 to 50 000 fattening places). This led to the incidence of algal scums and enhanced growth of filamentous algae. By 1982, algal blooms were causing deoxygenation on a scale that restricted trout to the top 5 metres of the water column, whilst dead trout were observed on the lough bed. Low oxygen levels in cold hypolimnion water have been suggested as the cause of the disappearance of arctic char (Champ 1977).

Studies of the invertebrate fauna of Lough Sheelin (Dodd and Champ 1983) indicated a virtual disappearance of the previously abundant mayfly *Ephemera danica*, and drastic reductions in the numbers of caddis-flies (*Phryganea* spp, *Mystacides* spp, *Leptocerus* spp, *Cloeon simile* and *Centroptilum luteolum*). By 1979, the main diet of 'rising' trout was emerging chironomids. An integrated catchment management plan for farm waste, instigated in the early 1980s (Section 3.7.7), is now allowing a recovery of the biological community, although signs of improvement were only recently observed.

Lenat (1984) has studied the effect of agricultural run-off on water quality and the effectiveness of recommended agricultural management techniques. Well managed and poorly managed catchments were compared with control sites in various geographic regions. Physico-chemical measurements showed that agricultural activity had an adverse effect on water quality but these influences could be minimised by good management practice. Differences found in the species richness of benthic macroinvertebrate communities indicated that erosion control could also mitigate the impacts of agricultural run-off. Poorly managed sites had the least stable benthic communities. Ephemeroptera, Plecoptera, Trichoptera and Coleoptera were considered to be the most intolerant taxa. Collector-gatherers, scrapers and filter feeders were considered to be most tolerant of agricultural run-off, suggesting that the addition of particulate organic matter augmented food supplies. Agricultural run-off was demonstrated to cause water quality changes which affected the stream biota, but this effect could be reduced by using recommended erosion control techniques.

Some of the most serious pollution incidents occurring in England and Wales are briefly described each year in the annual farm waste report (NRA/MAFF 1990), giving the total number of fish killed. The most serious incident occurring in 1989 was a rupture of an earthen-banked pig slurry store in NRA Anglian Region, where 3 million gallons of slurry killed 10 500 fish and produced measurable effects over a distance of 60 km. This is an extreme case, but numerous incidents occur each year where hundreds of fish are killed due to farm wastes entering watercourses and causing pulses of intense pollution.

It should be noted that incident statistics are summarised in the annual farm waste report into categories of 'serious' or 'minor', depending on whether an incident affects one or more of a variety of water uses; this means that it is not possible to extract information from the farm waste report on the scale of biological impact nationally (i.e. how pollution incidents generally relate to biological effect), since effects on the biota are combined with effects on other water uses. This is a serious drawback to the current format of reporting farm pollution incident statistics. Moreover, the farm waste report does not deal with chronic farm pollution, where biological communities may be continually impoverished without readily observable catastrophic effects.

An example of a targeted biological study is that on the **Wyre** in NRA North West Region (NRA internal report), where biotic indices (Biological Monitoring Working Party score, Average Score Per Taxon) indicated highly impoverished invertebrate communities at a number of sites (BMWP scores between 15 and 30, ASPT scores between 3 and 5) between 1987 and 1989. Active discharges from farms were found upstream of a number of impoverished sites; other sites with poor biological communities were likely to be recovering from episodic farm pollution events, which are unlikely to be discovered by pollution control staff.

The Nature Conservancy Council (NCC 1991) reported that in a selection of regions in England and Wales, 71 Sites of Special Scientific Interest (SSSIs) are known to have been affected by pollution from livestock wastes between 1986 and 1990. Of these, 19 sites were streams or rivers, 8 were ditches, 31 were lakes and 13 were mires. Seventy percent of sites suffered from slurry pollution, often from land run-off. In addition to these sites, another 16 SSSIs have eutrophication problems where livestock pollution is suspected. It should be noted that the true number of sites of high ecological interest affected by farm pollution is likely to be much higher than this for a number of reasons: firstly, owing to the difficulties of adequately protecting riverine SSSIs, the NCC has until recently restricted designation to only a few sites; secondly, the NCC report does not consider the whole of England and Wales; lastly, even within the regions considered, monitoring effort is not sufficient to cover all sites of concern.

3.5 Preventative strategies and remedial measures

There is a need for the development of appropriate preventative strategies and remedial measures for the control of agricultural non-point source pollution. Models which incorporate the determination of dissolved pollutant concentrations in run-off and estimation of pollutant delivery to surface waters are considered to be useful screening devices prior to the design of catchment management plans.

The choice of preventative strategy that can be applied is largely dependent on the risk posed by the waste concerned. Such strategies have therefore also been grouped according to the three waste management phases of storage, treatment and disposal.

The responsibility for the implementation of preventative measures originates with the farmer, who must be provided with both the necessary advice (through consultation and development of the code of practice), and the incentive (both in terms of awareness of the problem through education, and financial aid) to tackle the problem at source. Such measures need not be costly to be effective; for example, the covering of slurry storage tanks to prevent the ingress of rain-water (Mannebruck 1986), which can also reduce odour problems, should be encouraged.

3.5.1 Storage

The cost-effectiveness of a range of farm waste handling facilities has been addressed in a separate report (Brewer 1990). Recommendations for the design and location of farm waste storage facilities are detailed in a number of ADAS publications (MAFF 1987, 1989d,e,f,g,h). In these booklets it is stressed that careful consideration should be given by the farmer to avoiding pollution from manures and slurries, and a reminder of the pertinent legislation is found at the rear of each leaflet. The choice of facility depends upon the livestock type and nature of the slurry produced (MAFF 1989d). While the advisory guidelines are detailed within the code of practice, the farmer has until recently only been liable in the event of spillage, with no legal obligation to obtain approval for the adequacy of his storage facilities. The recent introduction of regulations to control pollution from farm storage facilities will largely redress this problem (DoE 1991), although some of the required specifications now imposed fall short of the standards imposed on the continent (Sections 3.6 and 3.7).

3.5.2 Treatment

As stated in Section 3.3.2, the use of treatment facilities provides a potentially useful tool to the farmer for the disposal of excess farm waste material, given increasingly intensive operations and ever greater quantities of wastes being produced.

A wide range of systems has been developed for slurry treatment. Most trials, however, have proved disappointing and it is considered that as yet there is no system which is able to provide a final effluent which is suitable for safe direct discharge into a river. Two processes have been widely applied:

1. **Anaerobic digestion** - This method produces a gas which has the potential of being a fuel source to the farmer given the appropriate facilities. Nevertheless, the cost of installation and operation of facilities able to make use of this fuel as yet outweigh the potential benefits and are not encouraging to farmers. Experimental units have proved to be both expensive and problematic (Hobson 1984); a high degree of supervision is required. Anaerobic digestion of slurries has not been widely employed and only about ten installations exist in the U.K. at present. Anaerobic digestion is practised widely on the continent, in both small scale (individual farms) and large scale (communal) plants (Section 3.6).

2. **Aerobic systems** - These have been extensively researched in the U.K., but WRc studies in 1975 yielded no encouraging conclusions (Beck 1989a). Generally, aeration of slurry is used to reduce smell as a priority rather than polluting strength (Brewer 1990), although attempts have recently been made to reduce the polluting power of dirty water. Aerobic systems have also been considered to be particularly problematic for the handling of poultry wastes (Riley 1988).

The use of reed bed systems for the treatment of concentrated agricultural wastes has been considered. Gray *et al* (1990) have assessed experimental horizontal-flow reed beds, considering the quality of agricultural wastes before and after their movement through the bed. It was concluded that the use of artificial reed beds would provide a potentially low-cost means of treating organic agricultural wastes with BOD levels up to 3000 mg l^{-1}, the resulting effluent being suitable for disposal to land but not to watercourses. The use of reed beds for the treatment of poultry manure wastes has also been shown to be promising, with up to 80% BOD removal efficiency (Vymazal 1990).

Aerobic treatment has been considered to be a viable option for the control of waste from intensive pig farming operations. Primary treatment of piggery wastes by lagooning may make such treatment more effective.

Poultry manure is problematical since water content can be high (depending upon waste management design). The dewatering of chicken manure by vacuum filtration, and the removal of moisture from poultry waste by electro-osmosis have therefore received attention.

Harrod (1989) has recommended that on-farm treatment be adopted for the control of farm wastes. This has the cost advantage of no initial transport requirements, although does little in itself to reduce non-point discharges. Integrated on-site treatment plants for piggery wastes are commercially available (Brown and Gibbs 1986). Guiver (1989), however, has indicated that farmers are not likely to employ on-site treatment unless methods are both foolproof and cheap, with few maintenance requirements.

An alternative to on-farm treatment is the establishment of treatment plants which serve a number of farms. Although not in use at present in this country, government-subsidized communal treatment plants are in operation in Europe (Section 3.6).

Chemical treatment of waste pesticides has been assessed by Johnson and Harris (1989). Since many pesticides are only partly soluble, flocculation of suspended matter followed by the use of activated carbon, for the removal of the relatively small amount of dissolved pollutants remaining, was recommended. Potential costs are nonetheless high, although this process is considered cost-effective for dealing with 'very difficult' toxic pesticides. Such treatment provides an effluent which may be disposed to public sewer. Only small amounts of solid waste are produced, so the cost of disposal to the waste contractor is low.

Waste materials otherwise disposed of are potentially useful as a fuel and therefore power, and power generation systems are commercially available (e.g. SCS Biotechnology 1989). The formation of integrated 'rural energy centres', which combine treatment and power generation, has been considered by Locke *et al* (1985), with the responsibility for the establishment of such centres obviously resting with agricultural organisations. Such projects represent potentially profitable business investments and are therefore areas worthy of further consideration by the farming community and NFU. Methanogenic fermentation (Klinger and Marchiam 1986) has been recommended as a solution for ecological problems involved with the accumulation of animal wastes, as well as providing a profitable end-product. Biofiltration (Noren 1986) has also been suggested, which has the advantage of reducing odours and ammonia volatilisation.

Harrod (1989) has considered the problems associated with straw based wastes due to their bulk when handling. There are advantages associated with the use of straw-based wastes in terms of pollution prevention, since the high proportion of dry matter makes them much less mobile and therefore less likely to gain access to watercourses in large quantities. Composting represents an opportunity for farmers to obtain a profitable end-product from excess slurry production. While composting may represent too costly an investment for an individual farmer, it nevertheless may be attractive to a consortium of interested individuals. This approach has already proved to be successful in the Netherlands (Fane 1989, see Section 3.6). Kroodsma (1986) has illustrated a process for the composting of poultry manure. These products are normally treated like liquid manures, but when air dried and heated, were found to produce a dry, odourless manure of at least 55% dry matter, suitable for application to most soils. Gonzalez *et al* (1989) have further described options available for composting procedures.

Incineration of farm wastes is likely to become more feasible as technological improvements are made, but it is very unlikely that this will prove to be an economical or practical proposition in the foreseeable future.

3.5.3 Disposal

Disposal options are constrained by three factors:

- o the persistence of the waste applied;
- o the mobility of pollutant constituents upon and within the soil;
- o the direct toxicity of those pollutants and the toxic effects of their degradation products.

Recommendations for the reduction of pollution risk during the disposal phase must therefore address each of these factors. With the exception of pesticides, the persistence of wastes does not pose a major problem, since most are highly biodegradable. However, most organic wastes are highly mobile and are therefore prone to direct run-off following their application to land. The main exception to this is manure, which has a much higher dry matter content.

The majority of farm wastes are highly toxic as has been already established in Section 3.2. The toxicity of pesticides is complicated by the occurrence of toxic breakdown products, which may present a greater hazard than the parent substance (both toxicity and persistence may be higher). Disposal options for the waste types of most concern are dealt with separately below.

Slurry

The efficient use of slurry in land application requires care to be taken during both storage and handling (Suss 1989). It is important that the farmer should maximise the fertiliser value of slurry, whilst minimising pollution risk. Nitrogen and phosphorus losses should therefore be kept to a minimum, and it may be necessary to treat the slurry before its application.

Farmers have a vested interest in ensuring a high soil retention of nutrients from slurries when fields are dressed. Retention will be influenced by the methods of collection and preparation of the slurry, nature of the soil, land topography, slurry composition, application method, and the weather conditions during and after spreading. Besson *et al* (1986) have compared nitrogen losses from different slurry types when applied to the land. For cattle slurries, losses were found to be greatest for methane-digested slurries, with a better nitrogen retention from stored and aerated slurry.

The adverse effects of incorrect slurry application have been reviewed by Vetter and Steffens (1989). It was noted that their application was rarely optimal, often owing to the vast quantities that are produced from intensive agricultural production. The problems associated with incorrect application are numerous, and have been summarised in Table 3.3 (Hojovec 1989).

Table 3.3 - Problems of incorrect applications of slurry (Hojovec 1989)

Soil	Nutrient enrichment (phosphorus, zinc, copper, nitrogen and potassium). pH too high after the long-term use of large poultry slurry dressings. Salt accumulation in dry areas. Excessive loosening of soil after large applications. Compaction during wet conditions or by heavy slurry tankers. Anaerobiosis after incorporation to wet soils.
Crops	Yield decrease by over-fertilization (e.g. lodging of cereals). Negative effects on plant quality (e.g. decrease in settled yield of sugar beet; decrease in starch content and keeping quality of potatoes, magnesium deficiency in green fodder and therefore diet of grazers after excessive potassium dressings). Increased weed competition. Negative effects on grassland sward composition. Scorching due to surface dressing, especially in hot conditions. Smothering of plants, especially after slurry dressings with high dry matter content. Risk of direct contamination (decrease in fodder intake by grazers and poor silage quality). Risk of spreading animal and plant pathogens.
Water	Groundwater pollution, especially nitrate. Nitrate, ammonium, phosphate, copper and organic enrichment from run-off; also enrichment following erosion of phosphorus and copper from soils, especially on steep slopes and bare compacted soils.
Atmosphere	Odour. Ammonia volatilisation, and other emissions.

The timing of application is also important in order to maximise nutrient uptake by plants. The proportion of nitrogen lost due to leaching from slurry applied to grassland in different seasons has been calculated (Table 3.4, Archer personal communication). High losses sustained during the autumn and winter suggest that spring is the best slurry application time (Demuynck 1984); since plant growth is maximal at this time; however, it should be remembered that land is likely to be at field capacity during spring, so the risk of surface run-off is still high.

Table 3.4 Percentage nitrogen lost to leaching from slurry applied to crops and grassland (Archer personal communication)

			Nitrogen leached expressed as % of total manure N applied	
			Arable	Grass
Farmyard manure	-	autumn	15	10
	-	winter	10	5
	-	spring	10	5
Cow/pig slurry	-	autumn	30	10
	-	winter	15	10
	-	spring	5	5

The timing of slurry production, as well as the timing of its disposal, is an important consideration; the practice of overwintering dairy cattle in animal housing, for example, will concentrate slurry production within small areas at this time, and should be considered when assessing disposal options.

It is often difficult to apply at rates lower than 15 m^3 ha^{-1}, although even this rate may be too high for the application of highly concentrated slurries such as poultry and pig manure (Vetter and Steffens 1989). It is therefore likely to be necessary to encourage the development of equipment which is capable of applying slurry at lower rates.

On arable land, it is important to maximise the incorporation of nutrients within the soil, so that losses are minimised; for this reason, incorporation should be carried out directly following spreading. It has been recommended that spread slurry should be incorporated within the soil within 2-3 hours after application. Such incorporation should not involve the deep-ploughing of slurry, but benefits to the soil should be maximised by tilling the top 5-10 cm, thus making nutrients available to the roots of young crops. Gaseous losses are very low after such shallow incorporation. Soil compaction should be avoided through, for example, the use of low ground pressure tyres and umbilical supply equipment. Plant damage can be caused by application rates being too high, dressing when crops are at a susceptible growth stage, when slurry is applied to frozen ground, or where application is followed by dry or hot weather or deep frost.

The relative merits to the farmer of different land spreading options have been described in detail (MAFF 1985a). Climatic and soil conditions, land use types and the sensitivity of the environment to rates of application must be considered within any proposed guidelines. MAFF recommend that a maximum slurry spreading limit of 50 m^3 ha^{-1} is adhered to where any risk to surface waters is identified, and that there should be 3 week intervals between applications; however, the basis of this application rate is not clear.

In contrast, Vetter and Steffens (1989) recommend that single dressings per growth period, using undiluted slurry of 10% dry matter not exceed the following values:

cutting areas: 15-20 m^3 ha^{-1} cattle slurry, or 10-15 m3 ha^{-1} pig slurry;

grazing areas: 5-10 m3 ha^{-1} cattle slurry (up to 15 m3 ha^{-1} in autumn).

The timing of land disposal of slurry in the U.K. is often restricted by current farming practices (though not by law). Two to four silage cuts are made each year, and it is considered undesirable to spread slurry until after the last cut, since the farmer fears that grass yield may be reduced. If the last cut is late in the year, as is usual, rainfall and low or non-existent soil moisture deficits further restrict spreading. However, if slurry is applied little and often through the growing season, it is likely that the effect on grass yield will be negligible (Section 3.7). Alternatively, soil injection of slurry provides a method of supplying nutrients to grassland during the growing season without risking any loss in grass yield through smothering and scorching.

Water-logged and frozen land is unsuitable for spreading as slurry cannot be efficiently incorporated within the soil. Water-logging can also occur due to excessive application, therefore preventing the incorporation of surplus slurry. Problems can also be encountered in excessively dry weather, when cracks may open up in clay soils due to shrinkage; these may provide rapid access to watercourses via under-drainage systems. Avoiding such situations when applying slurry is an essential part of any preventative strategy.

In order to minimise pollution risk from the land disposal of slurry, it is necessary to develop a clearly defined programme of spreading which ensures: a uniform rate of application over the entire available land area (not just the most accessible fields); and application to specific fields at a time when those fields are best suited to the acceptance of slurry, considering factors such as soil type, slope, rainfall and trafficability.

Silage liquor

Current recommendations for the minimisation of pollution risk from silage liquor are:

- o to prevent effluent production by wilting (Bastiman 1976);
- o be prepared to store large volumes and have necessary disposal facilities (Beck 1989a).

Of these options, the widespread trend is to increase storage capacity, since if silage is left to wilt for short periods only it is possible to harvest more frequently and enable greater quantities to be produced. The feeding of silage liquor to cows directly is currently receiving attention as a potential disposal option (Beck 1989a). Current practice is to dilute and spread thinly to land.

Pesticides

The traditional practice of disposal of waste sheepdip to soakaway is now discouraged (except on low permeability soils well away from aquifers). Farmers are now advised by the NRA to spread thinly to land following dilution of waste residues with slurry. This supposedly renders organophosphorus insecticides harmless in a matter of weeks through a combination of photochemical and bacteriological degradation, coupled with immobilisation. Care should nevertheless be taken to prevent rapid run-off to water sources; spreading on heavily drained land, compacted or frozen soils, or on steeply sloping land should be avoided.

Advice on pesticide disposal in the MAFF code of practice (MAFF 1990) essentially parallels that given by the NRA and emphasises the need to gain approval from the NRA for disposal to land or soakaway.

3.5.4 Catchment management

The effectiveness of riparian environments in the control of non-point source pollution through the amelioration of run-off quality has been investigated using a retention-time model (Phillips 1989). Specific combinations of soil, topography and vegetation characteristics were compared in their ability to filter nitrates from agricultural run-off. Although all typical riparian forests protected water quality, a wide variation was found in their effectiveness and a need for flexibility in determining buffer zone width was emphasised.

Howells (1971) has described pollution from run-off from agricultural and urban areas with respect to the control options available. These have included the use of appropriate conservation practices (for example, a well-vegetated strip alongside all streams would reduce phosphorus, pesticide, and suspended solid loadings to streams), treatment of run-off, and control of land use to make it compatible with water-use classification schemes.

Riparian buffer zones have been widely reported to produce reductions in pollutant load (suspended solids, nitrogen, phosphorus, BOD) from land run-off. Smith (1989) reports suspended solids, particulate phosphorus and nitrogen, dissolved phosphorus and nitrate concentrations in run-off to be >87%, >80%, >85%, >55% and >67% respectively lower from retired riparian pasture than from grazed riparian pasture. In the US and elsewhere, grant-aided programmes have been implemented for the widespread development of riparian buffer zones (see Section 3.7). In the U.K., no such programmes exist, except for the Set-Aside scheme which is EC-funded and only relates to land under intensive arable production; even with this scheme, the setting aside of riparian land is only offered as a suggestion rather than actively encouraged by extra funds or provisos in grant award (there is now a new Water Fringe option within the Set-Aside scheme). Research into the effectiveness of buffer zones at retaining nitrogen is currently being part-funded by WRc at Oxford Polytechnic [this has now been completed].

Hillman and Whyman (1984) used data collected on the high-flow response of a small river in southern England, chemograph responses and rainfall run-off relationships to establish a predictive catchment water quality model. Such a model can be reasonably accurate and it can be augmented to include a number of water quality parameters with only marginal increases in the requirement for data collection. More complex models using data on land management, farming practices, topography, geology and pedology are considered possible and some of these factors are included in the evaluation of run-off response. Agricultural management plans can be incorporated within a comprehensive catchment water quality management plan. Such plans, generated by a conflict of resource development and resource conservation, have been applied to effect an optimal solution to environmental, domestic, industrial, commercial and agricultural needs through the use of pollution studies, supported by computerised data storage and retrieval systems and ecological models (e.g. Lee and Adan 1983).

Braden *et al* (1989) have attempted to quantify the important link between farm management practices and implications for aquatic communities in receiving waters. They have produced a theoretical optimisation model which incorporates farm management practices, pollutant transport to watercourses and likely effects on fish habitat suitability (using the US Fish and Wildlife Service Habitat Suitability Index approach). The model deals with the effect of sediment and pesticide inputs, but the principle could be applied to organic pollution from livestock. The output from the model is a trade-off solution between crop yield and habitat suitability, and a management plan can be constructed from the optimal solution for each land unit.

A comprehensive management plan was developed for improving the water quality of the Siuntionjoki watercourse, Finland (Tamminen *et al* 1989). Inventories of pollution loads and wildlife species inhabiting the river and of the natural vegetation were compiled, and the effects of competing interests and activities on water quality and pollution abatement were assessed. The investigation involved consultation and collaboration with all interested parties, local authorities and nature conservationists, and the development of an agreed procedure for the evaluation of future developments from the standpoint of water quality protection was devised.

Osteen *et al* (1981) advocate the merits of water-based land management (WBLM) (i.e. relating the choice of land management practices to water use and water quality goals) as an important method of controlling agricultural non-point source pollution. However, its implementation is problematical because of the greater expense that would be incurred, and administrative and political complications.

In developing a method of monitoring pollution from animal wastes, the natural fluorescence of solid samples from cattle and piggery wastes, and of water samples from streams receiving livestock-contaminated run-off, was investigated (Lakshman 1975). The fluorescence of pollutants was stable and persisted with no significant decay. Samples from areas uncontaminated by animal wastes showed very weak or no fluorescence. The total carbon, total organic carbon, and total inorganic carbon contents of the polluted samples correlated with the intensity of fluorescence, confirming that the technique is useful for evaluating water quality and detecting pollutants on land. This fluorescence is strong enough for the technique to have potential for aerial remote sensing.

SENSMOD (Jolankai 1986), a system of sub-models, has been developed to follow the fate of pollutants of non-point source origin from small farming units (or individual point sources) through the entire system of a catchment until a downstream point of interest is reached.

The impact of agricultural management practices on both groundwater and surface water quality has been addressed Shoemaker *et al* (1990), who stresses the need for improved mathematical models for a realistic appraisal.

3.5.5 Farm waste management planning

Despite the necessity to control the adverse impacts of farm wastes, there is also a requirement to maintain agricultural efficiency. There is little doubt that modern agricultural practices could actually increase crop yield to the detriment of water quality (McWilliams 1984). Research is necessary to establish the most appropriate and effective methods of agricultural practice, and appropriate legislation and incentives are needed to encourage their adoption. Much research has therefore been concentrated on the development of agriculturally efficient and environmentally safer practices, but little of this research has investigated the adverse effects of these wastes upon the water quality of receiving streams (Ayers and Westcot 1985).

Svoboda (1989) has described a computer-based farm waste management system. This method requires a knowledge of the magnitude of waste production and the capacity for storage and/or treatment. Through a series of question/answer and calculation routines, results are presented in the form of tabulated information and graphs for a number of design and strategic options available.

Cole *et al* (1988) have compared the relative risk to surface waters of different pollution sources, using an inventory of pollution threats identified within a particular catchment from regional and national data, and a knowledge of pollutant travel and dispersal in watercourses. There is a need to extend this approach to produce a nationally accepted procedure for assessing, as objectively as possible, the risk of farm management activities. This is discussed further in Section 4.3.2.

Ultimately, there is no likelihood of significant reductions in farm pollution in the U.K. until an integrated approach to farm waste management is adopted, which involves co-operation and liaison between the farmer, MAFF and the NRA. The development of farm waste management plans for individual farms, including waste handling procedures, waste storage capability, waste spreading schedules (detailing where and when, weather permitting, waste is to be spread), and maintenance schedules for waste management equipment and facilities, is central to such an integrated approach.

A large disincentive to the development of management plans is the charging policy of MAFF's advisory branch, ADAS. Advice and consultation must be offered free to the farmer, as it is in other countries (Section 3.7), for such an approach to be effective. Ultimately, farm waste management plans need to be given legislative force, which would effectively extend the spirit of the new regulations on farm waste storage to all areas of farm waste management.

3.6 Legislation and incentive schemes

3.6.1 Legislation

Section 111 of the 1989 Water Act provided for the establishment of Water Protection Zones (WPZs) where specified activities may be restricted; these powers have not yet been used. Nitrate Sensitive Areas (NSAs) are special cases of such protection zones, and 10 pilot NSAs have been set up within which agricultural practices (including the timing and rate of livestock waste application to land) are restricted. However, NSAs differ from normal WPZs in that restrictions are purely voluntary, with financial incentives for compliance. Further designation of NSAs is planned, but this type of protection zone only deals with areas with vulnerable potable supplies. Ecological protection has not been a consideration, and the NCC have recently urged WPZ designation to protect ecologically sensitive catchments, such as lake and mire SSSIs (NCC 1991).

As already mentioned, the Government has recently released new regulations on the storage of slurry, silage and fuel oil (DoE 1991). The NRA are empowered with the enforcement of these regulations, and can serve notice for improvements on farmers with installations that do not comply with the specifications laid down. These regulations will be a useful tool in pollution prevention since they allow prosecution on the basis of risk (as defined by the regulations), but their usefulness has been restricted by the leniency of some of the required specifications. Moreover, farmers have the right of appeal against improvement notices, which is likely to introduce a considerable delay into the implementation of remedial measures. These regulations are discussed further in Section 3.8.

In 1986, 128 prosecutions arising from farm incidents were recorded in England and Wales, with an average fine of £276. In 1987 the Lord Chancellor encouraged the imposition of higher fines, suggesting a starting point of £2000 (the maximum fine) which would be reduced according to mitigating factors. However, in 1987, 225 prosecutions, with average fines of £205 were recorded. Subsequent increases were noted in 1988 but the NRA was not satisfied with the low penalties enforced. In 1989, 163 prosecutions were recorded and the average fine rose to £500. However, there were marked regional variations, with average fines ranging from £105 in NRA Yorkshire Region to £904 in NRA Severn Trent Region.

Under the Environmental Protection Act (1990), the maximum penalty in the magistrates court for pollution offences has been increased from £2000 to £20 000, but it remains to be seen whether this change will be used effectively.

3.6.2 Incentive schemes

The **Farm and Conservation Grant Scheme (FCGS)**, which succeeded the Agricultural Improvement Scheme (AIS) in 1989, pays for 50% of the cost of pollution prevention measures approved by the NRA. However, there are several deficiencies in the current format of grant allocation, including the non-eligibility of clean water separation systems (which are crucial to pollution prevention). The AIS offered 60% of costs in less favoured areas and 30% in favoured areas, in contrast to the current allocation of 50% in all areas. Less favoured areas are generally more at risk from farm pollution, with land typically having heavy soils, high rainfall and high landslopes; grant aid has effectively been reduced by 10% in these areas since 1989. Less favoured areas also usually produce oligotrophic waters, particularly sensitive to pollution from farm wastes (NCC 1991).

There is therefore justification in having a graded system of grant allocation, with higher incentives paid in high risk areas. The payment of 50% of costs has been criticised generally by the NCC as being too low to provide a significant incentive to farmers, given the current economic climate; they suggest that 75% would be more appropriate. Such a change could be assimilated within the existing grant aid budget if allocation were graded.

Other relevant incentive schemes are more catchment-based than the FCGS (which focuses on waste collection and storage), comprising the designation of **Nitrate Sensitive Areas** (NSAs) (MAFF 1990) and **Environmentally Sensitive Areas** (ESAs) (MAFF 1989). Both of these schemes aim to extensify agricultural practices within designated 'sensitive' zones. The former is government-funded and is primarily designed to reduce nitrate levels in drinking water supplies. The latter is EC-funded and is primarily concerned with landscape and conservation. Both schemes may stipulate restrictions on livestock numbers depending upon the nature of the designated zone, farmers are not obliged to enter either scheme. So far, 10 pilot NSAs and 10 ESAs have been designated [this has now been increased]. The **Set-Aside** scheme, another EC-funded venture, only applies to intensive arable farming.

3.7 International approaches to farm pollution control

3.7.1 Introduction

In order to make comparisons between farm pollution control activities in the U.K. and abroad, a review of international approaches was conducted. This was largely undertaken through telephone discussions and correspondence with relevant governmental bodies in Europe and elsewhere, which yielded most of the literature cited.

3.7.2 European combined initiatives

Control of pollution from livestock farming in Europe as a whole mainly stems from the need to protect soil quality and prevent nutrient contamination of surface and groundwaters. At present, control measures are left to individual member states; however, there is a proposed EC Directive on the control of nitrate from diffuse sources (CEC 1989) [this has now been approved]. This directive would require Member States to designate vulnerable zones, and within these to apply certain catchment management restrictions. Regarding livestock, there would be requirements to:

- o restrict the number of livestock according to the land available for manure/slurry spreading (see Table 3.5);

- o introduce rules for manure spreading and for the storage of manure/slurry during prohibited periods.

The proposed directive has the objectives of avoiding: nitrate levels in freshwaters which would conflict with legitimate water uses; and eutrophication of surface, estuarial, coastal and marine waters. The system of Nitrate Sensitive Areas in the U.K. has similar aims to the proposed directive, i.e. the extensification of agriculture within designated zones, but the U.K. system involves voluntary grant aid rather than statutory compliance. The ultimate effects on U.K. agriculture of this directive could be widespread and obligatory [the U.K. is required to issue draft regulations by 1995].

Table 3.5 Maximum livestock numbers per hectare of land available for slurry spreading, as proposed by EC Directive COM(88)708

Livestock type	Maximum no of livestock ha^{-1}
Dairy cows	2
Young stock or beef cattle	4
Fattening pigs	16
Sows with piglets	5
Turkeys, ducks	100
Laying hens	133
Young hens, 0-16 weeks	285

NB Numbers are not cumulative

In addition to this, the Ministerial Declaration at the 1987 London Conference on the North Sea agreed to take effective national steps in order to reduce nutrient inputs into areas of high pollution risk in the North Sea (Paris Commission 1989). The aim of the declaration is to reduce inputs by c 50% between 1985 and 1995. The Paris Commission felt that it was not possible to realise this aim without very stringent measures in the agricultural sector.

3.7.3 The Netherlands

Legislation

a. Waste production and land application

Control of pollution from livestock in Holland is mainly provided by the Soil Protection Act (1987) and the Act on Manure and Fertilisers (1986), which control slurry/manure production and disposal. The objective of the Dutch strategy is to reduce animal waste production so that the amount applied to the soil does not exceed the nutrient requirements of the crop. This should ensure that there is no nutrient/mineral accumulation in the soils and little potential for contamination of ground - and surface waters.

Every livestock farmer has a non-negotiable waste production quota, set in terms of phosphate load since nutrient enrichment of their waters is the primary concern. If part of a production quota is transferred from one farmer to another, that part of the quota is reduced by 30%. The quota indirectly sets an upper limit to the number of livestock that a farmer can keep on his land. Quotas for individual farms will be reduced in phases up to the year 2000. This is intended to force farmers to adopt practices that produce manure with a low phosphorus content, rather than to force reductions in livestock numbers. However, a limit on livestock density (which is likely to be as little as 3 dairy cows ha^{-1} or its equivalent) is to be set for land-dependent livestock farming (i.e. non-intensive) in order to strengthen the dependency on the land for grazing cattle (Dutch Ministry of Agriculture (DMA) 1990).

In addition, maximum application rates for land-spreading are specified, which become progressively more restrictive until the year 2000 (see Table 3.6). At present, the application rates are much higher than the calculated requirements of the soil (van Boheemian 1987) and the phased increase in spreading restrictions is to allow a transition period towards environmentally sound application rates. The values in Table 3.6 can be roughly converted to slurry volume using a phosphate content of 2 kg m^{-3} for undiluted cattle slurry, and 4 kg m^{-3} for undiluted pig slurry (both assuming 10% dry matter) (ADAS 1985). In addition to these restrictions, application of dirty water is restricted to 50 m^3 ha^{-1} yr^{-1} on grassland and 25 m^3 ha^{-1} yr^{-1} on arable land (van Boheemian 1987). In 1991/92, areas of conservation interest sensitive to eutrophication will be designated, and allowable phosphate loadings will be much lower.

Land application is prohibited at different times of the year according to soil type, as indicated in Table 3.7.

Table 3.6 - Maximum application rates of slurry/manure to land in the Netherlands (Dutch Commodity Board for Feeding Stuffs (DCBFS) 1989)

PERIOD	GRASSLAND	SILAGE MAIZE FIELDS	ARABLE LAND
up to 1:1:91	250	350	125
1:1:91 to 1:1:95	200	250	125
from 1:1:95	175	175	125
by 2000	final standard	final standard	final standard

Values are in kg P205 ha^{-1} yr^{-1}.

Table 3.7 Prohibitions on land application of slurry/manure in the Netherlands (Basset *et al* 1990).

SOIL TYPE/LAND USE	PROHIBITION PERIOD
Grassland	1 October to 30 November (and 1 January to 15 February on snow-covered ground)
Sandy soil	
a) silage maize/arable	Harvest (c 1 July) to 31 October
b) silage maize/arable followed by after-crop (i.e. winter crop)	1 October to 31 October
Clay soil	Under discussion

These restrictions will become more severe in phases similar to those governing maximum application rates, and by 1995 the spreading ban will run from September to February (Dutch Ministry of Agriculture 1990). At present, slurry spread to arable land must be ploughed in within 48 hours; however, from 1991 it must be ploughed in directly after application, and from 1995 all slurry spread to grassland must be injected. These latter stipulations are largely designed to reduce atmospheric emissions of ammonia (it is intended to reduce emissions by 70% compared to 1980 levels) and odour, although they are also likely to be effective in reducing the risk of direct run-off; it should be noted, however, that the risk of direct run-off is lower anyway due to the low-lying nature of the land in the Netherlands.

New legislation is being developed which will set a maximum allowable nitrogen concentration in the top metre of soil, which will further restrict land application (of both organic and inorganic fertilisers) (personal communication Dutch Ministry of Transport and Public Works). This is an approach used in some areas of Germany, and will be introduced in the Netherlands in phases starting in the early 1990s.

The above controls give rise to large manure surpluses in central and south-eastern areas of the country (van Boheemian 1987). In 1988, total manure/slurry production was estimated at 96 million tonnes, with the theoretical surplus totalling 14 million tonnes. (Dutch Ministry of the Environment 1990.) In 1985, approximately 55% of the theoretical surplus was transported to other farms for land-spreading (Voorburg 1988). It is expected that total production will be 82 m tonnes in 1991, with a surplus of 15 m tonnes, and 81 m tonnes in 1995, with a surplus of 16 m tonnes.

Under the legislation, livestock farmers with a waste production of greater than 125 kg P_2O_5 ha^{-1} are required to keep records of slurry/manure produced and where it is to be disposed, thus indicating whether a surplus exists and how large it is. Farms producing more waste than this are prohibited from expanding; this restriction will most affect those types of farming not tied to the land (mainly pig and poultry farming). Farmers pay a 'surplus levy' of up to 0.5 guilders kg^{-1} of phosphate over the reference production of 125 kg ha^{-1}; this tax is reduced if a bilateral waste disposal contract is signed with an arable farmer, if the waste is transported off the farm by government-approved means, or if a low-phosphate feeding system is used. The income from this tax is used to develop manure surplus disposal strategies in the form of the Manure Action Programme (MAP, which is additionally funded directly by the government).

Many bilateral agreements have now been made between farmers. There is even a private company that organises the long distance transportation and disposal to arable land of chicken manure. This is economically viable because of the highly concentrated nature of chicken manure, but long distance transfers are not really feasible for cattle or pig slurry in their original consistency.

The effectiveness of Phase 1 restrictions is currently being evaluated, and Phase 2 restrictions will be amended accordingly.

b) Slurry storage

From 1990 there will be measures requiring minimum slurry storage capacities under the Soil Protection Act. This is in addition to other legislation stipulating design criteria, ammonia emissions, and a minimum slurry storage capacity of 6 months (rising to 9 months in certain areas depending upon local circumstance). As part of the National Environmental Policy Plan, slurry/manure stores built after 1987 will have to be covered by 1991 in order to reduce ammonia emissions; this will also have the effect of reducing dilution by rainfall.

Monitoring and enforcement

The monitoring of compliance with waste storage requirements is undertaken by local authorities. Enforcement of the relevant legislation within the Soil Protection Act is the joint responsibility of the Ministry of Agriculture and the Ministry of the Environment. Two main methods of enforcement are adopted:

- Book-keeping - the records of surpluses kept by farmers are checked against receipts given by arable farmers or processing plants upon delivery of manure. This ensures that all surplus manure is disposed of outside of the farm. A 'minerals accounting system' was introduced in 1990, which looks at the net flux of minerals (N, P, K) at the farm level with the aim of controlling losses. The possibility of using this system in a regulatory way, which would link up with manure/slurry accounting, is currently being investigated (DMA 1990).

- Farm visits - AID, the relevant monitoring authority within the Ministry of Agriculture, undertakes site visits to ensure compliance. This includes audits of the number of livestock kept on the farm; if the number is too high for the waste production quota set for the farm (estimated by using average outputs per unit of livestock) the farmer will be instructed to reduce his herd size. AID do not assess pollution risk on farms, but merely enforce the appropriate legislation.

In addition to the above enforcement methods, the civil police enforce the prohibition periods on the application of manure to land.

Water pollution control authorities comprise 40 - 50 regional water authorities, responsible for small rivers and tributaries, and a National Water Authority responsible for major rivers such as the Rhine. They are not involved in the enforcement of any of the legislation mentioned above and only become involved with farms once a water quality problem has been identified. However, the Water Pollution Control Act (1970) makes provision for the setting up of buffer zones, for which the water authorities would be responsible.

Waste control

The Manure Action Programme is directing effort into three main areas: the improvement of feed quality; the improvement of manure/slurry quality and facilitating acceptable manure storage and disposal (for which the Foundation National Manure Bank is responsible); and centralised manure treatment and processing.

Regarding feed quality, research is focusing on improvements in: digestibility; the elimination of anti-nutritional factors (anfs) which block protein digestion; reduction in phosphorus content (especially through the addition of phytases which improve phosphorus digestibility); and reductions in heavy metal content (DCBFS 1990). Nutrient balances have shown that inputs of nitrogen, phosphorus and potassium to Dutch dairy farms greatly outweigh outputs, such that there is a net accumulation in the soil which can leach into ground- and surface waters. There is concern, however, that reducing waste phosphorus outputs will allow compliance with current regulations without necessarily reducing nitrogen (or indeed whole slurry) outputs.

Regarding manure storage, subsidy schemes worth 50 m guilders have been implemented for the enhancement of storage capacity. Such subsidies will help to alleviate the problem of storage for increasingly long periods over the autumn/winter period. The development of a manure application (to land) plan for each farm will be encouraged by the DMA.

The transportation of surplus slurry/manure to areas with a deficit is handled by the Manure Boards, which received a legal status in 1987 (van Boheemian 1987). The boards are obliged to remove surpluses from farms upon payment of a standard tariff. They subsidise the transport costs if the waste is taken from an area with a surplus to one with a deficit, and if it involves a distance of over 75 km for poultry waste and 50 km for other wastes.

A research programme was initiated in 1985 aimed at the development of large scale animal waste processing techniques. Field testing of these techniques commenced in 1988, aided by a subsidy scheme worth 51.5M guilders from the MAP, and the Dutch government has since been working to relieve the bottlenecks impeding large scale processing. It is intended that by 1994 there will be an industrial manure processing capacity of 6 million tonnes, with the responsibility for this resting with the Dutch agricultural sector. If this target is not met further legislative restrictions will be imposed.

At present, it seems unlikely that the manure processing industry will be able to handle even 1 million tonnes per annum by 1994. Voorburg (1988) concludes that 'slurry purifying', in which a final effluent is produced for discharge into receiving waters, is not practical since the effluent is too nutrient-rich. The Netherlands government now prefers evaporation of the liquid fraction to form a dry pelletized fertiliser, which can then be easily distributed over long distances to areas in which it can be safely utilised.

3.7.4 Denmark

Legislation

Statutory regulations concerning the control of pollution from livestock farming were introduced in Denmark in 1988 (Danish Ministry of the Environment 1989). These impose very tight and detailed restrictions on waste management and are enforced by the 275 local councils throughout the country.

a) Slurry and manure storage

The regulations include construction requirements concerning impermeability, strength and general design of storage facilities such as drainage to reception tanks. For farms holding more than 31 livestock units (LU - 1 LU is equivalent to one large dairy cow) a storage capacity of at least nine months of slurry production is required, although there may be smaller stipulations for farms with between 31 and 120 LU or in special cases on larger farms. Similarly, larger storage requirements may be stipulated. Surface water from adjacent areas and roofs should not run into solid manure yards. Field stacks of cattle and pig manure are forbidden.

There is a requirement for regular emptying of storage facilities (including silage and waste water tanks) to prevent overflow, and regular maintenance against wear and corrosion. Moreover, containers should be tested for effectiveness at intervals of at most 10 years.

b) Stock restrictions and waste application to land

Stocking density in Denmark is limited to 2 LU per hectare (Anderson *et al* 1990). In addition, the following maximum waste application rates must be adhered to:

Cattle slurry	2.3 LU ha^{-1} yr^{-1}
Pig slurry	1.7 LU ha^{-1} yr^{-1}
Other animals	2.0 LU ha^{-1} yr^{-1}

Slurry spread to tilled soil must be ploughed in within 12 hours of application. Also, slurry, silage liquor and waste water should not be applied where there is a risk of surface water contamination, although the regulations do not specify what type of land area constitutes a hazard. Application of slurry or silage effluent is not allowed between harvest and 1 November unless the area supports an established crop or is to carry a winter-sown crop. No application to bare frozen soil is allowed unless it can be ploughed in within 12 hours.

If more slurry is produced by a farm than can be disposed of on the available land, written agreements are required for the safe disposal of the surplus, to be approved by the local council. Disposal may be to other farms or to biogas plants, of which there are approximately 10 in Denmark. Biogas plants are considered to be a success in Denmark, taking not only animal waste but other waste materials including sewage sludge (after disinfection) (personal communication Danish Ministry of the Environment).

Monitoring and enforcement

There are c 80 000 farms in Denmark, 50 000 of which hold livestock. All of these farms were visited in 1987/88 by the local councils to assess the state of waste management practices in relation to the new regulations. Farmers were informed of remedial measures required and the deadlines for compliance. The deadline for compliance with storage capacity stipulations is the end of 1992, and 70% of farms are already complying. The penalties for non-compliance range from fines to imprisonment for up to one year.

It should be noted that only a small proportion of farms in Denmark have insufficient land to dispose of their own waste. Generally, animal waste does not need to be transported more than 3-4 km before it can be applied to the soil. The restrictions imposed, therefore, do not affect the farming community as much as they would in more intensively farmed countries, such as Holland.

Compliance with application rate restrictions is assessed by estimating total waste output (based on livestock holding and estimates of output per animal) and the extent of available land for disposal. If waste output is greater than the local disposal capacity of the holding, a known surplus will be produced. The farmer has to provide evidence that this surplus was produced and disposed of out with his holding via council-approved routes.

Twelve regional councils control the local councils and are responsible for the quality of surface and groundwaters. Regional councils are not directly involved with the monitoring of farm waste management practices, and only become involved when an environmental impact is observed. Compliance with the regulations should ensure that pollution risk is low, but where an immediate risk of leakage or overflow from a waste container is identified by a local council immediate remedial action or a cessation of use can be ordered. A farmer can be convicted for causing a **risk** of environmental damage, which gives the regulatory authorities much more power to order remedial measures than is the case in the U.K.

3.7.5 Switzerland

Introduction

In 1988, Switzerland produced 35 million m^3 of animal slurry, to be disposed of on 1 million ha of agricultural land along with an annual production of 200 000 tonnes of sewage sludge.

Legislation

Legislation on pollution control from livestock farming is contained within a draft chapter on agriculture (Swiss Government 1990) to be included in the Water Protection Act. In addition, the Law of Agriculture restricts livestock numbers, with the aim of maintaining farms as, or restoring them to, small traditional enterprises with sufficient land for their own waste disposal. With this objective in mind, it is the aim of the government to phase out intensive piggeries, which have no land for waste disposal. New constructions of intensive piggeries are banned.

a) Livestock restrictions

Table 3.8 gives the maximum numbers of livestock allowable on a single farm. Each figure shown is the maximum allowable number of that specific livestock type assuming that no other livestock types are present. Table 3.8 also gives the minimum amount of land legally required to dispose of the waste produced. Farms that do not comply with these requirements by the end of 1991 will, from the beginning of 1992, pay the taxes (per head of excess livestock) shown in the same table. There are lower exemption limits where all types of livestock can be kept on one farm without incurring extra taxation, thereby encouraging traditionally diverse farming practices.

b) Storage

A minimum storage capacity of 3 months production will be required, which the local authorities (cantons) can increase in mountainous areas or where climatic conditions are unfavourable. The requirement can also be lessened where livestock housing is only occupied seasonally. The local authorities are to fix dates for compliance with these requirements, but all farms should comply within 15 years.

Table 3.8 Maximum numbers of livestock allowed on a single farm in Switzerland (Swiss Government 1987, Strauch 1989)

Livestock type	Maximum number of head	Taxation on surplus (Swiss Fr per head)	Exemption limits (No of head)
Cattle	250	500	10
Fattening calves	200	200	10
Sows	150	500	10
Gilts	150	500	5
Weaner pigs for breeding	1000	100	10
Weaner pigs for fattening	1000	20	30
Fattening pigs	1000	100	60
Laying hens	12000	20	500

NB Numbers are not cumulative

c) Land spreading

Under the new draft regulations, each farmer will require a contract with the local authorities to apply his waste to land. Such contracts will define an 'area of exploitation' within which a farmer should be able to dispose of all of his slurry according to good agricultural practice. The farmer has to prove that the land surface within this area is suitable for the maximum allowable slurry/manure application rate of 3 cattle units ha^{-1} (1 unit = the annual production from one 600 kg cow). This maximum rate equates to c 54 m^3 ha^{-1} yr^{-1} of cattle slurry, using estimates of waste output in ADAS Booklet 2081 (MAFF 1986b). When this limit is reached, no more slurry or inorganic fertiliser may be applied. The maximum application rate can be reduced by the local authorities according to altitude, topography and soil type. Table 3.9 shows the guidelines used by the Cantons when assessing land suitability for slurry application, but these will have to be reviewed in the light of the new restrictions.

The land within the area of exploitation may be owned by or leased to the farmer, or confirmed as land where he has a contract to simply dispose of his slurry. If a farmer transports slurry out of this area he must have proof that he either owns or has the lease to at least 50% of the land used for slurry application, i.e. only up to 50% of the land used should be available through simple 'slurry application contracts'. This regulation is designed to prevent intensive piggery farmers from continuing their current practices without either leasing or buying a large proportion of the land required for disposal of their slurry.

The local authorities are to fix dates for compliance with these requirements, but all farms should comply within 5 years. It is envisaged that contracts to ensure sufficient land for slurry application will gradually disappear as intensive practices are phased out.

The draft legislation states that land spreading must be undertaken according to the most recent techniques and in a way which does not harm soil or water. It also allows for, in the interests of water protection, a complete cessation of land application where necessary, to be implemented over a transitional period of 5 years.

d) Waste treatment

The farmer will have a legal requirement to maintain and repair water treatment plants, which the local authorities will periodically inspect. Generally, the types of waste treatment being developed elsewhere in Europe are felt to be too sophisticated, costly and energy-consuming to be run on the small farms typical of Switzerland.

Table 3.9 Guidelines on land characteristics for the disposal of slurry in Switzerland (Swiss Office Federal de L'Agriculture 1979)

Physical factor concerned		Implications for slurry application
1. Intensity of rainfall and slurry spreading	a) High loading of very dilute slurry/dirty water	Spray evenly
	b) Heavy rainfall	Do not spray slurry
2. Infiltration characteristics	a) Non-cultivated soil	
	- uncovered, muddy or frozen surfaces	Do not spray slurry
	- loose soil	Spraying possible
	b) Cultivated soil	
	- frozen or snow-covered	Do not spray slurry
1. Intensity of rainfall and slurry spreading	a) High loading of very	
	- Limited infiltration	Spray evenly, reduced loads only
	- High infiltration	Standard load only
3. Topographic characteristics	a) Slope 10-25%	
	- thick vegetation	Standards loads
	- little or no vegetation	Reduced load only
	b) Slope 26-45%	
	- thick vegetation	Reduced load only
	- uncultivated	V reduced or no load
	c) Slope >45%	V reduced or no load

Standard load: isolated doses up to 60 m^3 ha^{-1}
Reduced load: isolated doses up to 40 m^3 ha^{-1}
V reduced load: isolated doses up to 25 m^3 ha^{-1}
The total annual quantity should be adminstered over 2 or 3 applications

3.7.6 Western Germany

Introduction

It is estimated that 211 million tonnes of animal waste is produced annually in West Germany, 80% of which comes from cattle (Van der Weghe 1990). Slurry accounts for *ca* 40% of this total, but regionally it can account for up to 70% (in the northern states).

Legislation

Each state develops a proportion of its legislation internally, and for this reason those states with the greatest farm pollution problems have produced specific ordinances controlling livestock waste disposal. The greatest concerns are nitrate levels in drinking water and eutrophication, rather than run-off of whole slurry.

a) Land application

Table 3.10 summarises the restrictions on slurry application in 4 German states. The 'Dung Unit' is widely used in West Germany as a means of expressing nutrient load. The exact definition of the dung unit varies, and Table 3.10 shows the variation between the 4 states considered. Nieder-Sachsen imposes a maximum slurry application rate of 2.5 Dung Units ha^{-1} yr^{-1} (3 DU until 31/12/92), and this rate can be reduced by the authorities if there is a pollution risk to surface or groundwaters. In Schleswig-Holstein, the maximum application rate is 2 DU ha^{-1} yr^{-1}, and the farmer has to prove to the authorities that he is not exceeding this rate. Application in this state is prohibited on certain types of ground, such as fallow land, frozen soil and protected biotopes. In Bremen, the maximum is again 2 DU ha^{-1} yr^{-1}, and this has to be split into several applications through the year with no one application exceeding 1 DU ha^{-1}.

Slurry application during the winter months is prohibited in all 4 states, the only exceptions arising (in 2 of the states) when winter crops are grown. In such cases very reduced applications are allowable. In Bremen, the regulations

Table 3.10 Restrictions on the land application of slurry in 4 German states (pers comm D Strauch, Universitat Hohenheim)

State	Neider-Sachsen	Schleswig-Holstein	Nordrhein-Westfalen	Bremen
Date of ordinance	1:2:90	1:8:89	1:6:84	1:5:89
Definition of Dung Unit (DU)	80 kg total N yr^{-1}	80 kg total N yr^{-1} or 60 kg total P205	80 kg total N yr^{-1} or 70 kg total P205	80 kg total N yr^{-1}
Maximum allowable application	2.5 DU (3.0 DU until 31:12:92)	2.0 DU	3.0 DU	2.0 DU
Periods of prohibition	16 Oct - 31 Jan for pasture harvest - 31 Jan for arable land	16 Oct - 31 Jan for pasture and covered soil 1 Oct - 28 Feb for arable land	1 Nov - 31 Jan for pasture and arable land with soil-covering winter crops 16 Oct - 14 Feb on uncult. arable land	16 Oct - 31 Jan 16 Oct - 14 Feb on marshy land 16 Sept - 26 Feb on uncovered soils*

* Except when winter crops are cultivated

state that even during the months when slurry application is allowed, there should be no application if the weather creates a risk of run-off to surface waters.

A number of states suffer from increasing levels of nitrate in drinking water, and water protection zones have been set up in states such as Baden-Wurttemberg to control nutrient applications (similar systems are in operation in other European countries such as the Netherlands). Three zones are identified, and slurry application is not permitted at any time in the inner and middle zones, whilst spreading is allowed between 1 October and 15 February in the outer zone (Strauch 1989). Compensation for loss of yield is paid to the farmer, and soils analyses are conducted after harvest to ensure that the recommended fertiliser applications have been followed. Control sites have been set up on different soil types to ensure a fair comparison with which to test compliance.

b) Storage capacity

The required minimum waste storage capacity across the country is 6 months of waste production, but farmers are now being requested to have sufficient capacity for 9 months production (van den Weghe 1990). Many farmers do not at present comply with the minimum required capacity.

Waste control

As in the Netherlands, greater nutrient utilisation in livestock feed is seen as an important step in the control of nutrient output. It is estimated that optimisation of feed composition and nutrition will allow a reduction in feed nutrient content of 25-30% (van den Weghe 1990). Research effort will also be invested in methods of drawing up nitrogen balance sheets in order to ensure that slurry application rates are compatible with soil requirements.

It is envisaged that stationary or mobile treatment facilities will be developed in areas of large slurry surpluses that can dry slurry into a reusable pellet form. 'Manure banks', and contracts for slurry disposal between farmers, already exist and the management structure for such activities is being developed further.

Anaerobic digestion and subsequent biogas production is a well-tested form of treatment for animal wastes in West Germany, but is only operated at a farm level. In 1986, 130 biogas systems were in operation, the majority of which handle under 150 dung units per year (equivalent to the production from 150 large cows), with 6 handling between 150 and 300 DU yr^{-1}, 4 handling 300-600 DU yr^{-1} and 2 handling over 600 DU yr^{-1} (Loll 1987). The plants can process slurry in its normal consistency. Biogas production in Germany is not profitable at present if viewed purely in economic terms (i.e. setting the revenue from energy production directly against capital and running costs), and it would appear that greater governmental subsidies are required for further expansion of the industry. The very important use of anaerobic digestion in odour control (emissions can be reduced by up to 90% - Voermans 1986) may help to win further subsidies for this method of waste control in Germany and other countries.

Proponents of biogas plants in Germany are pushing for communal systems handling far larger quantities of animal waste. This would serve to reduce overheads whilst greatly increasing energy output.

3.7.7 Other countries

Belgium

The Manure Decree (1985) prohibits slurry storage and spreading on land within protection zones and water supply zones. In other land areas, the maximum allowable land application rate is 4 animal units ha^{-1}, with a prohibition on land-spreading from September to the end of January. In the future, spreading will be prohibited on non-agricultural soils. A taxing scheme for manure surpluses similar to that operating in the Netherland is envisaged.

New constructions are required to have the capacity for 6 months waste production. Centralised manure banks will be available in the future.

Centralised manure banks will be a major route for the treatment and processing of manure surpluses. Advice is given to farmers on an individual basis on how to use animal waste efficiently and how to farm in non-polluting ways.

Sweden

Land-spreading is banned between 1 December and 28 February as from 1989 (Paris Commission 1989), and from 1995 will be further prohibited from 1 August to 30 November on land which does not hold a growing crop. Slurry/manure must be incorporated within the same day. Maximum stocking densities will be implemented shortly in order to restrict the amount of waste that requires disposal to land.

For all new installations, a minimum storage capacity of 8-10 months is required in the south of Sweden and coastal zones as far north as Stockholm from 1 January 1991. Existing installations must comply by 1995. In some areas there is a subsidy of 20% for increasing storage capacity.

Free advice is available for developing slurry/manure disposal plans on each farm. There are also plans to operate an approval system for waste spreading equipment.

The use of buffer zones is receiving research attention, and a plan for stream restoration incorporating improvements in agricultural run-off quality has been proposed by Petersen *et al* 1991. This involves the setting aside of 10 metre-wide buffer strips and planting fast-growing woody shrubs (possibly for harvest) to intercept run-off. The discharge outlets of tile drains are dug back away from the river bank to a distance of 8 metres, and the intervening land is made into a wetland area (a so-called 'riparian wetland horseshoe') for the reception of under-drainage run-off.

Petersen gives no experimental data on the removal efficiency of these horseshoe wetlands, although he does quote a number of references indicating efficient removal of nitrogen and phosphorus by wetlands in general. However, the role of wetlands in reducing phosphorus levels in run-off has recently been questioned (Gehrels and Malamoottil 1989). It is clear that the effectiveness of wetlands in improving run-off quality will ultimately depend upon site specific factors such as nutrient loading rate, residence time, vegetation type and management practices, and that generalised statements are therefore unhelpful. Whatever their role in nutrient removal, a wetland with sufficient capacity to avoid overland flow is very likely to reduce BOD loadings (by physical filtration and biodegradation) on receiving waters, and will therefore serve to maintain ambient oxygen levels: for instance, Knight *et al* (1987) report high BOD removal rates across a wide range of loading rates on a natural wetland).

Norway

The maximum land application rate stipulated in Norway is 2.5 animal units ha^{-1} yr^{-1}. Spreading is banned on frozen or snow-covered fields. Slurry/manure must be ploughed in immediately after spreading on tilled soils, except in the growing season. Plans to limit spreading to the growing season in sensitive areas are being considered.

Waste storage capacity must exceed that required for the longest period for which spreading is restricted. Silage effluents must be collected for use as fodder or fertiliser.

The development of individual waste management plans is offered to all farms. Subsidies are available for the establishment of buffer zones along vulnerable watercourses. For pollution prevention developments on the farm, farmers can get 35% of the outlay in grants, 10% in interest-free loans and 30% in low-interest loans.

USA

Farm pollution control programmes have been undertaken in 7 catchments throughout the USA under the Model Implementation Programme (MIP) (Brown *et al* 1989), initiated by the Department of Agriculture and the Environmental Protection Agency. The west branch of the Delaware river (New York State) was one of the catchments chosen, where phosphorus loads (particularly from livestock) into the eutrophic Canonsville Reservoir were causing problems. A survey of the 275 farms in the catchment was undertaken, targeting those farms with livestock buildings within 100 metres of a watercourse. The remedial measures applied involved the reduction of yard run-off volumes, particularly through clean water separation. Although phosphorus loads from yard drainage were reduced considerably, the limited monitoring performed and model simulations suggested that dissolved phosphorus losses from the catchment as a whole were only reduced by less than 5%.

Nutrient modelling showed that run-off from manure-spread fields contributed greatly to phosphorus loadings to the reservoir. Manure spreading schedules were subsequently developed, which stipulated when individual fields should be used according to their physical and soil characteristics, accessibility and the likelihood of rain. These schedules encouraged a more even spreading of waste over the whole farm, rather than the use of a few fields adjacent to the farmyard. Model predictions for one section of the catchment suggested that total phosphorus losses could be reduced by up to 35%. It is not clear how these results relate to the run-off of whole slurry and the likelihood of acute impacts on aquatic life, but the development of such manure spreading schedules is also likely to reduce BOD loads and consequently have an ameliorative effect on this type of impact.

Lake Okeechobee in Florida is another catchment suffering from eutrophication problems largely due to livestock farming. Three years ago the Florida Department of Environmental Regulation imposed certain requirements (Best Management Practices, BMP) upon livestock operations to combat this, with the main stipulation being the collection and spray irrigation of yard washings (Flaig and Ritter 1989). The South Florida Water Management District (SFWMD) is also sponsoring a buy-out programme for those farms that choose not to make the required changes to their operations. Several dairies have chosen to sell their holdings, and several more are giving this consideration.

A separate study was undertaken on a sub-catchment of Lake Okeechobee, the Kissimmee River, to assess the efficiency of wetlands buffering as a BMP for improving run-off quality from livestock areas (Goldstein 1986). Two wetlands were monitored, and it was found that they were both in near steady-state equilibrium with respect to total nitrogen, whilst removal of total phosphorus was better but decreasing as adsorptive capacity was gradually spent. No measurements were made of inputs to and outputs from the wetlands of ammonia and BOD.

In an experimental agricultural non-point source pollution control programme in the U.S.A. (Young and Shortle 1989), the major problem of nutrient run-off in spring, resulting primarily from the winter-spreading of manure, was effectively eliminated in 60 out of 62 participating farms by installing manure storage systems. New legislation (The Rural Clean Water Programme) allowed 75% cost-sharing of eligible costs, and representative farm models indicated significant economic benefits from installing both earthen pit storage systems or more expensive above-ground systems. The ability to store manure led to a change from daily to seasonal manure application times, which reduced annual costs by around 22% for labour and 30% for commercial fertiliser, meaning that earthen-pit systems would probably be profitable without cost-sharing. The benefits of this scheme were noted in the improved quality of receiving waters.

Research is currently being conducted into the use of constructed wetlands for the treatment of agricultural wastes, including slurry (Hammer 1989). Slurry from a 500-pig holding is being fed, after primary settlement, into an experimental wetland system to determine optimum design criteria.

The use of riparian buffer zones (Vegetative Filter Strips or VFSs) is expected to become much more popular in the US following their approval under the Conservation Reserve Programme (CRP) by the Department of Agriculture (Dillaha *et al* 1989). This programme has the objectives of reducing soil erosion, improving water quality and enhancing wildlife habitat. Soil erosion is a major concern in the US and is probably the main motivation for the incorporation of VFSs into the CRP.

Farmers receive between \$75 and \$200 ha^{-1} yr^{-1} over the 10 year contract period, and the VFSs they construct must meet certain requirements (concerning width and planting of vegetation, for example). Funding for VFSs is also available from 'State non-point source pollution control programmes' such as the Chesapeake Bay Programme in Virginia. Farmers receive \$0.33 $metre^{-1}$ of VFS, but must refund this if the strip is not properly maintained.

Construction specifications similar to those laid down by the federal programme must be observed. Both programmes specify that the strip must be designed to filter sheet flow. A survey was undertaken by Dillaha *et al* (1989) to assess the effectiveness of VFSs installed so far in Virginia, and many were found to be breached by concentrated overland flows of run-off which were not filtered at all. These breaches were often associated with large fields where run-off could gather into significant flows with high momentum. Such problems can only be remedied by better design and regular maintenance.

Ireland

As discussed in Section 3.4.3, Lough Sheelin has suffered from nutrient enrichment largely due to intensive pig farming. A manure management plan was instigated in 1975 (Dodd and Champ 1983), the main actions being to:

- increase manure storage capacity to 6 months production;
- spread manure only through the growing season;
- check soil nutrient status regularly;
- base application rates on the nutrient requirements of the soil.

Following continual poor waste management practice, brought about largely by the sheer volume of slurry requiring disposal, a slurry transport scheme was initiated in 1980. Through the winter of 1980/81, 6.9 million gallons of slurry were exported from the catchment area, costing £2.23 (1980 prices) per 100 gallons in subsidy to defray haulage charges. In the first two years of operation, the scheme exported 29 million gallons of slurry at a cost in subsidies of £84 000, plus the labour cost of providing a full-time co-ordinator.

In 1982, 16 pig producers were participating in the scheme, with a combined nominal slurry production of 23 million gallons yr^{-1}; about 15 million gallons of this was exported. Slurry was transported using up to 10 large (25 m^3)

tankers, hauling over distances of up to 40 miles. Some of these tankers were owned by the large pig producers, whilst independent hauliers also became established.

Despite the large reductions in phosphorus level in feeder streams, static subsidy for the transport scheme was curtailed in 1984 (Champ 1990). Since then, efforts have centred on the development of slurry management plans, which are now in place for nearly all piggeries. These plans include a complete ban on spreading over the winter months. Restrictions have been extended to cover cattle farming operations in the catchment. In the longer term, it is hoped that anaerobic digestion will relieve the dependency on land-spreading, and state funds were made available in 1989 to construct a trial digester.

3.8 Regional attitudes to farm pollution control in the U.K.

Below is a summary of regional attitudes to various options for farm pollution control, derived from consultation with regional pollution control staff. Appendix A lists those who participated in the consultation. It should be pointed out that the views expressed by those interviewed do not necessarily reflect regional policy.

3.8.1 Legislation

Legislation governing water pollution control was generally thought by those pollution control staff interviewed to be adequate in England and Wales, but there was a lack of control over non-point source pollution which could not be rectified without more regulation of catchment activities. In contrast, there are numerous examples of restrictions on livestock density and slurry application to land in Europe, as described above. In many European countries slurry application is banned through the winter (see Section 3.2), whereas in the U.K. these are invariably the season of greatest activity, since livestock farmers do not want to compromise growth of grass/silage by application in spring and summer.

The Water Act (1989) has potential for catchment control through the designation of protection zones, but its usefulness will depend upon the strength of subsequent enabling legislation. The **Severn-Trent Region** is a special case in terms of catchment control, since the Severn-Trent Act (1983) allows for binding management agreements to be made between landowners and the NRA. Influence over agricultural planning was seen as fundamental to effective catchment control by the NRA regions, and their current position as statutory consultees did not provide sufficient influence. Agricultural exemptions from planning still gave cause for concern.

The new regulations to control silage, slurry and fuel oil installations were regarded as useful, but are thought by a number of NRA regions to be too lenient in some respects.

Wessex Region feel that although the Water Act (1989) has simplified the prosecution process, their powers to 'remedy or forestall' pollution suffer from a bureaucratic time delay which makes it very difficult to act in time to prevent incidents. **Yorkshire Region** would like to see an extension to the use of prohibition orders, where bans can be imposed on specific activities which threaten specified water uses. At present they can only be applied to land disposal of sewage and trade effluents, not agricultural waste.

Forth RPB felt that the power to serve improvement notices on prosecuted polluters was needed, which would allow subsequent prosecution on the basis of a maintenance of high pollution risk. Denmark has legislation that allows prosecution of any potential polluter on the basis of pollution risk (see Section 3.7.4). Such powers in the U.K. would be likely to have a great impact on farm pollution incident statistics.

3.8.2 Prosecution

It was widely felt that the low level of fines imposed by magistrates for farm pollution offences is a major hindrance to farm pollution control. Some persistent polluters prefer to be fined than invest in preventative measures, implying that at least for some farmers this may be the most cost-effective option. Fines awarded in **Yorkshire, North West, South West** and **Welsh Regions** and throughout **Scotland** and **Northern Ireland** were particularly low, although improvements had been seen in some NRA regions recently (in **Anglian Region,** where NRA staff have been increasing the awareness of magistrates, in **Thames** and in **Northumbrian Regions**). It remains to be seen whether the recent increase in the maximum level of fine from £2000 to £20 000 in the magistrate's court (Environment Act 1990) will be used effectively. The relatively recent moves to award costs to the NRA were seen as a positive step and provided a useful source of revenue; however, **Northumbrian Region** noted that this did not act as a deterrent to the farmer since payment of costs are often covered by insurance policies. In **Scotland,** costs cannot be recovered by RPBs since prosecutions are taken on by the Crown.

Thames Region noted that although financial penalties needed to be substantial, it was important that they did not interfere with the farmer's ability to pay for remedial measures. For this reason it was felt that fines should be closely linked to the ability to pay. Maximum publicity of successful prosecutions was considered a very effective deterrent in a number of NRA regions (e.g. **Anglian** and **Yorkshire**), since many farmers value their position in the local community and bad press can tarnish their image. This can also stir other farmers in the neighbourhood into taking remedial action. **Anglian Region** have contacted pressure groups on previous occasions to ensure good media coverage for successful prosecutions.

3.8.3 Education

The farming community's ignorance of the impact of farm pollution was seen as a major obstacle to improved river quality. Many farmers still believe that slurry helps the river by 'fertilising' it. Most NRA regions felt that farmers did not understand pollution control legislation, and **South West Region** thought that the code of good agricultural practice was not widely understood (this is currently being revised by MAFF in association with the NRA) [This has now been completed].

In addition, low financial penalties were felt to hamper the improvement of awareness through educational initiatives, since certain farmers were unlikely to take notice unless there was a real possibility of incurring a substantial financial penalty. For this reason, there was a mixed opinion on the usefulness of educational/publicity campaigns involving 'mailshots' and presentations at agricultural shows, since many felt that they were a case of 'preaching to the converted'. However, **NRA Anglian Region** have found presentations to groups of 'Young Farmers' to be useful in enhancing the awareness of the next generation of farmers.

Individual farm visits were seen as the ultimate educational tool, since they target the problem catchment/farmers and allow personal contact. **NRA South West Region** have found their region-wide campaign very useful in educating the farming community. **NRA North West Region** upgraded 40 km of the Wampool from NWC class 2 to 1B following an intensive campaign of farm visits, although this did coincide with drier weather which naturally leads to fewer farm pollution problems. However, follow-up visits showed that farmers had made waste management improvements. **Yorkshire Region** found water quality improvements (from NWC Class 4 to 2) on the Wiske following an intensive campaign which led to 34 grant-aided developments. Other NRA regions have found it difficult to discern water quality improvements following campaigns, largely because farm pollution is often localised in the unclassified headwaters of catchments.

The level of advice that should be given to farmers about required remedial measures causes difficulty for pollution control staff, since the NRA consider it important that their regulatory role is not compromised by being associated with detailed remedial plans. There is a danger, now that ADAS are charging for their advice on pollution prevention (although the first consultation is free), that farmers will be reluctant to seek independent advice and thereby risk causing pollution. The attitude of the NRA is perhaps not wholly consistent, since written approval is frequently given to pollution prevention plans under the Farm and Conservation Grant Scheme. A similar disclaimer to that used for grant aid approvals could be used when detailing advice on remedial measures.

The inclusion of courses on pollution prevention at agricultural colleges was seen as a good way of incorporating awareness into the farming community at an early stage, although most colleges contacted in England and Wales so far have apparently been non-committal about collaboration with the NRA. In comparison, the relationship between Scottish Agricultural Colleges (SACs) and river purification boards is at an advanced stage. The Scottish Farm Waste Liaison Group (SFWLG), responsible for the publishing of annual farm pollution statistics in Scotland (e.g. SFWLG 1989), consists of 2 representatives from the RPBs, 2 from the SACs, and 2 from DAFS.

3.8.4 Grant aid

The Farm and Conservation Grant Scheme, which pays for 50% of the construction costs of pollution prevention facilities if approved by the NRA, was generally seen in a favourable light. However, there are several important omissions that limit the usefulness of the scheme:

- The separation of clean water (mainly roof run-off from farm buildings) from the dirty water collection system, by appropriate guttering and channelling, is not eligible for grant aid since it is seen by MAFF as construction for clean water rather than polluted water. This is a very important omission from the scheme, since numerous pollution incidents are caused by excessive strain on waste storage capacity due to the incorporation of rain-water.

- Existing facilities in need of maintenance, repair or upgrading are not eligible for aid, which provides no incentive to the farmer to keep his waste management system in order. Routine waste management is vital to pollution prevention, and farmers are generally reluctant to spend time on what it seen as non-productive work.

- Grant allocation is not graded depending upon the importance or relevance of the work to pollution prevention. Schemes are either given 50% of construction costs or nothing at all. Graded allocation would provide greater incentive to undertake the most effective measures and would help to target aid at problem farms or catchments.

Yorkshire Region felt that more use could be made of EC Set-Aside money in pollution control; using it to restrict livestock numbers is one example. There is also plenty of scope for more encouragement of the setting up of buffer zones alongside riverbanks, although this is already a method of selecting Set-Aside land suggested to the farmer by MAFF. **Yorkshire** thought that there was also a need for rationalisation of Nitrate Sensitive Area and Set-Aside grants.

3.8.5 Waste treatment and management

Waste treatment was generally seen as a high risk option of disposal since farmers are reluctant to undertake the necessary routine maintenance work. Few farms have consented discharges to watercourses in any region, and applications for consent would not be looked upon favourably.

Weeping-wall stores were seen as a low-maintenance and effective method of solids separation. Low-rate irrigation systems are becoming increasingly popular throughout England and Wales for the disposal of dirty water and silage effluent, and when used properly are an effective method of disposal. However, the spray gun of such systems needs regular relocation to avoid over-application, and failure to do this has resulted in a number of land run-off incidents. Incidents have also been reported where the nozzle of the spray gun has been detached to allow a higher rate of application, thus destroying the principle of the system and again leading to over-application. In addition, such systems are abused by their use on under-drained land and by using them to apply slurry, for which they were not designed.

Clean water separation was seen as a vital part of waste management, and even though it was not currently included in the F&CGS, its investment could be rationalised by making the farmer aware of the savings to be made from lower required storage capacity and not causing pollution incidents.

A reversion to straw-based systems of waste management was seen as a beneficial pollution control measure. Slurry-based systems involve the storage of enormous volumes of highly mobile waste which have the potential to cause catastrophic pollution, and all too often realise that potential. The main obstacles to a large-scale reversion are: the cost of livestock housing modifications; the cost of straw (particularly transportation); and the reluctance of farmers to apply straw-based wastes at any time other than when the land was ready for ploughing and reseeding. The availability (and therefore cost) of straw has worsened due to the intensification of farming practices which has seen cereal (and straw) production and livestock production largely detached and geographically isolated from each other. This has led to a large surplus of straw in regions such as East Anglia, with a dearth in areas of intense livestock production (where it is most needed) such as the South West.

The change to big-bale silage production from clamps was welcomed in all NRA regions as it reduces the possibility of major silage liquor releases by compartmentalising it. In 1989, 10% of silage was produced by big-bale in England and Wales (Dowlex 1991), and the figure is likely to be higher for 1990. The highly visual nature of big-bales is causing aesthetic problems, however, and the Council for the Protection of Rural England has protested at their increasing presence (Dowlex 1991). Big-bales also produce large amounts of plastic waste which require disposal.

The development of the mobile 'Sentinel Plant' for the treatment of spent sheepdip was seen positively by pollution control staff, although there was concern that it was likely to be extremely costly to run. Problems with clogging during trials may be overcome by prefiltration.

The use of constructed wetlands and buffer zones for the protection of river quality were thought to be promising options by most NRA regions, although more investigation into their application was required.

3.8.6 Catchment management

Regulation of farming practices, including restrictions on livestock density and land spreading of wastes, was considered a prime necessity. Simple catchment management models were seen by some regions as a way of placing such catchment controls on an objective footing. **Yorkshire Region** saw simple models combining with catchment inventory databases that would have the ability to calculate waste production and land availability (for slurry spreading) for individual farms and would be able to continually update this information. Changes in pollution control staff can mean large losses of local knowledge, and such databases would help to minimise the impact of such changes. **South West Region** saw simple catchment models as the basis for Farm Waste Management Plans, where each farm would have an approved system of waste storage and disposal that would be monitored by the NRA. **Northumbrian Region** thought that the development of a simple computerised package that could place pollution risk assessment on a more objective footing would be useful.

3.9 Discussion

It is clear from the international initiatives outlined in Section 3.7 that the control of farm pollution abroad involves a far higher degree of central planning than in the U.K. This takes the form of: strict legislation governing high risk farming practices; tax penalties on undesirable activities; and the necessary infrastructure and funds to help and encourage farmers to comply with legislation.

International concern about livestock farming is directed largely at nutrient loadings to ground and surface waters, rather than the run-off of whole slurry leading to acute biological impacts. Nutrient loadings from this source have been given relatively little consideration in the U.K. compared to outputs from sewage treatment works and inorganic fertilisers, and the situation abroad may be an indication of the steps that will ultimately be necessary. Immediate pressure to control nutrient outputs from livestock farming comes from proposed EC legislation and also from the European agreement over inputs to the North Sea, as outlined in Section 3.7.2.

Waste production and disposal

There are no controls in the U.K. paralleling the restrictions and seasonal prohibitions on livestock waste applications to land that exist throughout the rest of Europe. Current U.K. policy is to encourage the farmer to calculate the nutrient requirements of his crop (using the voluntary Code of Good Agricultural Practice) and apply slurry accordingly. There is a recommended limit of 50 m^3 ha^{-1} per application (which can be repeated at 3-4 weekly intervals) where there is a 'risk' of water contamination (MAFF 1986b), although there is no quantitative indication of what might constitute a risk. In reality, pollution risk will only be proven if a pollution incident occurs.

Maximum allowable **annual** application rates in the European countries studied vary depending upon local circumstance. Roughly converted into volumes of undiluted cattle slurry using ADAS Booklet 2081 (MAFF 1986b), the **least** stringent restrictions in each country are as follows:

Denmark	2.3 livestock units	= 41 m3 ha^{-1} yr^{-1}
Switzerland	3 livestock units	= 54 m^3 ha^{-1} yr^{-1}
Germany*	2-3 dung units	= 32-105 m^3 ha^{-1} yr^{-1}
The Netherlands	175 kg P2O5 (from 1995)	= 88 m^3 ha^{-1} yr^{-1}

* Restrictions vary between German states

These restrictions show quite a wide variation, but it can be seen that they are roughly comparable to the recommended rate for **single** applications in the U.K. By the year 2000, the maximum rate in the Netherlands will be reduced further.

In addition to restrictions on application rate, the **Dutch** approach of limiting the allowable waste surplus from individual farms gives an indirect control over stocking density. **Denmark** takes a more direct approach and restricts stocking density to 2 livestock units (equivalent to 2 large dairy cows) per hectare. Stocking density restrictions will also be introduced in Sweden in the near future.

U.K. policy allows farmers to legally apply slurry at any time of the year, including through the winter season when the Soil Moisture Deficit is either non-existent or minimal and rainfall is maximal, giving maximum risk of surface run-off. Furthermore, unless a winter crop is planted, there is little nutrient utilisation through this period (on both arable soil and grassland), thus enhancing nutrient losses to ground and surface waters.

Spreading through the growing season on grassland, as is legally required by default on the continent due to winter prohibitions, is not favoured in the U.K. due largely to concerns over smothering and scorching by slurry leading to reduced yields. However, negative effects can be minimised or even effectively eliminated if suitable application rates are used, but it is unlikely that farmers would choose such an option unless forced to do so. Boxem and Remmelink (1987) found that although 20 tonnes ha^{-1} applications of slurry in the growing season reduced herbage growth by 17%, the effect on subsequent milk yield was minimal; applications of 10 tonnes ha^{-1} only reduced herbage growth by 5%. It is important to note, however, that although the risk of surface run-off is lower during the growing season, particularly dry weather can lead to cracking of clay soils and easy access to land drains.

As outlined in Section 3.7.3, regulations coming into force in **The Netherlands** in 1995 will require all slurry applied to grassland to be injected. This will eliminate the problems of scorching and smothering, and sub-surface sward damage can be minimised by injecting in Spring when the soil is moist (Prins and Snijders 1987).

Waste storage

The new U.K. regulations on the storage of slurry, silage and fuel oil (DoE 1991) give stronger powers to pollution control authorities, in that they provide a more objective platform for the risk assessment of storage facilities and permit prosecution for non-compliance irrespective of whether pollution has occurred. This is important, since it provides a legislative means of **preventing** pollution, rather than punishing polluters after the damage has been done. However, the level of risk which is deemed acceptable in the regulations, as defined by the required storage specifications, is higher than that in other European countries. For instance, the new minimum acceptable storage capacity in the U.K. is 4 months of waste production, as opposed to 6 months in **Germany** (9 months is recommended), 6 months in **The Netherlands** (rising to 9 months in certain areas), 9 months in **Denmark** and 8-10 months in southern **Sweden**. Small storage capacities lead to a higher risk of overflow, especially in areas of high rainfall or on farms where there is no clean water separation (see Section 4.3.2).

Danish legislation permits prosecution on the basis of pollution risk irrespective of whether the storage specification requirements have been breached. This allows the regulatory authorities more scope in assessing what actually constitutes a risk, which is a difficult phenomenon to quantify in legislative terms.

Waste control

Control of livestock waste by means other than legislation is practised abroad in the form of financial penalties for continuing undesirable practices, financial incentives for switching to desirable practices, and a high degree of centralised planning of waste management. In extreme cases, land buy-outs may be considered (e.g. **US**). In the U.K., financial incentive is the only option that is used, in the form of the Farm and Conservation Grant Scheme. Set-aside may be used as an incentive to set up riparian buffer zones, but this is not particularly encouraged by the U.K. government and the funds are derived from the European Commission in any case.

Examples of financial penalties imposed in Europe include the levy placed on manure surpluses in **The Netherlands**, which also acts as an 'inverted incentive' since the levy can be reduced if certain desirable practices are followed. Money raised from this initiative provides a proportion of the funds for the centralised infrastructure that deals with the country's manure surplus. **Belgium** is also developing a taxation system for manure surpluses. Another example of financial penalty is the **Swiss** system of livestock restriction, whereby a sizeable levy is due for each head of livestock on a farm above set limits.

Examples of financial incentives used in Europe include a number of grant aid schemes similar to the U.K. Farm and Conservation Grant Scheme, subsidised slurry transport (**The Netherlands**), and schemes to develop riparian buffer strips to ameliorate run-off quality (**U.S., Norway**).

Centralised planning of waste management takes many forms. A number of European countries have set up 'manure banks' (**The Netherlands, Germany, Belgium**), from where the utilisation or processing of surplus livestock waste is determined. Large scale processing plants have been constructed to handle surplus waste (**The Netherlands, Denmark**), often involving the production of biogas. **Germany** and **The Netherlands** are seeking to provide treatment systems (mobile ones in the case of Germany) which dry slurry to a readily storable and transportable pellet-form. Where centralised waste management is not offered, some European countries provide a free service that formulates farm waste management plans for individual farms (**Sweden, Norway**). In contrast, ADAS now charge for consultations concerning pollution prevention (although the first meeting is free).

In the U.K., there is no such centralised involvement in farm waste management as that described above. An example of where such involvement would be of benefit, apart from in the types of initiative mentioned above, is in assisting in the distribution of straw from areas with a surplus to those with a deficit, in order to make a return to straw-based waste management systems more feasible. Solid manure, the end-product of straw-based systems, carries a low pollution risk compared to slurry, due to the high mobility of the latter.

Monitoring and enforcement

It is clear from the description of practices abroad that the level of effort invested in the monitoring and enforcement of farm pollution controls is much higher in a number of other European countries than in the U.K. This is largely due to the fact that there are few U.K. controls against which to test compliance. What is also evident is the major role played by local councils in the enforcement of catchment-based legislation (waste storage, land application). In **The Netherlands**, a duty also falls on the civil police, to enforce prohibition periods on the land application of slurry. In the U.K., local authorities have no such role to play, and responsibility for any such enforcement (at present the only relevant legislation is the new regulations on the storage of slurry, silage and fuel oil) lies with the water pollution control authorities. This creates an extra resource burden which is not covered by current budget allocations.

The regulatory authorities in a number of European countries (**The Netherlands, Denmark, Switzerland**) base enforcement on the estimation of waste production on individual farms, and relate this to the amount of land available on each farm for disposal. This gives an indication of the waste surplus to be expected, which can then be checked against the amount of waste transported off the farm. In the U.K., there is at present no formal monitoring of the capacity of a farm to dispose of its own waste. In **The Netherlands** and **Germany**, this process is being taken one step further; the development of nutrient budgets on individual farms will help to maximise nutrient recycling and minimise losses to surface and groundwaters.

All of the European countries looked at in detail place an onus on the farmer to demonstrate safe disposal of his livestock waste. This takes the form of: legally binding contracts between farmers for slurry transfers (**The Netherlands, Denmark, Switzerland, Germany**); disposal via safe routes (e.g. biogas plants, 'manure banks') approved by the regulatory authorities (**The Netherlands, Denmark**) or receipt systems for slurry transfers (**The Netherlands, Denmark**). There may also be soil monitoring to check that allowable slurry application rates have not been exceeded (**The Netherlands, Germany**). There are no such systems of accountability organised in the U.K., where in effect pollution has to occur before improper disposal can be verified.

4. NATIONAL ASSESSMENT OF FARM POLLUTION

As with Section 3, this assessment was conducted in 1990 and much of the information reported on NRA activity will be somewhat out of date.

4.1 Introduction

A regular national assessment of farm pollution that has been available until recently is the annual farm waste report, produced jointly by NRA and MAFF (e.g. NRA/MAFF 1989). This has now been superseded by the new NRA annual pollution report, which will incorporate pollution incidents from all sources. An annual report on farm pollution is produced in Scotland by the Scottish Farm Waste Liaison Group (e.g. SFWLG 1989). Such annual reports are useful, but they have only a coarse geographical resolution (based on regional differences). There is a need for a national assessment with a finer resolution of farm pollution problems, down to the catchment level. Such an assessment needs to incorporate pollution **risk** in addition to details of observed impact, so that the results can be used proactively for preventative planning. Furthermore, the farm waste report deals only with readily observable intermittent problems in the form of pollution incidents, and does not consider the more insidious effects of farm pollution.

4.2 Methods

4.2.1 Consultation

A questionnaire was constructed and sent to regional NRA pollution control managers prior to discussions. The questionnaire (see Appendix B) dealt with a number of issues, including regional risk assessment practices and monitoring and archiving procedures, but the main objective was to extract any information available within each region on the distribution of livestock farming and, more particularly, pollution problems arising from it. The questionnaire was intended to act as a guide to discussion, but the interviewees were also requested to gather any quantitative information available prior to the meeting. The questionnaire was also sent to River Purification Boards (RPBs) in Scotland and to the DoE (N Ireland) for written responses.

4.2.2 Identification of high risk areas

Data collation

In order to gain an indication of the distribution of pollution risk arising from livestock farming activities, data on a number of factors contributing to risk were collected on a national scale. The factors considered, listed below, were felt to be the most important contributors to farm pollution risk (see Section 4.3.2) that were amenable to a national assessment of this nature.

- Livestock numbers - Computerised data on livestock numbers in England and Wales for 1988, together with information on agricultural land areas, were obtained from the ESRC Regional Research Laboratory for Scotland. These data were derived from parish-level MAFF Agricultural Census data, which were reduced onto a 1 km^2 national grid using a 3-way land classification. This effectively allowed urban areas to be located, and livestock numbers were subsequently divided up between the two remaining land classifications (moorland and 'other' agricultural land). Sheep numbers were divided up equally between these latter two land classes, whilst other livestock types were assigned to 'other' agricultural land only. In order to keep the database to a manageable size, data from this source were initially aggregated into 2 km^2 gridsquares. Livestock numbers in Scotland were obtained at the parish level directly from the DAFS 1988 Agricultural Census, whilst 1988 district level data were obtained for N Ireland from the Department of Agriculture (N Ireland).

- Landslope - a national distribution of landslope was formulated on an ARCINFO Geographical Information System using an irregular elevation matrix digitised from 1:500 000 scale topographic maps. Landslope values were generated from a Triangulated Irregular Network (TIN) created from the elevation matrix.

- Winter Rainfall Acceptance Potential (WRAP, an indicator of soil permeability) - the national distribution of WRAP was digitised onto ARCINFO from a 1:500 000 map.

- Rainfall - the contour distribution of average annual rainfall over the period 1941-1970 was digitised from 1:500 000 scale maps. A TIN was generated from points sampled along the contours, and a point-value grid was created from this with 1000 m^2 cells.

Data was stored in an ARCINFO Geographical Information System for subsequent analysis.

A catchment map of the U.K. was constructed (see Map 1, Map Annex), using maps supplied by NRA regional staff, quinquennial River Quality maps and Ordnance Survey maps, to act as a basis for the development of all distribution maps produced in this study. This was digitised into an ARCINFO coverage.

Data on farm pollution incidents from 1988 (a relatively wet year) and 1989 (a dry year) were sought from all NRA regions to gain a semi-objective picture of the distribution of intermittent pollution problems. The national grid reference, incident type and incident severity of all farm pollution incidents occurring in the calendar years 1988 and 1989 were requested. The incident types stipulated were essentially the categories used in the annual farm waste report, whilst the incident severities stipulated were either those used in the farm waste report or the new NRA national incident categories (see Appendix C for further details).

The pollution incident data gathered were used to modify the level of detail of the national catchment map to ensure that the map was consistent with the level of spatial heterogeneity exhibited by pollution incidents. The data were subsequently used to compare the distribution of known intermittent farm pollution problems with the formulated pollution risk maps.

Data were also sought from NRA regions on recent NWC downgradings known or suspected to be caused (at least in part) by farm pollution. This was meant to give an impression of the distribution of chronic farm pollution problems, which could then be compared with the distribution of pollution risk in the same way as the pollution incident data.

River biota provide an integrated picture of prevailing water quality (Section 5), and for this reason biological data would be a useful inclusion in a national assessment of farm pollution. However, biological data are not as readily available on a national scale as data on pollution incidents and NWC downgradings, with the consequence that any coverage of information would be very patchy. With the incorporation of biological assessment into the 1990 River Quality Survey, the inclusion of biological information in this type of assessment becomes a possibility for the future.

Data analysis

Using the ARCINFO database, values for each factor considered were assigned to each catchment.

i) Livestock

The total numbers of dairy cattle, beef cattle, pigs, poultry and sheep were calculated for each catchment in the national catchment map by aggregating values in individual 2 km^2 grid squares within each catchment. In order to allow comparison between catchments, these numbers were converted into livestock density, in terms of **total** catchment area. Whilst this indicates the national distribution of livestock farming activity, much of the pollution risk posed is related to the amount of organic waste produced, especially in relation to the amount of land available for its disposal.

Estimates of organic waste production from each livestock type (except sheep) were made for each catchment, based upon information on waste output in ADAS advisory booklets (MAFF 1985a and 1986b - see Appendix D). These estimates took into account the dependency of volumetric output on bodysize. Volumetric output was converted into a waste loading by dividing by the land area that was potentially available within each catchment for disposal (by land application). Available land was taken as the combined area of crops, fallow and pasture.

The output from each livestock type was then combined to give a total volumetric waste loading.

Although total volumetric waste loading gives an indication of the relative likelihood of run-off occurring due to over-application, it gives no indication of the polluting strength of the waste. Polluting strength varies with livestock type (see Section 3.2), and it is important to try and take this into account when combining livestock waste outputs to give an indication of pollution risk. Estimates of Biological Oxygen Demand (BOD), nitrogen and phosphorus output were calculated from estimated volumetric waste output, assuming typical waste compositions for each livestock type (see Appendix D).

It should be noted here that the estimates of waste volume and polluting strength made above take no account of farming practice or subsequent waste management, which will critically modify the pollution risk posed. The amount and nature of waste that is stored and ultimately requires land disposal will depend upon factors such as: the intensity of farming; the length of overwintering period (if applicable); methods of waste management (particularly the production of manure or slurry); the age of the waste and the amount of ammonia volatilisation. Since the above factors could not readily be accounted for in this analysis, it was decided to express waste output in terms of the daily production of undiluted fresh slurry (excreta and faeces).

The organic waste loading figures so produced give an assessment of pollution risk in each catchment due to the presence of livestock alone, which can then be refined by taking the risk due to landslope, soil permeability and rainfall into account.

Other important sources of pollution risk due to livestock farming arise from silage production and sheepdipping. The distribution of risk associated with silage has not been dealt with specifically in this analysis (although it will roughly follow the distribution of cattle), but an attempt has been made to indicate the distribution of risk of sheepdip pollution. For this, the assumptions (relevant to this analysis) made by Littlejohn and Melvin (1989) were followed. These were that:

- 300 sheep are dipped in one day in one dipper;
- the dip volume is 2700 litres;
- propetamphos is the active ingredient, at a working concentration of 320 mg l^{-1};
- a topping up of the dipper with fresh sheepdip is required through the day;
- a completely fresh volume of sheepdip is made up each day.

Using these assumptions, spent sheepdip production, in terms of both volume and 'propetamphos equivalents', can be estimated for each catchment (see Appendix D). Dividing this production by total catchment area gives a coarse relative indication of waste loading. The use of propetamphos also gives rise to large quantities of waste phenolic compounds, which can be simply estimated from the amount of propetamphos used (see Appendix D).

Looking at the distribution of waste sheepdip production in terms of only one pesticide (albeit a common one, accounting for perhaps up to 70% of U.K. sheepdip usage) is useful since it gives an indication of the scale of waste sheepdip production in terms of one of the main active ingredients; however, there are a wide range of sheepdips on the market with varying ecotoxicities (these have recently been reviewed by WRc - NRA Project Reference A19.53). Similar estimations could be performed for other sheepdip pesticides using the volumetric estimations of sheepdip usage and the active concentration of the pesticide concerned. If the relative usage of different formulations were known in a catchment the scale of waste production of each pesticide could be estimated in the same way. A more detailed study of sheepdip usage has recently been undertaken by WRc (NRA Project Reference A07.2(208)).

ii) Physical factors

For the national distribution of rainfall and landslope, which had been resolved into point-value grids, points falling within a catchment were averaged to give a mean rainfall and landslope. For the WRAP distribution, which was zonated, a mean value was assigned to each catchment by giving areal weightings to each area of WRAP class falling within a catchment.

iii) Overall risk

The relative importance of each of these risk factors considered is difficult to determine, and so a pragmatic approach to assigning weighting factors has to be taken in order to produce a combined index of risk.

Appendix D gives the list of weighting factors used and, as can be seen, a high weighting has been given to livestock activity compared to the physical factors considered. The combination of all 3 physical factors constitutes 20% of the value of the index. BOD was originally used as the parameter which describes total livestock activity (for the above reasons), and two different BOD factors were used to produce the livestock contribution to the index: 1) BOD production in relation to available land area; and 2) BOD production in relation to catchment area. Both of these measures are important, since the former gives an indication of intensity of risk, and the latter indicates the scale of risk throughout the catchment. Thus, one small dairy farm may present an acute risk in an otherwise non-agricultural catchment (i.e. waste loading is high), but the scale of risk over the entire catchment is not important (i.e. catchment waste production is low).

Pollution risk maps produced using BOD loading were very weighted towards the U.K. distribution of poulty. Since poultry rarely produce water quality problems of a significant nature, it was felt more appropriate to use total volumetric loading as the livestock risk factor.

Based on the application of similar weighting factors to those used in estimating organic pollution risk, it has been possible to produce a map of pollution risk due to waste sheepdip.

Limitations of the approach

The number of factors contributing to farm pollution risk that can be accounted for in a national analysis such as this are limited. Whilst the factors used in this study are very important to overall pollution risk, factors such as form and quality of management, chance events, proximity to watercourse, the presence of land drains (i.e. factors acting at the level of detail of individual farms or even fields) are equally important and cannot be incorporated into the analysis. It is therefore clear that this approach can only hope to direct attention to those catchments that have the most potential, on the basis of known factors, to give rise to farm pollution problems.

Mention should also be made of the level of accuracy of the digital data used in the analysis. From an error analysis, it has been estimated that 90% of grid cell values of average rainfall lie within 150 mm of the true average. The estimated accuracy of the digitised WRAP map is 500 metres, whilst 90% of point-values of height (from which landslope was derived) are estimated to lie within 50 metres of their true location. Concerning the base catchment map, the scope of the study allowed no more than a relatively crude estimation of watershed position, usually from small regional catchment maps copied free-hand onto 1:250 000 River Quality maps. Nevertheless, 90% of representative sample points on these boundaries are estimated to lie within 500 metres of the true watershed - the exception to this would be NRA Southern Region, for which no catchment map was available and where low drainage densities make estimation of watershed position from River Quality maps highly dubious.

For the above reasons it is evident that this analysis can only provide a general description of each catchment in terms of slope, soil permeability and rainfall. Although data on livestock are held at a finer resolution than catchment level (2 km^2 gridsquares), the catchment is treated as the smallest discriminable unit. It is not possible to accurately locate agricultural land within a catchment, although the total area of catchment occupied by agriculture is known. For this reason, and because of the limits to the accuracy of the physical data described above, it is not possible to link agricultural land to its physical characteristics. It may well be, therefore, that within a catchment most livestock farming is practised in relatively low risk areas (flat land, permeable soils), whilst potentially high risk areas are devoid of livestock; such a catchment could be designated as high risk in this analysis. This has to be accepted as one of the limitations of a wide-ranging assessment such as this.

Agricultural census data have their own limitations, since livestock are assigned to the parish in which the owner is registered, and this may not be the same parish in which they are held. This can lead to catchments appearing to be of high pollution risk when there are in reality very few livestock present.

Lastly, the limitations of the data on observed farm pollution used in this study should be noted. These limitations are particularly important since the validity of the pollution risk maps produced has largely been judged on these data. Pollution incident information is largely based on chance observation, usually by members of the public, and as such depends upon a clearly visible effect, public vigilance and public presence. This dependency means that it is inevitable that large numbers of incidents are never observed or reported, and that the efficiency of reporting is likely to decline with increasing remoteness from population centres. It also means that less visible forms of pollution, such as that from sheepdip, hardly feature in pollution statistics. Furthermore, incidents have to be verified by pollution control staff, and the inevitable time delay between reporting and attendance often means that real incidents are missed and cannot therefore be reported as such. All of these considerations lead to an underestimate of uncertain magnitude in the frequency of intermittent farm pollution problems and, more importantly to the identification of high risk areas, the likelihood of spurious geographical distributions of pollution incidents.

The usefulness of information on chronic farm pollution problems is limited in that the NWC classification system is not applied in headwater streams, where a high proportion of problems are to be found. Moreover, classification of a river stretch is based on a limited number of discrete samples through the year, which may not adequately reflect the prevailing water quality. Both of these factors lead to a likelihood of gross underestimation of farm pollution.

For the above reasons, it is important not to place too much emphasis on agreement between the distribution of farm pollution risk, as produced in this analysis, and existing data on observed pollution problems. However, such a comparison is useful as a general guide, and it can highlight areas where water quality problems are likely to occur but where little monitoring has been undertaken.

4.3 Results and discussion

All maps referred to below can be found in the separate Map Annex to this report.

4.3.1 Distribution of livestock farming activity

Map 1 shows the national catchment map used as the basis for all subsequent distribution maps in the map annex. Table 1 in the Map Annex is a list of catchments, by region, which cross-refers to the catchment codes in Map 1.

Unfortunately, owing to problems in obtaining appropriate district boundaries, it has not been possible to show livestock densities for Northern Ireland, although livestock data have been obtained for this area. The following discussion is therefore necessarily restricted to England, Wales and Scotland. Northern Ireland would be included in any subsequent analysis if further funding were available.

Dairy cattle

Map 2a shows the U.K. distribution of dairy cattle by river catchment. Hotspots of activity are apparent in south west Wales, south west and north west England, and around the Cheshire Plain. Very little activity is evident in eastern England or Scotland, although there are significant numbers of dairy cattle to the south of Glasgow. Areas of lower but significant activity generally border the above hotspots.

At a more detailed level, the highest densities of dairy cattle in the south west of Wales are located in the **Taf, Cynin** and **Cywyn** valleys (0.9-1.2 head ha^{-1} catchment area), with the **Eastern** and **Western Cleddau,** the middle and lower reaches of the **Teifi,** the **Gwili,** lower **Tywi, Nyfer, Gwaun** and surrounding coastal areas having slightly lower densities (0.6-0.9 head ha^{-1}).

Some of the catchments around the Cheshire Plain have the highest dairy cattle densities in the U.K. (>1.2 head ha^{-1}), these being the upper **Weaver, Gowy,** and

Northerbury Brook (Dee catchment). Adjacent catchments with slightly lower densities (0.9-1.2 head ha^{-1}) include the **Dane, Wheelock** and **Wincham Brook** (all in the main Weaver catchment), the middle reaches of the **Dee**, the **Sow, Churnet** and middle and lower reaches of the **Dove**. Slightly lower densities again (0.6-0.9 head ha^{-1}) are found in the upper tributaries of the **Severn** (including the **Morde, Perry, Roden** and **Rea Brook**).

In the south west of England (encompassing NRA South West and Wessex Regions), intermediate to high densities are widespread, with the greatest densities (0.9-1.2 head ha^{-1}) in the **Axe** catchment (near the Exe), upper **Tamar** (main river), the middle reaches of the **Stour** and its tributaries (**Cale, Lydden**), the **Mells** (Bristol Avon), **Hartlake** and **Sheppey** (Brue). Intermediate densities (0.6-0.9 head ha^{-1}) are found throughout the **Torridge** and **Exe** catchments, the tributaries of the upper **Tamar** (**Thrushel, Carey** and **Ottery**), the **Erme, Yeo, Isle, Axe** (Weston-Super-Mare), the upper **Bristol Avon** and upper and middle reaches of the **Frome**. There are hardly any catchments in the south west area which fall into the lowest density class (<0.3 head ha^{-1}), indicating significant dairy activity throughout.

In the north west of England, high densities (>1.2 head ha^{-1}) are found in the **Brock** catchment and middle reaches of the **Wyre**, with slightly lower densities (0.9-1.2 head ha^{-1}) on the lower **Wyre** and lower **Ribble**, the **Conder**, and also on the **Wampool** and **Waver** in the northern Lake District. The lower **Eden** (including the **Caldew** and **Petterill**), **Ellen**, middle reaches of the **Ribble**, upper **Aire, Darwen** and **Bela** all have intermediate densities (0.6-0.9 head ha^{-1}). Low but significant densities (0.3-0.6 head ha^{-1}) are found throughout the **Eden** catchment and the middle reaches of a number of catchments running eastwards across the Pennines to the east coast (**Aire, Nidd, Washburn, Wharfe, Ure, Swale, Wiske** and **Leven**).

In the south west of Scotland, intermediate densities (0.6-0.9 head ha^{-1}) are found on the **Irvine**, with lower densities on the **Ayr**, middle reaches of the **Clyde, Avon Water, Urr Water** and lower **Nith**.

Finally, low density clusters of dairy activity are evident in Gwent and the southern Welsh border area (e.g. lower **Usk** and **Wye**), and also in south east

England along the South Downs and adjacent catchments (e.g. middle reaches of the **Arun, Kurd, Adur West,** upper **Mole** and **Eden).**

Beef cattle

It is evident from Map 2b that beef farming activity is generally not as concentrated into specific areas as dairy farming, but areas of higher activity are still apparent. The distribution is again skewed towards the western side of the U.K. as with dairy farming activity, with the most notable concentration of activity in the south west of England, and other important concentrations in northern England (along the Scottish border), the border area between Wales and England, south west Wales, and (in Scotland) the Solway area and north eastern Perthshire.

In south west England, beef farming is concentrated in the NRA South West Region, in contrast to dairy farming which is also very important in NRA Wessex Region. The highest beef cattle densities (>0.6 head ha^{-1}) are found in areas of the **Tamar** catchment (the **Ottery, Inny** and lower reaches of the **Tamar**), with slightly lower densities (0.45-0.6 head ha^{-1}) in other areas of the catchment (the **Lynher, Thrushel, Lydd, Tiddy** and lower and middle reaches of the main river), and also in the **Camel** catchment, the **Fowey,** the middle reaches of the **Exe** (including the **Batherm**), **Tresillion** and **Allen, Carnon** and **Kennal Vale.** Virtually the whole of the remainder of NRA South West Region has intermediate densities of beef cattle (0.3-0.45 head ha^{-1}), including the **Torridge** catchment, the rest of the **Exe** catchment and the **Taw.**

Along the Welsh border, the highest densities (0.45-0.6 head ha^{-1}) are found in areas of the upper **Severn** (including the upper **Teme**), Intermediate densities are found throughout the **Wye** catchment (including the **Monnow, Lugg, Frome** and **Ithon),** the upper **Severn** (including the **Clun,** middle reaches of the **Teme, Onny, Corve, Rea, Camlad, Rea Brook** and **Tanat**) and lower **Usk**.

Catchments holding significant densities of beef cattle (0.15-0.3 head ha^{-1}) in south west Wales are essentially the same as those holding high densities of dairy cattle. Other noticeable concentrations in Wales are found on Anglesey (including the **Cefni** and **Braint)** and the **Leyn Peninsula,** on the south coast between Swansea and Cardiff (**Ogmore, Ely, Kenson** and **Cadoxton**), and in the **Clwyd** catchment to the north.

In northern England, the catchments of the northern Lake District hold the highest densities of beef cattle, with the **Wampool** having >0.6 head ha^{-1}, and the **Waver, Ellen** and lower **Eden** having between 0.45 and 0.6 head ha^{-1}. Intermediate densities (0.3-0.45 head ha^{-1}) are found in adjacent catchments (upper and middle reaches of the **Eden, Petterill, Caldew,** lower **Derwent, Cocker, Irthing, White Lyne** and **Black Lyne**) and other catchments stretching east across to the Northumbrian coast (lower reaches of the **North** and **South Tynes, East** and **West Allen,** upper reaches of the **Wansbeck** and **Blyth,** lower **Aln** and lower **Coquet**). There are also a number of catchments with intermediate densities (0.3-0.45 head ha^{-1}) running from the south of NRA Northumbrian Region down into the Vale of York (the **Browney, Gaunless,** middle reaches of the **Tees,** middle and lower reaches of the **Swale, Wiske,** lower **Ure** and the upper **Ouse**).

A few small catchments around the western Peak District area hold intermediate densities of beef cattle (**Sow, Blithe,** lower **Dove, Hilton Brook, Wye** (tributary of the Derwent) and **Amber**), and there are also similar densities in a cluster of catchments to the south of Birmingham (**Avon** headwaters, upper **Soar, Sence, Eye** and upper **Welland**).

In Scotland, the Solway area holds a concentration of beef cattle, with a high density (0.45-0.6 head ha^{-1}) in the **Urr** Water catchment and intermediate densities (0.3-0.45 head ha^{-1}) in catchments such as the lower **Nith** and **Cairn Water,** the **Water of Fleet** and the **Dee,** the **Bladnoch** and the **Water of Luce.** Further north in Perthshire, there are high densities in the **Ythan** and **Ugie** catchments and intermediate densities in the **Cowle Water** catchment, upper **Don,** lower **Dee, Urie** and **Deveron.**

Pigs

Map 2c shows the U.K. distribution of pigs by river catchment. From this, it is clear that pig rearing is concentrated in two main areas along the east coast, Yorkshire (Vale of York and eastwards to Humberside coast) and East

Anglia, with minor but significant concentrations in the north west of England, the Midlands, Thames Valley, Somerset, Dorset, and a small pocket of activity in Perthshire. Very little activity is evident along the west coast of the U.K.

In the Yorkshire hotspot (all in NRA Yorkshire Region), highest densities (class 5, >3 ha^{-1}) are found in the upper **Ouse** catchment, the **Foulness** and **Mires Beck, Stream Dyke** and around **Spurn Head**. Slightly lower densities (2-3 ha^{-1}) are found in the lower **Calder,** lower **Ure, Foss,** lower **Derwent,** the **Beck** and the **Hull**. Catchments holding intermediate densities (1-2 ha^{-1}) are the **Wiske,** the middle and lower reaches of the **Swale** and **Nidd,** middle reaches of the **Ure, Gypsey Race,** middle **Ouse** and lower **Aire**. A number of adjacent catchments have low but still significant densities (class 2, 0.5-1 ha^{-1}).

In the East Anglian area of activity (all within NRA Anglian Region), densities of >3 ha^{-1} are found in the **Dove** and **Chickering Beck** and parts of the **Deben** and lower **Wissey** catchments, whilst slightly lower densities (2-3 ha^{-1}) are located in the middle reaches of the **Waveney,** the upper **Alde** and **Ore, Deben, Gipping, Tas** and upper reaches of the **Little Ouse.** Catchments with intermediate densities (1-2 ha^{-1}) are the upper **Waveney,** the **Thet,** upper **Yare** and **Tiffey,** upper **Wissey, Tud,** upper **Bure, Blyth** and **Walpole,** and tidal reaches of **Deben** and **Orwell.** As in Yorkshire, a number of catchments bordering this hotspot of activity have low but significant densities (class 2, 0.5-1 ha^{-1}).

In the small area of activity in the north west of England (within NRA North West Region), the middle reaches of the **Wyre** hold densities of 2-3 ha^{-1} (class 4), whilst the lower **Wyre,** tidal **Ribble** and lower **Douglas** have intermediate densities (1-2 ha^{-1}). To the south of this area, a number of catchments hold low but significant pig densities.

In the Thames Valley, the **Lambourne, Pang** and **Sulham, Ock,** and some of the upper and middle reaches of the Thames itself hold intermediate densities (1-2 ha^{-1}), with the predominantly agricultural catchments to the north and west holding low but significant densities (0.5-1 ha^{-1}).

Further west in NRA Wessex Region, intermediate densities are found in the **Semington Brook** catchment, **Cam Brook, Mells, Cary** and **King's Sedgemoor Drain, Cerne** and **Frome.** High densities are found in some small catchments to the south of this area (upper **Frome** and **Piddle,** and the **Simene**).

To the west in NRA South West Region, the **Otter** has intermediate densities (1-2 ha^{-1}) and a number of adjacent catchments have slightly lower densities (0.5-1 ha^{-1}), but there is generally very little activity.

In the Midlands, a few isolated catchments hold densities of 1-2 head ha^{-1} (upper **Soar** and **Witham** and some reaches of the **Trent**), and a number of catchments (particularly to the south of the Yorkshire hotspot and in the Cheshire Plain area) hold slightly lower but significant densities.

Poultry

It is evident from Map 2d that poultry farming activity has a much more scattered distribution than other livestock types, with isolated catchments of high density (>40 ha^{-1}) in north Wales (**Braint**), the Midlands (**Hilton Brook, Rothley Brook**, upper reaches of the **Great Ouse**), and north east England (**Castle Eden Burn, Leven**). However, small clusters of catchments with significant activity are evident in East Anglia, Lancashire, Lincolnshire/Nottinghamshire and around the Forth Estuary, and there are belts of catchments with lower densities running north-to-south along the Welsh border country, and also east to west in south west England. A number of catchments in south east England also have significant densities.

Catchments with the highest density (>40 head ha^{-1}) in the East Anglian area of activity are the middle reaches of the **Waveney,** the **Dove** and **Chickering Beck,** the upper **Ant** and the **Tas,** whilst the upper reaches of the **Little Ouse** have slightly lower densities (30-40 ha^{-1}). Intermediate densities (20-30 ha^{-1}) are found in the upper **Waveney,** the **Thet** and the **Gipping**, and a number of adjacent catchments have lower but still significant densities (10-20 ha^{-1}).

In Lancashire, the lower **Douglas** and the middle reaches of the **Wyre** have high densities (class 5 - >40 head ha^{-1}), and slightly lower densities (30-40 ha^{-1})

are found in the **Darwen** and **Lostock**. Intermediate densities (20-30 ha^{-1}) are found in the **Crossens** and **Yarrow** catchments.

In the Lincolnshire/Nottinghamshire area, the lower **Maun** has a high density (class 5 - >40 ha^{-1}), whilst class 4 (30-40 ha^{-1}) densities are found in the lower reaches of the **Trent,** and the lower **Ancholme.** The upper **Maun, Slea** and **Hobhole** drainage area have intermediate densities (20-30 ha^{-1}), and a number of adjacent catchments have densities of 10-20 ha^{-1}.

It should be noted that due to the lack of large scale clustering of poultry farming activity in specific areas, it is likely that large poultry farms will occur in catchments in isolation, such that when poultry numbers are averaged out over the catchment low densities are indicated even though localised pollution risk may be high. This is unfortunate but unavoidable in an assessment on this scale.

Sheep

Map 2e shows high sheep densities over most parts of Wales, stretching into the border country; indeed, only the cattle farming area in south west Wales has low densities. In comparison, the rest of the U.K. has relatively low concentrations of sheep. The next most important area is the Pennines, with lower but significant densities in the adjacent areas of the Lake District and Yorkshire Dales. Southern Scotland also has low but significant densities, but further north all catchments are placed in the lowest category, even though sheep farming is an important activity in these areas. The south west of England has a concentration of sheep in the Exmoor area and a smaller concentration in the Dartmoor area; in addition, much of the rest of the south west has low but significant sheep densities. Smaller concentrations occur in Kent (around Romney Marsh), the Peak District, the Yorkshire Wolds and the northern Cotswolds.

In Wales, the upper reaches of the Wye (including the **Irfon** and **Ithon**) have the highest sheep densities in the U.K. (>4 ha-1), along with tributaries of the upper Severn (upper **Teme,** the **Sence** and middle reaches of the **Soar**) and the **Elwy** (a tributary of the Clwyd). Adjacent catchments have slightly lower sheep densities of 3-4 ha^{-1} (middle reaches of the **Tywi**, upper **Usk**, **Monnow**, upper **Teifi**, **Ystwith**, **Wye headwaters**, upper **Dovey**, **Severn headwaters** and several upper Severn tributaries (**Banwy**, lower **Vyrnwy** and **Tanat**), **Dysynni**, upper reaches of the **Dee** including the **Alwen** and **Ceiriog**, upper **Clwyd**, lower **Conwy**, **Gwyrfai** and **Seiont**) but still higher than nearly all other catchments in the U.K. Similar densities are found on Anglesey (including the **Alaw**), and the **Llyn Peninsula**. Catchments between the Welsh uplands and lowland areas tend to have intermediate sheep densities (2-3 ha^{-1}).

In the Pennines, a number of catchments have intermediate densities of 2-3 ha^{-1}: the **Eamont**, **Lowther**, **Kent**, **Rawthey**, upper **Eden**, **Greta**, middle reaches of the **Lune**, **Wenning**, upper and middle reaches of the **Ribble**, upper **Hodder**, upper **Aire** and middle reaches of the **Wharfe**. Adjacent catchments in the Yorkshire Dales (including the upper **Swale**, and the upper and middle reaches of the **Ure** and **Nidd**), the Lake District (including the upper and middle reaches of the **Eden**, the **Petterill**, **Caldew**, upper **Derwent** and upper **Leven**) and up into the Cheviots (including the upper and middle reaches of the **Tees**, the **North** and **South Tynes** and the **Coquet**) all have lower but significant densities (1-2 ha^{-1}).

In southern Scotland, significant densities (1-2 ha^{-1}) are found throughout the **Tweed** catchment (including the **Teviot**, **Ettrick** and **Yarrow Waters**, **Blackadder** and **Whiteadder Waters**), in some areas of the Solway region (including **Urr Water**, lower **Nith** and **Cairn Water**, **Annan**), the upper and middle reaches of the **Clyde**, and the **Esk** (Forth region).

In the south west of England, catchments draining Exmoor with the highest sheep densities (3-4 ha^{-1}) are the **Barle**, **Yeo** and **Lyn**, with intermediate densities (2-3 ha^{-1}) in the much of the rest of the **Taw** catchment (**Mole**, **Bray** and tidal **Taw**), the **Batherm** (Exe tributary), and the **Cober** (which drains the adjacent Dartmoor). Lower but significant densities (1-2 ha^{-1}) are found over much of the remainder of NRA South West Region, including throughout the **Torridge**, **Tamar** and **Camel** catchments.

In Kent, the **Brede,** lower **Rother** and **Welland Marsh** have the highest densities (2-3 ha^{-1}), with the **Beult,** upper **Great Stour, Dour** and the middle and upper reaches of the **Rother** having lower but still significant densities (1-2 ha^{-1}).

Small clusters of catchments with densities of 1-2 sheep ha^{-1} occur in the Peak District (**Wye,** upper **Derwent, Manifold** and upper **Dove, Derwent reservoirs, Goyt, Sett, Etherow, Dean** and **Micker Brook**), northern Cotswolds (**Leam** and **Stowe, Stour,** upper **Cherwell, Tore,** upper **Nene, Wilton Nene, Avon headwaters** and **Sence**) and Yorkshire Wolds (middle reaches of the **Rye, Walmouth Beck, Costa Beck, Seven** and the **Esk**).

4.3.2 Pollution risk and its assessment

Farm visits to assess pollution risk are a vital aspect of farm pollution control. In addition to giving an indication of the likelihood of pollution problems, they fulfil an essential educational role and also serve to remind the farmer that his activities are under scrutiny.

Current practices and survey results

The level of effort put into, and degree of formalization of, farm pollution risk assessment varies considerably between regions, depending to a certain extent upon the perceived importance of farm pollution in each region relative to other sources. Individual farm visits are necessarily labour intensive and there is often little opportunity for pollution control staff to pursue such proactive work in the face of their large reactive workload. Farm and Conservation Grant Scheme assessments and responses to pollution incidents provide *ad hoc* opportunities for general risk assessment; in most RPB regions these are the only opportunities to undertake risk assessments.

It should be noted that the risk assessment is only the initial phase of pollution prevention, and that follow-up work to ensure preventative measures are taken is vital if improvements are to be made. Such follow-up work constitutes a large proportion of the effort used in farm surveys.

NRA South West Region have a regionwide farm campaign and have now visited over 6000 farms since 1984. **Anglian Region** also have a rolling programme of farm visits across the region, and **Northumbrian Region** plan to develop a regionwide programme. A number of NRA regions (**Severn-Trent, North West, Yorkshire, Thames and Welsh**) target their proactive effort by selecting catchments/areas where farm pollution problems are most acute. **Welsh Region** survey all potential polluters, rather than concentrating only on farms, in order to show farmers that they are not being singled out. **Wessex** and **Southern Regions** perform risk assessments on a more local and *ad hoc* basis, relying upon the knowledge of local pollution control staff rather than directing effort at a regional level.

Risk is assessed subjectively in all regions, based on various contributory factors which are not standardised between or within regions. The only real exception to this is **Forth RPB**, who have recently introduced a standardised farm questionnaire which is now used on all their farm surveys. This asks for a long list of details including: slurry and silage storage capacity, the likely fate of effluents, detailed questions on sheep dippers (construction, method of pesticide disposal, etc.), and the current impact on receiving waters.

Although risk assessment is subjective country-wide, it is performed by experienced pollution control staff whose knowledge should not be underestimated. Factors that are considered in most instances are given below:

Presence of discharges
Age, condition and type of storage facilities
Waste storage capacity (in relation to waste production)
Clean water separation
Proximity to watercourses
Topography
Soil type
Rainfall
Quality of management
Scope for containment of leaks/spillages
Presence of land drains

Downstream users
Waste handling facilities
Sheep dipper design

The area of suitable land for waste spreading in relation to waste production is not usually assessed, although general land suitability (in terms of topography and soil type) may be.

Farms are divided into 3 risk categories in a number of NRA regions (**South West, Yorkshire, Welsh, Thames and Severn-Trent**) and the type and timescale of follow-up action is dictated by these categories. In the other 5 regions, high risk farms are identified and singled out for follow-up action.

The following risk categories are used in **NRA South West Region**:

Red - farms that are polluting and have been reported to pollution control staff for a quick remedial visit.

Green - farms that are discharging but not causing pollution. This category consists of: those farms likely to be discharging but not causing pollution; those likely to cause pollution; and those which cannot be classified as blue.

Blue - farms that are unlikely to pollute.

Results up to the summer of 1990 showed that 465 farms out of the 6000 visited so far in their campaign had a red status, with a further 1333 having a green status.

NRA Severn-Trent Region undertook a major farm pollution risk survey in 1988, targeted at the major dairy areas of Shropshire, Staffordshire, Warwickshire and Leicestershire. One thousand and sixty-eight farms were visited, and each was given a series of grades ('satisfactory', 'doubtful' or 'unsatisfactory') on a number of different types of pollution risk (including land disposal, farm housing, drainage etc.). It was concluded that 39% of farms surveyed had unsatisfactory or dubious slurry systems, whilst 36% had unsatisfactory or dubious silage systems. Taking all types of pollution risk into account, only one third of farms were considered satisfactory.

NRA North West Region have recently undertaken targeted farm pollution risk surveys on the Weaver (1987), Brock (1987) and Gowy (1988) and Wampool catchments. In the Gowy catchment, pollution risks were divided into 9 categories (e.g. slurry stores, silage, land run-off, etc.) and each farm was classed as 'satisfactory' or 'unsatisfactory' in each category. Out of 157 farms visited, 84 (54%) are to be sent or have been sent letters requiring remedial action. Visible impacts on water quality in the Gowy catchment are mainly restricted to unclassified headwater streams. A number of tributaries in the Brock catchment vary between NWC class 1B and 4 due to the fluctuating impact of farm discharges. Two hundred and nine working farms were visited in the 1987 survey and were classed according to the NRA/MAFF Farm Waste Report categories for pollution incidents. Forty-two percent (87) of farms were producing some sort of discharge, with 26% having leaking silos, 24% discharging parlour/dairy washings, and 22% having overflowing storage/reception tanks. These 87 farms were sent letters requiring remedial action, and a total of 97 farms required follow-up visits within different timescales.

NRA Yorkshire Region has conducted recent risk surveys on the Wharfe, Wiske, Upper Derwent and Upper Dearne. The Wharfe survey consumed 3 man years of effort (6 staff each occupied for 6 months), and involved visits to 300 farms. It was found that one third of farms posed a serious pollution risk, whilst another third posed an intermediate risk.

NRA Northumbrian Region conducted a small catchment study of sheepdip locations and found high risks of pollution due to the close proximity of dippers to watercourses and the presence of plugged drains with direct access to the river. **NRA Welsh Region** still finds a high number of sheepdips with direct plugged access to adjacent watercourses. The risk of sheepdip pollution varies with the stipulations laid down by MAFF each year. In 1990, only one dipping period was required, meaning a halving of sheepdip usage and in many cases, where spent sheepdip is not emptied until the next dipping period, more pesticide degradation prior to disposal. **Tweed RPB** are currently engaged in a study of sheepdips in their region, and initial indications are that a relatively high proportion (*c* 40%) of dippers could cause pollution [this study has now been completed]. They advise that the dipper is emptied immediately after use and the spent dip is diluted and spread thinly to land. In reality dippers are invariably not emptied until the next dipping period. Disposal of sheepdip to land is now recommended by the majority of regions, except in aquifer protection zones. In areas where there is no risk to groundwater, some regions still allow soakaways.

Standard risk report forms are completed in 4 of the NRA regions (**Southern, South West, Welsh** and **Yorkshire**), and one RPB region (**Forth**), and in one NRA region (Chichester district of **Southern Region**) the information is subsequently entered onto a computer archive. **South West Region** keeps a computerised archive of farm details, including risk category. **Yorkshire Region** also maintains an archive of 6000 farms.

Discussion

Most farm pollution problems in surface waters occur on heavy soils where surface run-off is high; in dry conditions such soils crack and can give wastes easy access to watercourses via land drains. The high risk of contaminated run-off on such soils is compounded by sloping ground and high rainfall. High rainfall also puts greater strain on waste storage facilities if clean water is not separated or storage is not covered (which is often the case). Areas of high rainfall also suffer from fewer suitable opportunities to spread waste to land, meaning that waste has to be stored for longer periods, again leading to high risks of undersized storage facilities.

The above considerations of risk are all related to physical factors that can be quantified relatively easily from mapped information. Pollution problems arising include overflowing slurry stores and reception pits, yard run-off, and land run-off following land application. There are other less predictable risk factors, mentioned in Section 4.2.2, which can only be quantified through farm visits. These include quality of management, farm practices (e.g. yard washing), design and age of waste management systems, the presence of land drains, and the proximity of farm buildings and storage facilities to watercourse access. Such factors can give rise to the whole spectrum of farm pollution problems.

Land drains can short-circuit the normal and more difficult route of wastes into watercourses, and their presence is often unknown to pollution control staff. Under-drainage is frequently installed on heavy soils prone to waterlogging, where pollution risk is already high. A key consideration in pollution risk from sheepdip is dipper design, and in particular whether there is a drainage area that collects sheepdip dripping off fleeces and returns it to the dipping tank. Contractors using mobile dips are an increasing cause of concern in many areas, since they have the potential to dump spent sheepdip injudiciously and quickly move out of the area.

Certain methods of waste handling give cause for concern and have to be considered high pollution risks. The use of mobile dips above is one example, and third party arrangements for the disposal of pig slurry are a similar concern.

Approaches are being developed for a more objective assessment of farm pollution risk. The Potable Abstraction Risk Index (PARI) is to be further developed within the NRA R&D Programme (Reference NRA Project A17(90)1) for the assessment of point-source discharges, such as storage failure or sheepdip overflow/discharge. It considers the amount of pollutant likely to enter a river, the likely available dilution and highest acceptable environmental concentration of the pollutant. The index is given by:

$$I = \frac{W}{(A \times Q95 \times C)}$$

where: W is the weight of pollutant in kilograms,

Q95 is the 95 percentile exceedance flow of the river in m^3/s,

C is the highest acceptable environmental concentration,

and A is a factor which varies depending upon the likely duration of pollutant discharge (A = 3600 for a discharge lasting 1 hour).

The index can be modified by assigning weightings to factors affecting risk. These would include installation design, age, visible condition and capacity in relation to the storage volume required (particularly relevant for slurry storage).

For dispersed pollution sources, two methods for assessing the potential polluting load from land spread with animal slurry and sludge are being developed at WRc, so that areas of farmland can be classified as high or low risk with respect to the pollution of receiving waters (NRA Project Reference A17.007). One method uses basic land use characteristics, such as soil type, slope and distance from watercourse, together with a subjective ranking of the importance of each factor, to allow each land unit to be given a risk classification. This approach is ideally suited to an analysis using Geographical Information Systems (GISs), and the method has been applied to some catchments.

The alternative method being developed involves the formulation and solution of flow and pollutant transport equations in a water flow and quality simulation. This yields more objective results in terms of cause and effect than the method previously described, but has a considerable data requirement and would not readily be applied to many catchments. The best way forward would appear to be to combine the two approaches so that the simulation method is used to refine the GIS methodology, which may then be more widely applied. The estimation of the capacity of a farm to accept the slurry it produces is a key consideration in determining the risk of over-application. The combination of these two approaches offers a way of determining this parameter.

4.3.3 Identification of high risk areas

Data collation

In the present study, the collation of a complete national coverage of observed farm pollution problems, with which to compare formulated distributions of risk, was hampered by the difficulty in obtaining data from regional NRA archives.

Regarding pollution incidents, computerised archive systems are at various stages of development and sophistication around the NRA regions. **Thames** and **North West Regions** have highly interactive systems, with the former being menu-driven and the latter requiring some degree of computer literacy. They are both based on river catchments, which is useful for pollution prevention work. **Yorkshire** and **South West Regions** have good systems, but the former is perhaps not particularly interactive and the latter has some difficulty with some types of data extraction due to the historical 'extract' programmes inherited from South West Water Authority. All four of these regions were able to provide full lists of pollution incident data from 1988 and 1989. South West were, however, unable to supply severity grades due to the form in which the data were requested (i.e. individual incidents by national grid reference).

Welsh Region have recently commissioned a new pollution incident archive, but historical data is not available on the system. However, district pollution control staff managed to extract all of the required incident information from individual files.

Wessex Region have a pollution incident archive, but it was essentially designed for operational management in the water industry and was not useful for this data collection exercise. A new archive is being developed but it will be some time before it is operational. For this reason, the only information that could be gathered without great difficulty from Wessex concerned those incidents where formal samples were taken and a decision was taken centrally as to whether prosecution proceedings should be initiated. This amounted to just over half of the serious incidents recorded in the Wessex region for both 1988 and 1989, and can therefore only be taken as a very crude indication of the true distribution of incidents.

Northumbrian Region have recently set up a computerised archive based on that of Yorkshire Region, but was not operational in time to be of benefit to this study. However, regional staff manually extracted the data required from individual files. **Anglian Region** have also recently set up a regionwide archive, but again it was not operational in time for this study. Eastern Area managed to extract data on a catchment basis from incident files.

Severn-Trent Region have recently set up an incident archive, and **Southern Region** are in the process of doing so, but again these were not available in time for this study. It was deemed to be too time-consuming for staff in either region to extract the required information from individual files, meaning that no incident data have been collected from these regions.

With respect to NWC class downgradings attributable to farm pollution, only 4 regions (**Severn-Trent**, **Yorkshire**, **Welsh** and **Anglian**) provided data, whilst there were no known downgradings in **Northumbrian Region**. The data received are given in Appendix E.

Data analysis

All maps mentioned in this section have been overlayed with available pollution incident data from 1988.

i) Dairy cattle

Map 3a shows the distribution of organic waste production from dairy cattle in relation to the amount of land potentially available for disposal (i.e. the loading). The pattern is very similar to that in Map 2a, suggesting that the density of dairy cattle in relation to the area of land potentially available for waste disposal is relatively constant between catchments. Farm pollution incident data from 1988 are superimposed for comparison of pollution risk and observed impact.

Catchments with the highest waste loadings have loading rates in the region of 40 litres day^{-1} ha^{-1} of potentially available land. For an overwintering period of 4 months, the loading rate of stored undiluted waste on the available land would be 6.4 m^3 ha^{-1}, assuming that all waste is stored as slurry. If the slurry is diluted on a 1:1 ratio (this is normal practice for ease of handling) the loading rate would amount to 12.8 m^3 ha^{-1}.

Importantly, this loading rate assumes that all agricultural land within a catchment is available for disposal and would not pose a pollution risk. The major obstacles to such an assumption are that: the farmer wishing to dispose of slurry may not own the land that is potentially available for disposal; and that physical factors such as rainfall, soil permeability and landslope will not permit safe application on all potentially available land. Physical catchment characteristics will be considered later in the section.

The pollution incidents indicated on Map 3a are those attributable to cattle generally (i.e. dairy and beef) rather than dairy cattle as a whole. This means that comparisons with waste loadings derived purely from dairy cattle are not likely to yield good relationships. Also, it should be borne in mind that data for NRA Severn Trent and Anglian Regions are not available, and that regional differences in the way that data is archived means that data is not readily comparable between regions. Notwithstanding these considerations, the distribution of pollution incidents does broadly reflect the distribution of waste loading from dairy cattle.

In the Cheshire Plain area, pollution incidents are concentrated into catchments with high waste loadings. The most notably affected catchments are the upper **Weaver, Northersbury Brook** and the middle reaches of the **Dee,** with the majority of the incidents being caused by leaking slurry stores, silage liquor and farmyard drainage. Pollution from silage liquor is not dealt with in this risk analysis directly, but it can be seen that it follows the same pattern as cattle slurry problems.

The catchments to the south and east of the worst affected catchments have high dairy cattle waste loadings, but no pollution incident data is available for comparison since they fall within NRA Severn Trent Region. However, information on NWC class downgradings is available (Appendix E) for Severn Trent Region. This shows that none of the catchments with very high waste loadings (class 5 - **Blithe, Churnet, Tean** and the middle and lower reaches of the **Dove**) have any recently reported downgradings attributable to farm pollution. Of the catchments in the next highest waste loading category (class 4 - upper **Terne, Sow,** upper **Trent, Roden, Perry,** upper reaches of the **Severn, Manifold,** upper **Dove, Hilton Brook,** and **Ecclesbourne**), the **Roden** is the only catchment with reported downgradings (8.7 km, including Soulton Brook).

Further north, the middle reaches of the **Ribble** have a high number of pollution incidents, along with the lower reaches of the same river and middle reaches of the **Wyre**. The area first mentioned is in a high (class 4) waste loading class, and the other two areas are in the highest class (class 5). Again, those incident types featuring most are slurry store discharges, silage liquor leaks and farmyard drainage. On the Wyre, dairy problems combined with discharges from intensive piggeries have led to a downgrading in NWC class to class 3. Some catchments further away from the main concentration of waste loading in this area also have high pollution incident densities: the upper **Dearne** (class 4 waste loading), the middle and lower reaches of the **Calder** (only class 2 waste loading), the **Colne** (class 3) and the middle reaches of the **Nidd** (class 3) are examples of this. In contrast, some catchments with high waste loadings (class 4 or 5) in the main area of activity have low pollution incident densities: examples are the **Conder,** lower **Lune, Bela** and **Keer**. These catchments tend to be located along the coastal strip and the low number of incidents may be related to some ameliorating physical factor (such as reduced landslope); however, NRA North West Region have identified the **Lune** as a problem catchment, and the absence of recorded pollution incidents here may be due to gaps in the availability of national grid reference data for some incident records.

Further east, the **Wiske** has an intermediate waste loading but has no recorded pollution incidents for 1988; however, this river was downgraded in 1987 from NWC class 2 to class 4 due to dairy farming. It has since recovered after an intensive campaign by NRA Yorkshire Region, but 24.1 km are still downgraded (NWC class 2, Appendix E). In the same area, NRA Northumbrian Region have recorded high ammonia levels in the **Skerne** and **Leven** which may be due to dairy or beef cattle; these catchments both have low but significant waste loadings from dairy cattle (class 2).

In the northern Lake District, the **Wampool** and the **Waver** have the highest waste loadings but have relatively low pollution incident densities compared with the upper and middle reaches of the adjacent **Eden** catchment, which have intermediate-to-low waste loadings from dairy cattle.

In south west Wales, high pollution incident densities reflect high waste loading more accurately, with the highest number of incidents occurring in the **Taf** (class 5), **Cynin** and **Cywyn** (class 5), and the **Eastern** and **Western Cleddau** (both class 4). The majority of incidents are again due to either slurry stores, silage liquor or farmyard drainage. The Eastern and Western Cleddau also have recent reported NWC downgradings (see Appendix E) due to cattle (either beef or dairy): 14.6 km on the former (mainly on the **Syfynwy**) and 6.1 km on the latter (including the **Anghof**).

In Gwent and the southern Welsh border area there are catchments with high numbers of pollution incidents but relatively low waste loadings: these are the lower **Usk** (class 3 waste loading), **Monnow** (class 1), **Frome** (class 2) and lower **Lugg** (class 2). Recent NWC downgradings due either to beef or dairy cattle in the same area have been reported (see Appendix E) in: the lower **Usk** (Olway Brook, 8.0 km), the **Monnow** (Trothy, 12.9 km; Worm Brook, 16.2 km; Jury Brook, 5.8 km), lower **Wye** (Garren Brook, 13.8 km; Rudlace Brook, 4.0 km; How Caple Brook, 5.8 km), **Frome** (Lodon, 18.0 km; Hackley Brook, 5.0 km), lower **Lugg** (Withington Lakes, 5.2 km; Withington Marsh Brook, 6.8 km; Moreton Brook, 8.8 km; Wellington Brook, 4.8 km; Bodenham Brook, 6.1 km), upper **Lugg** (Main Ditch, 13.0 km; Ridgemoor Brook, 5.6) and the middle reaches of the **Wye** (Enig, 6.9 km). This represents a severe impact on minor tributaries in the lower and middle reaches of the main Wye catchment, and cannot readily be explained by the intensity of dairy farming alone.

In the south west of England, catchments in NRA Wessex Region show few incidents due to lack of data availability, as explained previously. Those incidents which are shown do not generally fall into the main concentration of waste loading (centred on the Lydden, middle Stour, Cale, Meels, Hartlake and Sheepey), but a number fall into catchments to the north east of this area in catchments with intermediate loadings (e.g. upper and middle reaches of the **Bristol Avon**). NRA Wessex Region identified catchments such as the **Yeo, Brue, Cary, Somerset Frome** and **Bristol Avon** as having the worst problems, mainly from dairy farming. With the exception of the Bristol Avon, all of these catchments have high waste loadings from dairy cattle (class 4 - 30 to 40 litres day^{-1} ha^{-1}).

Further west in NRA South West Region, incidents are relatively evenly spread over all catchments with intermediate to high waste loadings, with reduced numbers in catchments with low (class 1 or 2) loadings. Numbers of incidents occurring in areas with high loadings (class 4 or 5) do not appear to be any greater than those occurring where loadings are intermediate (class 3). The three main causes of incidents throughout the area are again slurry stores, silage liquor and farmyard drainage.

In the south east of England, the **Mole** has a high incident density, with an intermediate loading of dairy cattle waste. Other catchments with intermediate waste loadings in this area fall within NRA Southern Region, for which no pollution incident data are available. Southern Region report that the majority of farm problems are caused by dairy farms on clay, including the **Uck,** and upper **Medway** (**Eden, Beult** and **Teise**). The Uck has one of the highest waste loadings in the region (class 3) whilst the upper Medway tributaries fall within class 1 or 2.

ii) Beef cattle

As with dairy cattle, the general distribution of waste loading from beef cattle (Map 3b) is similar to the distribution of livestock density. However, the situation with beef cattle in NRA South West Region appears more acute in a number of catchments when viewed in terms of waste loading: the **Bovey, Tavy, Walkham** and **Fal** all have intermediate beef cattle densities (class 3), but are all in the highest waste loading category (class 5). A number of adjacent catchments have changed from class 3 in terms of stocking density to waste loading class 4. Interestingly, this area appears not to suffer greatly from cattle-related pollution incidents, unlike nearly all other areas of NRA South West Region.

The area of activity in the Scottish border region is also more important when viewed in terms of waste loading. Again, although waste loading is high in a number of catchments, few cattle-related pollution incidents have been reported. Catchments in class 4 (20-30 litre day^{-1} ha^{-1}) in this area include the **Wampool, Waver** and **Esk** in NRA North West Region, and the lower reaches of the **North** and **South Tyne** in NRA Northumbrian Region. No cattle-related pollution incidents were reported in 1988 in any Northumbrian catchment with a class 4 designation; high ammonia levels in the **Skerne** and **Leven** are possibly caused by dairy and/or beef cattle, as mentioned previously, but these catchments have only intermediate beef cattle waste loadings.

Generally, the relationship between beef cattle waste loading and pollution incident distribution is poor, in contrast to the situation with dairy cattle. Beef cattle waste loading around the Cheshire Plain is low (class 2 in most catchments), but there is a high concentration of incidents in this area. Similarly, catchments with a high waste loading (class 4) in the Welsh border area (middle reaches of the **Wye** and **Usk**) have few or no recorded cattle-related incidents (no data is available for adjacent Severn Trent catchments - the **Clun,** upper **Teme, Rhiw** and parts of the upper **Severn**). However, NWC downgradings of minor tributaries are extensive in the southern Welsh border area (see above discussion on dairy cattle), where waste loadings are intermediate or high (lower and middle **Wye, Monnow,** lower **Usk,** lower and upper **Lugg, Frome**).

It would appear from these observations that dairy cattle farming is far more important than beef cattle farming as a cause of pollution incidents. This may be due to a tendency to house beef cattle in deep litter, which presents a far lower pollution risk than slurry.

There are a number of catchments with high waste loadings in the Solway and Perthshire regions of Scotland, generally reflecting high livestock densities rather than a low catchment availability of land for disposal. Pollution incident data are not available from these areas for comparison with the distribution of waste loading.

iii) Pigs

Map 3c shows the distribution of waste loading from pigs. Again, the distribution of waste loading is similar to the livestock density distribution, with highest loadings apparent in clusters of catchments in East Anglia, Yorkshire, and to a lesser extent Lancashire and the Thames Valley. It is immediately apparent that waste loadings to land potentially available for disposal are very high in catchments in the main hotspots of activity, up to 75-100 litres day^{-1} ha^{-1}. One catchment in NRA Wessex Region has a loading of more than 100 l day^{-1} ha^{-1} (**upper Frome**).

Such high loadings are particularly problematic since the majority of pigs are intensively reared and thus housed throughout the year. This results in the entire annual production of waste passing through the farm's waste management system of storage and disposal. For catchments with a loading of 100 litres day^{-1} ha^{-1}, this means that 36.5 m^3 ha^{-1} of fresh undiluted pig waste has to be spread to all agricultural land within the catchment each year. Assuming all waste is produced as a slurry and allowing for a typical 1:1 dilution, this amounts to a waste loading of 73 m^3 ha^{-1} yr^{-1}. Again, this figure assumes that all potentially available land is suitable for spreading, which will certainly not be the case, and is also available to the farmer with slurry to be disposed of. This latter assumption is particularly inappropriate for intensive pig farming, as the amount of land attached to piggeries is invariably small in relation to the livestock holding.

In East Anglia, catchments in waste loading class 4 (75-100 litres day^{-1} ha^{-1}) are the same as those in the highest pig density class from Map 2c (**Dove** and **Chickering Beck,** and parts of the **Deben** and lower **Wissey** catchments). Catchments with class 4 loadings (50-75 litres day^{-1} ha^{-1}) again follow high pig densities: the upper and middle reaches of the **Waveney,** the **Thet, Tas,** upper reaches of the **Little Ouse, Blyth** and **Walpole,** upper **Alde** and **Ore, Deben, Gipping,** and to the north the upper Bure. Mapped pollution incident data are not available for these catchments for visual comparison, but data have been supplied by NRA Anglian Region in terms of the number of incidents occurring in each major catchment (not the catchments used in this study). In 1988, the **Waveney** suffered 8 pig-related incidents (none serious), whilst the **Dove** and **Chickering Beck** suffered 9 (2 serious). The majority of these incidents were due to leaking slurry stores and land-run-off in equal measure. The frequency of incidents in these catchments would appear higher than in any other catchments on Map 3c, although the proportion of serious incidence is higher elsewhere. The Gipping and Deben suffered 3 and 4 pig-related incidents in 1988 respectively (all minor). In terms of recent NWC downgradings, the **Dove** and **Waveney** catchments have a combined total of 13 km downgraded to class 3.

In Yorkshire, the lower **Calder** and **Spurn Head** have the highest waste loadings (class 4, 75-100 litres day ha^{-1}), and again the pattern generally follows that of livestock distribution (Map 2c). Few pig-related pollution incidents are evident within this area of activity, but it should be noted that nearly all of these are classed as serious. The majority of incidents occurred in catchments with either class 3 or 4 waste loadings.

In Lancashire, the middle and lower reaches of the **Wyre** have waste loadings of between 50 and 75 litres day^{-1} ha^{-1} (class 3), as does the adjacent tidal **Ribble**. Only 2 pig-related pollution incidents in the Wyre were reported in 1988 (both serious), but much of the Wyre is downgraded to NWC class 3 due mainly to pollution associated with pig farming. To the south east, high waste loadings (class 4) are also evident in the Medlock.

In the Thames Valley, the **Pang** and **Sulham, Blackwater, Bourne, Chertsey Bourne** and **Crane** have intermediate waste loadings (class 3). If this is compared to livestock distribution (Map 2c), it can be seen that for all of these catchments except the Pang and Sulham these very significant waste loadings are due to a relatively low availability of land for disposal. The majority of pigs in the Thames Valley are kept further upstream (Map 2c), where waste loadings are somewhat lower (generally class 2). It can be seen from Map 3c that in this area no pig-related pollution incidents were reported in 1988. The low frequency of incidents is largely due to the practice of 'free range' rearing, which is predominant in NRA Thames Region.

It should be noted that any catchment in Map 3c falling into any class other than class 1 has a very significant volumetric waste loading on a catchment-wide basis. In volumetric terms, the bottom of pig waste loading class 2 is equivalent to the middle of dairy cattle class 3 or the middle of beef cattle class 4.

iv) Poultry

Map 3d shows the distribution of volumetric waste loading due to poultry. It can immediately be seen that the volumes of waste involved are considerably lower, than for cattle or pigs, but it should be remembered that the polluting strength of this waste is considerably greater. The distribution of waste loading again largely reflects livestock distribution (Map 2d), showing a scattered distribution but with clusters of catchments with high waste loadings in East Anglia, Lincolnshire and Lancashire and elsewhere to a lesser extent. Poultry farming does not give rise to many pollution incidents, and this is evident from Map 3d; however, it should be remembered that mapped incident data are not available for some NRA regions containing clusters of catchments with high waste loadings (NRA Severn Trent, Anglian and Southern Regions).

In Lancashire, pollution incident data are available from NRA North West Region, and a number of poultry-related incidents are evident in the main area of activity (waste loading classes 3 - 5). In East Anglia, information supplied on a catchment basis (not shown on Map 3d) indicates that only 2 incidents were reported in 1988 in the main area of activity, one on the Blyth and one on the Deben.

v) Total volumetric waste loading

Map 3e shows the combined volumetric waste loading from dairy and beef cattle, pigs and poultry by river catchment. It is immediately evident that total loading is highly concentrated into relatively small areas. The highest total loadings (>60 litres da^{-1} ha^{-1}) are found in clusters of catchments in the south west of England (Dorset, Somerset and east Devon), the Cheshire Plain and Lancashire. Clusters of catchments with generally intermediate loadings are found in south west Wales, south west England (Cornwall and west Devon) and Yorkshire, whilst catchments with lower but still very significant loadings are evident in East Anglia, south east England and south west Scotland.

Fortunately, the combination of high pig and poultry waste loadings with high dairy and beef waste loadings is rare, and for this reason total waste loadings are not as high as might have been the case. The most significant mixing of cattle farming with pig and poultry farming is in Lancashire, centred on the **Wyre** and lower **Ribble** catchments. As can be seen, the distribution of all farm pollution incidents generally fits the distribution of total waste loading well.

vi) Total BOD loading

Map 4a shows the distribution of waste loading to agricultural land in terms of BOD content, and the influence of poultry waste, and to a lesser extent pig waste, is immediately evident (see Maps 3c and 3d). The emphasis has largely moved away from the west of the U.K. towards the east, especially East Anglia, Lincolnshire and Yorkshire, with south west Wales and much of Devon and Cornwall hardly rising above the lowest loading class. Catchments around the Forth Estuary in Scotland also become very important. However, the cluster of catchments in Lancashire centred on the **Wyre** and lower **Ribble** maintains its high waste loadings.

Since considering waste loading in terms of the land area available for disposal can lead to an over-importance of essentially non-agricultural catchments with a small amount of intensive agriculture, BOD production with respect to the entire catchment area was also considered (Map 4b). Although there are a number of catchments which change class relative to each other, changes are generally restricted to adjacent classes (i.e. 3 to 4 or 4 to 5) and the general pattern is similar to that in Map 4a.

It is evident that this distribution of waste loading does not fit the pattern of occurrence of pollution incidents as well as when waste loading was expressed in terms of volume (Map 3e). This is largely due to the very low occurrence of poultry-related incidents, which highlights the importance of the potential of a pollutant to gain access to watercourses. Poultry waste is highly concentrated but has a relatively high solids content, making it comparatively immobile; slurry, although not as high in polluting strength, is very mobile and gains access to watercourses very easily.

vii) Total nitrogen loading

Map 4c shows the distribution of total nitrogen loading due to cattle, pigs and poultry. Catchments with highest loadings are scattered, but broadly match those catchments with high poultry waste loadings. Highest loadings are in excess of 400 g N day^{-1} ha^{-1}, which over a year equates to 146 kg N ha^{-1} if all waste is disposed of on available agricultural land within the catchment. Some states in Germany impose nitrogen restrictions (Table 3.10), with the range in maximum application rates for the 4 States considered being 160-210 kg total N yr^{-1}. Highest loading rates in the U.K. are comparable to this range without accounting for the proportion of agricultural land that is not available for waste disposal due to ownership or physical characteristics.

viii) Total phosphate loading

Map 4c shows the distribution of phosphate loading from cattle, pigs and poultry to agricultural land. As with nitrogen, this follows the distribution of poultry and pig waste loadings much more closely than loadings due to cattle, owing to the concentrated nature of the former. Catchments with the highest (Class 5) loadings, such as the middle **Waveney, Tas, Dove** and **Chickering Beck** in East Anglia and the middle **Wyre, Darwen,** lower **Douglas** and **Lostock** in north west England, take more than 400 g P205 day^{-1} ha^{-1}. If it is assumed that all of this is disposed of within the catchment, this amounts to an annual loading of 146 kg P205 ha^{-1} - again, the actual loading is likely to be higher since not all agricultural land will be suitable or available for disposal.

This loading can be compared to standards being adopted by other European countries (see Section 3.7). Of the two German states in Table 4.10 which lay down phosphate restrictions, Schleswig-Holstein is the more stringent with a maximum allowable loading of 120 kg P205 ha^{-1} (2 Dung Units), although a greater loading would be acceptable if the total nitrogen loading did not exceed 160 kg N yr^{-1}. The Netherlands, although allowing very high phosphate loadings at present, are working towards maximum loadings of around 125 kg P205 ha^{-1}. Denmark allows a maximum loading of 2.3 Livestock Units ha^{-1} $year^{-1}$ for dairy cattle, which equates to roughly 67 kg P205 ha^{-1} $year^{-1}$. From these comparisons it would appear that a number of U.K. catchments exceed the loading rates that would be acceptable in certain areas of the continent, and that a significant number of other catchments are also likely to do so. Although the risk of phosphate enrichment of surface waters and subsequent phytostimulation will depend upon soil mobility and bioavailability of this phosphorus, there is a clear risk of significant river contamination.

ix) Sheepdip waste loading

Map 5a shows the estimated volumetric loadings of waste sheepdip by river catchment (in terms of total catchment area), based on assumptions made by Littlejohn and Melvin (1989). It can be seen that catchments in the Welsh uplands suffer loadings of over 48 litres ha^{-1} of catchment from one dipping period alone. If all sheepdip used in these catchments was propetamphos (a very widely used compound), Map 5b shows that these loadings could amount to up to 24 grammes of propetamphos ha^{-1} of catchment. However, without detailed knowledge of disposal practices, degradation rates, transport processes and dilution rates it is not possible to sensibly predict likely riverine concentrations. Pesticide transport is being modelled by WRc (NRA Project Reference A14.002) and a review of sheepdip practices has also recently been undertaken (NRA Project Reference A07.2) - these areas of research, in conjunction with the kind of estimates made in this study, may well allow such predictions.

Reported sheep-related pollution incidents in 1988 are shown in Maps 5a and 5b. The scarcity of such incidents is very unlikely to reflect the true impact of sheepdip contamination on river biota, but more likely to reflect the insidious nature of pesticide impact and the general remoteness of sheep farming. It should be noted that the relatively low estimated waste sheepdip loadings evident in Scotland should not be considered to represent a lack of pollution risk. Sheep farming is a very important land use in much of Scotland and, as discussed in Section 4.3.2, Tweed RPB have found that a high proportion of dippers in their area are likely to cause pollution (personal communication Ian Currie, Tweed RPB).

x) Physical factors

Maps 6a, 6b and 6c show the U.K. distribution of land-slope, annual rainfall and winter rainfall acceptance potential (WRAP) by river catchment. The westerly and northerly distribution of high land-slope is evident from Map 6a, and a similar distribution of high annual rainfall is apparent from Map 6b. Map 6c highlights the low permeability peat soils of upland areas to the west and north, and also clay soils in lowland areas such as the Weald (south east

England). The distributions of these three factors, all important contributors to farm pollution risk, have been used in combination to produce a broad assessment of the vulnerability of catchments to surface run-off (Map 6d). As can be seen, there is a strong northerly and westerly component to this vulnerability.

xi) Organic pollution risk

The combination of total waste loading and surface run-off potential, into a single measure of organic pollution risk (Figure 4.7), clearly highlights the coincidence of high waste loadings with high run-off potentials. On the Cheshire Plain, the lower run-off potentials are compensated by the very high waste loadings, giving catchments in the area a high pollution risk. In other hotspots, such as South West Wales, waste loadings are not quite as high but a higher run-off potential gives them a similar pollution risk to the Cheshire Plain. The pollution incident data available match the distribution of pollution risk quite closely.

xii) Sheepdip pollution risk

Map 7c shows the combination of waste sheepdip loading and physical factors to produce a distribution of sheepdip pollution risk for surface waters. This shows the majority of Welsh catchments having the highest pollution risk, combining high sheep densities with high rainfall, high landslopes and low soil permeability. Much of northern England (particularly to the west), southern Scotland and south west England also show relatively high pollution risk.

5. DEVELOPMENT OF RAPID APPRAISAL METHODS

5.1 An Indicator system for West Wales (Spring)

5.1.1 Introduction

The effective control of surface water pollution from the dairy and beef industries is hampered by the large number of farms involved and their distribution over large areas with poor accessibility. In the U.K. in 1989, dairy farming was practised at 56 300 holdings and beef farming at 72 400 (MAFF, 1990). There is therefore a need to prioritize and target pollution control effort, and the detection of impacted streams using a rapid biological assessment has been proposed as part of a pro-active strategy (Seager, Jones & Rutt, 1992).

Biological methods based on the benthic macroinvertebrate fauna are particularly appropriate because they have a proven ability to detect the effects of organic pollution, both chronic (e.g. Hynes, 1960; Hawkes, 1963; Pearson & Penridge, 1987) and episodic (Seager & Abrahams, 1990; Willemsen *et al.*, 1990). Intermittent pollution events typically generated by rainfall can be a feature of agricultural catchments (Schofield, *et al.*, 1989).

A variety of biotic indices have been developed in Europe for use in the assessment of organic pollution (Metcalfe, 1989) but these can be time-consuming to apply and are generally inappropriate for accurate use in the field. Systems based upon indicator taxa derived from TWINSPAN classification (Hill, 1979a) have the advantage of requiring identification and enumeration of fewer taxa and produce a simple site grading. The potential value of indicator systems has been demonstrated in studies of acidification in upland streams (Wade, *et al.*, 1989; Rutt, *et al.*, 1990a).

West Wales was chosen as a region in which to develop a protocol which could be adopted nationally because of the intensity of dairy farming in the area and its potential impact upon valuable salmonid spawning streams (Howells &

Merriman, 1986). These streams are naturally homogeneous in both physiography and chemistry, such that observed differences in fauna are likely to reflect the impact of pollution rather than natural variability. This section describes the development of a rapid biological assessment system based upon TWINSPAN classification, for the detection of farm pollution in the field in winter and early spring. This period was chosen because farmers commonly experience their most severe waste management problems during wet winter weather when stock is housed rather than on pasture. Earlier work by Reynolds (1989) had developed a system for use in the summer months when pollution from silage effluent tends to predominate.

5.1.2 Study Area

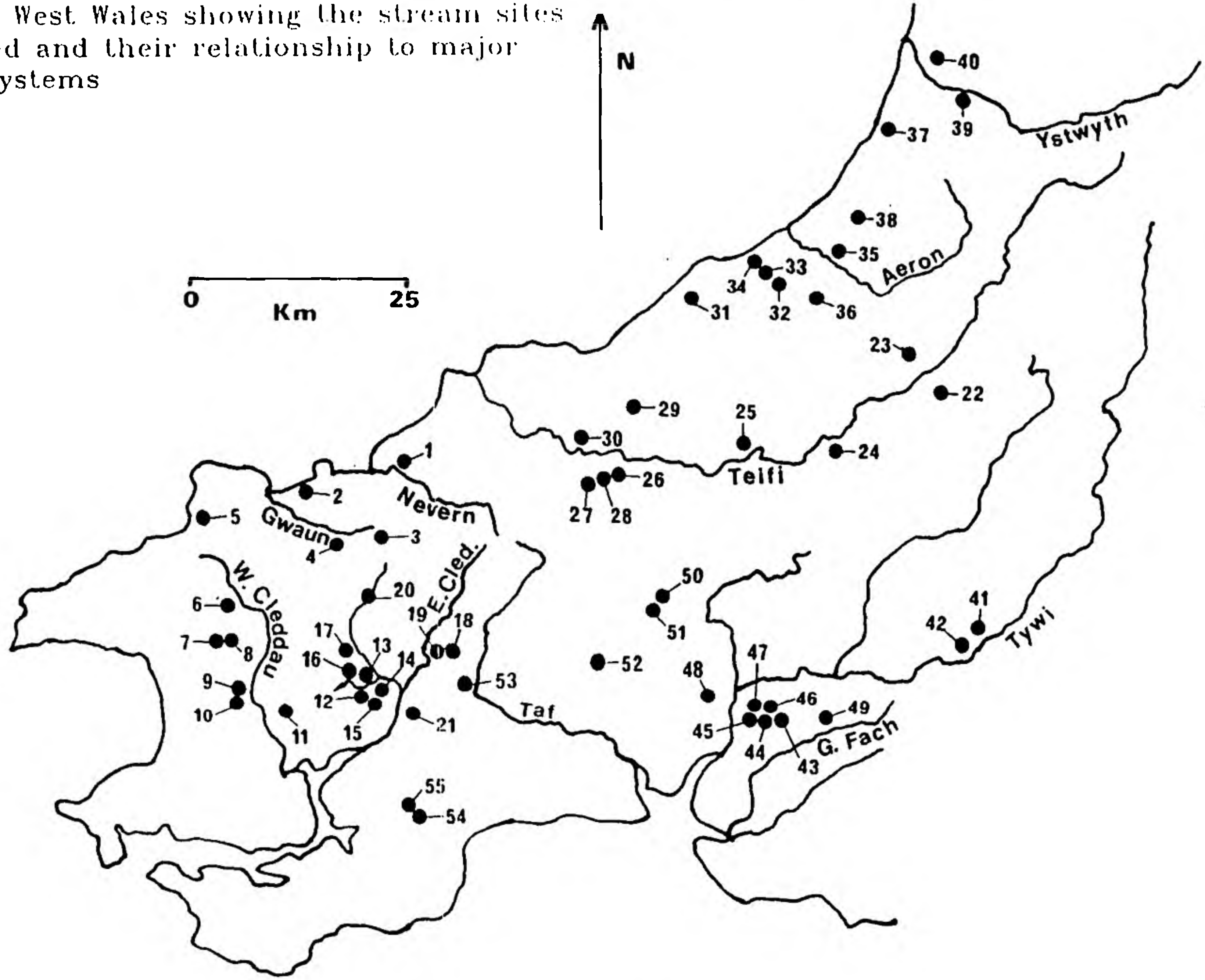
The fieldwork for this study took place in the county of Dyfed, West Wales (Figure 5.1). The majority of the land surface is gently undulating with few areas rising above 400 m O.D. It is underlain principally by Palaeozoic sediments mainly of Ordovician and Silurian age with some Cambrian, Devonian and Carboniferous rocks in the extreme south and west. Annual rainfall is relatively high for the U.K. (1200-1600 mm) leading to lush growth of pastures and the bulk of the land area is devoted to livestock production. Dairy and beef farming predominate with some sheep rearing on the higher less fertile soils. There is a high density of rivers in the area including the Tywi and Teifi which support nationally important stocks of migratory salmonids.

5.1.3 Methods

Site selection

After discussion with NRA pollution control staff, 55 sites on streams in West Wales were selected for survey (Figure 5.1). Sites were chosen to include a wide range of pollution impact, pollutant type (e.g. parlour washings, lagoon overflow, silage effluent) and geographical area. Sites were restricted to those likely to support populations of salmonids and unlikely to be affected by other sources of pollution found in rural catchments e.g. sewage treatment

Figure 5.1 Map of West Wales showing the stream sites sampled and their relationship to major river systems



works. Streams were mostly of first, second or third order (Strahler, 1957). Stream widths were in the range 1.0-6.0 m, most sites being between 2.0 and 4.5 m.

Fieldwork

Fieldwork was carried out between 27 February and 6 April 1990. The following procedure was used at each site:

1. A spot water sample was taken which was later analysed for a range of standard 'sanitary' determinands such as dissolved oxygen, Biochemical Oxygen Demand (BOD), inorganic nutrients and suspended solids (Data in Appendix F1).

2. The occurrence of microbial heterotrophic growth known as 'sewage fungus' (Curtis, 1969) was recorded and percentage cover both above and below large stones noted. Small representative samples of the growth were collected for microscopic examination (Data in Appendix F2).

3. Substratum composition and width were recorded (Data in Appendix F3).

4. Benthic macroinvertebrates were collected by kick sampling (Furse *et al.*, 1981) in riffles using a net with 1.0 mm mesh and 230x255 mm frame. Three separate one minute kick samples were taken such that the relative merits of 1, 2 and 3 minute sampling periods for use in the development of an indicator key could be assessed. Samples were preserved at the laboratory in 4% formaldehyde solution within two to eight hours of collection.

5. Lengths of stream in the range 30-60 m were electrofished semi-quantitatively to assess the populations of trout (*Salmo trutta*) and salmon (*Salmo salar*). One run was employed and fishing was carried out between riffles without the use of stop nets. Fish were identified and measured before being returned to the stream. All age classes were pooled to give a minimum total trout density calculated per 100 m^2. Minimum population estimates derived in this way have been found to

correlate well with the results of quantitative sampling based on catch-depletion (Strange, *et al.,* 1989). In July 1990 habitat information was collected to enable the calculation of predicted salmonid densities which could be compared with the observed populations using the HABSCORE model developed by NRA Welsh Region (Milner & Wyatt, 1991). Raw data are presented as Appendix F4.

Invertebrate sample processing

Macroinvertebrate samples were sorted and identified in the laboratory. Two of the three sets of 55 replicate one-minute samples, were identified to species or genus level except for difficult taxa such as chironomids and simuliids which were left at family level. The third set was identified to family level only. Two biotic indices commonly used in the United Kingdom, the Biological Monitoring Working Party (BMWP) Score and the Average (BMWP) Score Per Taxon (ASPT), were employed. The BMWP system allots a score in the range of 1 to 10 to a variety of common invertebrate families according to their tolerance of organic pollution. A quality assessment is derived by totalling the scores for all taxa present: high scores indicate good quality (Metcalfe, 1989). ASPT is derived by dividing the BMWP score by the number of scoring taxa which may reduce bias due to variation in stream size and sampling method (Armitage *et al.,* 1983). The two indices were evaluated for 1,2 and 3 minute sampling by appropriate combination of data from the three separate samples. Invertebrate data are available as Appendix F5)

Data derived from maps

Altitude, catchment area, distance from source and stream gradient for each site were estimated from Ordnance Survey 1:50,000 scale maps (Data in Appendix F3).

Data analysis

The invertebrate samples from the 55 sites were classified using a multivariate technique known as TWINSPAN (Hill 1979a) which has been widely used in freshwater ecology (e.g. Wright *et al.,* 1984; Ormerod *et al.,* 1987). Four separate TWINSPAN analyses were run with two different sampling periods - one minute and two minute - and with taxa grouped at two different taxonomic levels - 'species' and 'family' (Table 5.1). At both 'species' and 'family' level certain taxa were grouped at higher taxonomic levels. Percentage sewage fungus cover above stones was included in all four data sets. TWINSPAN generates an indicator key which can be used to classify new sites into the derived groups. Where TWINSPAN is run using quantitative data, cut levels are specified such that different abundance categories of a taxon are entered separately into the analysis as 'pseudospecies.' In this case 'pseudospecies' were set to be logarithmic abundance categories i.e. 1-9, 10-99, 100-999, etc. A TWINSPAN option was invoked which prevented abundance category 1 'pseudospecies' (1-9 individuals) appearing as indicators. This can simplify the keys and reduce the possibility of chance occurrence of single individuals leading to misclassification when the derived keys are applied in practice.

The data sets were ordinated by DECORANA (Hill 1979b) which gave a spatial representation of site affinities.

It was necessary to determine whether the derived TWINSPAN groups reflected different degrees of organic pollution as opposed to other factors which might determine fauna. Due to logistic and financial considerations water quality information on the selected sites was restricted to the single spot samples taken during survey work. Data obtained by continuous monitoring would have been the ideal due to the episodic nature of much pollution from farms. However, between-group differences in the means of a range of biotic and abiotic variables were tested by analysis of variance (Sokal & Rohlf, 1981) after transformation where necessary to normalize distributions. Substratum composition data was converted to a single variable (phi) using the methods of Wright *et al.* (1984). A trout Habitat Utilization Index (HUI) derived from HABSCORE was another of the variables examined. By comparing the significance levels of the tests between TWINSPAN analyses (SPP1, SPP2, FAM1 and FAM2) and taking into account the relative ease of identification of the indicators, it was possible to determine which of the four keys would be most effective in distinguishing different degrees of organic pollution.

Once the optimal key had been selected, the distribution of different invertebrate taxa between the stream groups was investigated using Chi-squared (Chi^2) tests (Sokal & Rohlf, 1981). Analyses were carried out at different levels of

Table 5.1 Data sets classified using TWINSPAN. Number of taxa includes percentage cover of sewage fungus entered as a separate variable

Data set	Sampling period (minutes)	Taxonomic level	No. of taxa
SPP1	1	'species'	81
FAM1	1	'family'	56
SPP2	2	'species'	92
FAM2	2	'family'	56

abundance e.g. *Baetis* spp. (1 or more individuals in a one-minute kick sample), *Baetis* spp. (2) (10 or more individuals) and *Baetis* spp. (3) (100 or more individuals).

Pollution Sources

Between 24 April 1990 and 17 May 1990, the majority of the streams found to be affected by pollution were investigated in order to identify the likely sources. Further information was obtained from NRA Pollution Control staff. The ideal approach would have been visits to all farms upstream of all fifty-five sites to accurately assess polluting discharges. However, resources were not available for such a major undertaking.

5.1.4 Results

The number of macroinvertebrate taxa entered into the TWINSPAN analyses varied from 56 for the 'family' level data sets FAM1 and FAM2 to 92 for data set SPP2 (Table 5.1). Sewage fungus communities were of variable composition but were most frequently dominated by the colonial bacterium, *Sphaerotilus natans*, with the phycomycete, *Leptomitus lacteus*, and the sessile protozoan *Charchesium* often abundant (Appendix F2). Whatever the actual composition of the growths, provided the constituents were essentially heterotroph, percentage cover (above stones) was entered into the analysis as an extra variable.

Initial TWINSPAN analyses produced indicator keys which incorporated taxa not readily identifiable in the field. These taxa were masked out as potential indicators and subsequent re-analysis resulted in more practical versions one of which is illustrated (Figure 5.2). Taxa to the left of each division line score -1, taxa to the right score +1 - the net total for a site indicates whether it will be placed to the left or right of the division. The even pattern of division indicates that there are no marked outlier sites in the data set.

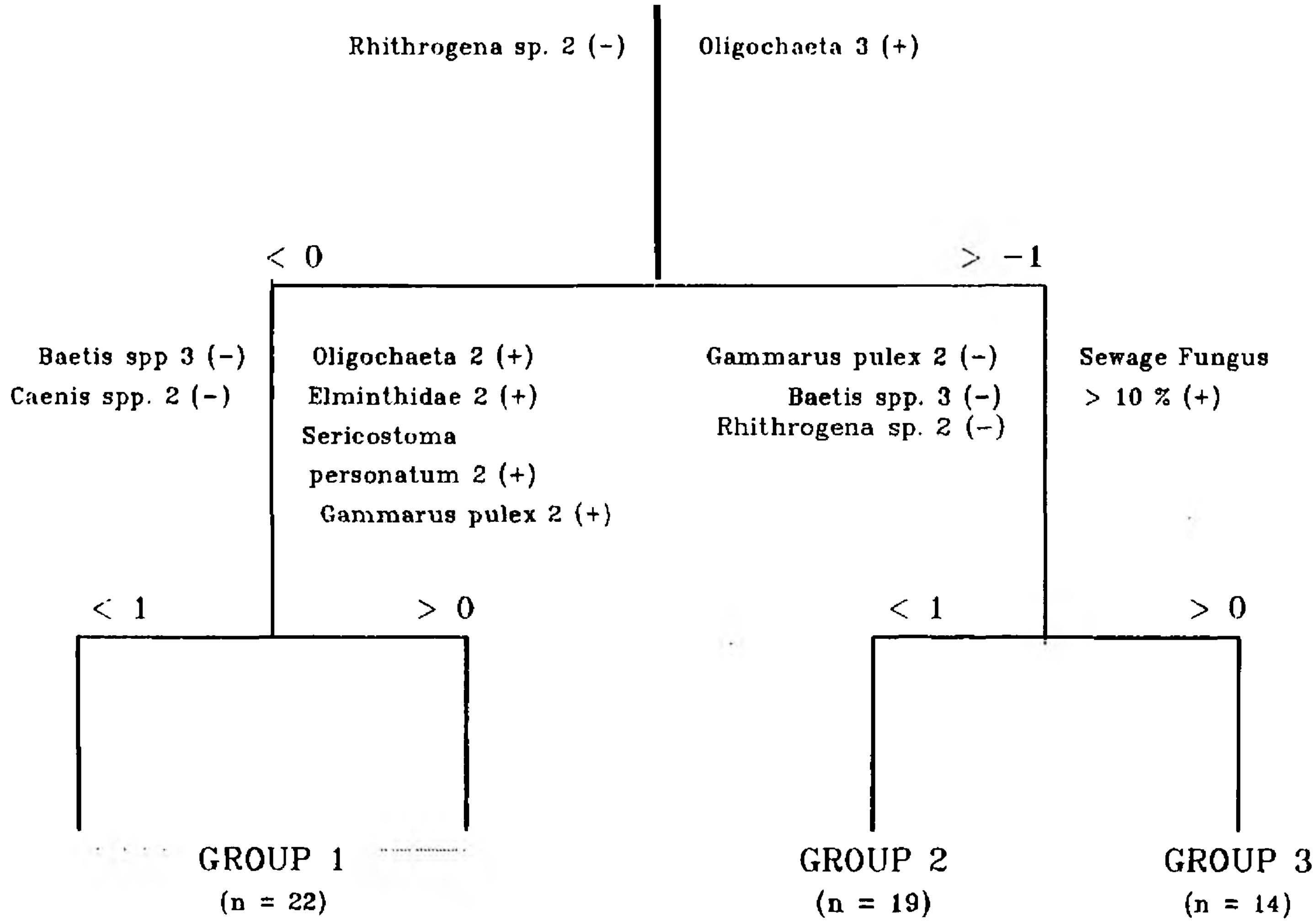
Preliminary assessment of the stream groups indicated that in each key the two site groups to the left of the initial division were very similar in biotic and physicochemical terms. This was illustrated by the marked overlap of these groups when projected on the first two axes of DECORANA ordination space (e.g. Figure 5.3) which in the case of SPP1 accounted for 82% of the variance in the data set. These pairs of groups were thus combined so as to produce three groups (1,2 and 3 in Figure 5.2).

For all four TWINSPAN classifications (SPP1, FAM1, SPP2, and FAM2) there were very significant differences ($p < 0.001$) between the three groups in a number of variables which might be considered to reflect the effects of farm pollution i.e. BMWP score, ASPT, ammoniacal Nitrogen concentration and minimum total trout density (Table 5.2). There were less significant differences in dissolved oxygen concentration and BOD, with no significant difference in suspended solids.

There were also significant differences in a number of variables not directly reflecting agricultural pollution i.e. distance from source, conductivity, pH and total hardness (Table 5.2) but there were no significant differences in altitude, slope and width.

There was thus little to choose between the four analyses in their ability to discriminate between different levels of pollution (Table 5.3). This was reflected in the consistency with which the four analyses allocated sites to the groups. Across the four analyses, 37 sites (67 %) remained entirely

Figure 5.2 TWINSPAN indicator key derived from the biological analysis of 55 sites in West Wales (SPP1 dataset).



Numerals after the taxon names indicate log abundance category
(1 = 1-9, 2 = 10-99, 3 = 100-999)

faithful to one of the three groups whilst 11 sites (20 %) moved between the two polluted groups (Groups 2 and 3). Only 6 sites (10.9 %) moved between Groups 1 and 2 whilst only a single site moved between the two extremes (Groups 1 and 3).

This consistency meant that the simplest and easiest key could be selected which was that derived from analysis of one minute samples processed to 'species' level - SPP1 (Figure 5.2). Sites in Group 1 only had sewage fungus in two cases, generally had BMWP scores (one minute samples) in excess of 100, ASPT values greater than 6.1 and supported healthy populations of trout (Tables 5.3, 5.4 , Figure 5.4). Sites in Group 3 generally had substantial growths of sewage fungus, BMWP scores less than 100, ASPT less than 6.0 and very small or non-existent trout populations. Group 2 sites tended to be intermediate (Tables 5.3, 5.4, Figure 5.4).

Table 5.2 Relationships between TWINSPAN groups from the four analyses and biotic and environmental variables

Variable	Transformation	SPP1	FAM1	SPP2	FAM2
Pollution dependent variables					
Dissolved Oxygen	-	**	**	**	***
BOD5	-	*	**	**	N.S.
Ammoniacal nitrogen	log	***	***	**	**
Suspended solids	log	N.S.	N.S.	N.S.	N.S.
BMWP-1mk	-	***	***	***	***
BMWP-2mk	-	***	***	***	***
BMWP-3mk	-	***	***	***	***
ASPT-1mk	-	***	***	***	***
ASPT-2mk	-	***	***	***	***
ASPT-3mk	-	***	***	***	***
Min. Trout density	log	***	***	***	***
Pollution independent variables					
Altitude	-	N.S.	N.S.	N.S.	N.S.
Slope	-	N.S.	N.S.	N.S.	N.S.
Width	-	N.S.	N.S.	N.S.	N.S.
Dist. from source	-	*	*	**	**
Catchment Area	-	*	*	**	N.S.
pH	-	***	**	**	***
Conductivity	-	***	**	***	*
Total Hardness	-	***	***	***	N.S.

Asterisks indicate probability level from analysis of variance - * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.
N.B. Chemical data are based upon the results of a single spot sample.

Table 5.3 **Relationships between TWINSPAN groups from data set SPP1 and biotic and environmental variables**

Variable (units)	Group 1 (n=22)	Group 2 (n=19)	Group 3 (n=14)	F	p
Pollution dependent variables					
BMWP-1mk	127	102	64	23.95	< 0.001***
	(101-154)	(75-130)	(38-91)		
BMWP-2mk	150	122	88	29.72	< 0.001***
	(132-168)	(97-148)	(61-116)		
BMWP-3mk	157	134	102	19.82	< 0.001***
	(136-179)	(109-146)	(66-135)		
ASPT-1mk	6.3	5.6	5.4	17.64	< 0.001***
	(6.1-6.6)	(5.1-6.0)	(4.7-6.2)		
ASPT-2mk	6.4	5.7	5.7	22.01	< 0.001***
	(6.2-6.7)	(5.3-6.1)	(5.2-6.2)		
ASPT-3mk	6.4	5.8	5.9	16.60	< 0.001***
	(6.2-6.7)	(5.4-6.2)	(5.4-6.4)		
Oxygen (mgl-1)	11.0	10.7	10.0	5.68	0.006**
	(9.9-12.2)	(10.1-11.3)	(9.2-10.8)		
B.O.D. (mgl-1)	0.9	1.5	1.4	4.42	0.017*
	(0.2-1.7)	(0.8-2.1)	(1.1-1.8)		
Ammoniacal N + (mgl-1)	0.02	0.12	0.11	12.04	< 0.001***
	(0.01-0.06)	(0.03-0.47)	(0.02-0.54)		
Solids + (mgl-1)	6.1	7.4	7.9	1.52	0.229
	(3.7-10.1)	(4.7-11.5)	(4.8-13.2)		
Min Trout + (per 100 m2)	14	6	2	20.08	< 0.001***
	(8-25)	(2-18)	(1-5)		
HABSCORE HUI (Std. Devs.)	-1.4	-2.8	-4.6	20.27	< 0.001***
	(-2.4,0.4)	(-4.5,-1.0)	(-6.2,-3.1)		

Table 5.3 continued

Variable (units)	Group 1 (n=22)	Group 2 (n=19)	Group 3 (n=14)	F	p
Pollution independent variables					
Altitude (m)	85 (28-141)	50 (19-81)	82 (22-142)	2.87	0.065
Slope (%)	3.3 (0-7.4)	1.5 (0.5-2.5)	2.7 (1.2-4.3)	2.37	0.103
Width (m)	3.2 (2.0-4.4)	2.8 (1.7-3.9)	2.4 (1.5-3.2)	2.89	0.065
Dist. from source (Km)	5.0 (2.5-7.6)	4.0 (1.5-6.5)	2.6 (1.2-3.9)	4.91	0.011*
Catchment + Area (Km2)	6.7 (3.3-13.4)	4.6 (2.0-10.6)	3.1 (1.4-6.8)	4.30	0.019*
Substratum (phi)	-4.2 (-5.9,-2.4)	-3.6 (-5.2,-2.0)	-4.3 (-5.8,-2.7)	0.93	0.400
pH	7.1 (6.6-7.5)	7.4 (7.2-7.7)	7.4 (7.3-7.5)	7.93	0.001***
Conductivity (uScm-1)	153 (115-192)	234 (147-321)	175 (133-216)	9.55	< 0.001***
Hardness (mgl-1)	48 (32-64)	77 (49-105)	57 (40-75)	10.05	< 0.001***

Variables marked (+) were log-transformed prior to analysis. Group values are means with standard deviation ranges (+ or - 1 S.D.) in brackets.
F statistic (F) and probability values (p) are from analysis of variance.
Other conventions as for Table 3.

N.B. Chemical data are based upon the results of a single spot sample.

The results of the Chi² tests (Table 5.5) supported the hypothesis of a relationship between stream group and degree of pollution. A number of pollution-sensitive taxa showed preference for Groups 1 or Groups 1 and 2, whilst a few tolerant taxa such as the leech *Helobdella stagnalis*, oligochaetes and chironomids showed a preference for Groups 2 and 3 (Table 5.5).

For most sites in Groups 2 and 3 it was possible to identify probable sources of pollution either during the assessment work in April and May 1990 or from discussion with NRA Pollution Control staff. The information obtained might be considered biased in that no assessment could be made in catchments which appeared unpolluted. It is included because it gives an indication of the range of discharge types involved (see Table 5.4).

Following short field trials in January and February 1991, it was felt necessary to incorporate sewage fungus at two extra stages in the system. These modifications eliminate the need for invertebrate assessments in cases of gross pollution and ensure that sites cannot be classified into Group 1 if sewage fungus is found to be present in visible quantity at a site either above or below stones. Sewage fungus is a definite indicator of organic pollution and its presence even at low abundance should prevent sites being classified in the unpolluted group. These modifications would result in 2 sites from Group 1 being placed in Group 2 and 6 sites in Group 2 being placed in Group 3 (Table 5.4). The possibility of using the family Heptagenidae as a whole an indicator rather than *Rhithrogena* was also investigated so as to ease identification in the field. Examination of raw data from the 55 sites showed that this would not lead to any differences in classification, so the simplification was made. The final form of the key was prepared as a flow chart (Figure 5.5).

5.1.5 Discussion

During the development of the indicator system, TWINSPAN classification was successful in defining three groups of different biological status as reflected in differences in biotic indices (BMWP score and ASPT), trout density and sewage fungus cover. These groups were suggested to represent three different levels of impact from organic wastes. A selected indicator key (based on one-minute kick samples) incorporated indicator taxa whose distribution and abundance have often been found to be influenced by organic pollution. For example in an intensive study upstream of Site 4 on Pontfaen Brook, Rhithrogena semicolorata (the most abundant heptagenid mayfly), *Baetis* spp. (baetid mayflies), and *Gammarus pulex* (the freshwater shrimp) all showed drastically

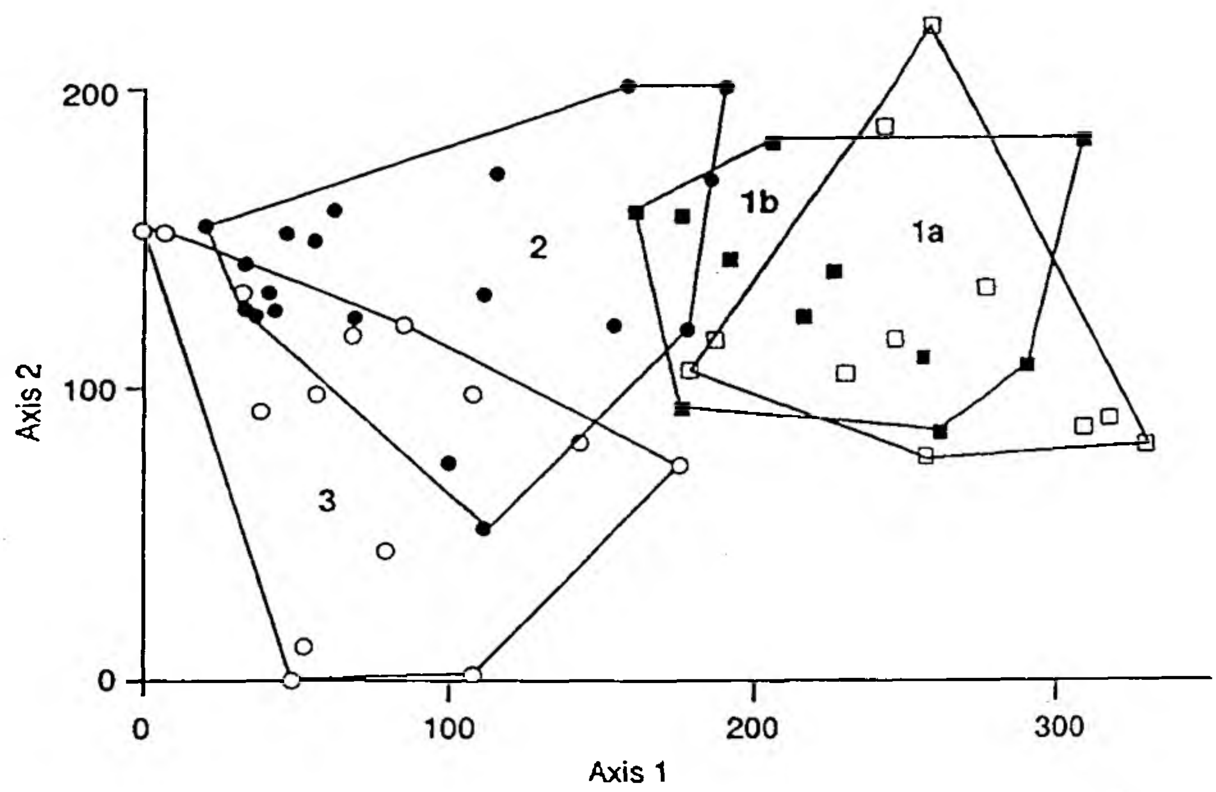
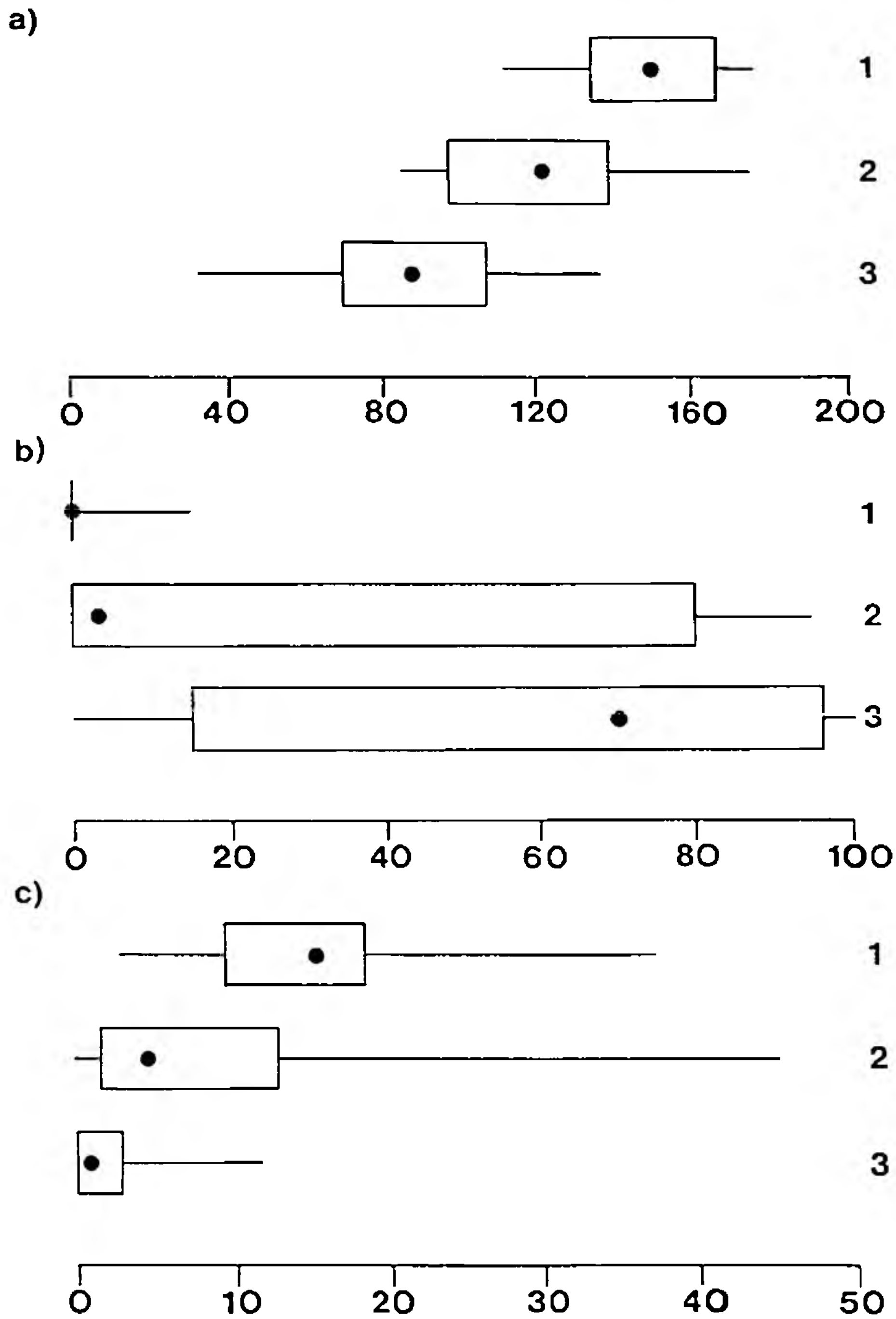


Figure 5.3 Ordination of 55 sites in West Wales by DECORANA. Numbered polygons denote site groups generated by TWINSPAN.

Figure 5.4 Distribution of a) BMWP score, b) sewage fungus cover (%) and c) minimum total trout density (No/m^2) within each TWINSPAN group.



NB Medians (dots), ranges ('whiskers') and first and third quartiles (boxes) are shown.

Table 5.4 Biological characteristics of stream sites in West Wales arranged by TWINSPAN group. The sources of pollution thought to be responsible for the observed biological impacts are given

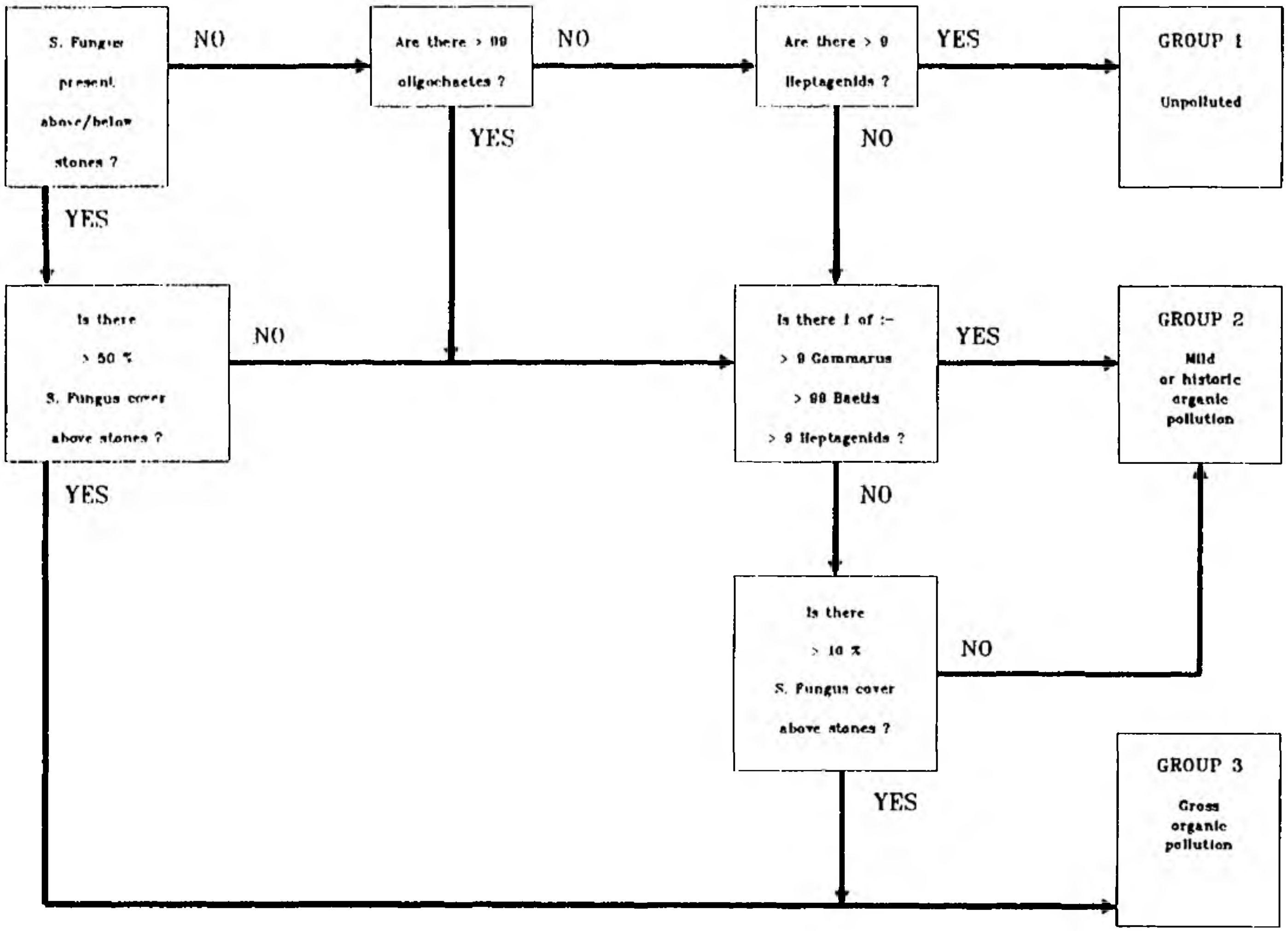
SITE	STREAM	NGR	BMWP -1mk	ASPT -1mk	SEWAGE FUNGUS(%)	MIN TROUT (/100m2)	POSSIBLE POLLUTION SOURCE
GROUP 1							
3	Afon Cwmau	SN037339	154	6.7	0	17.3	
6	W. Cleddau Trib	SM933276	168	6.5	0	13.2	
7	Nant y Coy BrooK	SM921242	100	6.3	0	30.7	
9	Camrose Brook	SM939191	103	6.1	0	16.3	
13	Deepford Brook	SM049200	139	6.0	5_2	3.9	Fish Kill in April 1988
19	Afon Rhydabil	SN107232	119	6.3	0	16.7	
20	Syrfnwy	SN047269	126	6.3	0	5.1	
22	Nant Eiddig	SN593452	148	6.4	0	10.1	
23	Nant Creuddyn	SN567492	148	6.2	0	16.5	
24	Afon Iar	SN500414	170	6.5	0	19.9	
25	Afon Cerdin	SN421415	78	6.0	0	9.8	
29	Afon Dulais	SN315467	142	6.2	0	15.9	
31	Afon Soden	SN373568	109	6.4	0	5.7	
35	Nant Cilcennin	SN500600	110	6.5	0	12.9	
36	Afon Feinog	SN466565	120	6.3	0	20.5	
37	Afon Carrog	SN562719	98	6.5	0	12.8	
38	Afon Arth	SN541630	157	6.8	0	14.7	
39	Nant Adal	SN624749	136	6.5	0	18.0	
40	Nant Paith	SN604787	101	6.3	0	36.9	
48	Nant y Ci	SN386187	107	5.6	0	2.8	
51	Pontgarreg Fach	SN316275	108	6.4	15_2	21.9	Leaking lagoon
55	Cresswell Trib.	SN096077	155	6.5	0	8.4	
GROUP 2							
5	Aberbach Stream	SM895361	147	5.7	< 1	12.5	Yard run off
8	Nant y Coy Brook	SM922242	116	5.8	80_3	44.8	Chronic silage effluent discharge
10	Knock Brook	SM938191	81	4.8	80_3	14.9	Whey spread to land
11	Fenton Brook	SM973174	97	5.7	0	0	Fish kill in 1989 and yard runoff
14	Deepford Brook catchment	SN072198	89	4.9	95_3	5.9	Various - intensive dairying

Table 5.4 Continued/...2

SITE	STREAM	NGR	BMWP -1mk	ASPT -1mk	SEWAGE FUNGUS(%)	MIN TROUT (/100m2)	POSSIBLE POLLUTION SOURCE
15	Cotland Mill	SN054193	118	5.9	0	12.9	Possible yard runoff
16	Holmes Stream	SN042208	78	4.9	70_3	13.4	Inadequate storage of slurry
30	Afon Hirwaun	SN258424	73	5.6	0	4.5	Variety of intermittent discharges
32	Drywi	SN445585	81	5.8	5	3.2	Uncertain - history of inputs
34	''	SN438593	104	5.8	0	1.5	Uncertain - history of inputst
41	Nant Coch	SN661252	75	5.0	10	1.1	Deliberate discharges from lagoons
42	Gurrey Fach	SN632231	94	5.5	0	4.1	Silage effluent and Yard runoff
43	Nant Cwmffrwd	SN445165	127	6.0	3	0	Leaking slurry stores
44	'' ''	SN443165	111	5.8	5	0	Lagoon leakage and yard runoff
45	'' ''	SN422174	133	6.1	1	7.9	Parlour & dairy washings
47	Nant y Glaston	SN422175	68	4.8	85_3	4.2	Dungstead washings
49	Gwendraeth Trib	SN491162	97	5.4	80_3	32.3	Farm identified - source uncertain
53	Afon Rhydbennau	SN150196	167	6.4	1	10.3	
54	River Creswell	SN096076	93	5.8	0	1.5	Sewage treatment works
GROUP 3							
1	River Gammon	SN083400	32	5.3	100	2.7	Spreading of slurry, whey etc.
2	Aberbach stream	SM997386	60	5.5	100	4.0	Yard washings & whey from pig farm
4	Pontfaen Brook	SN027329	99	6.6	80	2.5	Parlour washings and silage effluent
12	Churchill Brook	SN050197	53	4.4	60	11.5	Overflowing slurry lagoon
17	Slade Brook	SN037224	55	5.0	30	6.1	Dairy & yard washings and road runoff
18	Afon Rhydabil	SN116232	86	6.1	0	0	Poorly contained parlour washings
21	E. Cleddau trib	SN119181	55	5.0	20	1.2	Slurry spreading & yard runoff ?
26	Durog	SN292382	106	5.9	0	0	Uncertain - history of pollution
27	Durog	SN277375	108	6.0	80	0	Whey spread to land
28	''	SN283378	59	5.9	100	0	Whey spread to land
33	Drywi	SN440592	55	5.5	0	1.5	Uncertain - history of pollution
46	Nant y Glaston	SN425175	18	3.6	95	0	Dungstead washings
50	Pontgarreg Fach	SN321280	49	5.4	80	0	Farm identified - source uncertain
52	Afon Fenni	SN251221	65	5.9	30	0	Fish Kill in 1989 and lagoon overflow

Subscripts for Sewage Fungus cover denote changes in group occasioned by modifications to key.

Figure 5.5 Adaptation of the TWINSPAN indicator key for use in the rapid assessment of farm pollution in streams of West Wales .



reduced density below a chronic discharge of silage effluent whilst oligochaetes showed a large increase (Section 7.1). The same pattern was observed in a comparison between a control stream and a stream affected by more episodic inputs of silage liquor (Section 7.2). In general the taxa shown to be associated with Group 1 such as certain stoneflies, mayflies and caddis (Table 5.5) have been recognized by other workers as indicators of good water quality in stony streams (e.g. Hynes, 1960, Hawkes, 1963). Oligochaetes (here mainly Naididae) and chironomids which showed increased abundance in the polluted Groups 2 and 3 have often been shown to have increased population densities in association with organic pollution in both temperate streams (Hynes, 1960; Hawkes, 1963; Learner *et al.*, 1971; Nuttall & Purves, 1974; Suckling, 1982; Seager & Abrahams, 1990) and in the tropics (Pearson & Penridge, 1987). These taxa contain many generalist feeders which benefit from increased organic loading.

Table 5.5 **Frequency of occurrence of selected invertebrate taxa in the three TWINSPAN groups generated by analysis of the SPP1 data set**

TAXON	GROUP 1 (n=22)	GROUP 2 (n=19)	GROUP 3 (n=14)	Chi^2	p
Taxa associated with Group 1					
Leuctra spp.	*****	**	***	17.7	< 0.001
Protonemoura spp.	***	*	*	10.0	< 0.01
Chloroperla spp.	*****	**	***	13.1	< 0.01
Hydropsyche spp.	*****	**	**	21.8	< 0.001
Sericostoma personatum	****	*	*	16.2	< 0.001
Brachyptera risi (2)	***	**	**	6.9	< 0.05
Isoperla grammatica (2)	***	*	*	7.9	< 0.02
Hydropsyche spp. (2)	***	*		17.2	< 0.001
Rhithrogena sp. (3)	**	*		10.9	< 0.01
Amphinemeura spp.	****	**	**	8.9	< 0.02
Perlodes microcephala	**	*		7.2	< 0.05
Leuctra spp. (2)	***	*	*	13.7	< 0.01
Chloroperla spp. (2)	***		*	17.3	< 0.001
Rhithrogena sp.	*****	****	****	6.6	< 0.05
Isoperla grammatica	*****	***	**	8.9	< 0.02
Taxa associated with Groups 1 and 2					
Ecdyonurus spp.	****	***	*	15.3	< 0.001
Caenis spp.	***	*		7.1	< 0.05
Hydraena gracilis	****	***	*	17.2	< 0.001
Rhithrogena sp. (2)	*****	***		32.0	< 0.001
Elminthidae (2)	***	**		8.9	< 0.02
Gammarus pulex	****	*****	***	13.9	< 0.01
Elminthidae	*****	****	**	10.8	< 0.01
Rhyacophila dorsalis	*****	****	**	15.3	< 0.001
Plectrocnemia spp.	***	***		9.3	< 0.01
Pisidium spp.	***	***	**	9.4	< 0.01
Nemoura spp.	***	****	*	10.6	< 0.01
Potamopyrgus sp. (2)	**	***	*	8.8	< 0.02
Gammarus pulex (2)	**	****	*	11.3	< 0.01
Ancylus fluviatilis	****	***	*	12.0	< 0.01
Dicranota spp.	*****	***	***	7.9	< 0.02
Baetis spp. (3)	****	***	*	11.6	< 0.01

Table 5.5 continued

TAXON	GROUP 1 (n=22)	GROUP 2 (n=19)	GROUP 3 (n=14)	Chi^2	p
Taxa associated with Group 2					
Asellus spp.	*	**	*	6.8	< 0.05
Habrophlebia fusca		***		26.1	< 0.001
Potamopyrgus spp.	***	****	***	6.0	< 0.05
Taxa associated with Groups 2 and 3					
Helobdella stagnalis	*	***	**	10.4	< 0.01
Oligochaeta (3)	*	*****	***	27.1	< 0.001
Chironomidae (3)	*	****	**	10.9	< 0.01
Oligochaeta (2)	***	*****	*****	14.6	< 0.001
Taxa not showing Group association					
Gyrinidae	***	**	*	5.9	N.S.
Glossiphonia complanata	*	**	**	5.5	N.S.
Oligochaeta	*****	*****	*****	1.4	N.S.
Baetis spp.	*****	*****	*****	0.0	N.S.
Chironomidae	*****	*****	*****	0.0	N.S.
Lumbricidae	***	***	***	0.4	N.S.
Brachyptera risi	****	****	****	0.7	N.S.
Limnephilidae	****	***	***	4.5	N.S.
Paraleptophlebia spp.	**	**	*	1.6	N.S.
Erpobdella octoculata	**	***	*	5.0	N.S.
Chironomidae (2)	****	*****	*****	5.5	N.S.
Simuliidae (2)	***	****	***	1.3	N.S.
Simuliidae	****	*****	*****	3.3	N.S.
Ceratopogonidae	**	***	***	1.8	N.S.
Baetis spp. (2)	*****	*****	****	4.0	N.

Asterisks indicate percentage occurrence (* 1-20 % of sites in group, ** 21-40 %, *** 41-60 %, **** 61-80 %, ***** 81-100 %).

For certain taxa abundance categories are treated as different taxa i.e., (2) indicates > 9 individuals in a 1 minute kick sample, (3) > 99. Chi^2 values and associated probabilities are given.

Despite the evidence above, there is a possibility that the observed TWINSPAN groupings might reflect the influence of factors other than farm pollution. There were certainly significant differences between the three groups in pH, conductivity and hardness. These differences are unlikely to be cause-effect because the trends are opposite to those more commonly found i.e. diversity is generally lower in conditions of lower hardness and pH (Rutt *et al.*, 1991).

They may arise because the cleaner sites in Group 1 were often in less intensively farmed catchments with poorer soils and softer waters. Differences in catchment area and distance from source may reflect the fact that farm pollution in West Wales tends to have most influence on small streams with low dilution capacity (Howells & Merriman, 1986).

The mechanisms which may account for differences in the composition of invertebrate assemblages between the TWINSPAN groups are likely to vary according to the periodicity and intensity of the pollution inputs. Continuous discharges which elevate the mean BOD to even a little above the 0.5-2 mg l^{-1} typical of unpolluted streams can result in growths of 'sewage fungus' in well-aerated conditions (Water Pollution Research Laboratory, 1969; Quinn & McFarlane, 1988; Seager *et al.,* 1992). Respiration of such growths can lead to oxygen deficiency within the substrate and cause habitat disruption and loss of niches for specialist feeders (Curtis, 1969). Salmonids will also be affected due to loss of invertebrate food and smothering of spawning gravels (Smith & Kramer, 1963). By contrast intermittent discharges such as daily inputs of parlour and dairy washings, or more irregular yard run-off generated by rainfall, are likely to affect stream fauna more directly. Low dissolved oxygen and increased ammonia concentrations can affect sensitive taxa by mortality (Surber & Bessey, 1974, McCahon *et al.,* 1991; Williams, *et al.,* 1986) and by sublethal stresses such as the initiation of a drift response (Section 9, Edwards *et al.,* 1991) and disruption of mating behaviour (McCahon *et al.,* 1991). *Rhithrogena* may be especially sensitive to low dissolved oxygen due to the fact that it uses its gills as a suction device to maintain its position in areas of high current velocity (Jaag & Ambuhl, 1964).

It is envisaged that the methodology described here may fit into a co-ordinated strategy to tackle farm pollution in England and Wales. Areas for the use of indicator keys can be targeted using risk assessment maps produced by overlaying factors such as organic waste loading, rainfall and soil type using an ARCINFO Geographical Information System (Section 4). New indicator keys would have to be developed for some areas due to regional variation in fauna. Besides the use in identifying watercourses affected by organic inputs, it is suggested that rapid biological appraisal may be used to discover pollution sources and then to assess the success of remedial action following programmes of farm visits by the pollution control authorities (Seager *et al.*, 1992). The system is designed for rapid investigation of catchments as a cost-effective means of prioritising effort. More intensive biological and chemical survey work is clearly necessary where a deeper understanding of pollution impact is required.

5.2 An indicator system for Devon (Spring): 'Family' level

5.2.1 Introduction

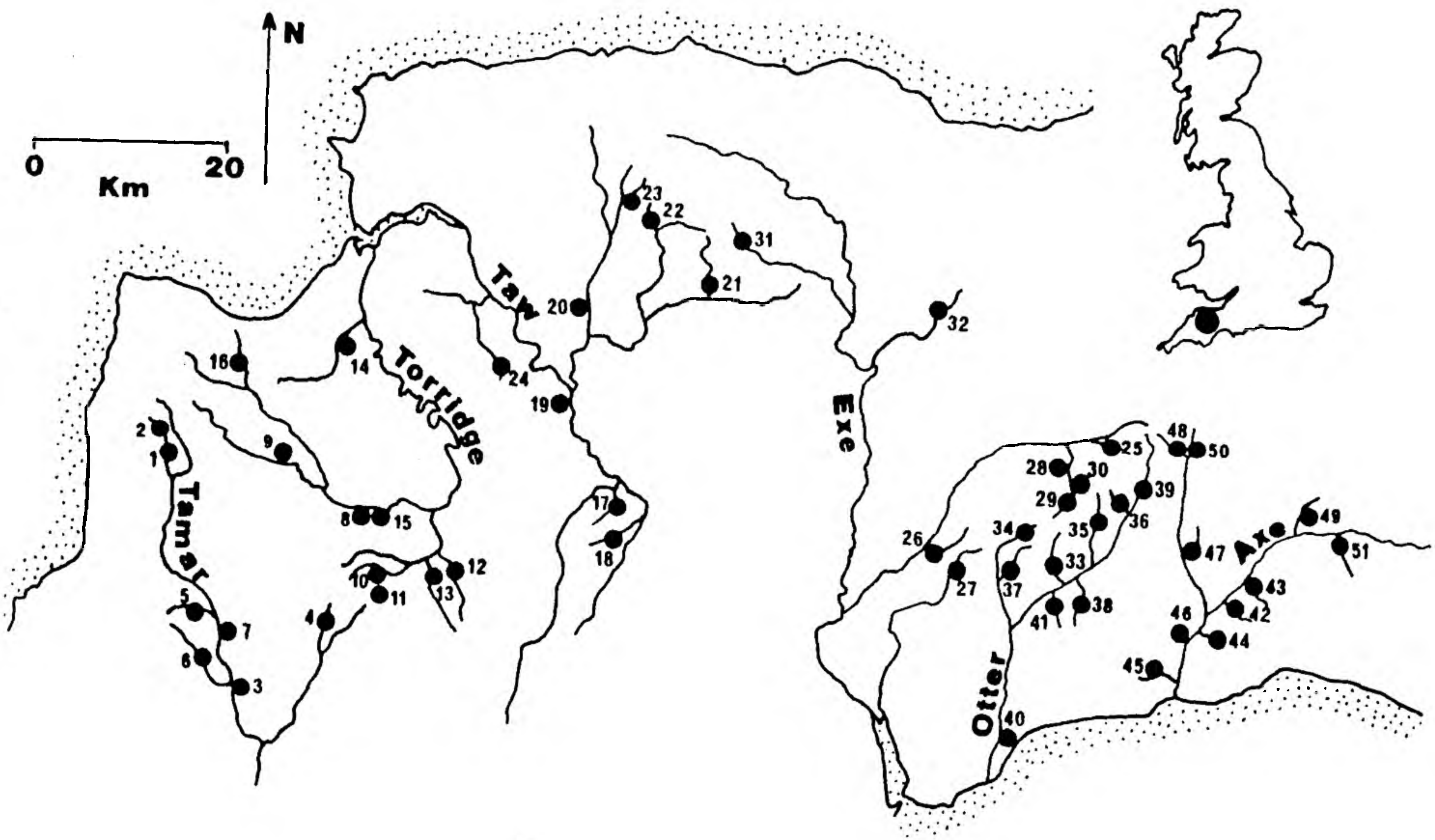
In order to test the feasibility of rapid appraisal as part of a national strategy for tackling farm pollution (Seager, *et al.*, 1992), it was decided to test the Welsh key in another NRA region (Section 6.2). This also offered the opportunity to develop a separate key specifically for a new region. NRA South-west Region offered their co-operation with this exercise. This region was suitable for study because it was confirmed as a high risk area with many recorded pollution incidents during the national assessment phase of this project (Section 4).

5.2.2 Methods

Site selection

Fifty-one sites within six catchments in Devon were selected in accordance with the criteria used in west Wales. It was important to ensure that a representative sample of the streams of the region was taken covering the range of geographical area. The six catchments lie in an area where dairy farming is the predominant land-use (Figure 5.6). The majority of subcatchments had densities of dairy cattle in the mid range (0.6-0.9 head/Ha catchment) whilst densities of beef cattle were in the mid to high range (0.6-1.2 head/Ha catchment). Consequently the total volumetric waste loading to the study catchments tended to be in the higher categories found during the national assessment exercise (Section 4). The three catchments in the West (Tamar, Taw

Figure 5.6 Map of North Devon showing the 51 sites sampled and their relationship to major river systems.



and Torridge) are underlain predominantly by the Carboniferous Culm measures and some sediments of the middle Devonian. The other three catchments (Exe, Otter and Axe) lie upon more recent sediments - Permian basal breccias, Triassic mudstones and greensand of the upper Cretaceous. Site selection within catchments was based upon information from the NRA Red List Farms file and from the local knowledge of NRA river wardens. Sites were chosen so as ensure a range of impact from farm pollution and were unlikely to be affected by pollution from non-agricultural sources. Streams were mainly of first or second order (Strahler, 1957) and in the range 0.8 to 4.0 m in width.

Fieldwork

Field-work was carried out between 25 February and 12 March 1991. The procedure at each site was as follows:

1. A spot water sample was taken at all but two sites. The samples were later analysed by the South-west Water laboratory at Countess Weir, Exeter under contract to the NRA (data in Appendix F6).

2. The percentage cover of 'sewage fungus' both above and below large stones (> 6 cm) was recorded.

3. Other environmental data such as substratum composition - classed as bedrock, boulders/cobbles (> 6 cm), pebbles/gravel (2 mm to 6 cm), sand, silt/clay - and width were noted (Data in Appendix F7).

4. A single 1 minute kick sample was taken from a riffle at each site to sample the invertebrate population. Samples were fixed back at the laboratory in 4% Formaldehyde solution.

Invertebrate sample processing

Macroinvertebrate samples were processed in the laboratory by both WRc staff and biologists from NRA South-west Region. Animals were identified to a level established during the development of the west Wales indicator key. Identification was generally to genus or family level which is rapidly achieved and yet sufficient to ensure the production of a workable indicator key (see Section 5.1). Data were stored on a data archive system known as BAETIS which has been developed within NRA Welsh Region (NRA, 1992). Raw data are presented as Appendix F8.

Data analysis

The invertebrate samples from the fifty-one sites were classified using a multivariate classification technique known as TWINSPAN (Hill, 1979a) and ordinated using DECORANA (Hill, 1979b). Both of these techniques have been widely used in freshwater ecology in the interpretation of complex data sets (e.g. Wright *et al.*, 1984; Ormerod *et al.*, 1987).

This subsection describes analysis of the data at the 'family' level with oligochaetes (other than Lumbricidae) and Hydracarina grouped at a higher taxonomic level (see Section 5.3 for treatment of data at species/genus level). Percentage sewage fungus cover above stones was included as a variable in the analysis. TWINSPAN generates an indicator key which can be used to classify new sites into the derived groups. Where TWINSPAN is run using quantitative data, 'cut' levels must be specified such that different abundance categories of a taxon are entered into the analysis as separate 'pseudospecies'. In our case 'pseudospecies' were set to be logarithmic abundance categories i.e. 1-9, 10-99, 100-999, etc. A TWINSPAN option was invoked which prevented category 1 'pseudospecies' (1-9 individuals) appearing as indicators. This can simplify the keys and reduce the possibility of chance occurrence of single individuals leading to misclassification when the derived keys are applied in practice.

The DECORANA axes were compared to a range of biotic and abiotic variable by product moment correlation after transformation where appropriate. This assisted in the interpretation of the TWINSPAN analysis.

To determine whether the derived TWINSPAN groups reflected different degrees of farm pollution, between-group differences in the means of a range of biotic and abiotic variables were tested by analysis of variance after transformation of data where necessary.

Pollution Sources

Details of the status of each site showing some evidence of pollution were given to NRA pollution control staff such that they could provide information on the pollution sources responsible either from existing records or investigation in the field. Such information is useful in validating the approach.

5.2.3 Results

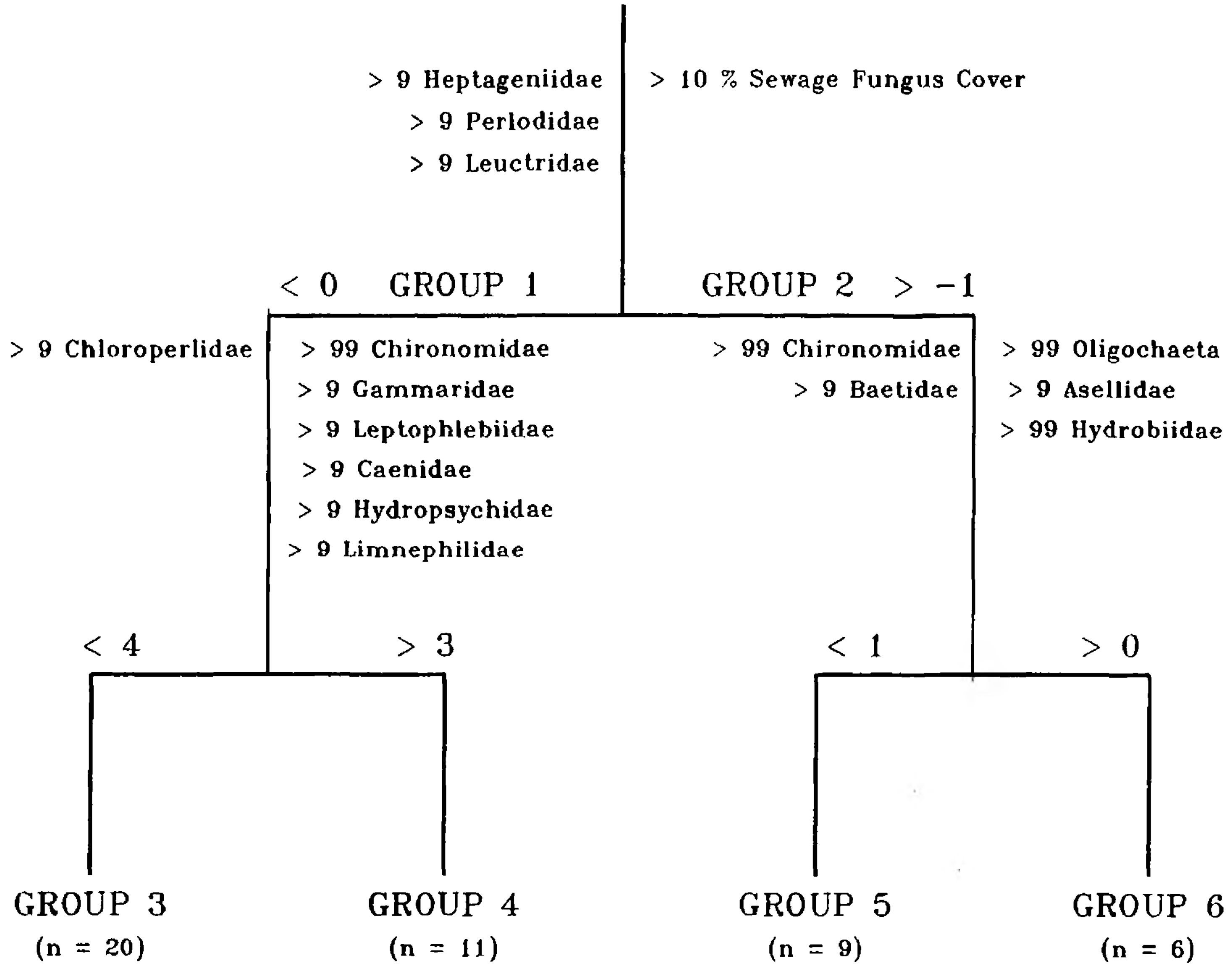
Generation of an indicator key

Out of the initial 51 sites surveyed, macroinvertebrate samples for three sites were mislaid so that TWINSPAN classification was carried out on the remaining 48 samples. The initial classification produced a very small site group consisting of two sites (26 and 37) at the second level of division so these were excluded from further analysis to ensure a more even pattern of division. Subsequent TWINSPAN analysis generated an indicator key incorporating taxa which were not suitable for enumeration in field samples. These taxa (Planariidae, Ancylidae, Sphaeridae, Lumbricidae and Elminthidae) were masked out as potential indicators which generated a more practical system (Figure 5.7). Taxa to the left of each division line score -1, taxa to the right + 1. The net total for a site indicates whether it will be placed to the left or right of the division.

Interpretation of the indicator key

The indicators for the initial division of the data set into two groups of 31 and 15 sites was on the basis of the abundances of heptagenid mayflies, leuctrid and perlodid stoneflies and the percentage cover of sewage fungus. The three negative indicators are all insect families held to be sensitive to organic pollution having BMWP scores of the highest category (10). All three have shown markedly reduced population densities at sites subject to farm pollution during intensive studies carried out for this contract (Section 7). Sewage fungus, the positive indicator is a well known indicator of organic

Figure 5.7 Indicator key for Devon derived from TWINSPAN analysis of macroinvertebrate data from 46 sites.



pollution (Curtis, 1969). The nature of the indicators thus suggests that this initial division reflects the influence of organic pollution. This is supported by analysis of variance using the two groups as treatments (Table 5.6).

Table 5.6 Relationships between TWINSPAN groups at first level of division and environmental and biotic variables

Variable (units)	Group 1	Group 2	F	p
Pollution dependent				
BMWP	155 (132-177)	94 (63-126)	56.48	<0.001***
ASPT	6.4 (6.0-6.8)	5.2 (4.4-5.9)	49.12	<0.001***
BOD (mg l^{-1})	1.5 (0.4-2.6)	1.8 (0.3-3.3)	0.68	0.413
Ammoniacal N[1] (mg l^{-1})	0.04 (0.01-0.12	0.09 (0.02-0.38)	4.29	0.044*
Pollution independent				
Altitude (m)	126 (71-171)	85 (35-136)	5.82	0.02*
Gradient (%)[1]	1.5 (1.0-2.4)	1.5 (0.8-2.8)	0.00	0.998
Width (m)	2.1 (1.3-3.0)	1.7 (1.1-2.4)	2.75	0.104
Distance from source (km)	2.8 (1.2-4.3)	2.2 (1.0-3.5)	1.44	0.236
pH	7.4 (7.0-7.8)	7.5 (7.2-7.8)	1.12	0.296
Conductivity (µS cm^{-1})	201 (122-280)	295 (176-415)	9.50	0.004
Alkalinity[1] (mg $CaCO_3$ l^{-1})	28 (13-61)	48 (28-81)	4.95	0.031*
Mean substratum class	2.8 (2.4-3.1)	2.7 (2.4-3.0)	0.37	0.548

Notes for Table 5.6

[1] denotes variables which were log-transformed prior to analysis. Group values are means with standard deviation ranges (±1 S.D.) in brackets. F statistic (F) and probability (p) are from analysis of variance. Asterisks indicate level of probability: * <0.05, ** <0.01, *** <0.001.

There were highly significant differences between the two groups in BMWP score and Average Score Per Taxon (ASPT) (Armitage *et al,* 1983) and a less significant difference in ammoniacal nitrogen concentration. There were also significant differences in conductivity, alkalinity and altitude which may reflect a tendency for dairy farming to be occurring more on richer soils. Correlations between site scores on DECORANA axis 1 and BMWP score, ASPT and ammonia were all highly significant ($p < 0.001$) and it was this axis which reflected the initial division of the sites (Figure 5.8, Table 5.7). There were also significant correlations with conductivity, altitude and alkalinity. TWINSPAN identified other 'Pseudospecies' preferential to the left-hand side of the division (Group 1) in addition to the indicator taxa. These tended to be those of pollution sensitive mayflies (Heptageniidae, Ephemeridae, Caenidae, Leptophlebiidae), stoneflies (Perlodidae, Leuctridae, Chloroperlidae, Taeniopterygidae), caddis (Rhyacophilidae, Goeridae, Sericostomatidae, Odontoceridae) and beetles (Gyrinidae, Elminthidae). Asellidae and oligochaetes were amongst the few taxa showing preference for Group 2.

The indicators at the second level of division showed less obvious relationship to organic pollution and this was reflected in the results of analysis of variance with four groups (Table 5.8). There were still highly significant differences in BMWP score and ASPT (although values of F are lower). Groups 3 and 4 and 5 and 6 showed marked overlap for BMWP score, although discrimination was still quite good for ASPT (Figure 5.9). Differences in width, pH, conductivity and alkalinity were all very significant (Table 5.8). Scores on DECORANA Axis 2, which reflected the second level of division (Figure 5.8), correlated only weakly with BMWP score ($p < 0.05$), moderately with ammoniacal Nitrogen ($p < 0.01$) and were not related to any other variables except pH. It is suggested that the second level of division does not relate strongly to farm pollution, other more natural factors becoming of equal importance.

Figure 5.8 Ordination of 46 sites in Devon by Decorana. Numbered polygons denote site groups generated by TWINSPAN.

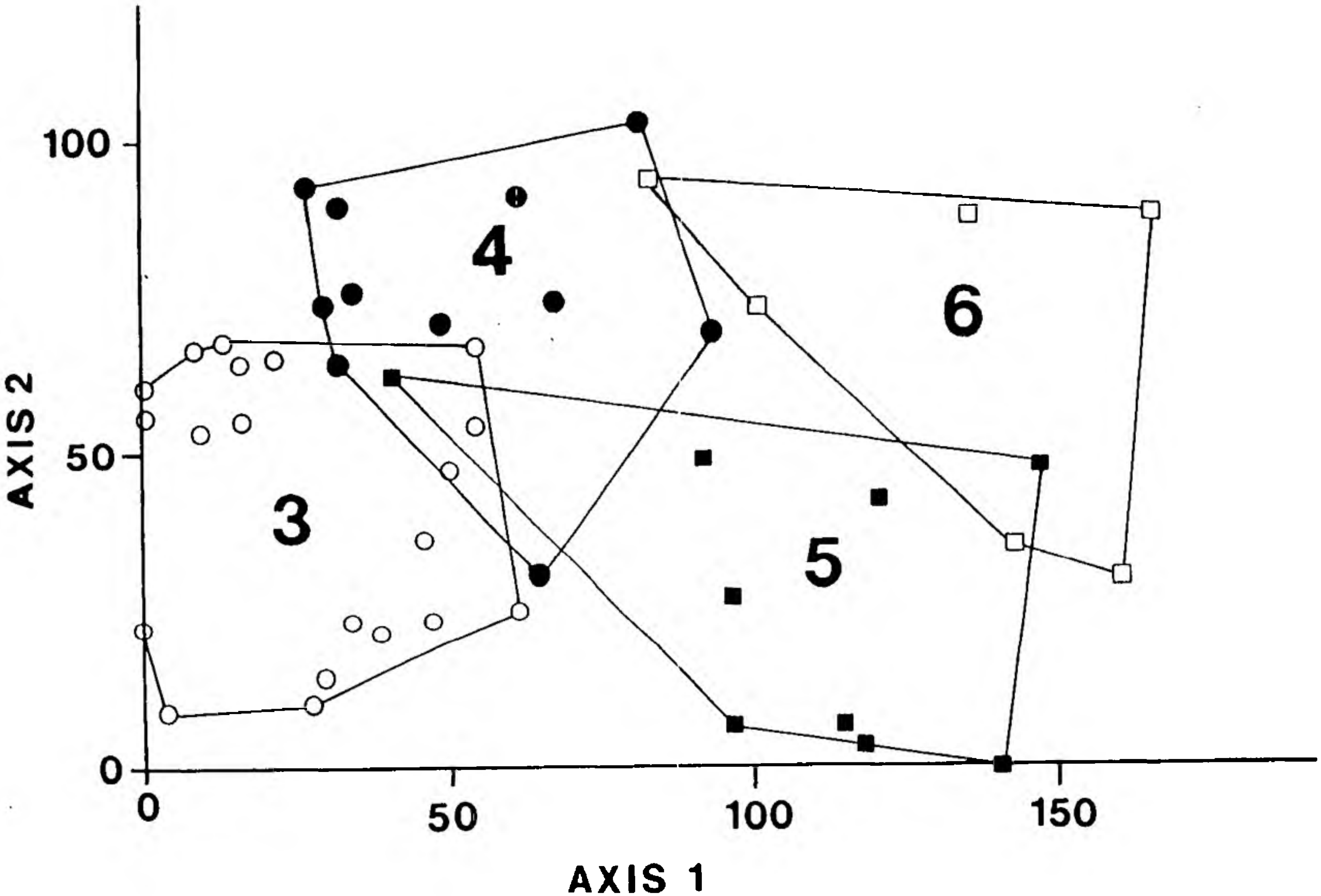
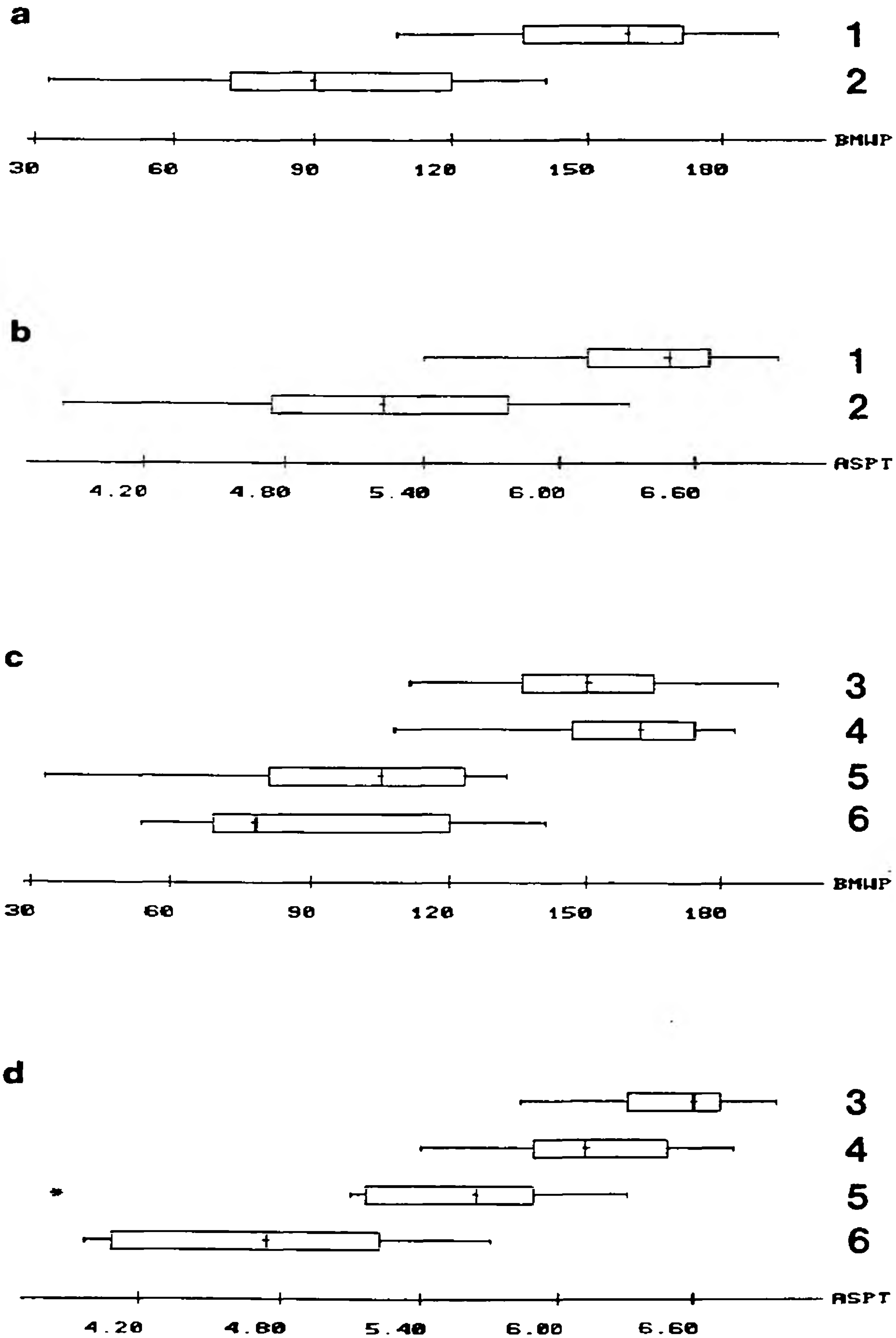


Figure 5.9 Distributions of biotic indices within TWINSPAN groups derived from data from Devon: a) BMWP for Groups 1 & 2; b) ASPT for groups 1 & 2; c) BMWP for Groups 3,4,5 & 6; d) ASPT for Groups 3,4,5 & 6.



A key for use in the field

On the basis of the above evidence it is suggested that just two TWINSPAN groups are used as the basis for a key for Devon (Figure 5.10). To produce a flow chart for use in the field sewage fungus has been included at an extra stage in the system such that Group 1 sites with some sewage fungus present

Table 5.7 Product-moment correlation coefficients between first axes derived from DECORANA and biotic and environmental variables

Variable	n	Correlation coefficients DCA1	DCA2
BMWP	46	-0.789 ***	0.356 *
ASPT	46	-0.927 ***	0.086
BOD (mg l^{-1})	44	0.176	-0.119
Ammoniacal N (mg l^{-1})[1]	44	0.466 ***	-0.414 **
Altitude (m)	46	-0.445 **	0.176
Gradient (%)[1]	46	-0.114	-0.035
Width (m)	46	-0.311 *	-0.018
Distance from source (km)	46	-0.176	-0.037
pH	44	0.255	0.392 **
Conductivity (µS cm^{-1})	44	0.556 ***	0.072
Alkalinity (mg $CaCO_3$ l^{-1})[1]	44	0.472 ***	0.243
Mean substratum class	46	-0.117	-0.447 **

Asterisks indicate level of significance (* $p<0.05$, ** $p<0.01$, *** $p<0.001$)
[1] denotes variables log-transformed prior to analysis

(1b) can be distinguished from sites without (1a). Sites in Group 1b can then be said to show evidence of some mild organic input and could be investigated further by surveying further upstream. On this basis only 7 of the 31 sites in TWINSPAN Group 1 can be placed in Group 1a reflecting a persistent low level of organic pollution in the remaining catchments. There is no marked difference in fauna which appears to relate to this enrichment (Table 5.9). It should be stressed that this is very much a preliminary version of an indicator key for Devon and further analysis and testing will lead to refinements.

Table 5.8 Relationships between TWINSPAN groups at the second level of division and environmental and biotic variables

Variable (Units)	Group 1 (n=20)	Group 2 (n=11)	Group 3 (n=9)	Group 4 (n=6)	F	p
Pollution dependent						
BMWP	153 (130-175)	159 (135-182)	96 (63-129)	91 (59-124)	18.33	<0.001***
ASPT	6.5 (6.2-6.9)	6.2 (5.7-6.6)	5.5 (4.8-6.2)	4.8 (4.1-5.4)	23.48	<0.001***
BOD (mg l^{-1})	1.6 (0.3-2.9)	1.3 (0.8-1.8)	1.8 (0.2-3.3)	1.9 (0.3-3.5)	0.35	0.791
Ammoniacal N[1] (mg l^{-1})	0.04	0.04	0.09	0.11	1.41	0.283
Pollution independent						
Altitude (m)	137 (83-191)	105 (53-158)	86 (53-120)	85 (12-158)	2.81	0.051
Gradient (%)[1]	1.5 (0.9-2.3)	1.7 (1.1-2.5)	1.7 (1.0-2.9)	1.3 (0.6-2.7)	0.54	0.658
Width (m)	2.4 (0.8-3.2)	1.6 (0.9-2.4)	1.4 (1.0-1.9)	2.2 (1.4-2.9)	5.34	0.003**
Distance from source (km)	2.9 (1.3-4.5)	2.6 (1.2-4.0)	2.1 (0.7-3.5)	2.4 (1.3-3.6)	0.60	0.621
pH	7.2 (6.9-7.5)	7.7 (7.5-7.9)	7.5 (7.2-7.7)	7.6 (7.1-8.1)	7.15	0.001**
Conductivity (µS cm^{-1})	169 (117-222)	259 (170-348)	241 (206-276)	418 (261-576)	13.95	<0.001***
Alkalinity[1] (mg $CaCO_3$ l^{-1})	20 (10-39)	52 (29-92)	41 (29-56)	69 (33-146)	9.07	<0.001***
Mean substratum class	2.8 (2.5-3.1)	2.6 (2.3-2.9)	2.8 (2.5-3.2)	2.5 (2.2-2.7)	2.27	0.094

[1] Denotes variables which were log-transformed prior to analysis. Group values are means with standard deviation range (±1 S.D.) in brackets. F statistic (F) and probability (p) are from analysis of variance. Asterisks indicate level of probability - * <0.05, ** <0.01, *** <0.001.

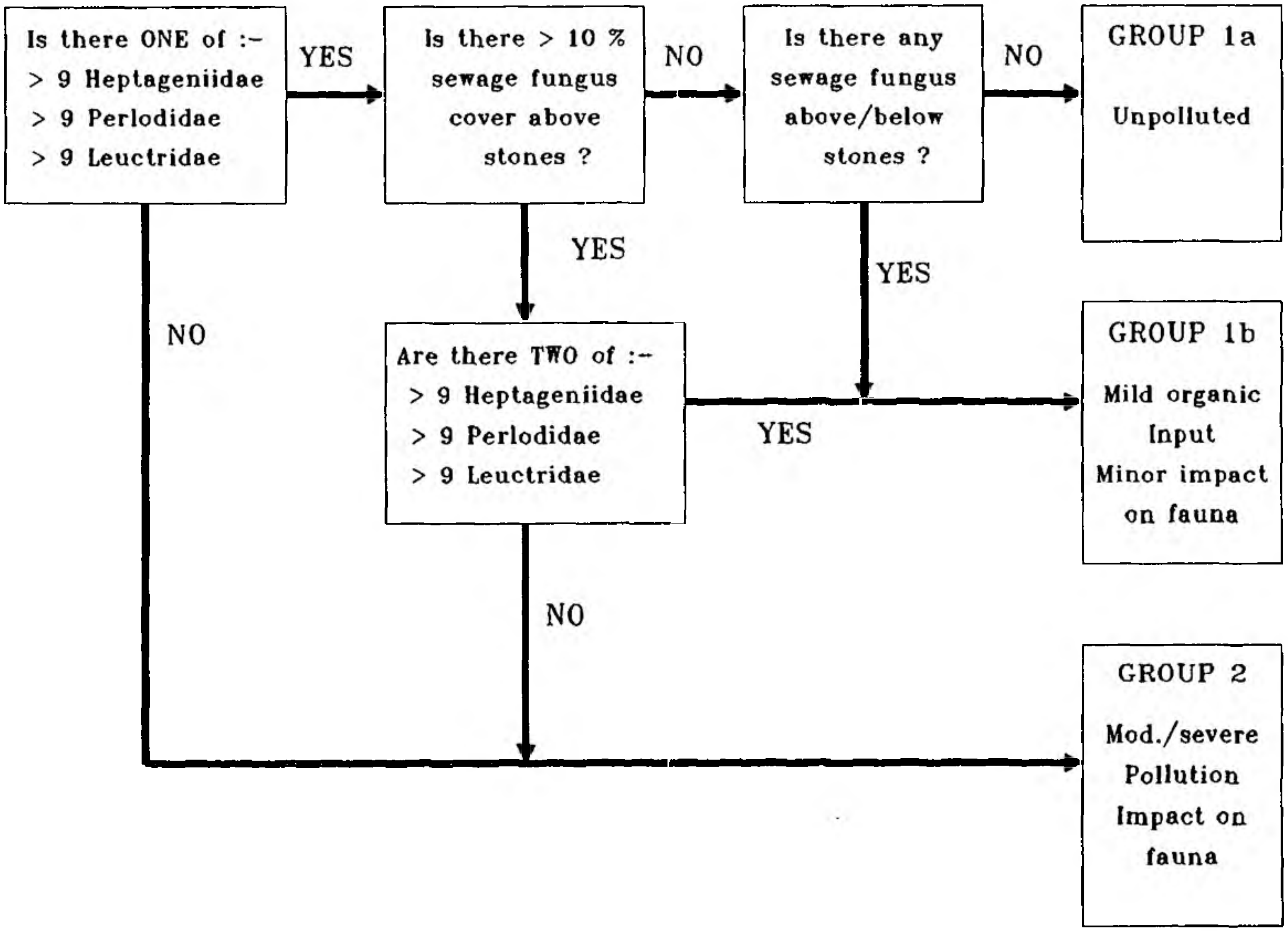
Table 5.9 Biological characteristics of stream sites in Devon arranged by pollution group. Information on possible pollution sources provided by Pollution Control staff are given

SITE	CATCHMENT	N.G.R.	BMWP	ASPT	% FUNGUS COVER ABOVE	% FUNGUS COVER BELOW	POSSIBLE SOURCES OF POLLUTION
GROUP 1a							
21 Trib of Yeo	Taw	SS792276	160	6.7	0	0	
22 Lyddicombe Bottom at Fyldon	Taw	SS738337	175	7.0	0	0	
31 Dane's Brook	Exe	SS819321	122	6.4	0	0	
39 Otter below Sweetlands Farm	Otter	ST216096	163	6.5	0	0	
44 Trib. of Axe below Trill	Axe	SY286959	174	6.4	0	0	
48 Yarty near Moorseek	Axe	ST245134	194	6.9	0	0	
49 Trib. of Axe at Chalkway	Axe	ST376075	134	6.7	0	0	
GROUP 1b							
4 Trib. in Witherdown Wood	Tamar	SX428953	129	6.8	40	20	No problem when site visited on 14.6.91
5 Trib. d/s Ogbeare Hall	Tamar	SX308096	134	5.8	2	10	
6 Tala Water near Boyton	Tamar	SX312919	189	6.8	2	5	
7 Trib. North of Northcott Hamlet	Tamar	SX334934	153	6.7	2	2	
9 Trib. of Waldon	Torridge	SS389106	162	6.0	30	10	Inadequate waste containment
10 Trib. of Lew below Whiddon Farm	Torridge	SX479999	147	6.4	7	15	Dirty Water runoff
11 Trib. of Lew East of Patchcott	Torridge	SX478994	150	6.3	5	10	Dirty water problem. LRI now installed
12 Medland Brook near Northwood	Torridge	SS552002	141	6.7	0	2	
13 Trib. of Lew West of Westacott	Torridge	SS532002	112	6.6	1	1	
16 Trib. near Stroxworthy	Torridge	SS344198	150	6.3	1	0	Domestic sewage and dirty water
24 Trib. below Deptford Farm	Taw	SS598192	140	5.8	0	15	Dirty water problem - L.R.I. proposed
25 Trib. of Culm at Stapley	Exe	ST186134	144	6.6	0	< 5	
28 Trib. of Madford near Abbey	Exe	ST141107	177	6.1	0	< 5	
30 Trib. of Madford D/S Gorwell	Exe	ST153094	186	6.9	0	5	
32 Trib of Batherm D/S Raddington	Exe	ST011257	135	5.4	0	< 5	No known problems, six farms upstream
33 River Wolf at A373 road bridge	Otter	ST129022	162	6.5	1	15	Many historic problems in catchment
35 Trib. of Otter at Luppitt	Otter	ST173066	183	6.8	?	10	A few historic problems - none recently.
36 Trib. of Otter at Upottery	Otter	ST199078	160	6.4	0	5	S. fungus in trib. below Chapelhayes Farm
38 Trib. of Otter at Blannicombe	Otter	SY161985	160	7.0	5	5	
40 Gissage at Gittisham	Otter	ST135983	135	6.1	5	50	Yard runoff from several farms
41 Trib. of Axe at Sector	Axe	SY311982	172	6.6	1	3	Waste containment problems from 2 farms
42 Trib. below Bagley Hill Farm	Axe	ST321010	162	5.8	0	5	Yard runoff - grant approved for L.R.I.
45 Trib. below Whitwell Farm	Axe	SY238925	110	5.5	0	< 2	Yard runoff - L.R.I. installed
47 Trib. of Yarty at confluence	Axe	ST259037	184	6.1	0	5	Very cloudy when sampled - no follow-up

Table 5.9 Continued/...2

SITE	CATCHMENT	N.G.R.	BMWP	ASPT	% FUNGUS COVER Above	Below	POSSIBLE SOURCES OF POLLUTION
GROUP 2							
1 Lamberal Water below Forda	Tamar	ST274109	106	5.9	40	50	No problem noted by warden in May 1991
3 Small trib. near Crossgate	Tamar	SX346884	92	5.1	60	40	
8 Trib. near Black Torrington	Torridge	SX463056	108	5.7	35	25	Dirty water overflow and domestic sewage
14 Trib. of Yeo at Looseham Bridge	Torridge	SS438217	58	5.3	10	15	Yard water and abuse of LRI system
15 Smithsland Stream below farm	Torridge	SS473059	83	5.2	30	20	Dirty water irrigation
16 Trib. of Taw at Aller Bridge	Taw	SS705067	35	3.9	85	90	Pig slurry & dirty water from dairy farm
17 Gissage Lake	Taw	SS706037	124	5.9	5	25	Problem unknown, no follow-up.
19 Trib. below Churchlands Farm	Taw	SS660165	133	6.3	0	0	Problem unknown, no follow-up.
27 River Clyst North of Clyst Hydon	Exe	ST038019	56	4.0	10	75	Yard runoff problem & other possibles
29 Trib. of Madford near Dunkerswell	Exe	ST149083	86	4.9	0	0	Immediately D/s Dunkerswell STW
34 Trib. of Otter at Broadhembury	Otter	ST101049	125	6.0	0	0	Historic problems, no recent reports
40 Otterton Brook near North Star	Otter	SY087858	74	4.6	40	40	No file on stream, no reported pollution
46 Trib. of Axe at Whitford	Axe	SY262956	141	5.2	< 1	10	Yard runoff
50 Buckland Stream below Blindmoor	Axe	ST263136	120	5.7	0	0	Slurry containment problems at two farms
51 Temple Brook at Greenham	Axe	ST410047	70	4.1	50	20	Intermittent discharge from large farm

Figure 5.10 Indicator system for the rapid assessment of farm pollution in Devon.



Pollution Sources

Information from pollution control staff in NRA South-west Region has been received for polluted sites on all catchments. The information received was of variable quality, some of it being rather out-dated, and only one attempt was made to follow-up reports of pollution where there was no suspected cause (Table 5.9).

5.2.4 Discussion

TWINSPAN analysis was successful in producing an apparent segregation of polluted from unpolluted sites at the first level of division as indicated by differences between groups in pollution related variables such as ASPT, BMWP score and ammoniacal Nitrogen. There was no significant difference in BOD but this is not surprising since water quality information came from just one spot sample. Subsequent TWINSPAN divisions did not seem to relate closely to organic loading. This may be related to the low number of sites found to be suffering from significant pollution resulting in poor discrimination at this end of the spectrum - the data set from which the West Wales key was derived had far more polluted sites (Section 5.1). Analysis of data at the species/genus level has provided more discrimination (Section 5.3) but it may be profitable to sample more polluted sites in Devon to derive a better key.

5.3 An indicator system for Devon (Spring): 'Species' level

5.3.1 Introduction

Section 5.2 described the development of an indicator key for use in the rapid assessment of farm pollution in Devon (NRA South-west Region). The methodology employed was similar that used in Section 5.1 to develop a similar bankside method for West Wales. TWINSPAN classification of the macroinvertebrate data was undertaken only at the 'family' level. It was suggested that the greater taxonomic penetration might yield a more discriminating key which would more valuable in the pollution control strategy for which it was developed (Seager, *et al.*, 1992). The current section describes analysis of the Devon data set at the species/genus level.

5.3.2 Methods

The methodology was as for Section 5.2 except that TWINSPAN classification was carried out on macroinvertebrate data grouped at the species or genus level where possible (data in Appendix F9). Substratum (phi) was calculated using the methods of Wright *et al.* (1984). Also the distributions of different invertebrate taxa between TWINSPAN groups were investigated using Chi-squared tests (Chi^2). Analyses were carried out with due allowance for different levels of taxon abundance; e.g. *Baetis* spp. (2) indicated > 9 individuals in the sample, whilst *Baetis* spp. (3) indicated > 99 individuals.

5.3.3 Results

Generation of an indicator key

Initial TWINSPAN classification of the macroinvertebrate data for the fifty-one sites produced an indicator key which incorporated taxa which were unsuitable for enumeration in bankside samples. These taxa were masked out as potential indicators in subsequent analysis. This process was repeated until an indicator key was derived which was suitable for use in the field (Figure 5.11). Taxa excluded were *Potamopyrgus jenkinsi, Ancylus fluviatilus, Erpobdella octoculata, Paraleptophlebia* spp, *Chloroperla* spp, *Elminthidae* and *Dicranota* spp.

Characteristics of stream TWINSPAN groups

Preliminary assessment of the stream groups suggested that the two groups to the left of the initial division were similar in biotic and physicochemical terms. DECORANA reflected this similarity by marked overlap of the site groups

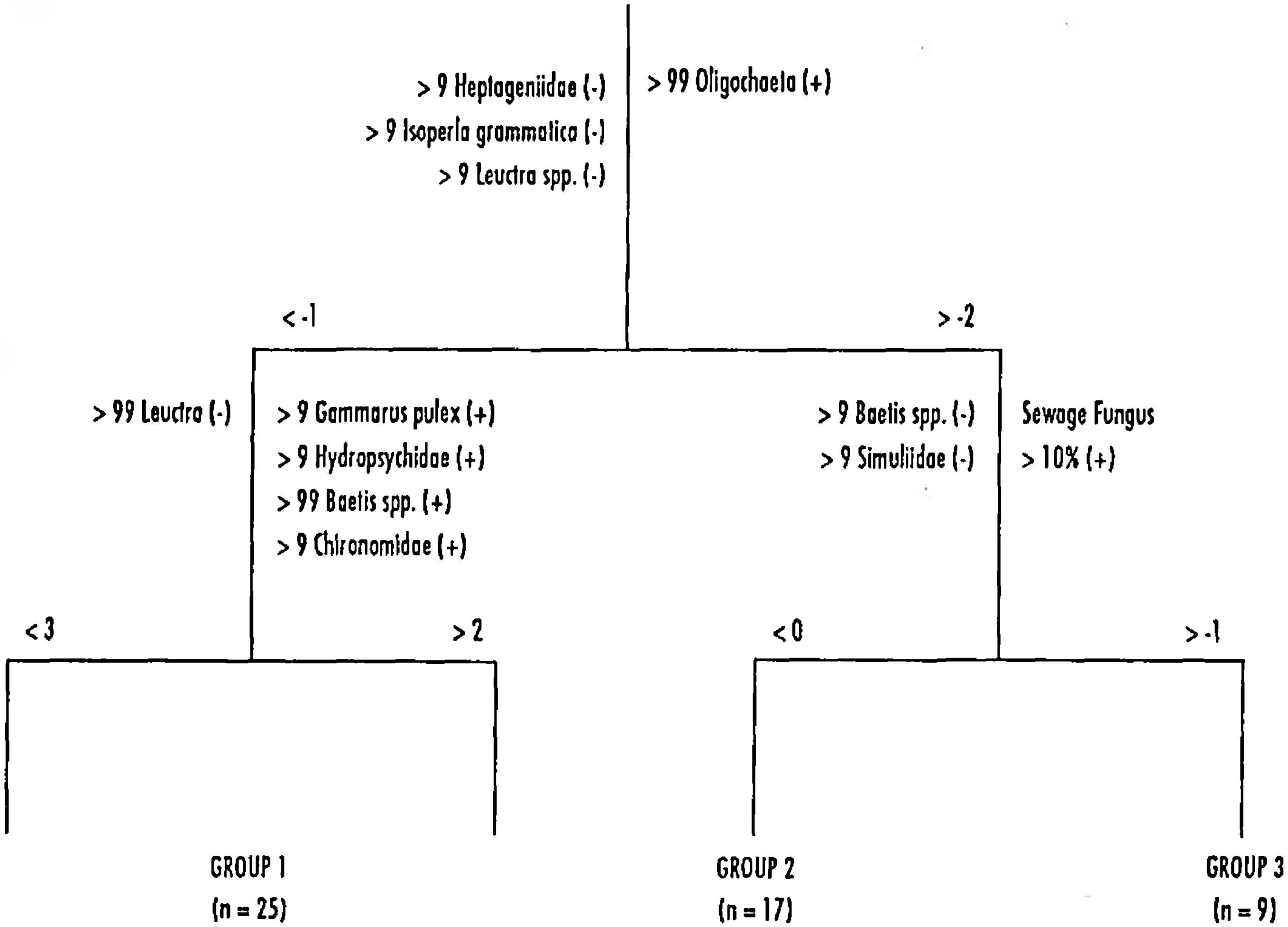


Figure 5.11 Indicator key for Devon derived from TWINSPAN analysis of macroinvertebrate data from 51 sites

in ordination space (Figure 5.12). These two groups were therefore combined so as to produce three groups (1, 2 and 3) in Figure 5.11.

There were highly significant differences between the three groups generated in the means of variables which might reflect the effects of farm pollution (Table 5.10, Figure 5.13). These were BMWP Score ($p < 0.001$), ASPT ($p < 0.001$), ammoniacal nitrogen ($p = 0.002$). There was no significant difference in BOD.

There were also significant differences for altitude ($p = 0.008$), conductivity ($p = 0.002$) and alkalinity ($p = 0.046$) but not for gradient, width, distance from source, substratum composition or pH (Table 5.10).

Analysis of the distribution of macroinvertebrate taxa between the three TWINSPAN groups by Chi^2 tests indicated that there were a number of taxa in addition to the indicators showing preference for the less polluted groups (1 and 2) (Table 5.11). These tended to be those of pollution sensitive stoneflies (*Protonemoura, Amphinemeura, Perlodes microcephala, Chloroperla, Bracyptera risi, Isoperla grammatica*), caddis (*Silo pallipes, Odontocerum albicorne, Hydropsychidae, Sericostoma personatum, Rhyacophila dorsalis*) and beetles (*Elminthidae and Gyrinidae*). There were a few pollution-tolerant taxa showing preference for the more polluted groups (2 and 3) (Table 5.11). These included the leeches *Erpobdella octoculata* and *Glossiphonia complanata, oligochaetes* and Asellus.

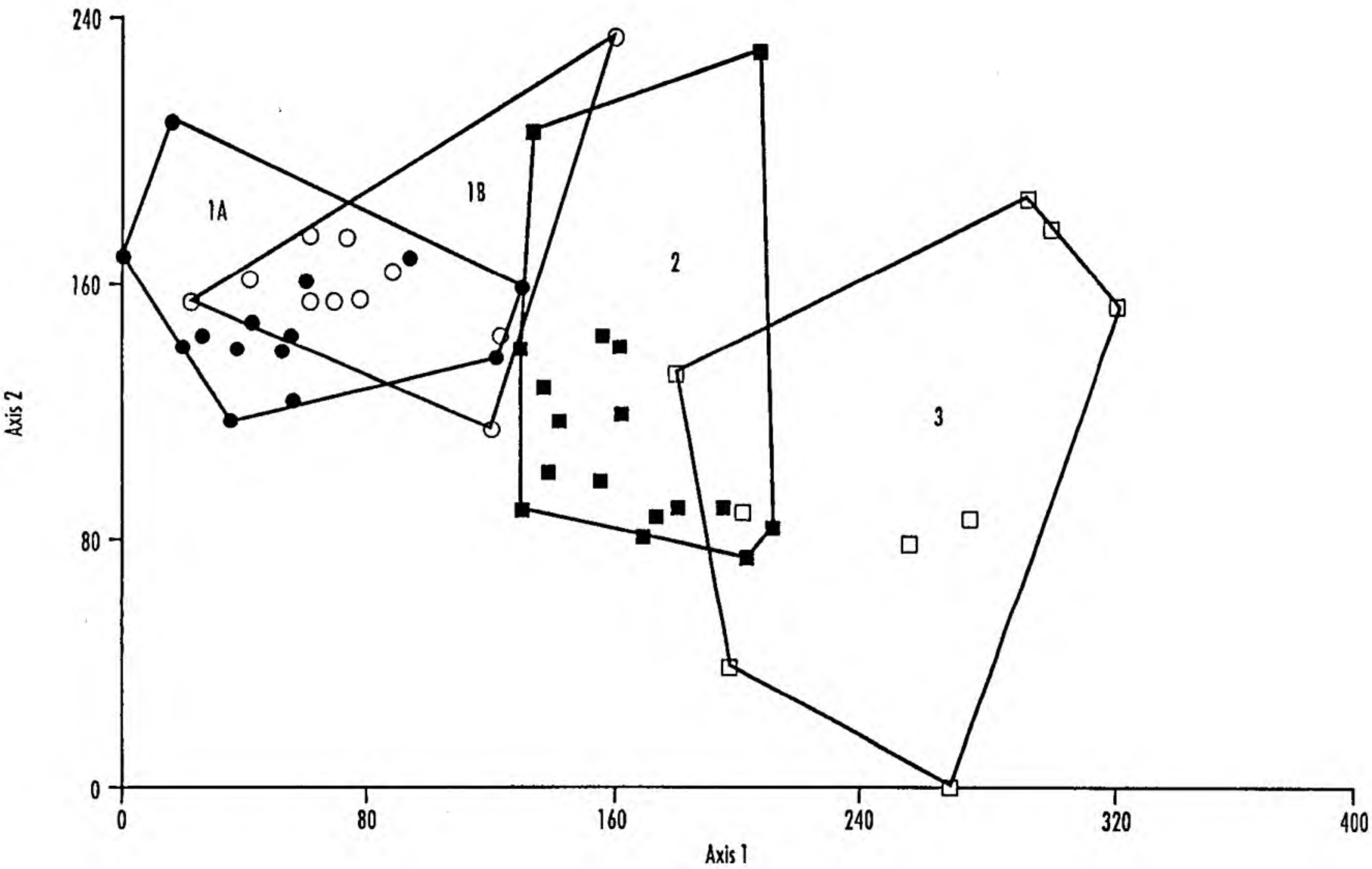


Figure 5.12 Ordination of 51 sites in Devon by DECORANA. Numbered polygons denote site groups generated by TWINSPAN

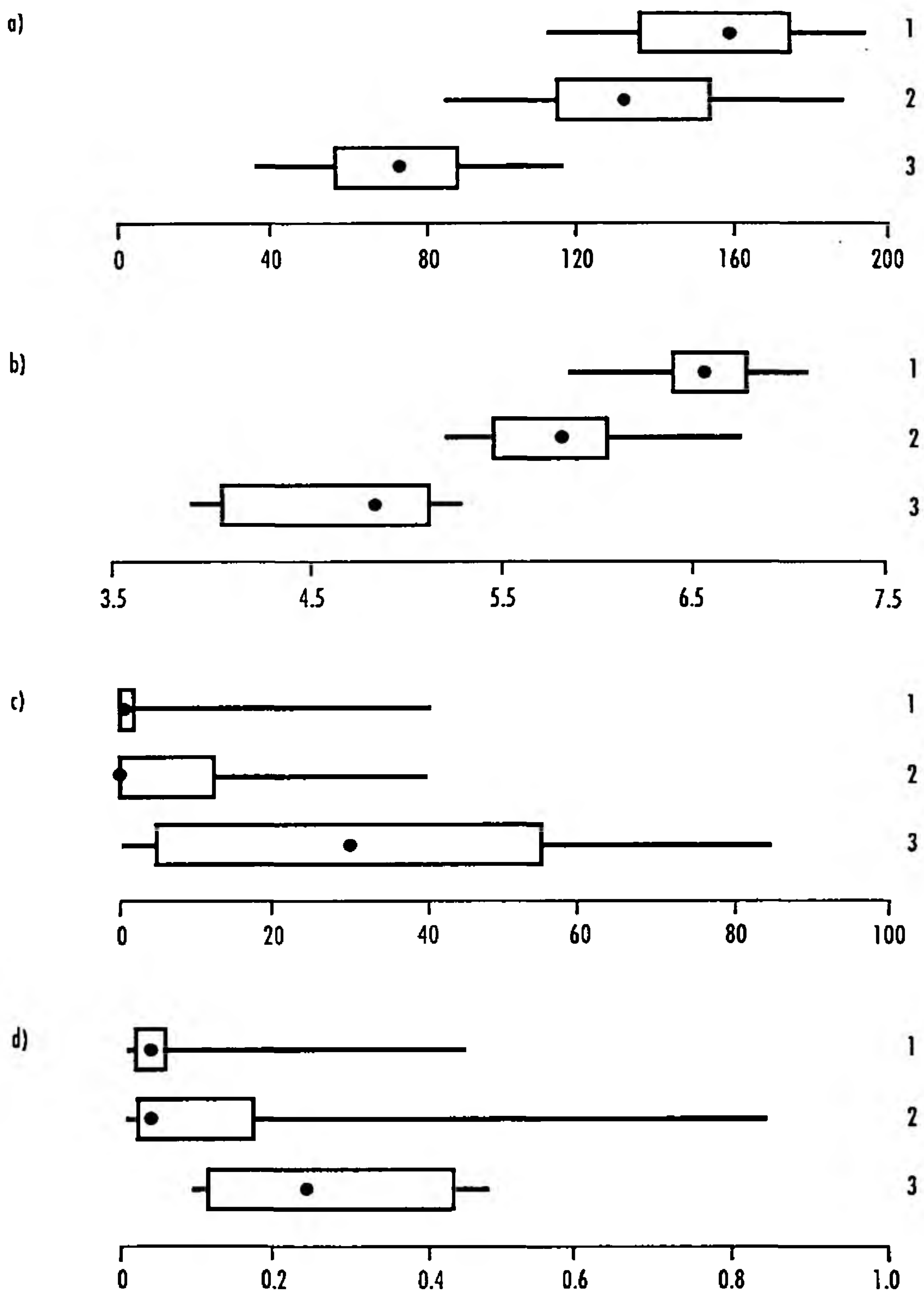


Figure 5.13 Distribution of a) BMWP score, b) ASPT, c) % sewage fungus cover and d) Ammoniacal Nitrogen (mg l-1) concentration within TWINSPAN groups derived from the 51 sites in Devon

Table 5.10 Relationships between TWINSPAN groups and environmental and biotic variables for the Devon Spring data set

Variable (Units)	Group 1 (n=25)	Group 2 (n=17)	Group 3 (n=9)	F	p
Pollution dependent variables					
BMWP	157 (135-178)	134 (108-159)	74 (51-98)	41.17	< 0.001***
ASPT	6.6 (6.2-6.9)	5.8 (5.4-6.2)	4.6 (4.1-5.2)	84.16	< 0.001***
BOD (mgl^{-1})	1.5 (0.3-2.6)	1.6 (0.4-2.8)	1.8 (0.7-3.0)	0.33	0.724
Ammoniacal N + (mgl^{-1})	0.04 (0.01-0.10)	0.05 (0.01-0.2)	0.21 (0.1-0.4)	7.24	0.002**
Pollution independent					
Altitude (m)	134 (79-189)	102 (54-150)	72 (51-98)	5.35	0.008**
Gradient (%) +	1.6 (1.1-2.4)	1.5 (0-2.3)	1.2 (0-3.3)	0.71	0.496
Width (m)	2.3 (1.4-3.1)	1.9 (1.3-2.4)	1.6 (0.9-2.3)	3.17	0.051
Distance from source (Km)	2.9 (1.3-4.5)	2.5 (1.4-3.7)	2.9 (0.3-5.6)	0.25	0.779
pH	7.3 (6.9-7.8)	7.5 (7.1-7.8)	7.5 (7.2-7.7)	0.87	0.426
Conductivity ($uScm^{-1}$)	190 (103-276)	252 (151-352)	327 (224-430)	6.99	0.002**
Alkalinity + (mgl^{-1} $CaCO_3$)	26 (11-63)	39 (22-67)	53 (33-85)	3.31	0.046+
Substratum (phi)	-4.4 (-5.7,-3.0)	-3.6 (-5.3,-2.0)	-3.9 (-6.4,-1.4)	0.95	0.392

+ Denotes variables which were log-transformed prior to analysis. Group values are means with standard deviation range (+ or - 1 S.D.) in brackets. F statistic (F) and probability (p) are from analysis of variance. Asterisks indicate level of probability - * < 0.05, ** < 0.01, *** < 0.001.

Table 5.11 Frequency of occurrence of selected invertebrate taxa in the three TWINSPAN groups

TAXON	GROUP 1 (n=25)	GROUP 2 (n=17)	GROUP 3 (n=9)	Chi^2	p
Taxa associated with Group 1					
Heptageniidae (2)	*****	*		29.3	< 0.001
Heptageniidae (3)	***			22.1	< 0.001
Brachyptera risi (2)	**	*		8.5	< 0.02
Leuctra spp. (2)	***	*		10.6	< 0.01
Perlodes microcephala	****	*		23.1	< 0.001
Isoperla grammatica (2)	****	*		24.0	< 0.001
Chloroperla spp.	*****	**		23.4	< 0.001
Chloroperla spp. (2)	***	*		16.8	< 0.001
Protonemoura spp.	**	*		7.1	< 0.05
Amphinemeura sp.	***	*	*	12.1	< 0.01
Odontocerum albicorne	**			11.4	< 0.01
Taxa associated with Groups 1 and 2					
Ancylus fluviatilis	****	*****	**	9.2	< 0.02
Ancylus fluviatilis (2)	**	***	*	7.9	< 0.02
Heptageniidae	*****	*****		37.2	< 0.001
Paraleptophlebia spp.	****	****	**	8.7	< 0.02
Baetis spp. (2)	*****	*****	**	13.1	< 0.01
Baetis spp. (3)	***	***		7.4	< 0.05
Brachyptera risi	****	***	*	14.1	< 0.001
Leuctra spp.	*****	****	*	23.4	< 0.001
Isoperla grammatica	*****	****	*	21.9	< 0.001
Gyrinidae	*****	***	*	15.2	< 0.001
Elminthidae	*****	*****	***	11.3	< 0.01
Elminthidae (2)	****	****	**	7.7	< 0.05
Silo pallipes	**	*		6.5	< 0.05
Sericostoma personatum	*****	***	**	13.6	< 0.01
Sericostoma personatum (2)	**	*		6.5	< 0.05
Rhyacophila dorsalis	*****	****		29.4	< 0.001
Hydropsychidae	*****	****	***	8.8	< 0.02
Hydropsychidae (2)	****	**		15.9	< 0.001
Dicranota spp.	*****	*****	**	17.0	< 0.001
Simuliidae (2)	*****	*****	***	6.2	< 0.05
Wiedemania spp.	**	*		8.5	< 0.02

Table 5.11 continued

TAXON	GROUP 1 (n=25)	GROUP 2 (n=17)	GROUP 3 (n=9)	Chi^2	p
Taxa associated with Group 2					
Habrophlebia fusca		***	*	13.1	< 0.01
Psychodidae	*	***		7.6	< 0.05
Taxa associated with Groups 2 and 3					
Potamopyrgus jenkinsi	***	*****	****	9.0	< 0.02
Pisidium spp.		**	**	9.9	< 0.01
Oligochaeta (3)	*	***	**	13.1	< 0.01
Glossiphonia complanata	*	****	***	15.3	< 0.001
Asellus spp.	*	***	****	14.9	< 0.001
Chironomidae (2)	****	*****	*****	8.4	< 0.02
Taxa associated with Group 3					
Erpobdella octoculata	*	**	****	6.7	< 0.05
Asellus spp. (2)		*	***	11.4	< 0.01
Dytiscidae	*	*	***	13.1	< 0.01
Taxa not showing association with Groups					
Polycelis spp.	**	***	***	2.1	N.S.
Polycelis spp. (2)	*	*	**	0.6	N.S.
Potamopyrgus jenkinsi (2)	**	****	***	3.5	N.S.
Lymnaea peregra	*	**	*	4.9	N.S.
Pisidium spp.	***	****	*****	2.6	N.S.
Oligochaeta	*****	*****	*****	0.0	N.S.
Oligochaeta (2)	****	*****	*****	4.5	N.S.
Gammarus pulex	*****	*****	*****	2.1	N.S.
Gammarus pulex (2)	***	****	***	2.1	N.S.
Ephemera danica	***	***	**	1.8	N.S.
Paraleptophlebia (2)	**	**		4.4	N.S.
Caenis spp.	**	*	*	4.0	N.S.
Nemoura spp.	***	****	***	1.3	N.S.
Gyrinidae (2)	**	*		4.2	N.S.
Hydraena gracils	***	***	*	4.7	N.S.
Rhyacophila dorsalis (2)	**	**		4.4	N.S.
Agapetus spp.	**	*	*	0.8	N.S.
Plectocnemia sp.	***	***	*	3.7	N.S.
Limnephilidae	*****	*****	****	5.9	N.S.
Limnephilidae (2)	***	**	**	2.4	N.S.

Table 5.11 continued

TAXON	GROUP 1 (n=25)	GROUP 2 (n=17)	GROUP 3 (n=9)	Chi^2	p
Hydropsychidae (3)	*	*		3.4	N.S.
Dicranota spp. (2)	**	*		2.6	N.S.
Simuliidae (3)	*	**	**	3.2	N.S.
Chironomidae (3)	*	***	***	4.4	N.S.
Ceratopogonidae	*	***	**	3.5	N.S.
Chelifera spp.	*	**	**	2.0	N.S.

Asterisks indicate percentage occurrence (* 1-20 % of sites in group, ** 21-40 %, *** 41-60 %, **** 61-80 %, ***** 81-100 %).

For certain taxa abundance categories are treated as different taxa, i.e. (2) indicates > 9 individuals in a 1 minute kick sample, (3) > 99. Chi^2 values and associated probabilities are given.

A key for use in the field

In devising a flow chart for use in the field it was thought desirable to include sewage fungus at a further stage in the system such that sites in Group 1 with some sewage fungus present (1a) can be distinguished from those without (Figure 5.14). Sites in Group 1b can be said to show evidence of mild organic input and could be investigated by surveying further upstream. On this basis only 8 of the 25 sites in Group 1 could be placed into Group 1a, thus probably reflecting a persistent low level of organic pollution at the remaining sites.

Pollution Sources

The information provided by pollution control staff about possible pollution sources was of variable quality. In some cases the information was vague and out-dated, in other cases more detail was provided and recent site visits had been carried out. For many sites in Groups 2 and 3, there were possible sources for observed impacts on the stream biota (Table 5.12).

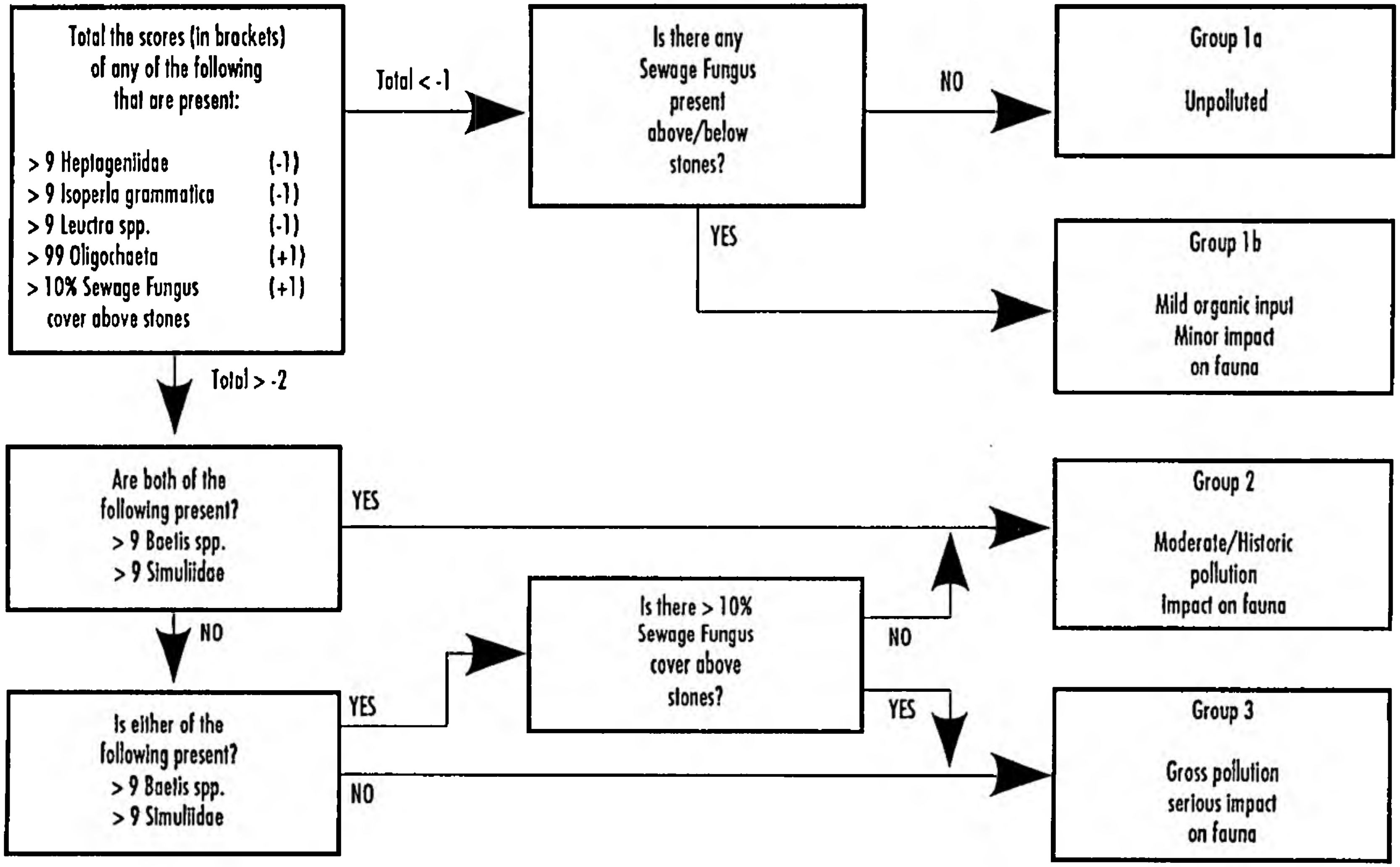


Figure 5.14 Indicator system for the rapid assessment of farm pollution in Devon

Table 5.12 Biological characteristics of stream sites in Devon arranged by pollution group. Information on possible pollution sources provided by Pollution Control staff are given

SITE	CATCHMENT	N.G.R.	BMWP	ASPT	% FUNGUS COVER ABOVE	BELOW	POSSIBLE SOURCES OF POLLUTION
GROUP 1a							
21 Trib of Yeo	Taw	SS792276	160	6.7	0	0	
22 Lyddicombe Bottom at Fyldon	Taw	SS738337	175	7.0	0	0	
23 Trib of Bray at Holewater	Taw	SS705354	163	7.1	0	0	
31 Dane's Brook	Exe	SS819321	122	6.4	0	0	
39 Otter below Sweetlands Farm	Otter	ST216096	163	6.5	0	0	
44 Trib. of Axe below Trill	Axe	SY286959	174	6.4	0	0	
48 Yarty near Moorseek	Axe	ST245134	194	6.9	0	0	
49 Trib. of Axe at Chalkway	Axe	ST376075	134	6.7	0	0	
GROUP 1b							
4 Trib. in Witherdown Wood	Tamar	SX428953	129	6.8	40	20	No problem when site visited on 14.6.91
5 Trib. d/s Ogbeare Hall	Tamar	SX308096	134	5.8	2	10	
7 Trib. North of Northcott Hamlet	Tamar	SX334934	153	6.7	2	2	
10 Trib. of Lew below Whiddon Farm	Torridge	SX479999	147	6.4	7	15	Dirty water runoff
11 Trib. of Lew East of Patchcott	Torridge	SX478994	150	6.3	5	10	Dirty water problem. LRI now installed.
12 Medland Brook near Northwood	Torridge	SS552002	141	6.7	0	2	
13 Trib. of Lew West of Westacott	Torridge	SS532002	112	6.6	1	1	
25 Trib. of Culm at Stapley	Exe	ST186134	144	6.6	0	< 5	
28 Trib. of Madford near Abbey	Exe	ST141107	177	6.1	0	< 5	Possible discharges from septic tanks.
30 Trib. of Madford D/S Gorwell	Exe	ST153094	186	6.9	0	5	
33 River Wolf at A373 road bridge	Otter	ST129022	162	6.5	1	15	Many historic problems in catchment
35 Trib. of Otter at Luppitt	Otter	ST173066	183	6.8	?	10	A few historic problems - none recently.
36 Trib. of Otter at Upottery	Otter	ST199078	160	6.4	0	5	S. fungus in trib. below Chapelhayes Farm
38 Trib. of Otter at Blannicombe	Otter	SY161985	160	7.0	5	5	
41 Gissage at Gittisham	Otter	ST135983	135	6.1	5	50	Yard runoff from several farms
42 Trib. of Axe at Sector	Axe	SY311982	172	6.6	1	3	Waste containment problems from 2 farms
47 Trib. of Yarty at confluence	Axe	ST259037	184	6.1	0	5	Very cloudy when sampled - no follow-up

Table 5.12 Continued/...2

SITE	CATCHMENT	N.G.R.	BMWP	ASPT	% FUNGUS COVER ABOVE	BELOW	POSSIBLE SOURCES OF POLLUTION
GROUP 2							
1 Lamberal Water below Forda	Tamar	ST274109	106	5.9	40	50	No problem noted by warden in May 1991
2 Lamberal Water nr. Broxwater	Tamar	SS267133	159	6.1	20	20	No problem noted by warden in May 1991
6 Tala Water near Boyton	Tamar	SX312919	189	6.8	2	5	
8 Trib. near Black Torrington	Torridge	SX463056	108	5.7	35	25	Dirty water overflow & domestic sewage
9 Trib. of Waldon	Torridge	SS389106	162	6.0	30	10	Inadequate waste containment
16 Trib. near Stroxworthy	Torridge	SS344198	150	6.3	1	0	Domestic sewage and dirty water
18 Gissage Lake	Taw	SS706037	124	5.9	5	25	Problem unknown, no follow-up.
19 Trib. below Churchlands Farm	Taw	SS660165	133	6.3	0	0	Problem unknown, no follow-up.
20 Trib. of Bray at Fullabrook	Taw	SS674257	85	5.3	0	3	
24 Trib. below Deptford Farm	Taw	SS598192	140	5.8	0	15	Dirty water problem - L.R.I. proposed
32 Trib of Batherm D/S Raddington	Exe	ST011257	135	5.4	0	< 5	No known problems, six farms upstream
34 Trib. of Otter at Broadhembury	Otter	ST101049	125	6.0	0	0	Historic problems, no recent reports
37 Trib of Tale at Haskin's Farm	Otter	ST083027	125	5.4	0	0	
43 Trib. below Bagley Hill Farm	Axe	ST321010	162	5.8	0	5	Yard runoff - grant approved for L.R.I.
45 Trib. below Whitwell Farm	Axe	SY238925	110	5.5	0	< 2	Yard runoff - L.R.I. installed
46 Trib. of Axe at Whitford	Axe	SY262956	141	5.2	< 1	10	Yard runoff
50 Buckland Stream below Blindmoor	Axe	ST263136	120	5.7	0	0	Slurry containment problems at two farms
GROUP 3							
3 Small trib. near Crossgate	Tamar	SX346884	92	5.1	60	40	
14 Trib. of Yeo at Looseham Bridge	Torridge	SS438217	58	5.3	10	15	Yard water and abuse of LRI system
15 Smithsland Stream below farm	Torridge	SS473059	83	5.2	30	20	Dirty water irrigation
17 Trib. of Taw at Aller Bridge	Taw	SS705067	35	3.9	85	90	Pig slurry & dirty water from dairy farm
26 River Weaver at B3181 Bridge	Exe	ST013033	116	4.8	0 +	0	Suspect farm upstream
27 River Clyst North of Clyst Hydon	Exe	ST038019	56	4.0	10	75	Yard runoff problem & other possibles
29 Trib. of Madford near Dunkerswell	Exe	ST149083	86	4.9	0	0	Immediately D/S Dunkerswell STW
40 Otterton Brook near North Star	Otter	SY087858	74	4.6	40	40	No file on stream, no reported pollution
51 Temple Brook at Greenham	Axe	ST410047	70	4.1	50	20	Intermittent discharge from large farm

+ Sewage Fungus strands on vegetation only.

5.3.4 Discussion

TWINSPAN classification was successful in defining three groups of different biological status as reflected in differences in BMWP scores, ASPT and ammoniacal nitrogen concentration. Group 1 sites generally had BWMP scores in excess of 140 and ASPT in excess of 6.4 and could be considered to represent largely unpolluted sites (Table 5.12, Figure 5.13). By contrast, Group 3 sites were the most polluted, with BMWP scores generally below 90, ASPT of all sites below 5.4 and frequently with substantial growths of sewage fungus. Group 2 sites were intermediate reflecting mild or perhaps historic pollution. There were significant differences between the three groups in altitude, conductivity and alkalinity but these are not thought to be major determinants of faunal composition in this study. The differences are likely to reflect the fact that unpolluted catchments tended to be at higher altitude and draining poorer soils where sheep farming rather than dairy farming will be the commonest land use.

The indicator key generated by TWINSPAN incorporated taxa whose distribution and abundance have been shown to be influenced by organic pollution both by work under this contract (Section 7) and by numerous other studies (e.g. Hynes, 1960; Hawkes, 1963). The mechanisms which may account for the absence of sensitive taxa from sites affected by organic pollution may involve both acute and chronic effects dependent on pollutant type. These relationships have been discussed more fully elsewhere (e.g. Rutt *et al.* 1993).

The TWINSPAN classification reported here was carried out at the 'species' or 'genus' level by contrast with the 'family' level analysis of the same data reported previously (Section 5.2). It would appear that the use of data at the 'species/genus' level has produced a more discriminating analysis, with divisions more closely reflecting the effects of farm pollution. The indicators are very similar to those found previously and present comparably few problems for field identification. It is recommended that future developments of indicator keys should be based on species/genus level data which provide more information for the classification process. Unsuitable indicators can always be masked out of analyses which can be repeated until a practical system is obtained.

5.4 <u>An indicator key for West Wales (Summer)</u>

5.4.1 Introduction

The indicator key developed for West Wales in the Spring appears to be applicable in the period December to May inclusive. The summer fauna of welsh streams is considerably different from the winter/spring fauna and certain farm discharges such as silage leakages and parlour/dairy washings tend to become more predominant or evident at this time. It was therefore considered necessary to develop a separate key for surveys of farm pollution during this period. An earlier key developed by Reynolds (1989) had pioneered the indicator approach but had the disadvantages of being inapplicable to smaller streams and was based upon three-minute kick samples which are time-consuming to process on the bankside. This subsection describes the development of a summer indicator system for West wales based upon one minute kick samples.

5.4.2 Methods

Site selection

Twelve of the fifteen catchments surveyed to test the Welsh Spring Key (Section 6, Figure 6.1) were revisited in July 1991 to establish whether any new farm discharges would be discovered by sampling during this period using the indicator system devised by Reynolds (1989). At 99 of the 146 sites one-minute kick samples were taken and 50 of the samples were later chosen for processing to ensure a range of pollution impact, a range of stream type and a range of geographical area.

Fieldwork

Fieldwork largely followed the methodology for the development of the Devon key (Section 5.2) but no spot water samples were taken. Sampling took place between 4 and 26 July 1991. Environmental data can be found in Appendix F10 and invertebrate data in Appendix F11.

Invertebrate sample processing

Macroinvertebrate samples were processed in the laboratory to species or genus level for most groups. Difficult taxa such as simuliids and chironomids were left at family level. BMWP score and ASPT were derived. Data were stored on the NRA Welsh Region data base known as BAETIS (NRA, 1992).

Data analysis

Data analysis followed the principles described in Section 5.2. Substratum composition data were converted to a single variable using the methods of Wright *et al.,* 1984).

5.4.3 Results

Initial TWINSPAN classification of macroinvertebrate data for 50 sites produced an indicator key that incorporated taxa which were unsuitable for enumeration in bankside samples. These taxa were masked out as potential indicators in subsequent analyses. This process was repeated until an indicator key was derived which was suitable for use in the field (Figure 5.15). Taxa excluded were *Elmis aenea, Hydraena gracilis, Limnius volkmari, Chloroperla torrentium, Oligochaeta, Potamopyrgus jenkinsi, Pisidium* spp., *Ancylus fluviatilis,* and *Dicranota* spp.

There were highly significant differences ($p < 0.001$) between the four groups, in the means of BMWP score and ASPT, biotic indices which tend to reflect the degree of organic pollution (Table 5.13). Probable pollution sources could generally be identified for sites in Groups 3 and 4 (Table 5.14). Of variables not dependent on organic pollution, differences in Distance from source, width and substratum composition were just significant ($p < 0.05$).

In devising of flowchart for use in the field, it was thought desirable to include sewage fungus at two further stages in the system (Figure 5.16) as for the West Wales spring key (see Section 5.1.4).

Figure 5.15 Indicator key derived from analysis of 50 sites sampled in West Wales in Summer 1991

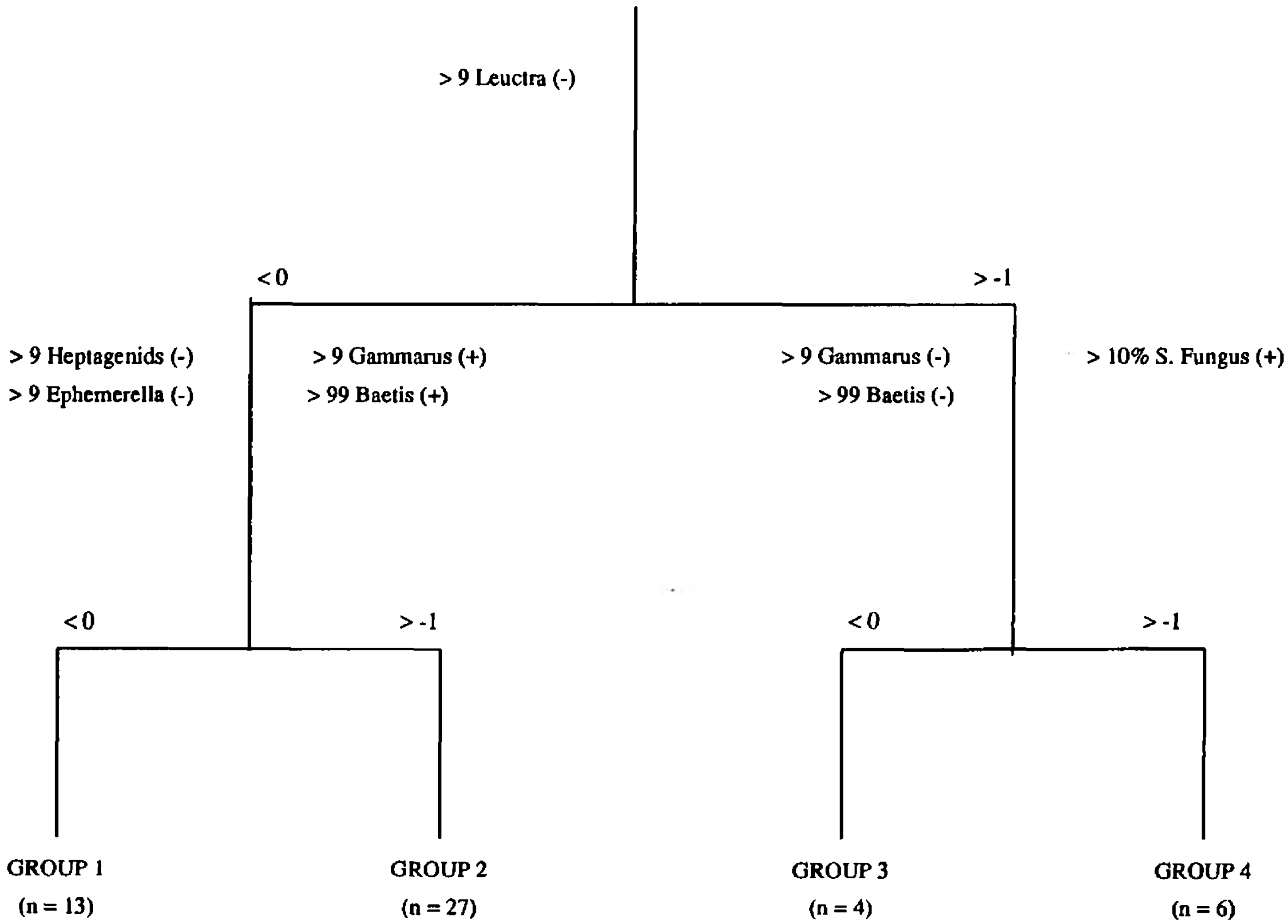


Figure 5.16 Flow chart of rapid appraisal for use
in West Wales in the summer.

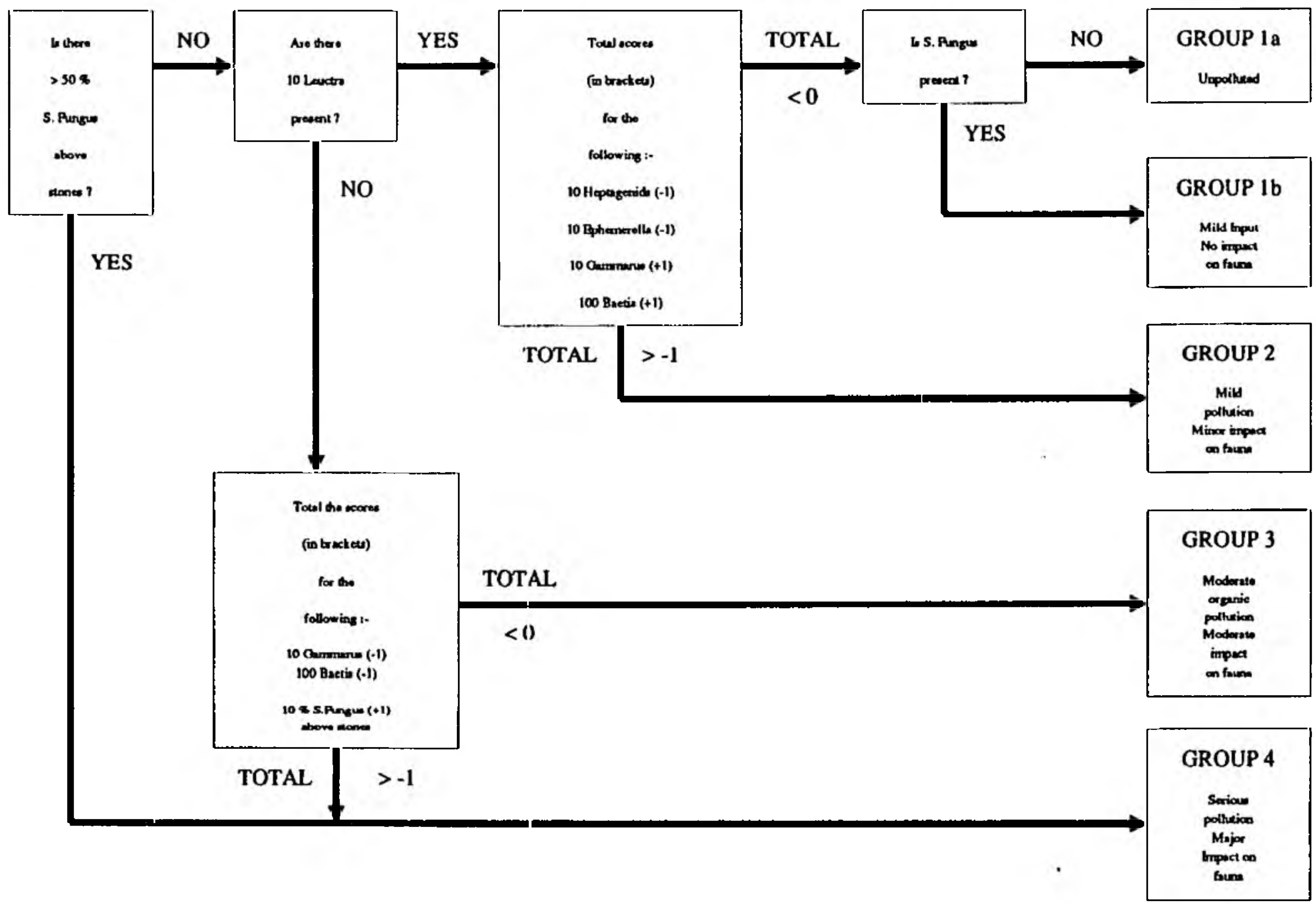


Table 5.13 Relationships between TWINSPAN groups and environmental and biotic variables for the Welsh Summer Indicator Key

Variable (Units)	GROUP 1 (n=13)	GROUP 2 (n=27)	GROUP 3 (n=4)	GROUP 4 (n=6)	F	p
Pollution dependent variables						
BMWP	127 (107-157)	108 (81-135)	91 (76-105)	46 (18-75)	15.13	<0.001***
ASPT	6.6 (6.4-6.8)	5.9 (5.4-6.3)	4.9 (4.5-5.2)	4.6 (3.5-5.6)	29.34	<0.001***
Pollution independent variables						
Altitude (m)	58 (30-86)	70 (20-119)	31 (0-64)	103 (50-155)	2.39	0.081
Dist. from source (Km)	4.5 (1.6-7.4)	2.8 (0.8-6.9)	3.2 (1.1-5.2)	1.1 (0.4-1.7)	3.85	0.019*
Width (m)	2.8 (1.5-4.2)	1.9 (0.6-3.3)	1.8 (1.3-2.2)	0.9 (0.6-1.3)	3.70	0.018*
Gradient + (%)	2.6 (1.0-6.8)	2.0 (1.0-4.0)	1.3 (0.5-3.2)	3.5 (2.0-6.4)	1.82	0.156
Substratum (phi)	-4.9 (-6.4,-3.5)	-4.2 (-6.7,-1.7)	-1.2 (-2.4,-0.1)	-3.5 (-5.5,-1.4)	3.26	0.03*

Group values are means with standard deviation range (+ or - 1 S.D.) in brackets.
F statistic
(F) and probability (p) are from analysis of variance. Asterisks indicate level of probability - * < 0.05, ** < 0.01, *** < 0.001.

Table 5.14 Biological characteristics of stream sites sampled in West Wales in July 1991, giving possible pollution sources

SITE	N.G.R.	BMWP	ASPT	% FUNGUS COVER ABOVE STONES	% FUNGUS COVER BELOW STONES	POSSIBLE SOURCES OF POLLUTION
GROUP 1						
11 Rhydafallen 1	SN096209	120	6.3	0	0	
12 Rhydafallen 2	SN087231	120	6.3	0	0	
17 Llanycefn 3	SN101230	145	6.6	0	0	
18 Llanycefn 4	SN099236	132	6.6	0	0	
35 Peris 8	SN514675	96	6.4	0	0	
36 Peris 7	SN529670	128	6.7	0	0	
37 Tigen 1	SN182268	125	6.6	0	0	
38 Tigen 2	SN192265	154	6.7	0	0	
39 Cynin Trib 1	SN278251	150	6.8	0	0	
40 Cynin Trib 2	SN279250	143	6.8	0	0	
41 Cynin Trib 4	SN284257	137	6.9	0	0	
45 Pibwr 7	SN426180	86	6.6	0	9	
49 Rhydw 5	SN439145	115	6.8	0	2	
GROUP 2						
5. Millin 9	SN005153	71	5.1	0	0	
6. Millin 12	SN001161	97	5.4	0	13	
7. Fenton 1	SM974174	105	5.5	0	1	
8. Fenton 2	SM997174	104	5.8	0	2	
9. Fenton 3	SN019173	103	5.2	0	7	
10. Fenton 7	SN014177	96	5.3	0	1	
13. Rhydafallen 3	SN086240	93	6.2	0	0	
14. Rhydafallen 5	SN083253	90	6.0	50	10	Silage effluent from polluted trib. u/s
20. Llanycefn 10	SN086274	118	6.6	0	0	
21. Duad 2	SN112388	93	5.8	0	0	
22. Duad 4	SN118393	144	6.5	0	0	
23. Duad 6	SN123393	123	6.2	0	0	
24. Duad 13	SN128412	109	5.7	43	6	Waste containment problems at farm u/s
25. Duad 12	SN127407	121	6.4	0	0	

Table 5.14 Continued/...2

SITE	N.G.R.	BMWP	ASPT	% FUNGUS COVER ABOVE STONES	BELOW STONES	POSSIBLE SOURCES OF POLLUTION
26. Cou 1	SN292422	111	6.2	0	0	
27. Cou 2	SN286434	93	5.5	0	0	
28. Cou 4	SN286436	76	5.4	0	0	
29. Cwerchyr 1	SN354412	109	6.1	0	10	
30. Cwerchyr 2	SN354419	103	5.7	0	0	
31. Cwerchyr 6	SN372432	136	6.5	0	0	
32. Cwerchyr 9	SN369449	135	6.8	0	0	
34. Peris 1	SN572664	118	5.9	0	0	
42. Pibwr 2	SN421180	117	5.3	0	5	
43. Pibwr 5	SN435181	152	6.3	0	0	
44. Pibwr 6	SN426180	122	5.8	0	0	
45. Pibwr 7	SN426180	86	6.6	0	9	
46. Rhydw 3	SN439145	106	5.6	0	0	
47. Rhydw 8	SN443154	133	6.0	0	2	Yard water from two farms upstream
GROUP 3						
1. Millin 2	SM993148	77	4.5	50	0	Silage input from trib. upstream
2. Millin 4	SM997156	79	4.6	1	4	Silage input from trib. upstream
4. Millin 7	SM991158	106	5.0	0	2	Cattle wade upstream
47. Rhydw B	SM429145	100	5.3	0	1	Silage effluent from overflowing pond
GROUP 4						
3. Millin 6	SN003155	28	4.0	100	100	Severe silage effluent discharge u/s
15. Rhydafallen 6	SN253078	54	5.4	100	56	Severe silage effluent discharge u/s
16. Rhydafallen 8	SN076268	25	3.6	0	20	Discharge from small STW
17. Llanycefn 6	SN097248	62	5.2	90	90	Lagoon overflow
33. Cwerchyr 15 u/s	SN347448	92	5.8	0	0	Parlour & dairy washings to spring
48. Rhydw 4	SN429148	17	3.4	42	14	Silage effluent from overflowing pond

5.4.4 Discussion

As for the other keys generated under this contract TWINSPAN classification was again able to generate an indicator key which allotted sites to groups in relation to their degree of organic pollution (Sections 5.1, 5.2, 5.3). Differences between groups in distance from source and stream width probably reflect the fact that the impact of farm pollution will be greatest on small streams which have low dilution potential, especially in the summer months.

The small number of sites in Groups 3 and 4 means that the discriminating power of the system is more suspect than has been the case for other keys. The system will receive testing in Summer 1992 by NRA staff and it may prove necessary to improve discrimination by expanding the data base to include more sites affected by pollution. The system should be applicable from July to September inclusive.

6. TESTING THE RAPID APPRAISAL METHOD

6.1 Evaluation exercise in West Wales

6.1.1 Introduction

Section 5.1 details the development of an indicator key for use in the assessment and detection of organic pollution from agriculture in west Wales. Such methods have advantages over chemical or visual inspection of watercourses as they have the potential for revealing historic or intermittent pollution. The key was derived from TWINSPAN analysis (Hill, 1979a) of biological data from fifty-five sites throughout this area of intensive dairy farming and is applicable in late winter and early spring. It is based upon assessing the abundance of indicator taxa in a one-minute benthic invertebrate sample and quantifying the growth of 'sewage fungus' (Curtis, 1969) on the stream bed (see methods). The key is rapid in operation and is intended as a bankside means of rapidly pin-pointing pollution sources, forming part of co-ordinated pollution control programme (Seager, *et al.,* (1992). Once polluting farms have been identified, farm visits can be arranged and remedial measures proposed. The indicator approach can be employed subsequently to assess the effectiveness of pollution control measures in improving water quality and biological status in each catchment. This section describes a large-scale exercise designed to test out this whole approach to the farm pollution problem.

6.1.2 Methods

Initial survey using rapid appraisal

After discussion with NRA pollution control staff, fifteen catchments were selected for survey (Figure 6.1). Fourteen were within the area of west Wales in which the indicator key was developed (local Pollution Control areas A, B and C). All but one of these catchments lay within areas having densities of dairy cattle of 0.6-1.2 head/Ha catchment and of beef cattle of 0.3-0.9 head/Ha

Figure 6.1 Map of West Wales showing the fifteen catchments surveyed and their relationship to major river systems.

AREA A

1. Millin Brook
2. Fenton Brook
3. Cartlett Brook (top)
4. Slade Brook
5. Rhydyfallen Stream
6. Llanycefn Brook
7. Nant Duad

AREA B

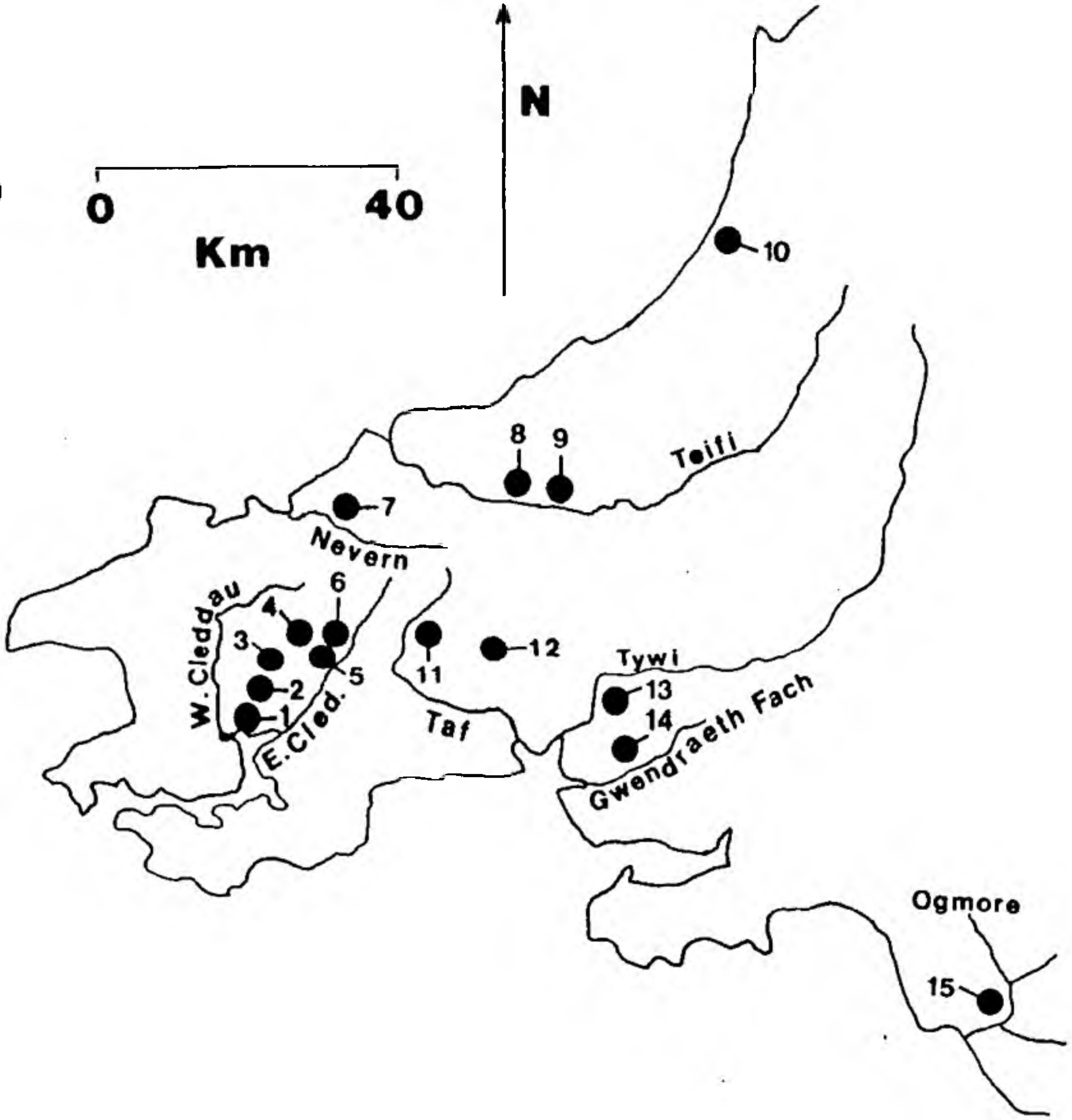
8. Nant Cou
9. Nant Cwerchyr
10. Afon Peris

AREA C

11. Afon Tigen
12. Trib of Cynin
13. Nant Pibwr
14. Nant Rhydw

AREA E

15. Nant Ffornwg



catchment as revealed by the national assessment work carried out for this contract (Section 4). Volumetric organic waste loadings to these were consequently high, giving rise to a high pollution risk. The remaining catchment, the Nant Ffornwg, was selected so as to examine the effectiveness of the system outside the area for which the key was originally designed (Figure 6.1). The density of dairy cattle was much lower here but there were 0.6-0.9 beef cattle/Ha catchment. For the fifteen catchments, stream order was between 1 and 3 (Strahler, 1957) and catchment area was in the range 1.8 to 16.3 Km^2 (Table 6.1).

Within each catchment, sites were selected so as to sample all significant tributaries and pin-point pollution sources. This resulted in a range from three sites on Slade Brook (the smallest catchment) to eighteen on the Nant Duad (Table 6.1). A total of 146 sites were examined in all.

Field-work was carried out between 7 February and 27 March 1991. The procedure at each site was as follows:

1. Environmental data such as substratum composition, stream width and depth were collected (data in Appendix G1).

2. The abundance of 'sewage fungus' was assessed by examining five large, submerged stones from riffles and estimating the percentage cover both above and below (data in Appendix G1).

3. A one minute kick sample was taken from a suitable riffle and the sample examined on the bankside to classify the site into one of three groups using the indicator key developed for West Wales (Figure 5.5). The sample was retained and fixed with 4% formaldehyde solution back at the laboratory.

A proportion of the sites were sampled by non-biologists. Five pollution control staff and one bailiff underwent two days training prior to the survey so that they could take part. This enabled an assessment to be made as to whether the method could be used routinely by such operators. They received one days training in the survey methods and were accompanied by a specialist biologist for their first day in the field.

Table 6.1 Characteristics of the study catchments grouped by Pollution Control Area

	River	N.G.R.	AREA (Km^2)	No. sites
AREA A				
1. Millin Brook	W. Cleddau	SM995140	13.1	14
2. Fenton Brook	W. Cleddau	SM974174	7.6	11
3. Cartlett Brook (top)	W. Cleddau	SN005220	6.7	5*
4. Slade Brook	E. Cleddau	SN036225	1.8	3
5. Rhydyfallen Stream	E. Cleddau	SN096207	10.1	9
6. Llanycefn Brook	E. Cleddau	SN103217	9.1	10
7. Nant Duad	Nevern	SN105389	16.3	18
AREA B				
8. Nant Cou	Teifi	SN294420	6.6	5
9. Nant Cwerchyr	Teifi	SN347400	15.6	15
10. Afon Peris	Peris	SN512675	13.7	9
AREA C				
11. Afon Tigen	Taf	SN183267	5.9	8
12. Trib of Cynin	Taf	SN278251	4.7	5
13. Nant Pibwr	Tywi	SN409178	14.4	13
14. Nant Rhydw	Gwendraeth Fach	SN435129	7.4	9
AREA E				
15. Nant Ffornwg	Ogmore	SN897815	9.2	7

* Only the top part of this catchment was covered comprehensively but a further five sites were covered lower down in the catchment and are included in the data analysis.

The macroinvertebrate samples were sorted in the laboratory with the assistance of NRA staff. The numbers of the indicator taxa Heptageniidae, *Gammarus*, *Baetis* and oligochaetes were recorded in order to compare the effectiveness of biologists and non-biologists in classifying sites using the key. A Biological Monitoring Working Party Score (BMWP) score was determined by a standardised 30 minute search through the sample (data in Appendix G1).

In order to test the effectiveness of the indicator key at distinguishing sites of different levels of organic pollution, BMWP score and ASPT (Average BMWP Score Per Taxon) were compared between indicator groups by analysis of variance. Both of these variables are useful indicators of the level of organic pollution (Armitage *et al.*, 1983). Other environmental variables were also compared in this way.

In response to comments from pollution control staff, the original indicator key was modified so as to classify sites into five rather than three groups by splitting the intermediate Group 2 into three. It was felt that this middle group was far two wide ranging in level of impact and needed to be divided. BMWP score and ASPT were again compared between groups to establish whether this more precise classification could be justified in terms of biological differences between the groups.

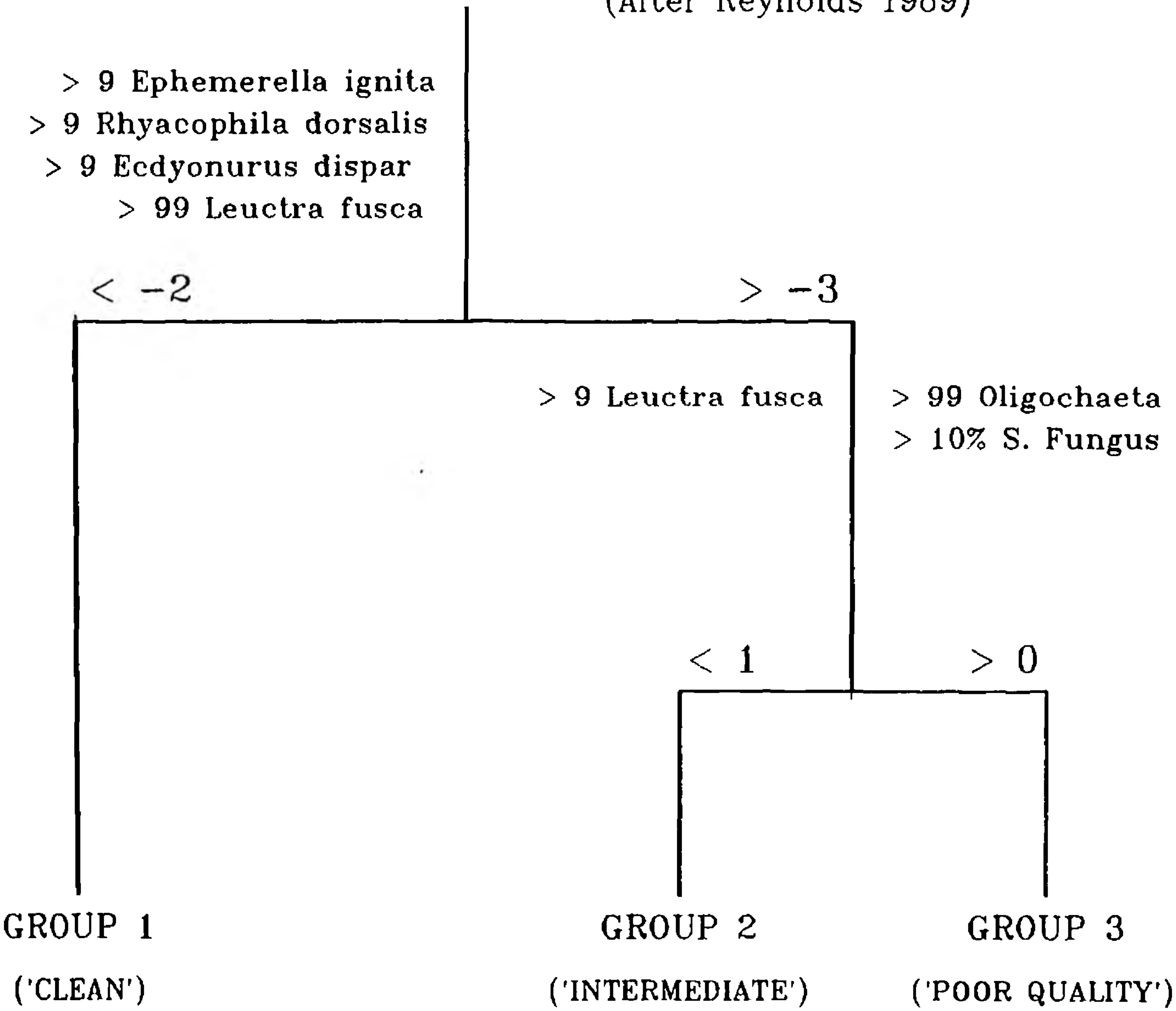
Programme of farm visits

For each catchment a map was produced showing the details of the sites sampled and farms proven or suspected of causing pollution (Appendix G2). This information was rapidly fed-back to the NRA Agricultural Liaison Officer as each catchment survey was completed. The Liaison Officer then initiated a programme of farm visits during which remedial measures could be agreed between the farmers and the NRA.

Summer Farm Pollution survey

Between July 3 and July 26 1991, twelve of the catchments surveyed in the February/March period were surveyed using an indicator key based on three minute kick samples (Figure 6.2) developed for summer use in west Wales by NRA staff (Reynolds, 1989). Insufficient time was available to survey all catchments so Slade Brook, Cartlett Brook and the Nant Ffornwg were not covered. Sampling was generally carried out at the same sites as for the February/March survey, some sites being omitted where lack of pollution allowed a lower sampling intensity on certain sub-tributaries. One minute kick samples were collected from most sites to enable the production of a new summer indicator key which would be more rapid in its operation and more comprehensive in its coverage of stream type (Section 5.4).

Figure **6.2** Indicator key for the rapid assessment of farm pollution impact in the summer in West Wales.
(After Reynolds 1989)



Follow-up survey using rapid appraisal method

The 146 sampling sites were revisited in Spring 1992 between 17 February and 3 March in order to assess any improvements resulting from remedial measures. One site proved too overgrown to be accessible but at the remaining sites sampling proceeded as described for the initial survey. The only difference was that the five group indicator key (Figure 6.3) was employed on the bankside and all sampling was carried out by WRc biologists. Catchment maps were again produced and sent to the NRA Agricultural Liaison Officer (Appendix G3). Due to small numbers found in Groups 2c and 3, differences between the groups in environmental variables and biotic indices were analysed using nonparametric Kruskal-Wallis tests (Sokal & Rohlf, 1981).

6.1.3 Results

Initial survey using rapid appraisal method

With the exception of six sites for which samples were mislaid a laboratory pollution group was determined by examination of the one-minute kick samples. There were ten discrepancies (only 7.1% of the total) between the laboratory pollution group and that derived in the field and the laboratory group was utilized in all further analyses. All of the discrepancies involved sites in Groups 1 and 2. Of the full 146 sites 28 fell into Group 1 (unpolluted), 96 into Group 2 (mild/historic pollution) and 22 into Group 3 (grossly polluted) - see Table 6.2.

Of the 140 sites available for comparison, forty-seven were sampled by non-biologists working individually and there were two discrepancies (4.3%) between field and laboratory grouping. Thirty-seven sites were examined by specialist biologists working individually and there was one discrepancy (3.1%). This suggested that non-biologists were proficient at identifying and enumerating the indicator taxa. Further training may be required in sampling site selection within the catchment. It is important to ensure that catchments are sampled intensively enough to find all pollution sources.

Figure 6.3 Adaptation of the indicator key to provide greater discrimination in the mid-range of pollution effect.

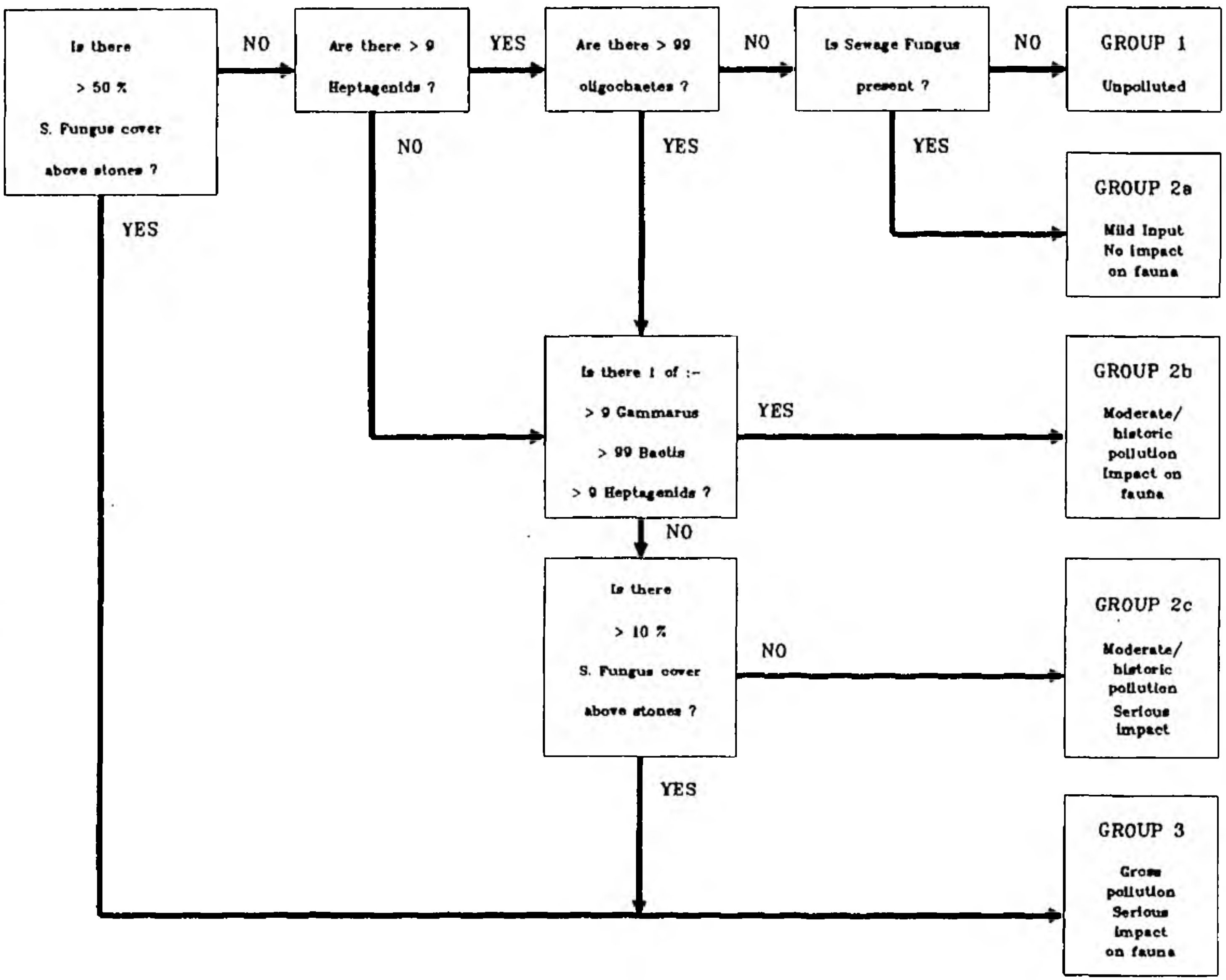
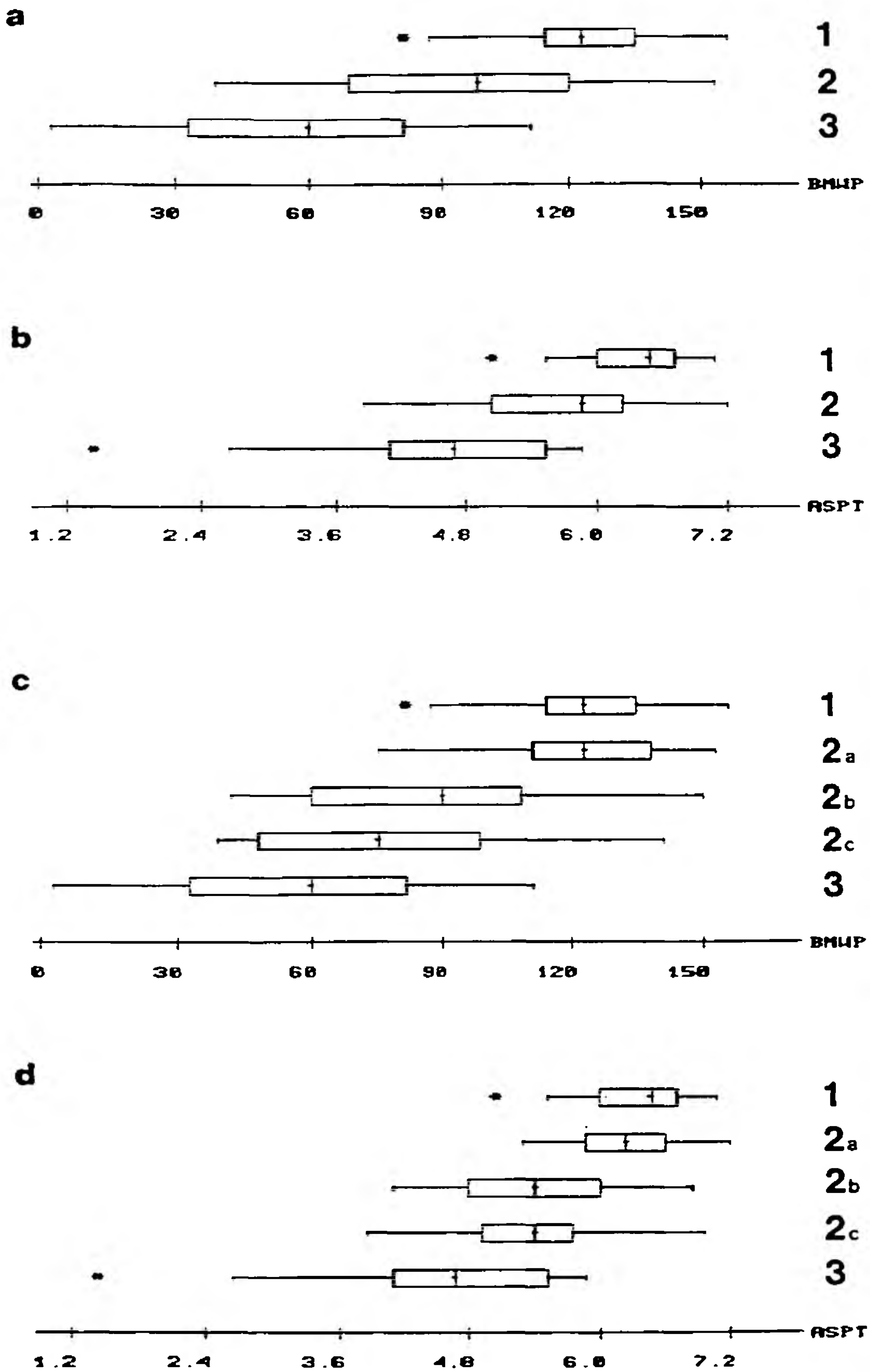


Table 6.2 Pollution groups and polluting farms grouped by catchment and pollution control area

	Pollution Groups			Polluting Farms	
	1	2	3	Definite	Possible
AREA A					
1. Millin Brook	-	7	7	6	2
2. Fenton Brook	-	9	2	4	-
3. Cartlett Brook	-	7	3	2	-
4. Slade Brook	-	1	2	1	-
5. Rhydyfallen Stream	3	5	1	1	-
6. Llanycefn Brook	-	9	1	2	-
7. Nant Duad	2	16	-	4	5
AREA B					
8. Nant Cou	1	3	1	2	2
9. Nant Cwerchyr	9	6	-	1	1
10. Afon Peris	4	5	-	1	-
AREA C					
11. Afon Tigen	4	4	-	-	-
12. Trib of Cynin	4	1	-	-	-
13. Nant Pibwr	-	10	3	3	1
14. Nant Rhydw	-	7	2	5	4
AREA E					
15. Nant Ffornwg	1	6	-	2	-
TOTALS	28	96	22	34	15

In response to comments from NRA pollution control staff, the indicator key was modified so as to split Group 2 which covered a wide range of impact into three groups (Figure 6.3). To determine whether these new groups were different in biological terms, the means of BMWP score and ASPT (Average Score Per Taxon) were compared between the pollution groups. The results showed that there were highly significant differences between groups both for the original three group key and for the modified five group key (Figure 6.4, Tables 6.3 & 6.4). It is noticeable that Groups 1 and 2a are very similar in the distributions of both BMWP score and ASPT indicating that the presence of a small amount of sewage fungus (which downgrades sites to Group 2a) does not markedly affect the fauna.

Figure 6.4 Distributions of BMWP score and ASPT within pollution groups: a) BMWP with three groups; b) ASPT with three groups; c) BMWP score with 5 groups; d) ASPT with five groups.



Medians (dots), ranges ('whiskers') and 1st and 3rd quartiles are shown. Asterisks indicate outliers.

Small growths of sewage fungus may, however, be an indication of more serious pollution upstream. Groups 2b, 2c and 3 show a progressive deterioration in biological quality. There were also significant differences for the five groups in altitude and substratum composition but these showed no clear pattern across the five groups (Table 6.4).

Overall thirty-four definite sources of farm pollution were found during the course of the survey. In addition there were a further fifteen farms which were situated such that they could not be eliminated as sources of pollution observed downstream (Table 6.2). The catchment maps produced marked the survey sites giving details of pollution group observed and sewage fungus cover above and below stones. Polluting farms were also illustrated (Figure 6.5, Appendix G2). These maps were sent to the NRA Agricultural Liaison Officer together with a short report giving a suggested order of priority for farm visits.

Table 6.3 Relationship between pollution groups indicated by original indicator key (three groups) and biotic indices

	n	BMWP score	ASPT
Group 1	28	123 (104-142)	6.4 (5.9-6.9)
Group 2	93	97 (63-142)	5.7 (5.0-6.5)
Group 3	19	59 (26-92)	4.6 (3.5-5.8)
F		25.29	27.89
p		<0.001	<0.001

Group statistics are means with standard deviation ranges in brackets. The F statistic (F) and probability value (p) are from analysis of variance.

Programme of farm visits

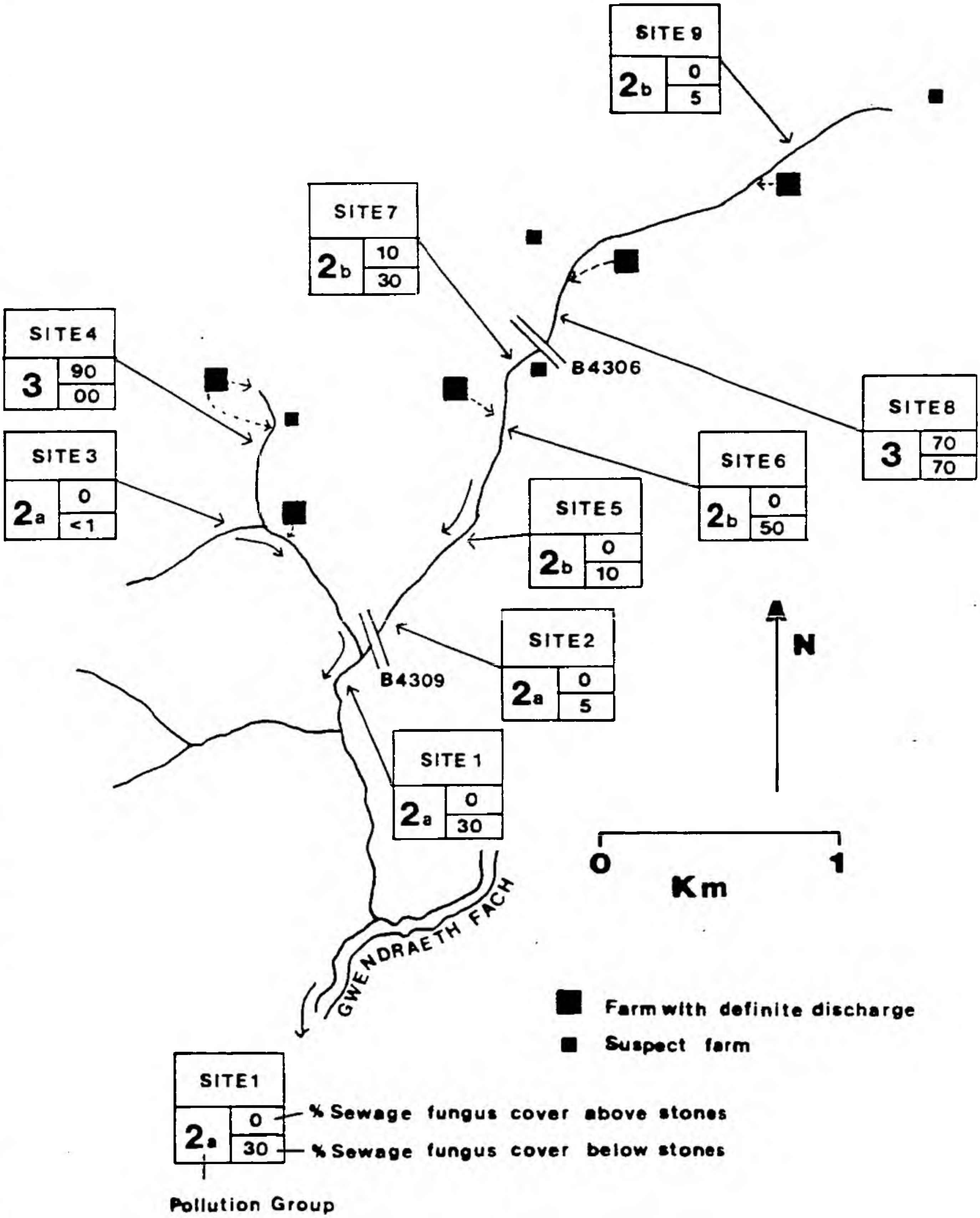
The programme of farm visits was initiated on 26 March 1991 and was still uncompleted on 30 August 1991. Leaving aside two farms in the Ffornwg catchment which were not nominated for follow-up work, only 18 out of 32

Table 6.4 Relationship between pollution groups indicated by revised indicator key (five groups) and biotic indices and environmental variables - West Wales, Spring 1991

Variable (units)	GROUP 1 (n=28)	GROUP 2a (n=30)	GROUP 2b (n=40)	GROUP 2c (n=23)	GROUP 3 (n=19)	F	p
Pollution dependent							
BMWP	123 (104-142)	122 (102-142)	89 (58-120)	76 (47-106)	59 (26-92)	27.3	< 0.001***
ASPT	6.4 (5.9-6.9)	6.3 (5.8-6.7)	5.5 (4.7-6.3)	5.4 (4.7-6.3)	5.4 (3.5-5.8)	23.2	< 0.001***
Pollution independent							
Altitude (m)	100 (61-140)	75 (31-120)	61 (27-95)	74 (31-117)	59 (12-106)	4.7	0.001**
Gradient (%)	4.1 (0-8.3)	1.9 (0.7-3.2)	2.6 (0.1-5.1)	3.2 (0-6.4)	2.6 (0.7-4.5)	2.3	0.063
Width (m)	2.0 (1.0-3.1)	2.7 (1.4-3.9)	2.1 (0.9-3.4)	1.9 (0.8-2.9)	2.0 (0.3-3.7)	1.7	0.152
Dist. from source (Km)	2.4 (0.6-4.1)	3.3 (1.1-5.5)	2.7 (0.3-5.1)	2.3 (0.2-4.4)	2.2 (0.1-4.2)	1.3	0.292
Substratum (phi)	-3.6 (-5.2,-2.0)	-4.8 (-6.4,-1.5)	-2.8 (-5.2,-0.3)	-2.7 (-4.9,-0.5)	-3.3 (-5.7,-0.9)	4.9	0.001**

Group values are means with standard deviation ranges in brackets. The F statistic (F) and probability value (p) are from analysis of variance.

Figure 6.5 Example of a catchment map produced to communicate findings to Pollution Control staff. Data from a survey of the Nant Rhydw on 14 and 20 February, 1991.



definitely polluting farms and 6 out of 15 suspect farms had been visited (Table 6.2). Coverage of five catchments was complete (Slade Brook, Rhydyfallen Stream, Llanycefn Brook, Nant Cou and Nant Rhydw) but no farms were visited in the Millin Brook catchment (eight farms reported) and coverage of the remaining six polluted catchments was incomplete. Of 17 pollution problems identified, remedial action had been agreed in 9 cases. In only two cases had formal letters had been sent to the farmers concerned (Table 6.5).

Due to the slow progress it was decided not to cover the Millin Brook catchment but even so the initial visits were not completed until 5 February 1992, too late for any remedial work to have been completed prior to the survey. The situation immediately prior to the follow-up survey was as reported in Table 6.6. Remedial work had only been confirmed as completed at ten farms. At all farms visited, information was recorded in a standardised format (Appendix G4).

Summer farm pollution survey

In all, 99 sites in the twelve catchments were surveyed using the summer indicator key. Of these 40 fell into the unpolluted Group 1, 13 into the grossly polluted Group 3 and 46 into the intermediate Group 2. With the exception of one possibility in the Cwerchyr catchment, there were no new sources of farm pollution discovered beyond those evident during the survey in February/March. Many of the problems identified during the previous survey were much less evident, although eight farms were still causing significant pollution. This apparent improvement was probably due to drier summer weather and the fact that livestock were being kept on pasture rather than housed in farm buildings.

Follow-up survey using rapid appraisal method

Comparing data for 1991 with that for 1992, it can be seen that the number of sites in the less polluted groups (1 and 2a) had increased markedly whilst the number of sites in the less polluted groups had been markedly reduced (Table 6.7).

Table 6.5 Progress with programme of farm visits as of 30 August 1991

	Polluting Farms (Definite/possible)	No. visited	Problems identified	Remedial Action agreed	Confirmation letters sent
AREA A					
1. Millin Brook	6 / 2	0 / 0	0 / 0	0 / 0	0 / 0
2. Fenton Brook	4 / -	2 / -	2 / -	2 / -	0 / -
3. Cartlett Brook	2 / -	1 / -	1 / -	0 / -	0 / -
4. Slade Brook	1 / -	1 / -	1 / -	0 / -	0 / -
5. Rhydyfallen Stream	1 / -	1 / -	1 / -	1 / -	0 / -
6. Llanycefn Brook	2 / -	1 / -	1 / -	1 / -	1 / -
7. Nant Duad	4 / 5	2 / 1	2 / 0	2 / -	0 / 0
AREA B					
8. Nant Cou	2 / 2	2 / 2	1 / 0	0 / -	0 / -
9. Nant Cwerchyr	1 / 1	0 / 0	- / -	- / -	- / -
10. Afon Peris	1 / -	0 / 0	- / -	- / -	- / -
AREA C					
13. Nant Pibwr	3 / 1	3 / 0	1 / -	0 / 0	0 / 0
14. Nant Rhydw	5 / 4	5 / 3	5 / 2	2 / 1	1 / 0
TOTALS	32 / 15	18 / 6	15 / 2	8 / 1	2 / 0

Table 6.6 Progress with programme of farm visits as of 12 February 1992

	Polluting Farms (Definite/possible)	No. visited	Problems identified	Remedial Action agreed	Work carried out
AREA A					
1. Millin Brook	6 / 2	0 / 0	0 / 0	0 / 0	0 / 0
2. Fenton Brook	4 / -	4 / -	3 / -	2 / -	0 / -
3. Cartlett Brook	2 / -	2 / -	2 / -	2 / -	2 / -
4. Slade Brook	1 / -	1 / -	1 / -	1 / -	1 / -
5. Rhydyfallen Stream	1 / -	1 / -	1 / -	1 / -	1 / -
6. Llanycefn Brook	2 / -	2 / -	2 / -	2 / -	0 / -
7. Nant Duad	4 / 5	4 / 5	3 / 2	3 / 2	0 / 0
AREA B					
8. Nant Cou	2 / 2	2 / 2	2 / 2	2 / 2	1 / 0
9. Nant Cwerchyr	1 / 1	1 / 1	1 / 0	1 / -	1 / -
10. Afon Peris	1 / -	1 / -	1 / -	1 / -	0 / -
AREA C					
13. Nant Pibwr	3 / 1	3 / 1	2 / 0	2 / -	0 / -
14. Nant Rhydw	5 / 4	5 / 4	5 / 4	5 / 4	2 / 2
TOTALS 12.02.92	32 / 15	26 / 13	22 / 8	21 / 8	8 / 2
(30.08.91	32 / 15	18 / 6	15 / 2	8 / 1	0 / 1)

Figures on the left-hand side of the slash refer to farms found to have a definite discharge, figures to the righthand side refer to farms where pollution was possibly occurring.

Table 6.7 Breakdown of sites surveyed in West Wales by catchment and indicator key group

	1991					1992				
Catchment	1	2a	2b	2c	3	1	2a	2b	2c	3
1. Millin Brook			4	3	7	4	1	9		
2. Fenton Brook +		1	4	3	2		8	2		
3. Cartlett Brook +		3	4		3	2	4	1	1	1
4. Slade Brook			1		2			2	1	
5. Rhydafallen Stream	3	1	2	2	1	7		2		
6. LLanycefn Stream		4	4	1	1	4	3	2	1	
7. Nant Duad	2	10	5	1		2	13	3		
8. Nant Cou	1		1	2	1	1		2	2	
9. Nant Cwerchyr	9		2	4		10	4			1
10. Afon Peris +	3	3	1	1		8				
11. Afon Tigen	4	1	1	2		4		3	1	
12. Trib. of Cynin	5					5				
13. Nant Pibwr		4	4	2	3	5	5	3		
14. Nant Rhydw		4	3		2	3	4	1		1
15. Nant Ffornwg			7					6		1
Totals	27	31	42	21	22	55	42	36	6	4

+ Catchments where one site was not comparable between years.

There were again highly significant differences between the groups in BMWP score and ASPT (Table 6.8). There were less significant differences in altitude, gradient and distance from source. Such differences may relate to the tendency for farm pollution to have most impact on small streams with low dilution capacity.

There were a number of study sites where marked improvements in stream quality (as measured by both indicator key and biotic indices) could be directly related to improved waste management at farms upstream (Table 6.9). There were two cases where there was no improvement in waste management and correspondingly no improvement in stream quality.

6.1.4 Discussion

The rapid appraisal method developed for use in west Wales has proved effective in pin-pointing sources of farm pollution in the fifteen catchments studied.

Table 6.8 Relationship between pollution groups indicated by revised indicator key (five groups) and biotic indices and environmental variables - West Wales, Spring 1992

Variable (units)	GROUP 1 (n=54)	GROUP 2a (n=43)	GROUP 2b (n=35)	GROUP 2c (n=6)	GROUP 3 (n=4)	H. Adj	p
Pollution dependent							
BMWP	127 (89-161)	130 (97-173)	96 (34-142)	84 (36-124)	42 (17-105)	44.1	< 0.001***
ASPT	6.4 (5.3-7.7)	6.4 (5.3-7.0)	5.5 (3.8-6.6)	5.5 (4.0-6.2)	3.9 (3.4-6.6)	52.8	< 0.001***
Pollution independent							
Altitude (m)	85 (10-210)	55 (15-195)	60 (5-210)	86 (15-100)	88 (55-160)	12.9	< 0.020*
Gradient (%)	2.9 (0.5-20)	1.3 (0.5-10)	1.4 (0.5-13.3)	3.3 (1.3-4.0)	4.0 (0.8-5.7)	16.4	< 0.010**
Width (m)	1.5 (0.5-5.0)	2.5 (0.5-5.3)	2.0 (0.5-8.0)	1.8 (0.3-3.2)	1.0 (0.6-3.0)	8.2	N.S
Dist. from source (Km)	2.0 (0.1-9.5)	3.3 (0.2-8.0)	1.5 (0.4-6.0)	1.2 (0.3-3.0)	0.8 (0.3-3.0)	15.0	< 0.010**
Substratum (phi)	-3.7 (-7.3, 2.0)	-4.0 (-7.3, 2.5)	-2.7 (-5.9, 4.6)	-3.6 (-6.4, 1.0)	-3.3 (-7.3,-2.7)	8.3	N.S.

Group values are medians with ranges in brackets. The H statistic (H) and probability value (p) are from Kruskal-Wallis tests.

Table 6.9 Sites where biological quality clearly reflected success or otherwise of remedial measures taken on farms upstream

Site	1992 Grp.	1991 Grp.	No. of farms	Problems resolved	Problems remaining	BMWP 1992	BMWP 1991	ASPT 1992	ASPT 1991
Improvements									
Cartlett 5	1	3	1	A		135	73	6.1	5.6
Cartlett 6	1	3	1	A		128	18	6.1	3.6
Slade 1	2b	3	1	ABC	D	101	72	5.9	5.5
Slade 2	2c	3	1	ABC	D	36	8	4.0	2.7
Rhydafallen 6	1	2c	1	BCD		113	43	6.3	4.3
Llanycefn 4	1	3	1	CD		118	*	6.6	*
Cou 3	2b	3	1	AC		89	21	5.6	4.2
Cwerchyr 14	1	2c	1	A		102	77	6.4	5.5
Pibwr 2	2a	3	1	A		118	84	6.2	5.6
Pibwr 9	2a	3	1	D	C	102	75	6.0	5.8
Rhydw 7	1	2b	3	AD(x3)		147	109	6.1	6.4
Rhydw 8	2a	3	3	AD(x3)		135	73	6.1	5.6
No improvement									
Cartlett 4	3	3	1		AD	24	25	4.0	5.0
Rhydw 4	3	3	2	CD	ABCD	17	3	3.4	1.5

N.B. Letters indicate nature of polluted: A = Slurry, B = Silage effluent, C = Parlour & dairy washings, D = Yard washings.

The method proved simple enough to be usable by non-biologists such as pollution control staff and a bailiff. It is estimated that eight to ten sites can be covered by a trained operator and the results rapidly disseminated.

Following comments from pollution control officers the key was adapted to be more discriminating in the intermediate range of pollution effect. The five groups in this new key showed differences in biological quality in terms of BMWP score and ASPT. The revised key requires no more identification ability than the three-group system and could be readily adopted for use in future surveys.

A less satisfactory aspect of the study was the limited time that the NRA agricultural liaison officer could afford to direct towards visiting problem farms highlighted by the rapid appraisal technique. This was due to the demands of other commitments and highlights the need for sufficient resources being made available to visit reported farms so that remedial measures can be implemented. Follow-up work must be comprehensive and thorough if improvements in water quality and biological quality of targeted catchments is to be achieved. It is also important that visits are conducted soon after the event, since many pollution problems temporarily resolve themselves with the onset of dry weather and consequently become more difficult to detect.

The use of the summer indicator key did not result in the discovery of any significant pollution sources beyond those found in February/March. It is suggested that effort is directed to carrying out rapid appraisal work in the February-April period, when farm pollution problems tend to be most evident. This approach also ensures that the farmers concerned are able to carry out remedial work during the summer months which are the most convenient for these operations. Catchments can then being resurveyed the following spring to assess improvement.

The follow-up survey indicated significant improvements in stream quality both at the catchment level and in relation to specific discharges. These improvements could often be related to remedial work; however, there were also marked improvements at Millin Brook where no farm visits took place. Either farmers here had taken action independently prompted by farm visits in neighbouring catchments or there were improvements due to drier weather conditions in the winter 1991-1992 which may also have affected the other catchments. Winter rainfall data (November to February) from 6 gauging stations within the sampling area showed an average 37% reduction in rainfall from the 1990/91 winter (a mean of 489 mm) to the 1991/92 winter (a mean of 308 mm). The winter of 1990/91, however, was not an exceptionally wet winter, with rainfalls at or below the long term average at each station (a mean of 549 mm).

Rapid appraisal techniques could be used as part of a rolling programme of farm pollution control. Effort could be targeted at a coarse level using the risk assessment techniques developed under this contract (Section 4) and selected subcatchments prioritized for rapid assessment on the basis of local knowledge, poor fish stocks or pollution incident records.

6.2 Evaluation exercise in Devon

6.2.1 Introduction

As a further test of the indicator key developed for West Wales, it was decided to carry out an evaluation exercise in Devon (NRA South-west Region.

6.2.2 Methods

The study area and sampling strategy have been previously described (Section 5.2). Fifty-one sites were sampled in 6 catchments (Figure 5.6).

The procedure on-site followed that described in Section 6.1.2 except that, in addition, a spot water sample was taken. Sampling was carried out by an Environmental Scientist from WRc and two Biologists from NRA South-west Region.

Following sample processing in the laboratory (according to the methods described in Section 5.2.2), each site was classified according to the West Wales indicator key with five groups (Figure 6.3).

To test the performance of the key, between Group differences in a range of biotic and environmental variables were examined by Kruskal-Wallis tests as group-size was sometimes small.

6.2.3 Results

There were highly significant differences ($p < 0.001$) in BMWP score, ASPT and ammoniacal nitrogen between the sites classifying into the five groups of the West Wales indicator key (Table 6.10). There were much less significant differences ($p < 0.05$) in alkalinity and conductivity (Table 6.10). These differences probably reflect the fact that pollution by organic farm wastes is less prevalent in areas of poorer soils and softer waters where sheep farming predominates.

6.2.4 Discussion

The West Wales key appears to adequately discriminate the biological quality of streams in Devon. It appears unaffected by confounding environmental factors or faunal differences between West Wales and Devon. This is unsurprising since the two areas are in close proximity and are similar in topography. It is interesting to note that for all indicator key groups BMWP score tended to be higher than in Wales which may reflect regional differences in faunal diversity (J. Murray-Bligh, personal communication).

6.3 Evaluation exercise in Yorkshire

6.3.1 Introduction

This section describes an evaluation of the West Wales indicator key (Sections 5.1, 6.1) in three subcatchments in NRA Yorkshire Region. The aims of the exercise were to determine whether the methodology was applicable to these areas, to identify types of water course for which it is inappropriate and to disseminate information about the technique.

Table 6.10 Relationship between pollution groups indicated by the West Wales indicator key (five groups) and biotic indices and environmental variable - Devon, Spring 1992

Variable (units)	GROUP 1 (n=7)	GROUP 2a (n=18)	GROUP 2b (n=15)	GROUP 2c (n=4)	GROUP 3 (n=6)	H. Adj	p
Pollution dependent							
BMWP	163 (134-194)	152 (112-186)	122 (56-189)	124 (85-133)	72 (35-129)	23.8	< 0.001***
ASPT	6.7 (6.4-7.1)	6.4 (5.4-7.0)	5.7 (4.0-6.8)	6.1 (5.3-6.4)	4.7 (3.9-6.8)	24.5	< 0.001***
Ammoniacal N (mgl-1)	0.02 (0.01-0.04)	0.04 (0.01-0.49)	0.12 (0.01-0.85)	0.02 (0.01-0.09)	0.12 (0.02-0.45)	10.9	< 0.050*
Pollution independent							
Altitude (m)	155 (40-215)	125 (60-180)	90 (10-185)	98 (50-300)	73 (30-160)	9.3	N.S.
Gradient (%)	1.7 (0.5-20)	1.6 (1.0-3.3)	1.6 (0.8-3.3)	1.3 (0.3-5.0)	1.9 (0.7-2.5)	1.2	N.S.
Width (m)	2.1 (0.9-4.0)	2.0 (1.2-3.5)	2.0 (0.8-3.0)	1.6 (0.1-2.0)	1.3 (1.0-3.0)	5.0	N.S.
Dist. from source (Km)	1.5 (0.1-9.5)	0.8 (0.2-8.0)	1.0 (0.4-6.0)	1.8 (0.3-3.0)	0.8 (0.3-3.0)	3.9	N.S.
Substratum (phi)	-3.9 (-5.5, 2.9)	-4.0 (-6.9,-1.8)	-2.9 (-6.6, 0.2)	-4.8 (-5.7,-3.2)	-5.1 (-6.4,-2.6)	4.5	N.S.
pH	7.5 (6.9-8.2)	7.4 (7.0-7.9)	7.4 (7.0-8.2)	7.4 (6.4-7.9)	7.5 (6.5-7.7)	0.9	N.S.
Conductivity (uScm-1)	172 (83-473)	187 (143-289)	281 (185-572)	211 (70-231)	255 (112-423)	13.2	< 0.020*
Alkalinity (mgl-1)	27 (9-185)	28 (13-76)	41 (25-156)	28 (5-66)	43 (5-56)	13.2	< 0.020*

Group values are medians with ranges in brackets. The H statistic (H) and probability value (p) are from Kruskal-Wallis tests.

6.3.2 Methods

Three subcatchments, in NRA Yorkshire Region were selected following discussion with Mr Brian Hemsley-Flint and Dr Elizabeth Chalk of NRA Yorkshire Region. These were as follows:-

1.	Sutton/Willow Beck	10 March 1992
2.	Danby Beck	10 March 1992
3.	Earby Beck	11 March 1992

In each subcatchment, sampling sites were selected so as to sample all major tributaries and to pin-point sources of organic pollution.

Sites were sampled by teams of two, consisting of a WRc Environmental Scientist and an NRA biologist or Pollution Control Officer. At each sampling site width, depth and substratum composition were noted and sewage fungus cover above and below large stones were recorded (data in Appendix G5). A one-minute kick sample was taken from a riffle at each site and the contents examined in a tray on the bankside. Sites were classified into one of five groups using the indicator system devised for use in West Wales in the winter and Spring (Figure 6.3). All macroinvertebrate families were recorded and BMWP score and ASPT were calculated.

Differences in biotic indices (BMWP score and ASPT) and environmental variables between the five indicator key groups were assessed using the Kruskal-Wallis test (a nonparametric analogue of analysis of variance) as group size was often small (Sokal & Rohlf, 1981).

6.3.3 Results

General

When data from all 40 sites was pooled Kruskal-Wallis tests revealed that there were significant differences ($p < 0.05$) in BMWP score and ASPT between the indicator key groups (Table 6.11). There was however quite large variation within the groups as revealed by the ranges. There were no significant differences in other environmental variables.

Sutton/Willow Beck

Fourteen sites in this subcatchment were sampled (Table 6.12, Figure 6.6). Six sites (7,9,10,12,13,14) fell into groups 1 and 2a where the impact of pollution is held to be minimal (Table 6.12, Table 6.13). BMWP scores and ASPT were generally rather low but this may be a natural feature of this catchment. Of the four sites in Group 2b, sites 4 and 11 were on rather small streams and a Group 1 fauna is perhaps unlikely to be present naturally. The remaining two sites (6 and 8) were below recent discharges of untreated sewage from which the stream was probably still recovering. Of the sites in Group 2c, Sites 1 and 3 were downstream of a number of large farms. Site 3 (BMWP score 48) had a particularly poor fauna when compared with Site 6 upstream (BMWP score 103) and a serious discharge from a pig farm (Isle Beck Grange, SE456776) has since been located. Site 5 on Car Dike had a very silty substratum which will very probably account for the poor classification. The only site in Group 3 (site 2) is downstream of known discharge from a turkey-processing plant.

Table 6.11 Relationship between indicator key group and biotic indices and environmental variable sacross the three catchments in Yorkshire

Variable (units)	GROUP 1 (n=7)	GROUP 2a (n= 5)	GROUP 2b (n=13)	GROUP 2c (n=12)	GROUP 3 (n=2)	H. Adj	p
Pollution dependent							
BMWP	86 (68-117)	89 (85-105)	73 (36-103)	50 (7-93)	38 (12-65)	11.5	< 0.050*
ASPT	6.5 (6.4-7.1)	5.9 (5.4-7.0)	5.4 (4.0-6.8)	5.8 (5.3-6.4)	3.7 (3.9-6.8)	10.62	< 0.050*
Pollution independent							
Altitude (m)	147 (40-215)	133 (60-180)	130 (10-185)	143 (50-300)	80 (30-160)	3.6	N.S.
Gradient (%)	2.0 (0.7-5.0)	1.3 (0.5-4.0)	0.9 (0.3-6.7)	1.4 (0.3-13.3)	1.6 (0.3-2.9)	1.0	N.S.
Width (m)	2.0 (1.3-4.0)	2.9 (1.3-4.0)	1.0 (0.6-4.5)	2.5 (0.7-6.5)	1.0	11.1	< 0.050*
Dist. from source (Km)	2.3 (0.7-6.5)	3.0 (2.5-8.5)	1.2 (0.4-14.5)	3.0 (1.0-18.0)	2.2 (0.3-4.0)	5.3	N.S.
Substratum (phi)	-4.0 (-5.4,-2.8)	-5.2 (-6.9,-4.1)	-3.6 (-7.1,-0.9)	-3.7 (-6.1,-6.3)	-4.2 (-6.4,-1.9)	4.2	N.S.

Group values are medians with ranges in brackets. The H statistic (H) and probability value (p) are from Kruskal-Wallis tests.

Table 6.12 Characteristics of sites sampled on 10/3/92 in the Sutton/Willow Beck catchment grouped by indicator key group

GROUP	Site	NGR	BMWP	ASPT	Comments
1	7	SE481808	68	5.2	Small stream but pebbly substratum
	10	SE491841	75	5.4	
	14	SE491872	80	6.2	Stream small and shallow
2a	9	SE481826	95	5.6	
	12	SE492856	89	5.9	Livestock farm upstream
	13	SE492856	105	6.2	Livestock farm upstream
2b	4	SE443762	36	3.6	Small stream, fine gravel substrate
	6	SE474775	103	5.4	Sewage pollution 4-6 weeks previously
	8	SE477809	49	3.8	Sewage pollution 4-6 weeks previously
	11	SE491841	87	6.2	Small stream < 1m in width
2c	1	SE426768	93	5.2	Pig farm discharge u/s. Fine gravel
	3	SE431769	48	4.8	Pig farm discharge upstream
	5	SE474774	7	2.3	Sandy/silty substratum
3	2	SE429766	12	2.4	D/S turkey processing plant

Figure 6.6 Sutton/Willow Beck Catchment

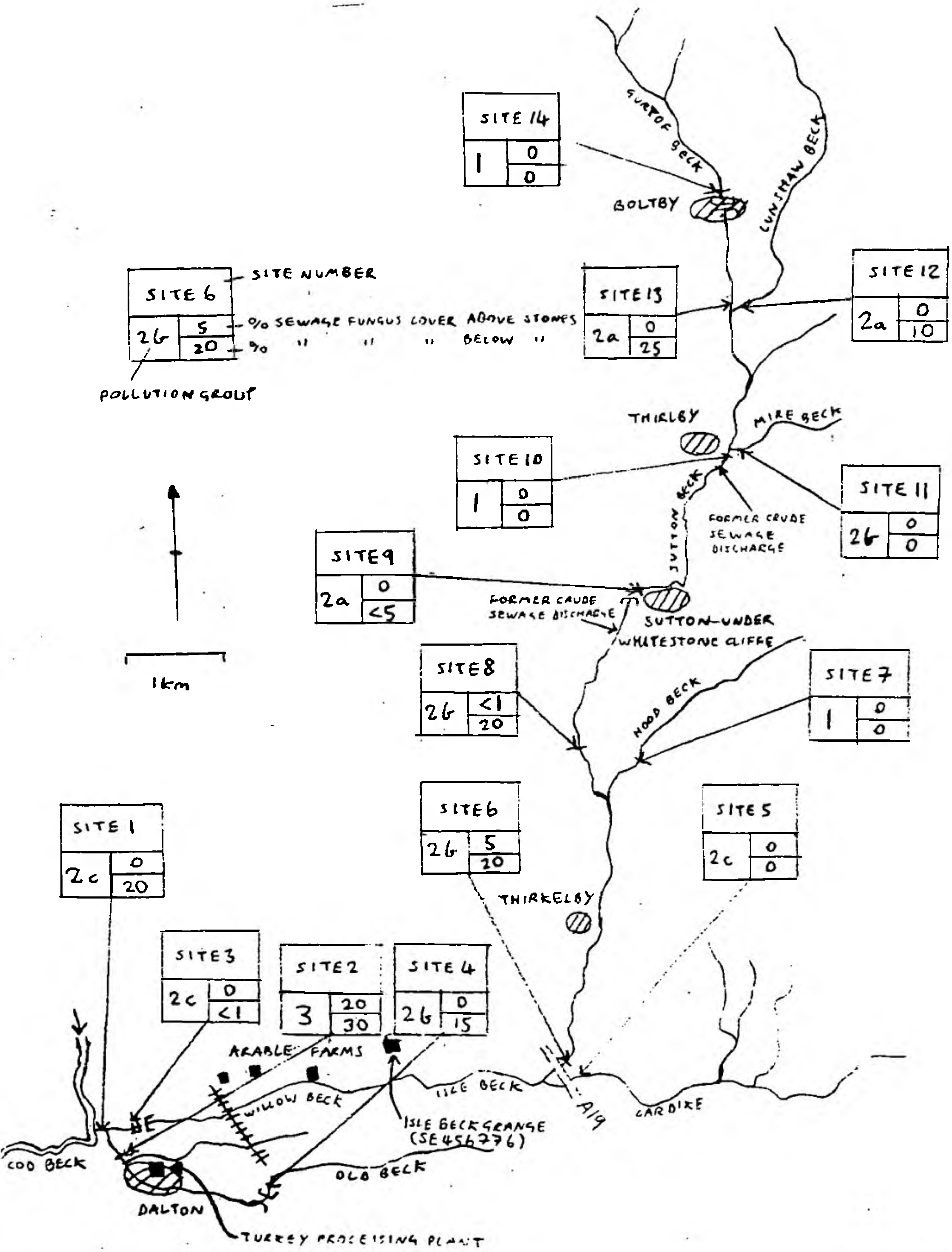


Table 6.13 Macroinvertebrate families recorded on 10/3/92 at sites on Sutton Beck grouped by indicator key group

	1			2a			2b				2c			3
Family	7	10	14	9	12	13	4	6	8	11	1	3	5	12
Heptageniidae	*	*	*	*	*	*		*		*				
Leptophlebiidae						*		*		*	*			
Ephemeridae								*				*		
Taeniopterygidae						*				*				
Leuctridae			*		*									
Perlodidae	*	*	*	*	*	*				*				
Chloroperlidae			*											
Odontoceridae											*			
Leptoceridae											*			
Lepidostomatidae								*						
Sericostomatidae				*		*					*			
Nemouridae		*	*		*	*		*		*				
Rhyacophilidae					*					*				
Glossosomatidae						*						*		
Limnephilidae	*	*		*	*	*				*		*		
Ancylidae				*				*	*					
Gammaridae	*	*	*	*	*	*	*	*	*	*	*			
Dytiscidae			*			*		*		*				
Hydrophilidae					*									
Elminthidae	*	*	*	*	*	*	*	*	*		*	*		
Hydropsychidae	*	*	*	*	*	*		*	*		*	*		
Tipulidae	*	*	*		*		*	*		*				
Simuliidae	*	*		*	*		*	*	*		*	*		
Planariidae	*	*		*							*			
Baetidae	*	*	*	*	*	*		*	*	*	*			
Hydrobiidae	*					*	*	*		*	*	*		
Lymnaeidae													*	
Planorbidae									*					
Sphaeridae		*					*		*		*			
Glossiphonidae						*	*		*		*		*	*
Hirudinidae								*						
Erpobdellidae								*	*		*			*
Asellidae				*			*	*	*		*	*		*
Chironomidae	*	*	*	*	*	*	*	*	*	*	*	*		*
Oligochaeta	*	*	*	*	*	*	*	*	*	*	*	*	*	*
Empididae			*		*									
Ceratopogonidae						*								

It was recommended that visits to the following premises were required in order to secure improvements in water quality:

1. The turkey processing plant at SE430765.
2. The farms between Sites 3 and 6 (located at SE439774, SE443775, SE449774, SE457779).

Danby Beck

Eight sites in this catchment were sampled (Figure 6.7, Table 6.14). There was only one site which fell into Group 1 (Site 4, Table 6.14, Table 6.15). The two sites in Group 2b and one in Group 2c are downstream of a formerly severe input from Church House Farm (NZ699063) which entered a tributary of the Beck. The remaining four sites in Group 2c (Sites 5-8) were at the top of the catchment. All had high ASPT but a restricted fauna lacking *Gammarus*, molluscs and mayflies which is characteristic of acidified streams (Table 6.14, Table 6.15).

Table 6.14 Characteristics of sites sampled on 10/3/92 in the Danby Beck catchment grouped by indicator key group

GROUP	Site	NGR	BMWP	ASPT	Comments
1	4	NZ693045	86	6.6	80% diatom cover
2b	1	NZ693083	81	5.4	85% diatom cover
	3	NZ694063	73	5.6	Former severe impact by farm waste
2c	2	NZ692073	91	6.1	D/S previously polluted tributary
	5	NZ693045	90	6.4	Acid stream fauna
	6	NZ694044	49	6.1	Acid stream fauna, silt cover on stones
	7	NZ691039	51	7.3	Acid stream fauna, thick cover of algae
	8	NZ690033	47	6.7	Acid stream fauna, silt cover on stones

Earby Beck

Eighteen sites in this catchment were sampled (Figure 6.8, Table 6.16). There were six sites which fell into Group 1 or 2a (1,5,12,13,15,17); all had relatively high BMWP scores and ASPT and a diverse fauna (Table 6.16, Table 6.17). Of the sites in Group 2b, four were on small streams which may not naturally support Heptagenidae (Sites 2,6,10,14). One site in Group 2c had a fauna characteristic of acidified conditions, while sites 4 and 8 on the main stream had a poor fauna (Table 6.16, Table 6.17) with poor biotic scores. The only site in Group 3 was downstream of a serious discharge from Moor Hall farm (SD911458) where Pollution Control staff have observed poor waste management.

Figure 6.7 Danby Beck Catchment

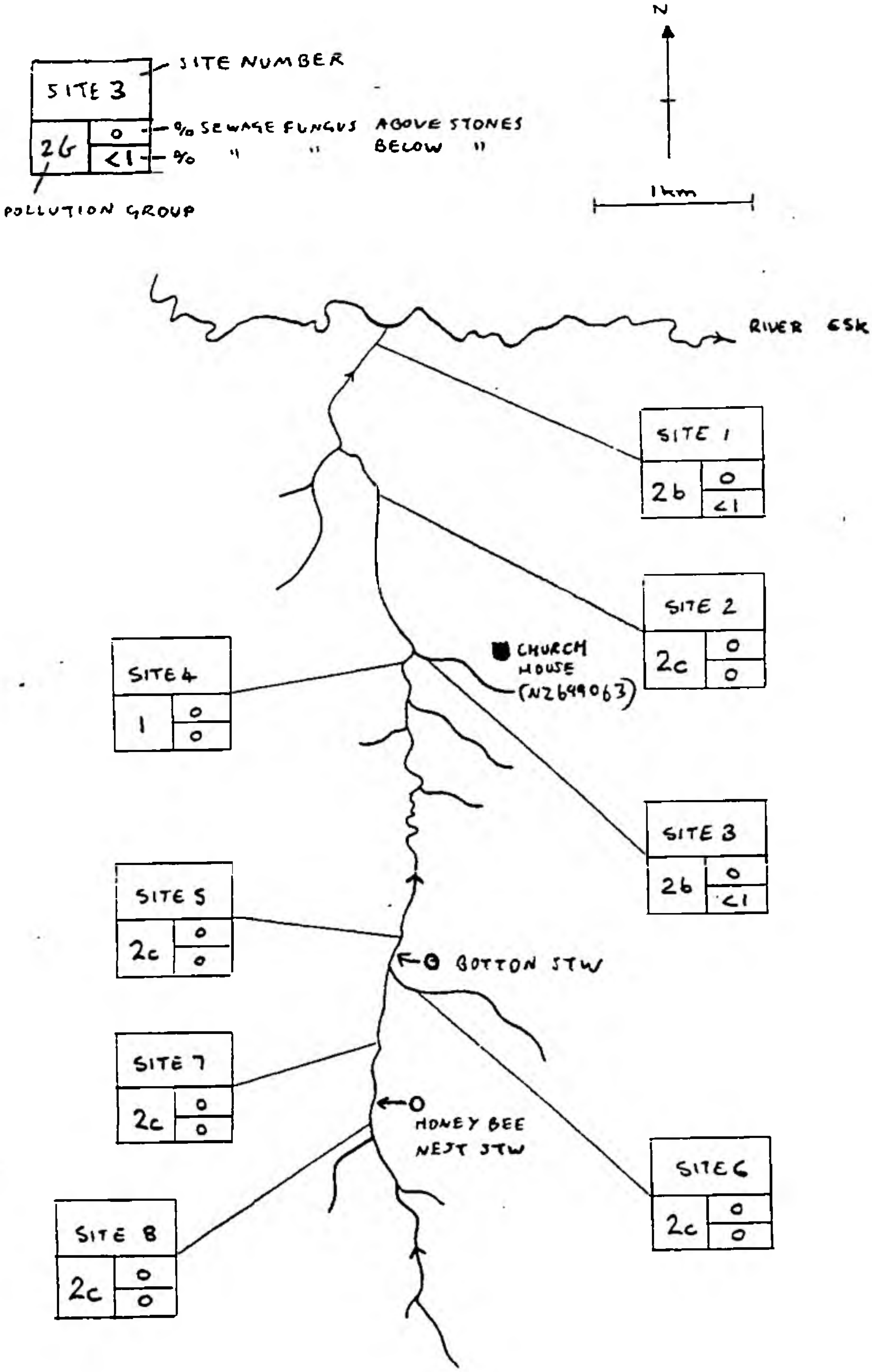


Figure 6.8 Earby Beck Catchment

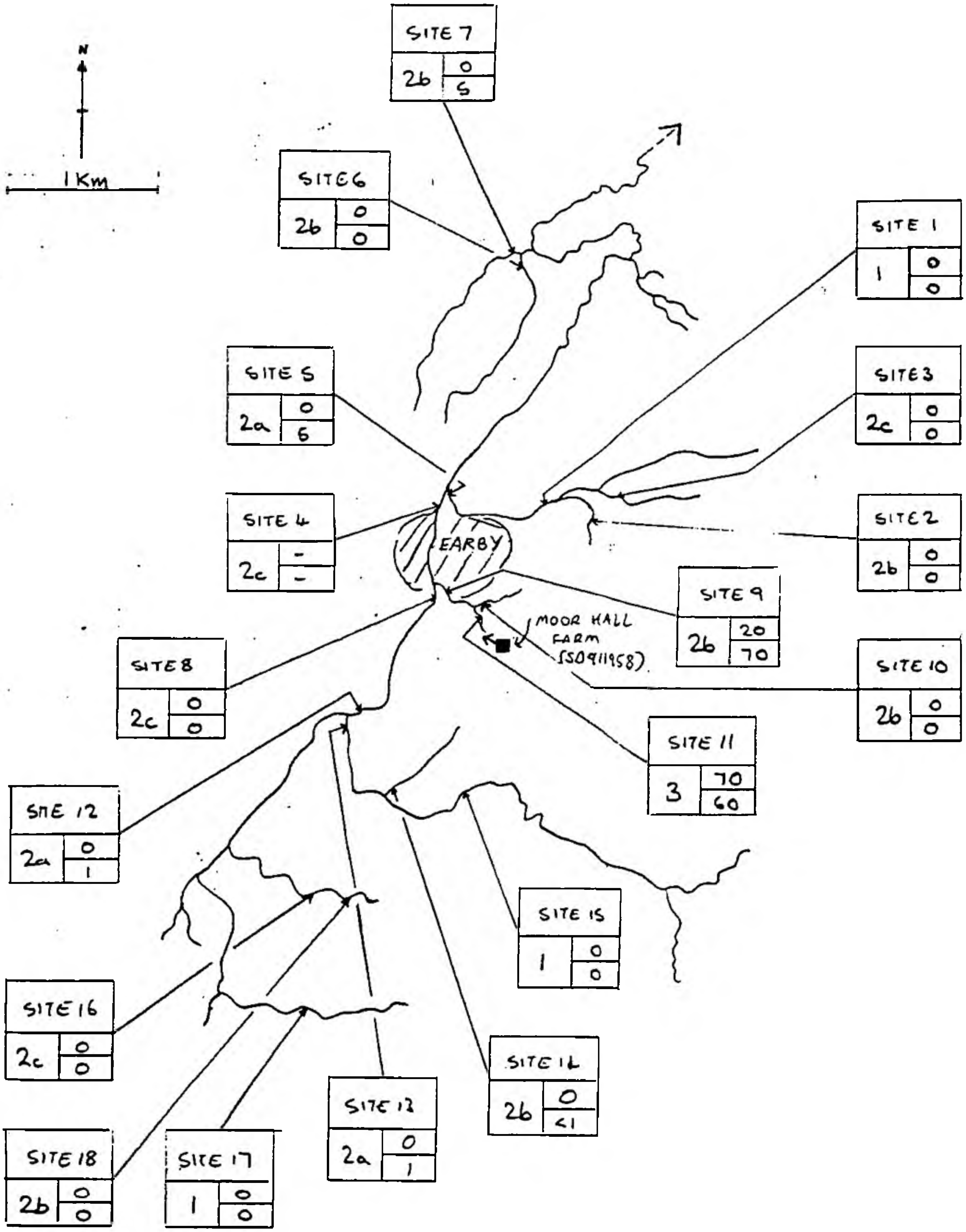


Table 6.15 Macroinvertebrate families recorded on 10/3/92 at sites on Danby Beck grouped by indicator key group

	1	2b		2c				
Family	4	1	3	2	5	6	7	8
Heptageniidae	*							
Taeniopterygidae	*	*	*	*	*			*
Leuctridae	*	*	*	*	*	*	*	*
Perlodidae	*		*	*	*			
Chloroperlidae	*				*	*	*	
Nemouridae	*	*	*	*	*	*	*	*
Rhyacophilidae	*	*		*	*	*	*	*
Polycentropodidae					*			
Limnephilidae		*	*	*	*	*	*	*
Ancylidae		*		*				
Gammaridae		*	*	*				
Dytiscidae					*		*	
Elminthidae	*	*		*				
Hydropsychidae				*				
Tipulidae	*	*	*	*	*	*		*
Simuliidae	*	*	*	*	*		*	
Baetidae	*	*	*	*				
Sialidae					*			
Hydrobiidae			*					
Sphaeridae			*					
Glossiphonidae		*		*				
Asellidae		*						
Chironomidae	*	*	*		*	*		
Oligochaeta	*	*	*	*	*	*		*
Muscidae		*						
Hydracarina		*			*			

Discussion

When data from all three subcatchments were pooled, there were significant differences between indicator key groups in biotic indices which tend to reflect degree of organic pollution. This tends to suggest that the indicator key is itself giving an indication of degree of organic pollution in these Yorkshire catchments. However there was a much higher degree of variability within groups as compared with similar analysis of data from Wales and South-west England (Sections 6.1, 6.2)and there were twelve sites at which the key was seen to suggest organic pollution when it was not thought to be occurring. The misclassifications were due to three main effects.

Firstly, low gradient sites such as Car Dike (Site 2c on Sutton Beck) where substratum particle size tended to be small, will not classify higher than Group 2b because the indicator family Heptagenidae will not be present even in unpolluted conditions.

Secondly, small streams may hold a reduced fauna having unsuitably low flow during dry weather. They may also have small particle size. Both of these influences may naturally reduce the abundance of indicators such as Heptagenidae, *Gammarus* and *Baetis*.

Thirdly, sites subject to acidification will not classify higher than Group 2c because heptagenids, *Gammarus*, and *Baetis* are all highly sensitive to acidic conditions. Such sites will tend to have somewhat reduced BMWP score but ASPT remains high because high-scoring taxa such as many stoneflies are less sensitive to acidification (see Table 6.14).

The latter two confounding influences were particularly common in the subcatchments studied, reducing the applicability of the rapid appraisal system. However such factors are likely to mitigate against the success of all objective biotic indices and a strong element of subjectivity is always likely to be involved in impact assessment work at such sites.

Despite the above drawbacks, the rapid appraisal system highlighted three serious discharges (one of which was previously completely unknown) and reflected the influence of other minor and historic pollution influences. It is suggested that, if operators are aware of the environmental limitations, there are areas of Yorkshire Region where dairy and beef farming is practised where the indicator key could be used unmodified. Such areas might be identified using the GIS system developed under this contract (Section 4) together with local knowledge of stream fauna.

Table 6.16 Characteristics of sites sampled on 11/3/92 in the Earby Beck catchment grouped by indicator key group

GROUP	Site	NGR	BMWP	ASPT	Comments
1	1	SD915469	117	6.5	
	15	SD908446	107	6.7	
	17	SD898432	109	6.8	
2a	5	SD907469	85	6.5	
	12	SD902453	89	5.8	
	13	SD901453	-	-	
2b	2	SD917467	70	5.4	Small stream
	6	SD913486	49	4.9	Small stream, may dry up intermittently
	7	SD911486	43	3.9	
	9	SD908461	91	5.4	Located discharge from Moor Hall farm
	10	SD910461	88	5.5	Small stream, < 1m in width
	14	SD905447	61	4.7	Small stream, < 1m in width
	18	SD901440	99	6.6	
2c	3	SD914478	47	5.9	Acid stream fauna, small stream
	4	SD907469	45	5.0	
	8	SD907462	54	4.2	
	16	SD899441	80	5.7	
3	11	SD909461	65	5.0	Below discharge from Moor Hall farm

The key could be used either as an extra tool for biologists or as a method usable by pollution control staff during catchment-wide farm pollution campaigns. The method offers a rapid means of identifying polluted streams and pollution sources and a convenient gauge of improvement following the implementation of remedial measures. A careful feasibility study would be required to determine whether useful indicator systems could be developed for more low lying areas of Yorkshire.

6.4 An evaluation exercise in Cheshire (NRA North-west Region)

6.4.1 Introduction

This section describes a evaluation of the West Wales indicator key (Section 5.1) in three catchments in Cheshire (NRA North-west Region). The

Table 6.17 Macroinvertebrate families recorded on 11/3/92 at sites on Earby Beck grouped by indicator key group

Group	1			2a		2b							2c				3
Family	1	15	17	5	12	2	6	7	9	10	14	18	3	4	8	16	10
Heptageniidae	*	*	*	*	*				*	*		*		*	*		
Leptophlebiidae	*			*	*												
Taeniopterygidae	*	*	*	*	*	*	*			*		*	*	*		*	*
Leuctridae		*	*	*								*				*	
Perlodidae	*	*	*	*	*	*			*			*	*			*	
Chloroperlidae	*	*	*														
Sericostomatidae	*	*													*		
Philopotamidae										*	*						
Nemouridae	*	*	*		*	*	*		*	*	*	*					*
Rhyacophilidae	*		*		*				*			*		*			*
Polycentropodidae			*	*		*	*		*	*	*		*			*	
Limnephilidae	*	*	*	*		*		*	*	*	*	*	*			*	*
Ancylidae								*	*								
Gammaridae	*	*	*	*		*	*	*	*	*	*	*				*	*
Gyrinidae		*															
Hydrophilidae		*	*							*		*					
Scirtidae											*						
Elminthidae	*	*			*		*					*			*		
Hydropsychidae	*	*	*		*				*	*		*			*	*	*
Tipulidae	*		*	*	*	*			*	*		*	*	*	*	*	*
Simuliidae	*		*		*	*	*	*	*	*	*		*			*	*
Planariidae												*				*	
Baetidae	*	*	*	*	*	*		*	*	*	*	*		*	*		*
Hydrobiidae	*				*	*		*	*	*	*				*		*
Lymnaeidae								*									
Planorbidae															*		
Sphaeridae						*		*	*	*	*				*		
Glossiphonidae											*						
Erpobdellidae				*			*	*	*					*	*	*	*
Asellidae					*		*							*	*		
Chironomidae	*	*		*	*	*	*	*	*	*	*	*	*	*	*	*	*
Oligochaeta	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
Veliidae																	
Ceratopogonidae				*													
Psychodidae									*								
Hydracarina				*					*					*	*		

aims of the exercise were to determine whether the methodology was applicable to these areas, to identify types of water course for which it is inappropriate and to disseminate information about the technique.

6.4.2 Methods

Three catchments in Cheshire were selected following discussion with Dr David Holland and Mr Frank Crossland. These were as follows:

1. Upper Weaver 17 March 1992
2. River Duckow 17 March 1992
3. Upper Dane 18 March 1992

In each catchment, sites were selected so as to sample all major tributaries and to pin-point sources of organic pollution.

Sites were sampled by teams of two, consisting of a WRc Environmental Scientist and an NRA biologist or Pollution Control Officer. At each sampling site, width, depth and substratum composition were noted and sewage fungus cover above and below large stones was recorded (data in Appendix G6). A one-minute kick sample was taken from a riffle at each site and the contents examined in a tray on the bankside. Sites were classified into one of five groups using the indicator system devised for use in West Wales in the winter and spring (Figure 6.3). All macroinvertebrate families were recorded and BMWP score and ASPT were calculated.

Differences in BMWP score and ASPT between the five indicator key groups were assessed using the Kruskal-Wallis test as (a nonparametric analogue of analysis of variance) as group size was often small (Sokal & Rohlf, 1981).

Previous surveys of the Weaver and Duckow catchments by NRA biologists have indicated poor water quality with no sites having an NWC class consistently better than class 2. Therefore, to give an idea of what fauna might naturally occur in the absence of pollution, RIVPACS was used to predict the fauna which might be expected at the bottom site sampled on the Upper Weaver (Cox *et al.*, 1991).

6.4.3 Results

General

When data from all 45 sites were pooled, Kruskal-Wallis tests revealed that there were significant differences ($p < 0.01$) in BMWP score and ASPT between the indicator key groups (Table 6.18). There was however quite large variation within the groups as revealed by the ranges (Table 6.18). There were less significant differences ($p < 0.05$) in altitude and substratum composition.

Upper Weaver

Eleven sites were sampled in this catchment (Table 6.19, Figure 6.9). There were no sites in either Groups 1 or 2a because none of the sites supported Heptageniidae which is the indicator taxon for these two groups (Figure 6.9, Table 6.20). A RIVPACS prediction for Site 1 gave only a 19.5 % predicted probability of occurrence of this family (Appendix G7). The sites in Group 2b were characterized by low BMWP scores (12-43), low ASPT (3.0-4.5), growths of diatom and filamentous algae and a substratum composed mainly of fine gravel, sand and silt. Only one definite farm discharge was located directly above any of these sites - that from Weaver Farm (SJ552510). Of the sites in Group 2c, one was some 2 Km below a discharge from Chorley Hall (SJ580498), the other had a very silty substratum. The only site in Group 3 (the most polluted group) was directly below a discharge from a farm in the village of Chorley (SJ574510).

River Duckow

Ten sites were sampled in this catchment (Table 6.21, Figure 6.10). As for the Upper Weaver the family Heptageniidae was absent from the system and consequently there were no sites in Groups 1 or 2a (Table 6.22). All sites were in Groups 2b and 2c. Fauna was taxa-poor throughout the catchment, the maximum BMWP score being 47 and maximum ASPT 3.9. The majority of sites were characterized by fine substratum.

Figure 6.9 Upper Weaver Catchment

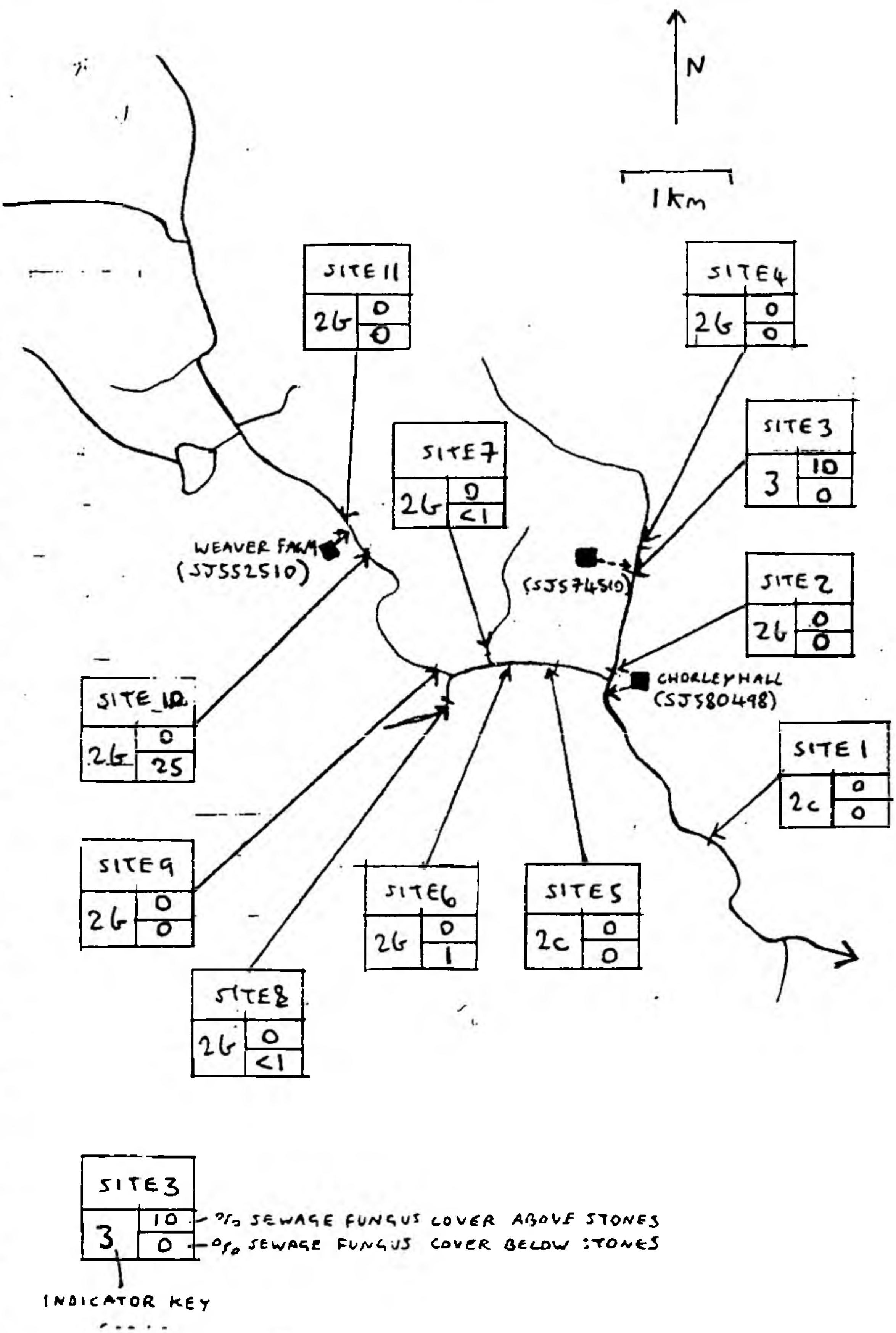


Figure 6.10 River Duck ow Catchment

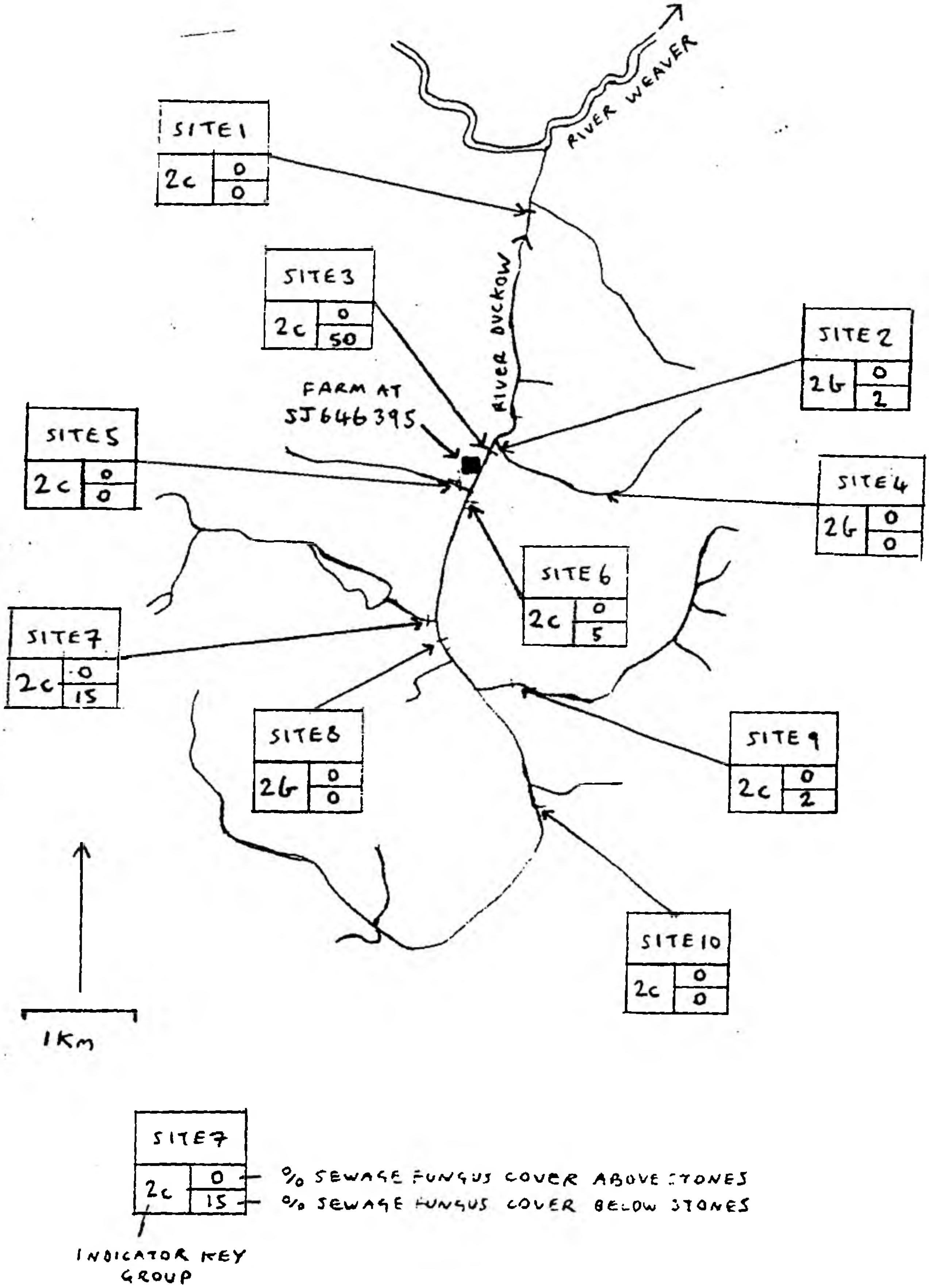


Table 6.18 **Relationships between indicator key group and biotic indices across the three catchments in Cheshire**

Variable (units)	GROUP 1 (n=7)	GROUP 2a (n= 1)	GROUP 2b (n=19)	GROUP 2c (n=17)	GROUP 3 (n=1)	H. Adj	p
Pollution dependent							
BMWP	82 (48-99)	104	43 (12-97)	28 (6-57)	17	18.5	< 0.001***
ASPT	6.8 (6.1-6.9)	6.5	3.9 (3.0-6.8)	3.6 (2.0-6.2)	3.4	18.3	< 0.010**
Pollution independent							
Altitude (m)	155 (140-200)	98	82 (59-255)	75 (50-210)	78	10.0	< 0.050*
Gradient (%)	3.3 (0.7-6.7)	0.6	0.8 (0.2-6.7)	0.7 (0.2-10.0)	0.4	1.0	N.S.
Width (m)	1.7 (1.2-13.0)	6.5	1.0 (0.8-4.5)	2.0 (0.7-5.0)	1.5	8.7	N.S.
Dist. from source (Km)	1.5 (1.0-16.0)	19.5	2.0 (0.6-7.5)	4.0 (1.0-11.5)	2.8	5.9	N.S.
Substratum (phi)	-5.0 (-6.1,-2.9)	-6.9	-2.4 (-4.8, 8.0)	-2.4 (-6.3, 8.0)	-0.6	11.4	< 0.050*

Group values are means with ranges in brackets. The H statistic and probability value (p) are from Kruskal-Wallis tests.

Table 6.19 Characteristics of sites sampled on 17/3/92 in the Upper Weaver catchment grouped by indicator key group

GROUP	Site	NGR	BMWP	ASPT	Comments
2b	2	SJ578498	12	3.0	Silty substratum
	4	SJ581511	20	3.3	Filamentous algae on stones
	6	SJ569498	43	3.6	Extensive algal cover
	7	SJ568499	27	4.5	
	8	SJ564495	26	3.3	Sandy substratum
	9	SJ564498	28	3.5	Extensive diatom growth
	10	SJ557503	41	4.1	Below input from Weaver farm
	11	SJ554513	43	3.6	Extensive diatom cover
2c	1	SJ586483	24	3.0	Below input from Chorley Hall
	5	SJ573519	31	6.2	Silty substratum
3	3	SJ580509	17	3.4	Below input from Chorley

Table 6.20 Macroinvertebrate families recorded at sites on 17/3/92 in the Upper Weaver grouped by indicator key group. Logarithmic abundance categories are shown (1= 1-9, 2=10-99, 3=100-999 etc.)

	2b								2c		3
Family	2	4	6	7	8	9	10	11	1	5	3
Taeniopterygidae										2	
Phryganeidae							1				
Nemouridae										1	
Limnephilidae										1	
Gammaridae	3	2	3	2	2	3	2	3	1		1
Elminthidae								1			
Hydropsychidae			3					1			
Tipulidae			1		1	1					
Simuliidae		2	2	2		1	2	2	2	2	2
Baetidae			2				2	2			
Sialidae							1				
Hydrobiidae		2	1		3			2	1		
Lymnaeidae								1			
Planorbidae			1								
Sphaeridae			1		1	1		2	1		
Glossiphonidae					1		1				
Erpobdellidae			2			1	1	2	1		
Asellidae	2	1	2	1	2	1	2	1	2		1
Chironomidae	3	2	2	2	2	2	2	2	2	2	3
Oligochaeta	2	3	1	1	1	1	3	2	1		3
Cladocera			1		2						
Copepoda									3		

It was recommended that visits to the following premises were required in order to secure improvements in water quality:

1. Chorley Hall Farm (SJ580498)
2. Farm at Chorley (SJ574580)
3. Weaver Farm (SJ552510)

No obvious farm discharges were identified but it was recommended that the farm at SJ646395 should be visited due to its proximity to the water course and apparently inadequate waste management.

Upper Dane

Twenty-four sites in this catchment were sampled (Table 6.23, Figure 6.11). By contrast with the Upper Weaver and Duckow, faunal richness was generally higher and Heptageniidae were present at 13 sites such that eight sites fell into the less polluted Groups 1 and 2a (Table 6.23, Table 6.24). Of the sites in Group 2b, two (17,24) were downstream of suspect farm discharges, one was on a small stream and one was characterized by slow flow and fine substratum (Table 6.23).

Table 6.21 Characteristics of sites sampled on 17/3/92 in the River Duckow catchment grouped by indicator key group

GROUP	Site	NGR	BMWP	ASPT	Comments
2b	2	SJ647396	47	3.9	
	4	SJ659391	29	3.2	Mainly sandy substratum
	8	SJ643376	18	3.0	Fine substratum
2c	1	SJ650420	25	3.6	'Concreted' substratum, algal cover
	3	SJ646395	38	3.5	Silty substratum
	5	SJ644390	22	3.1	Small tributary, fine substratum
	6	SJ645389	17	3.4	Mainly fine substratum
	7	SJ642379	27	3.9	
	9	SJ650372	12	2.4	Fine substratum
	10	SJ652362	6	2.0	Sandy/silty substratum

6.11 Upper Dane Catchment

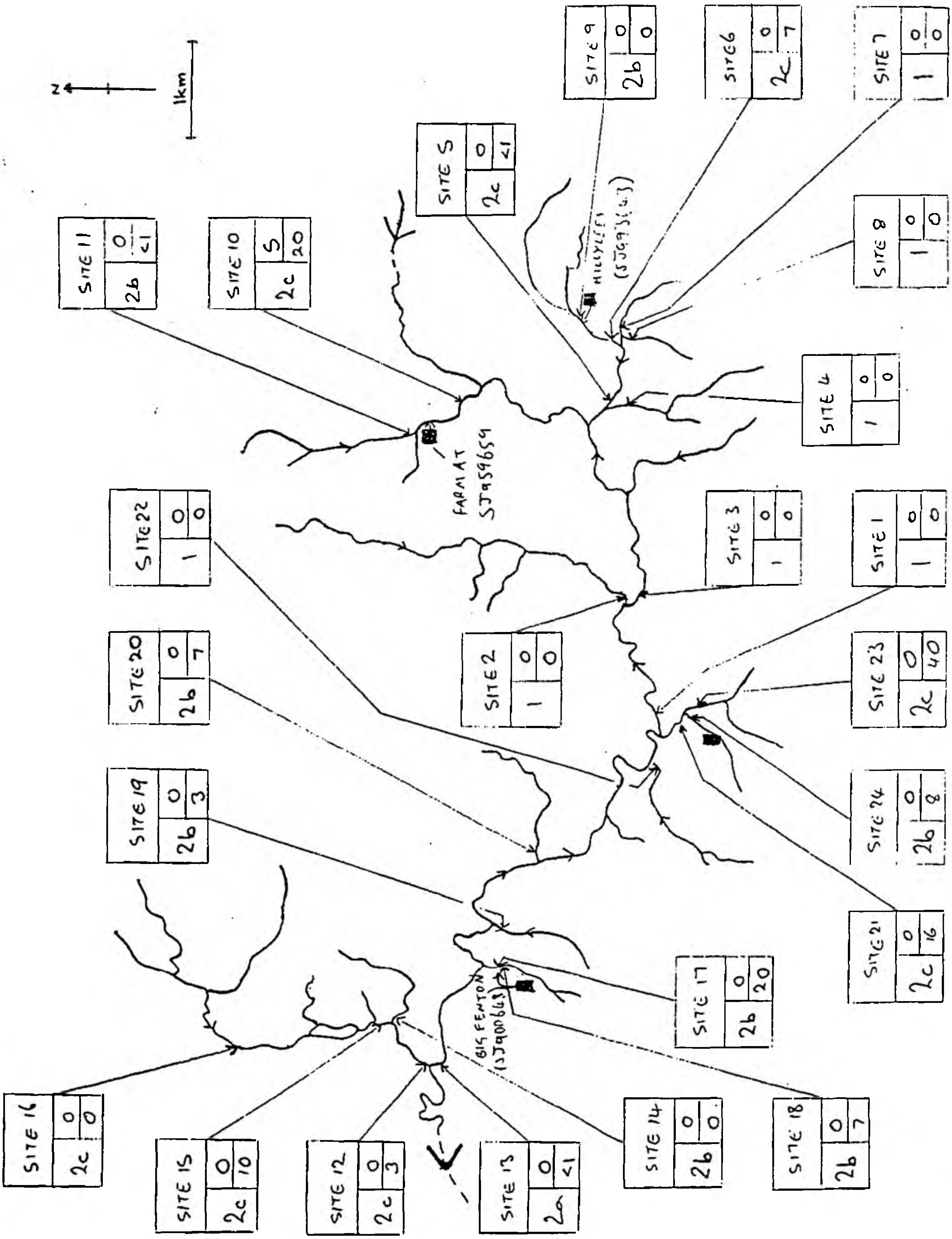


Table 6.22 **Macroinvertebrate families recorded on 17/3/92 at sites on the River Duckow grouped by indicator key group. Logarithmic abundance categories are shown (1=1-9, 2=10-99, etc.)**

	2b			2c						
Family	2	4	8	1	3	5	6	7	9	10
Nemouridae								2		
Limnephilidae										
Gammaridae	2	4	2	1	1		1	1		
Ancylidae					1					
Elminthidae	2									
Scirtidae	1									
Hydropsychidae	2			1	1					
Tipulidae	1	1				1				
Simuliidae	1			1		1	1	1		
Baetidae	2									
Hydrobiidae	2	1	1	2		1			1	
Lymnaeidae		1	1		1					
Planorbidae	1				1					
Sphaeridae		1	1		1	1	2			
Glossiphonidae		1			1					
Erpobdellidae					1	1		1	1	1
Asellidae	1	2		1	1			1	1	
Chironomidae	2	1	3	3	3	2	3	2	2	2
Oligochaeta	2	2	2	1	3	3	2	2	3	2
Cladocera							1			
Hydracarina				2		1	1			

Table 6.23 **Characteristics of sites sampled on 18/3/92 in the Upper Dane catchment grouped by indicator key group**

GROUP	Site	NGR	BMWP	ASPT	Comments
1	1	SJ930166	56	6.2	
	2	SJ943639	88	6.8	
	3	SJ946636	48	6.9	
	4	SJ963639	97	6.9	
	7	SJ971639	79	6.1	
	8	SJ970639	82	6.8	
	22	SJ924635	99	6.6	
2a	13	SJ892658	104	6.5	

Table 6.23 continued

GROUP	Site	NGR	BMWP	ASPT	Comments
2b	9	SJ973645	92	6.6	
	11	SJ959661	71	5.5	
	14	SJ896602	64	4.6	Small stream, < 1.0 m wide
	17	SJ902649	45	4.5	D/S input from Big Fenton
	18	SJ902649	57	4.8	
	19	SJ906650	97	6.1	
	20	SJ914648	42	3.8	Fine substratum
	24	SJ929632	84	6.5	Suspect farm upstream
2c	5	SJ963641	54	6.0	Possible oil pollution
	6	SJ972643	9	3.0	Oil pollution from Hillylees
	10	SJ964656	40	5.0	Serious discharge upstream
	12	SJ891659	57	4.1	Extensive diatom growth
	15	SJ895663	36	3.6	Extensive diatom growth
	16	SJ892677	32	4.6	Fine sand/gravel substratum
	21	SJ929634	56	4.7	Extensive diatom growth
	23	SJ930633	28	4.0	Diatom growth, farm U/S

Of the sites in Group 2c, there were three (5,6,10) where the poor quality could be linked to definite farm discharges (Table 6.23). Two were being affected by oil pollution from Hillylees (SJ973643), and one by a serious discharge of slurry from a farm at Wincle (SJ959659). Four more sites (12,15,21 and 23) had extensive diatom cover above stones and some sewage fungus below, suggesting some form of enrichment although no discrete discharges were located in the time available.

It was recommended that the following premises were visited:

1. Hillylees farm (SJ973643)
2. Farm at SN959659
3. Farms upstream of sites 23 and 24

Big Fenton Farm (SN9008648) had already been visited prior to the survey and remedial action has been agreed.

Table 6.24 Macroinvertebrate families recorded on 18/3/92 at sites in the Upper Dane grouped by indicator key group

Group	1							2a	2b								2c							
Family	1	2	3	4	7	8	22	13	9	11	14	17	18	19	20	24	5	6	10	12	15	16	21	23
Heptageniidae	2	2	2	2	2	2	2	2	1					1		2							1	1
Leptophlebiidae					1	1		1			1						1							
Ephemeridae																	1							
Taenioterygidae	1	2		1		1	1	1		1							1		1					
Leuctridae		2	2	2		2	2		2					1		1								
Perlodidae	2	2	2	1	2	2	2	2	2	2				1		1								
Chloroperlidae				1			1	1	2					1		1								
Sericostomatidae		1						1												1				
Nemouridae	1	1	1	1	1	1	1	1	1		1	1	1	2		1							1	
Rhyacophilidae		2		1	2				1															
Polycentropodidae							1			1		1	1	1		1								
Limnephilidae	2	1		1	1	1	1	1		1	1		1	1		1			1			1	1	
Ancylidae																				1				
Gammaridae				1	2	2	2		2	2	2	3	3	2	2	1	1	1	1	1		1	1	
Dytiscidae									1															
Hydrophilidae				1			1			2														
Scirtidae													1											
Elminthidae				1				1																
Hydropsychidae	2	2	1	2	1	2		1		2	1			1							1			
Tipulidae					1		1	1	1	2	1	1	1	1		1	1				1	1	1	
Simuliidae		2			2			2	2	2	2	1	1	2	1	1	1		2	2	2	2	2	3
Planariidae												2	2	1	1				1		1			
Baetidae	2	1	1	2	3	2	2	2	2	3	2	1	1	2	2	2	1		2	2	2	2	2	2
Sialidae										1														
Hydrobiidae											2	1	1							1			1	
Planorbidae																				1				
Sphaeridae								1			1				1					2	1			
Glossiphonidae											1			1	1					1			1	1
Erpobdellidae											1				1		1			1	1			
Asellidae															1					1	1	1	1	1
Chironomidae	1	1	2		2	2	2	2	2	2	1	2	2	2	2	2		1	2	2	2	2	3	3
Oligochaeta	1	1		1	1	1	1	1	1	1	1	2	2	1	2	1	1	2	4	1	2		3	3
Veliidae										1	1													
Ceratopogonidae	1										1							1						
Rhagionidae								1												1				
Muscidae																					1			
Hydracarina	1		1													1				1				

Logarithmic abundance categories are shown (1=9, 2=10-99 etc.).

Discussion

When data from all three catchments were pooled, there were highly significant differences between indicator key groups in biotic indices which tend to reflect degree of organic pollution. Differences in altitude, width and substratum composition were less significant ($p < 0.05$). These results tend to suggest that the indicator key is itself giving an indication of degree of organic pollution in these Cheshire catchments.

Six definite farm discharges were identified which could be linked to poor classification, but there were a large number of sites graded as 2b or worse for which no obvious pollution source was located. The majority of such sites were in the more lowland catchments (Upper Weaver and Duckow) which are unlikely to support Hetageniidae under unpolluted conditions. These sites tended to have a substratum of fine gravel, sand and silt and the predicted probability of Heptageniidae was only 19.5% for a typical site as determined by RIVPACS. There were also sites on small streams which may lack heptagenids due to low flow or fine substratum.

Thus, although it appears applicable on the higher ground which rims the Cheshire basin, due to the confounding environmental influences the Welsh system lacks discrimination in the lowland areas, although it is likely that other biotic indices are similarly less effective. If rapid appraisal is deemed to be a useful approach for this area of intensive dairy farming, a new system might be developed for this and the similar adjoining area of NRA Welsh Region. It is possible that Hydropsychidae, Elminthidae, Leptophlebiidae, Gammaridae and Baetidae together with sewage fungus might prove important indicators.

Rapid appraisal could be used either as an extra tool for biologists or as a method usable by pollution control staff during extensive surveys prior to a programme of farm visits. The method may be of use as a rapid means of identifying polluted streams and pollution sources and a convenient gauge of improvement following the implementation of remedial measures.

7. INVESTIGATION OF POLLUTION PROCESSES

This section describes a series of intensive studies designed to complement the work of Schofield and Bascombe (1990), on pollution processes operating in the Eastern Cleddan catchment.

7.1 Pontfaen Brook: A chronic discharge of silage effluent

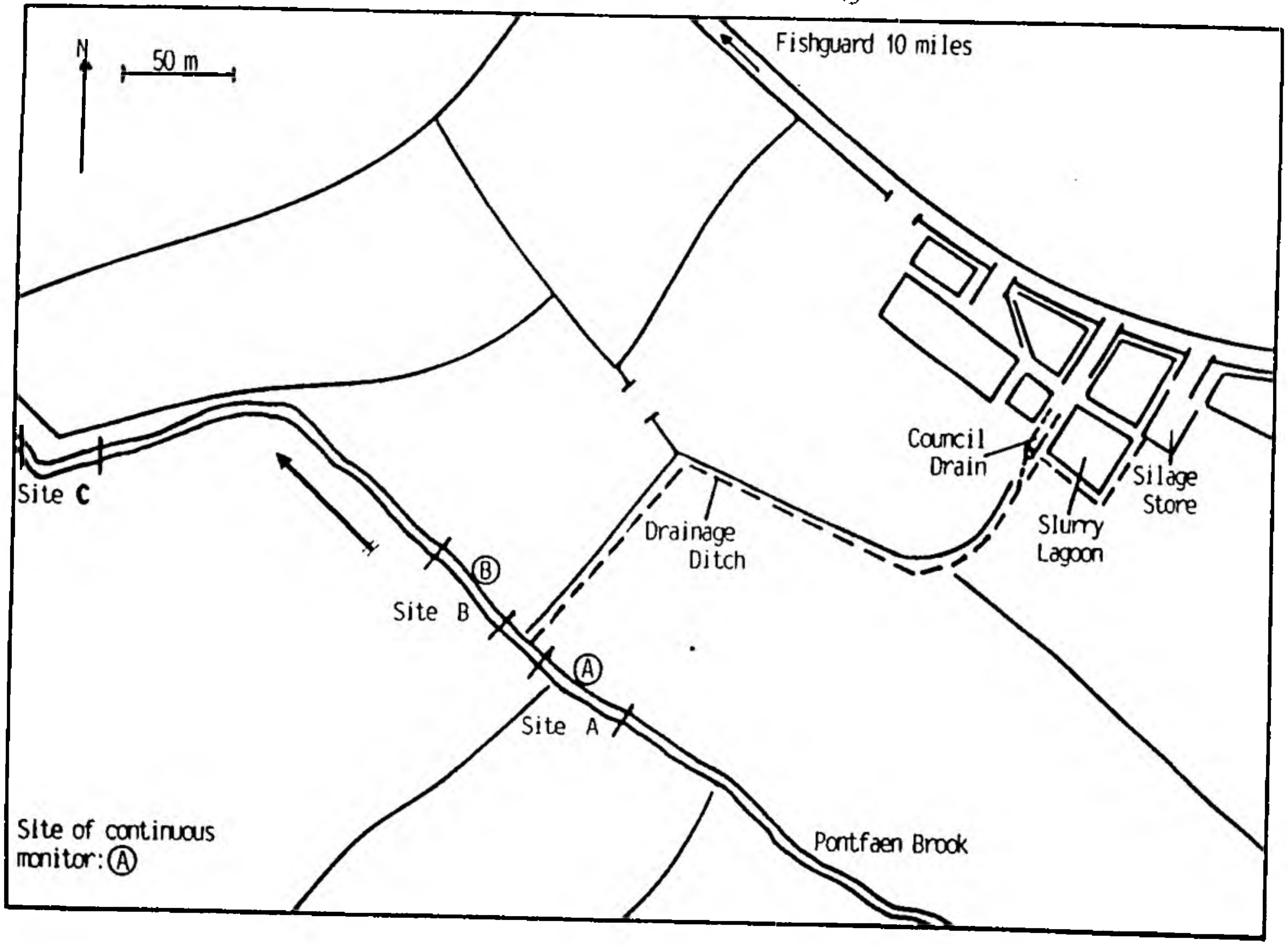
7.1.2 Study area

Pontfaen Brook, a tributary of the Afon Gwaun in south west Wales (NGR: SN032324), was identified during the development of the West Wales rapid appraisal system (Section 5.1) as being suitable for detailed study for the following reasons:

- It was obvious that the source of pollution was leaking silage effluent and so effects specific to this type of pollutant could be investigated.
- Pollutant entered the stream at a single point enabling polluted and unpolluted stretches of stream to be easily defined.
- The unpolluted upstream reference zone provided suitable habitat for salmonids and a diverse invertebrate assemblage.
- The position of the site, near to a road and surrounded by rough pasture, enabled easy access.

The study began in early June 1990. Effluent was initially observed leaking through a gap in the wall of a silage store at Trewern Farm (SN034325). It was carried 300 m along a drainage ditch which discharged into Pontfaen brook (Figure 7.1). Below this input an abundant growth of sewage fungus was observed. When remedial work carried out on 13 June 1990 failed to reduce this growth a second polluting discharge was discovered. A council road drain contaminated with silage effluent was found to be discharging to the drainage ditch after passing beneath a slurry lagoon (Figure 7.1). Most of the silage

Figure 7.1 Pontfaen Brook and the surrounding area.



effluent appeared to be entering the drain from an old cross-drain originally installed to drain a damp area adjacent to the silage store, an area which is now part of the slurry lagoon. This cross-drain was sealed up on 3 August 1990. However, a small input of silage effluent was still entering the road drain. It was apparently seeping through the floor of a second silage clamp between the road and the lagoon. Intensive survey work was completed by the beginning of December 1990 but sampling of macroinvertebrate populations continued until June 1991.

7.1.3 Methods

Water quality monitoring

Two pHOx 100 DPM continuous monitoring units were installed at Sites A and B (above and below the contaminated ditch) together with TINYLOG data loggers (Figure 7.1). Initially these were equipped to measure pH, temperature, dissolved oxygen and ammonium but conductivity was added later when new equipment became available (data in Appendix H1). From September 1990, real-time data were available from Site B following the installation of METEORBURST telemetry equipment. A rainfall recorder with TINYLOG logger was included at Site A (data in Appendix H2).

Weekly spot samples were taken from Pontfaen Brook at Sites A and B, from the drainage ditch and from the council drain. These were analysed for a number of standard determinands at the NRA Llanelli laboratory (data in Appendix H3). To complement the spot samples, stream, ditch and pipe flow were estimated by the depth-area method for the stream and using timed bucket fills for the ditch and pipe.

A Rock and Taylor autosampler was used to collect samples of stream water during the course of two episodes of heavy rainfall. This followed the observation of peaks in conductivity and ammonium ion concentration coupled with rainfall. For the first event on 5/6 October 1990 the equipment was hand-triggered but for the second on 23/24 November 1990 the sampler was triggered remotely using the METEORBURST telemetry system. The sampler was programmed to collect a sample every 30 minutes, up to a maximum of 48 samples.

A full set of samples was collected for the second event but equipment failure resulted in only 23 samples from the first covering the first 11 hours only. Samples were analysed at the Llanelli laboratory for pH, conductivity, total ammonia, Total Oxidised Nitrogen (TON), nitrite, orthophosphate and Dissolved Organic Carbon (DOC) (data in Appendix H4 and H5).

Biological monitoring

Three sites of approximately 40 m in length were selected (Figure 7.1). Two of these were in the vicinity of the continuous monitors, A and B, and a third, Site C, was located 250 m downstream, just upstream of a further input of silage effluent. Several aspects of the stream biota were examined:

1. Sewage fungus - Detailed visual estimations of the percent sewage fungus cover on the stream bed were made monthly at all three sites over 10 contiguous 2m sections of stream. Rough estimates of cover were made weekly. During the first three months, samples were examined microscopically in an attempt to identify and quantify the species of heterotrophic micro-organisms present.

2. Benthic macroinvertebrates - Three minute kick samples were taken every month from riffles at Sites A, B and C and the invertebrates identified to family level in order that BMWP (Biological Monitoring Working Party) scores could be calculated. In June only, five replicate cylinder samples were also taken, from riffles at Sites A, B and C, to allow population densities of individual species to be compared between sites (data in Appendix H6).

3. Fish populations - At the beginning of the study the three sites were electrofished quantitatively. Densities (100 m^{-2}) were calculated using the Seber and LeCren catch depletion method (Bagenal 1978). Detailed information on the habitat characteristics of the stream sections was also gathered so that the Welsh NRA HABSCORE system (Milner and Wyatt 1991) could be used to calculate expected densities of salmonids which were then compared with the observed catch (data in Appendix H7).

7.1.4 Results

Water quality

1. pH - Values of pH both above and below the input fluctuated between 6.0 and 7.5 as revealed by continuous monitoring, spot samples and auto samples. It is unlikely that this would have a direct deleterious effect on invertebrate populations or fish.

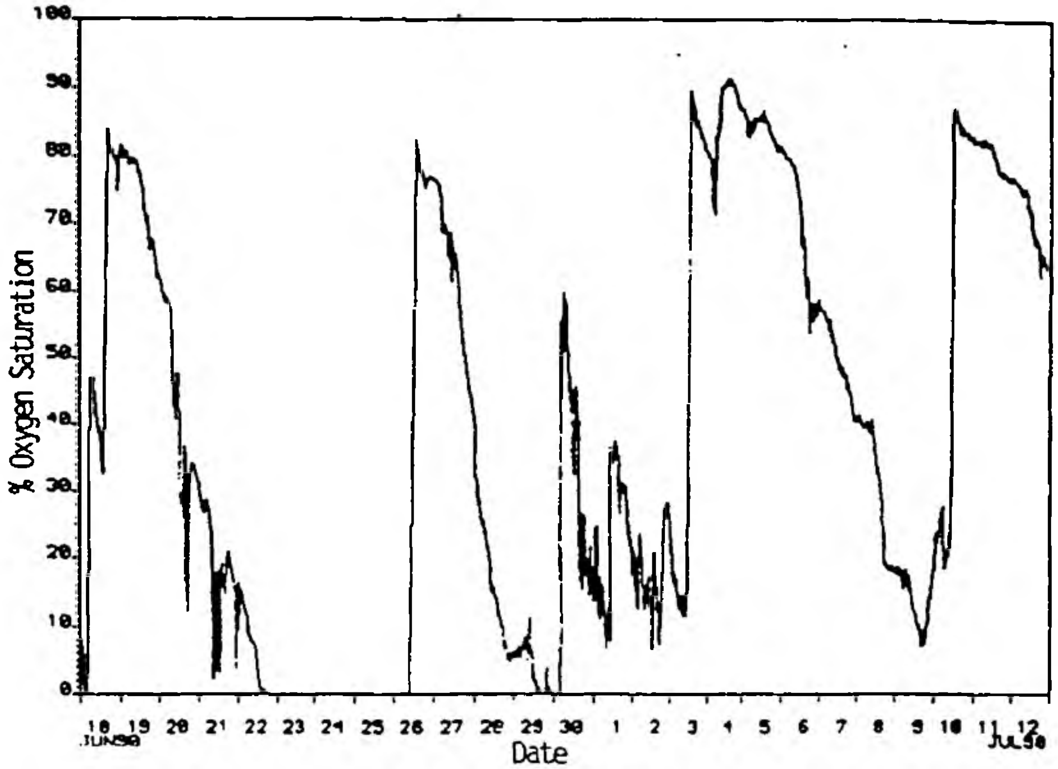
2. Dissolved oxygen - Growth of sewage fungus on the downstream DPM sensor head seriously affected dissolved oxygen measurement in the early weeks of the study (Figure 7.2). The pattern obtained was due to physical removal of the fungus during servicing and its subsequent regrowth. Copper-based anti-fouling paint was used in an attempt to prevent this. The downstream head was painted on 10 July but fouling still continued to affect the readings until the beginning of August (Figure 7.2). By this time the sewage fungus had disappeared from the stream as a whole. Dissolved oxygen data obtained from spot samples showed that there were relatively high oxygen concentrations at both Site A and Site B throughout the study (Figure 7.3). At neither site did the concentration fall below about 8 mg l^{-1}.

3. Conductivity - When conductivity data became available in early August it was apparent that at Site B (below the input) conductivity maintained a base level of 100-120 µS cm^{-1} but that during periods of high rainfall episodic peaks of up to 200 µS cm^{-1} were observed. At Site A (above the input) rather smaller peaks in conductivity were observed but the base level was comparable. The autosample data for the first event captured exhibited a peak of 187 µS cm^{-1} coupled to rainfall intensity but there was little observable increase during the second event.

4. Ammoniacal nitrogen - Spot sample data revealed that although high concentrations of ammoniacal Nitrogen were present in the council drain and in the ditch, concentrations in the Brook upstream and downstream of the input were similar and did not exceed 1.0 mg l^{-1} (Figure 7.4). Calculated concentrations of the toxic component - unionised ammonia -

Figure 7.2 Continuous monitoring of oxygen saturation at Pontfaen Brook, site B.

a) June 18 – July 12



b) July 12 – August 7

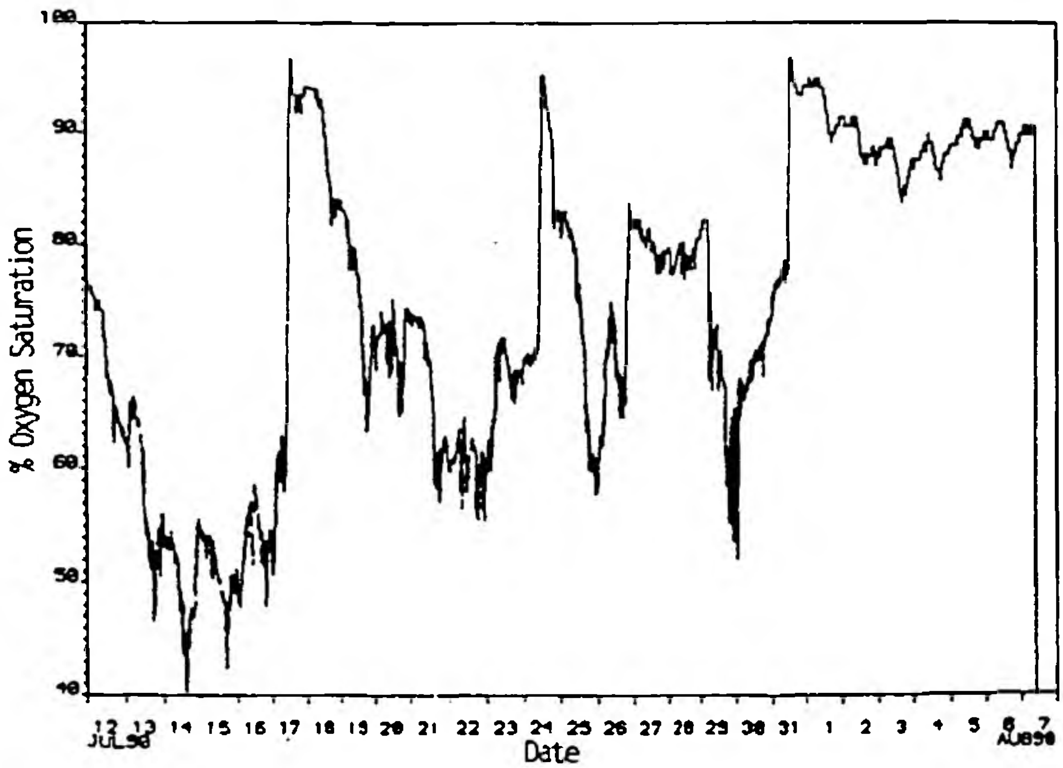
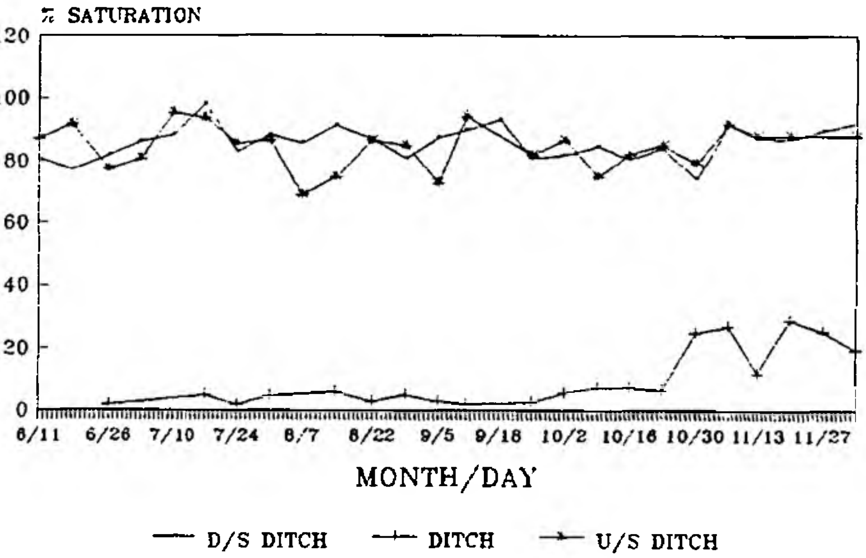


Figure 7.3 Dissolved oxygen levels at Pontfaen Brook from spot sampling.

a) Oxygen saturation (%)



b) Oxygen concentration (mg/l)

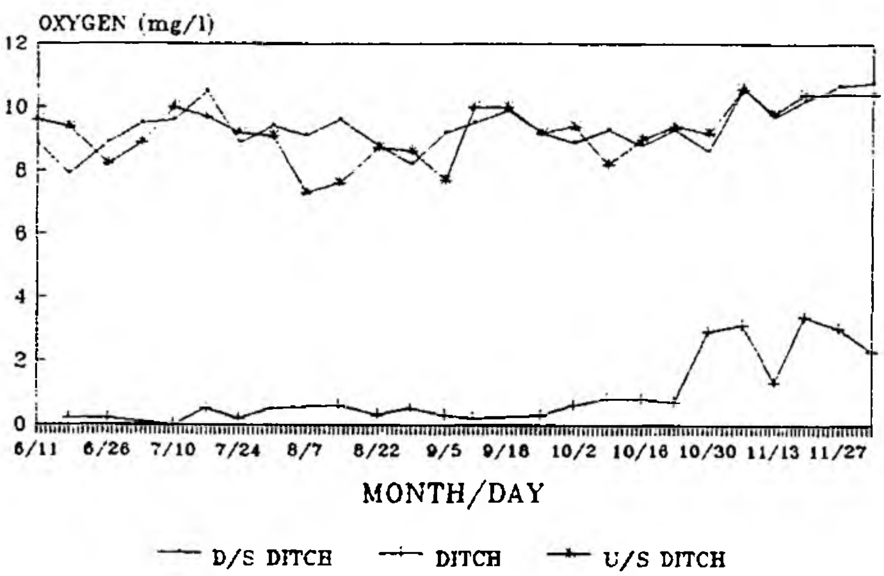


Figure 7.4 Ammoniacal nitrogen concentrations at Pontfaen Brook.

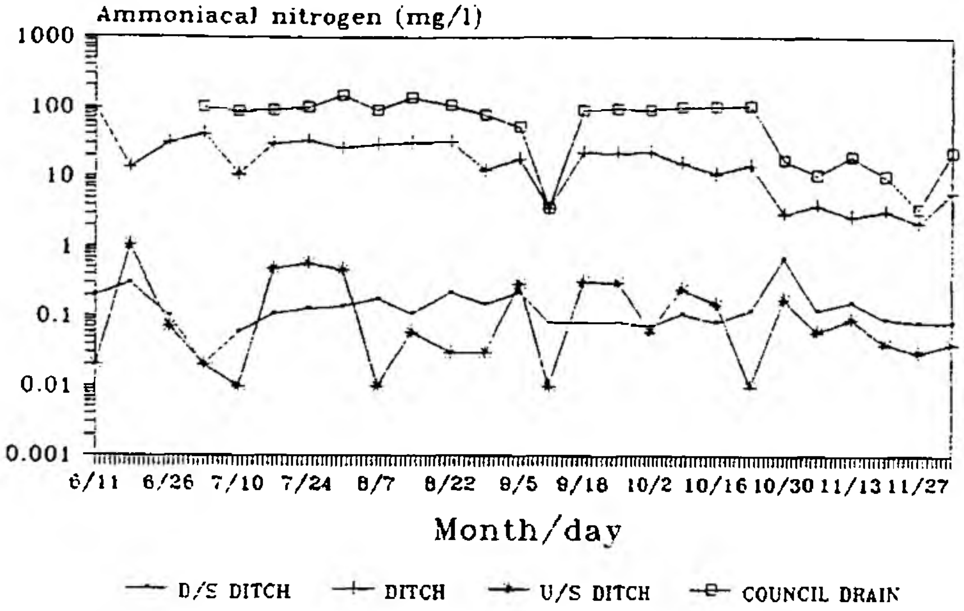
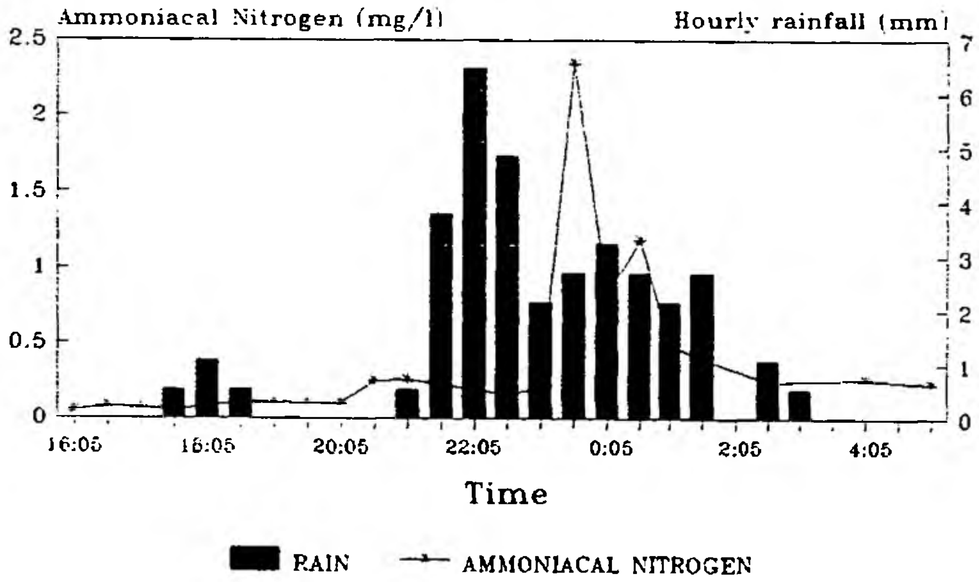


Figure 7.5 Ammoniacal nitrogen concentrations at Pontfaen Brook during event 1 (5/6 October).



did not exceed 0.004 mg l^{-1} in the stream. Continuous monitoring revealed divergences of ammonium ion concentration (effectively equal to total ammonia) from the base level during rainfall episodes at both Sites A and B. The insensitivity of the instrument at low concentration precluded accurate estimation of the peaks but they appeared small. During the two rainfall events maximum ammoniacal nitrogen concentrations recorded were 2.34 and 0.57 mg l^{-1} for the first and second events respectively (Figures 7.5, 7.6). In neither event did calculated concentrations of unionised ammonia exceed the maximum of 0.004 mg l^{-1} recorded during spot sampling.

5. BOD - The maximum BOD recorded in the ditch was 3521 mg l^{-1} recorded when first sampled on 11 June 1990. This large value corresponded to a BOD downstream at Site B of 13.6 mg l^{-1} as compared with 1.6 mg l^{-1} recorded at the upstream control Site A. BOD in the ditch exhibited a fairly steady decline during the course of the study leading to an equalisation of the BOD at Sites A and B by the end of July (Figure 7.7). In order to quantify BOD input to the stream the ditch BOD loading was calculated as the product of BOD and flow rate. The contribution this made to the downstream BOD was calculated as the product of the BOD and the ratio of the ditch flow to the main stream flow (Figure 7.8). Increases in ditch load and downstream BOD in September and October corresponded to increased rainfall (Figures 7.8, 7.9).

6. Other determinands - nitrite, orthophosphate and DOC concentrations all exhibited marked peaks lagging slightly behind maximum rainfall intensity as revealed by analysis of the automatic samples from both of the captured rainfall events. In the case of DOC peaks of 13.8 and 10.1 mg l^{-1} were observed as compared with initial concentrations of 2-3 mg l^{-1}. Levels appeared to be sustained considerably longer after maximum rainfall intensity than was the case for the other two determinands. Chloride exhibited a marked peak in concentration during the first rainfall event but this effect was very muted during the second event as was the case for conductivity.

Figure 7.6 Ammoniacal nitrogen concentrations at Pontfaen Brook during event 2 (23/24 November).

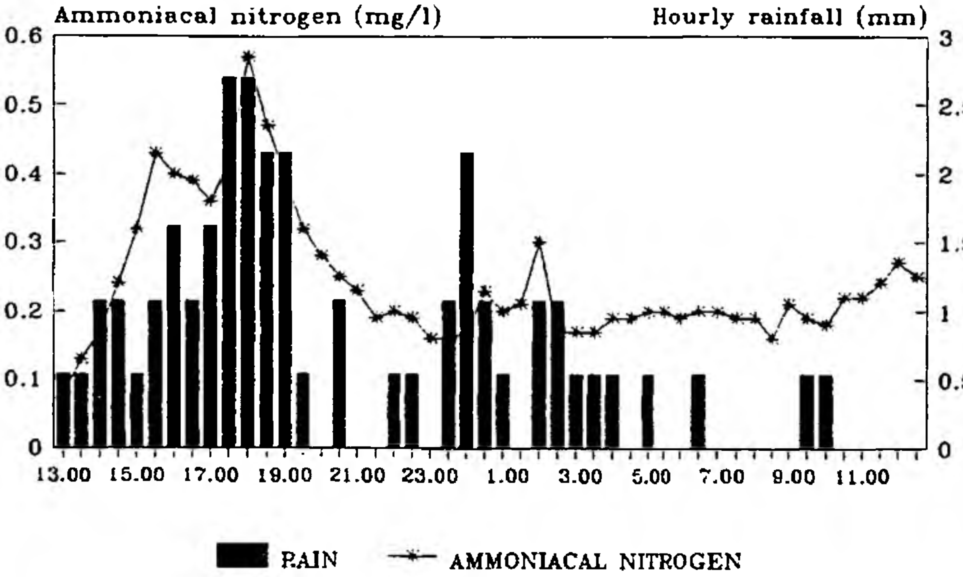
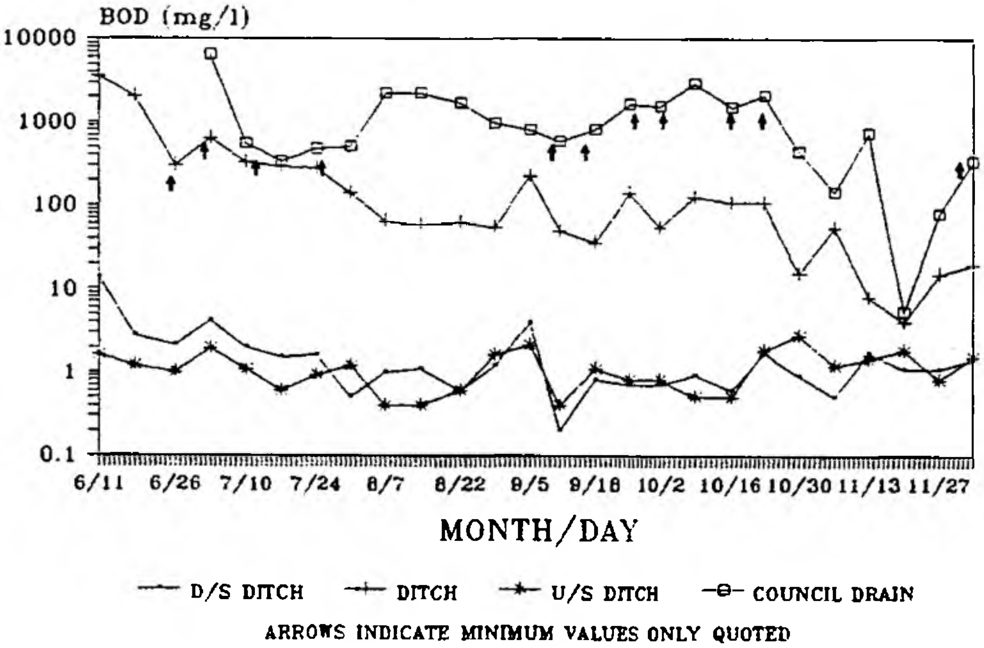


Figure 7.7 BOD concentrations at Pontfaen Brook.



Sewage fungus

At the beginning of the study there was close to 100% cover at both the downstream sites (B and C) with no fungus at the upstream site, A. Fungus remained abundant downstream until mid July after which it gradually declined until it had all but disappeared by early August except for a small quantity on vegetation immediately downstream of the input ditch. This decline corresponded to a reduction in BOD loading from the ditch and a reduction in the ditch contribution to downstream BOD (Figures 7.10, 7.11). Microscopic analysis revealed that the Phycomycete, *Leptomitus lacteus* and the colonial bacterium *Sphaerotilus natans* were the most common components of the fungus. As time passed the proportion of *L. lacteus* to *S. natans* decreased indicating an increase in localised oxygen concentrations within the sewage fungus (Table 7.1).

Table 7.1 Microscopic analysis of sewage fungus from Pontfaen Brook

Date	Site B Taxa present	% Abundance	Site C Taxa present	% Abundance
June 18	*Leptomitus lacteus*	96	*Leptomitus lacteus*	95
	Beggiatoa sp	1	*Beggiatoa* sp	1
	Thiothrix sp	3	*Thiothrix* sp	3
	Spirochaetes		*Fusarium* sp	
	Protozoa (many)		*Spirochaetes*	
	Bacteria (many)		Protozoa (many)	
	Diatoms (few)		Bacteria (many)	
July 10	*Leptomitus lacteus*	99	*Leptomitus lacteus*	60
	Sphaerotilus natans	1	*Sphaerotilus natans*	1
	Protozoa		*Zoogloea* sp	5
	Bacteria		Protozoa (few)	
Aug 7	*Leptomitus lactus*	65	Normal stream fungi,	
	Sphaerotilus natans	30	no protozoa or bacteria	
	Beggiatoa alba			
	Protozoa			
	Bacteria			

There was a small recurrence of sewage fungus at Site B in September following a period of very heavy rainfall. A larger recurrence occurred in October with a maximum abundance on 23 October (Figure 7.10). This corresponded to a more sustained autumnal increase in rainfall which continued throughout the remainder of the study period (Figure 7.9). The sewage fungus had died back completely by November. No sewage fungus was recorded at Site C after July 1990.

Macroinvertebrate populations

Initial sampling in June 1990 revealed a diverse invertebrate assemblage at Site A and impoverished assemblages at Sites B and C. At Site A BMWP score was 125 whilst Sites B and C scored 28 and 17 respectively (Table 7.2). Also two groups of organisms tolerant of low oxygen concentrations, oligochaetes and chironomidae, showed an enormous increase in density, from 76 and 55 m^{-2} at Site A to 17 700 and 11 700 m^{-2} at Site B (Table 7.3). These were also abundant at Site C with densities of 7000 m^{-2} for the oligochaetes and 6900 m^{-2} for chironomids. There was a contrasting reduction in several taxa intolerant of organic pollution, for example the mayfly *Rhithrogena semicolorata* and the stonefly, *Chloroperla torrentium* respectively, fell from 270 and 287 m^{-2} to zero at both Sites B and C (Table 7.3).

The invertebrate populations at the two downstream sites began to recover early in the study and by December the BMWP scores for Sites B and C (103, 101) were similar to that for Site A (106) (Figure 7.11). The recovery at Site B appears to be the more rapid probably because it is closer to the unpolluted reach of the stream which is a potential source of drifting colonists.

Analysis of the monthly macroinvertebrate data from the 3 sites by Detrended Correspondence Analysis (DECORANA) generated the two principal axes shown in Figure 7.12, which account for 81.3% of the variation in the data set. Sites B and C can be seen to move through ordination space over time and finally coverage with Site A by January. The first axis is quite closely related to BMWP score ($r = -0.87$, $p<0.001$), whilst the second axis perhaps reflects seasonal changes in fauna. When the site was visited on 26 June 1991, the stream was found to be severely polluted once more (Figure 7.11).

Figure 7.8 Ditch BOD load and its effect on downstream BOD in Pontfaen Brook.

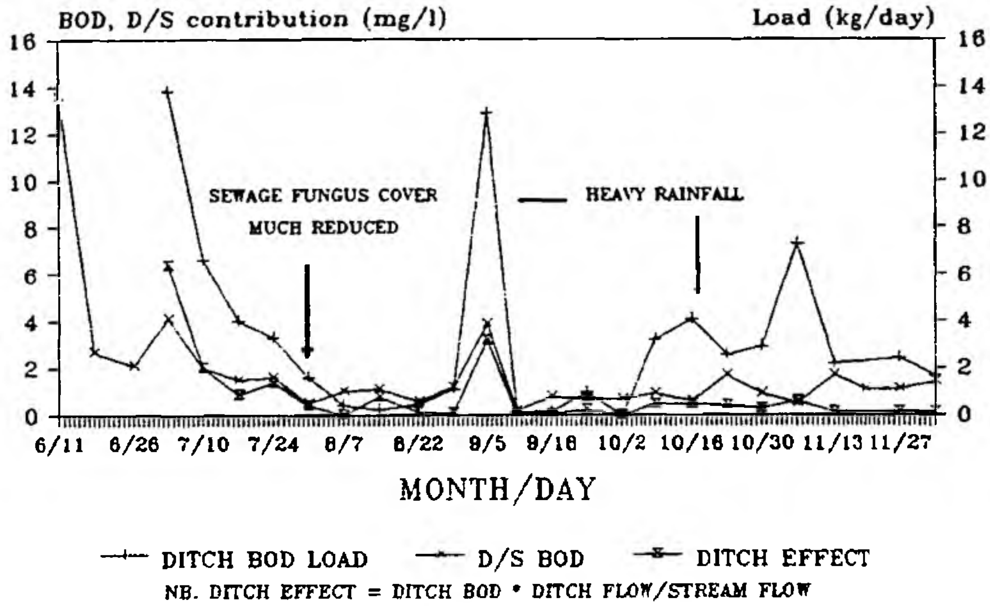


Figure 7.9 Daily rainfall at Pontfaen Brook, 18 June - 4 December 1990.

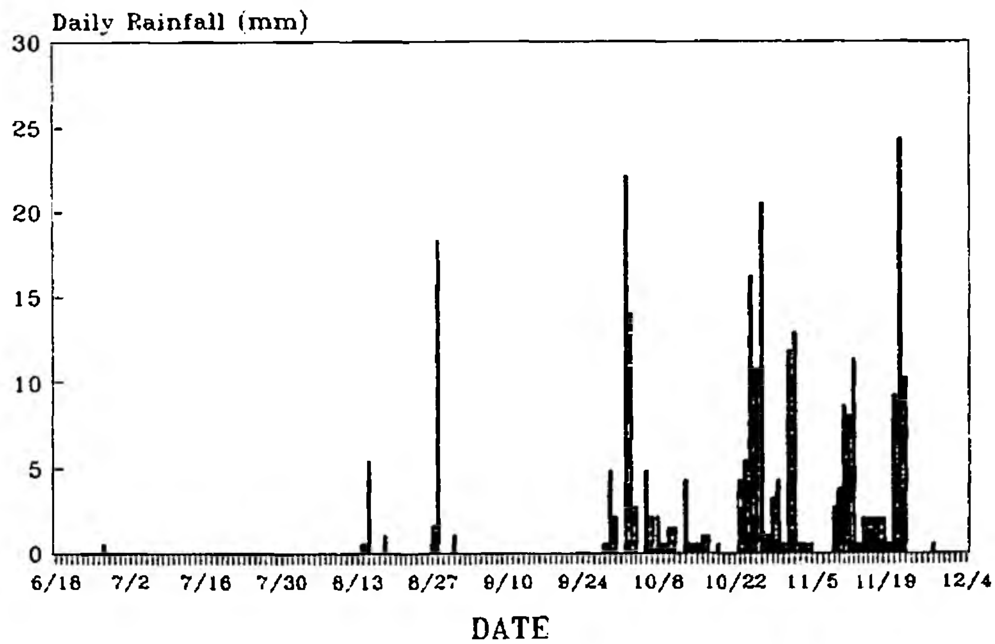
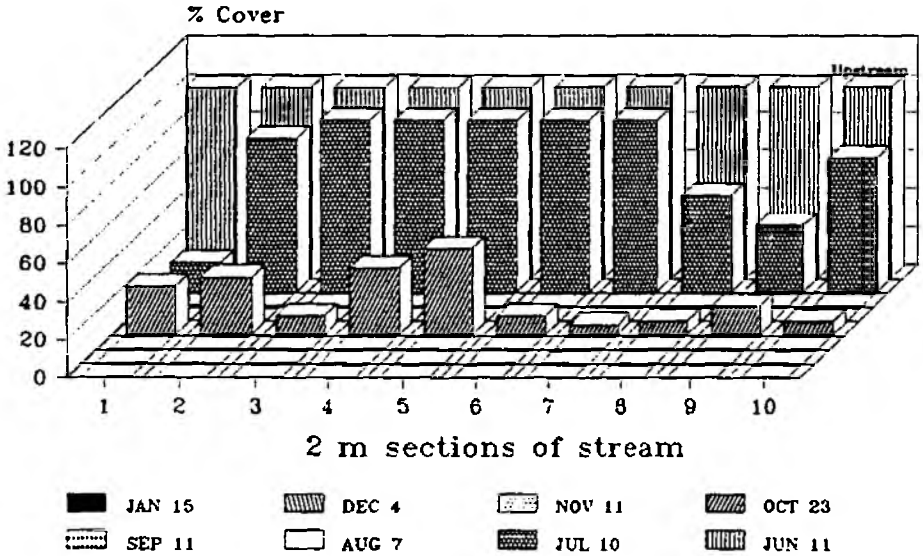


Figure 7.10 Sewage fungus growth in Pontfaen Brook.

a) Site B



b) Site C

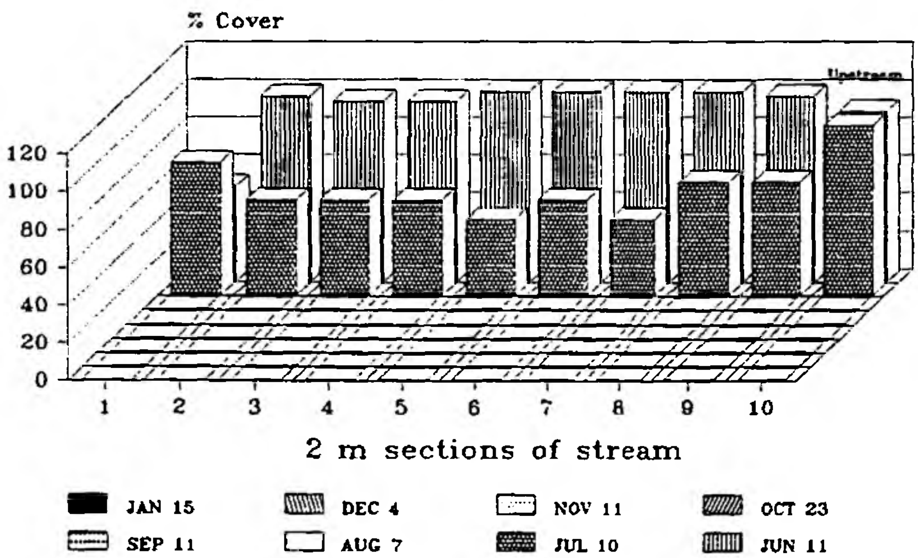


Figure 7.11 Recovery of the macroinvertebrate community of Pontfaen Brook and subsequent response to further pollution.

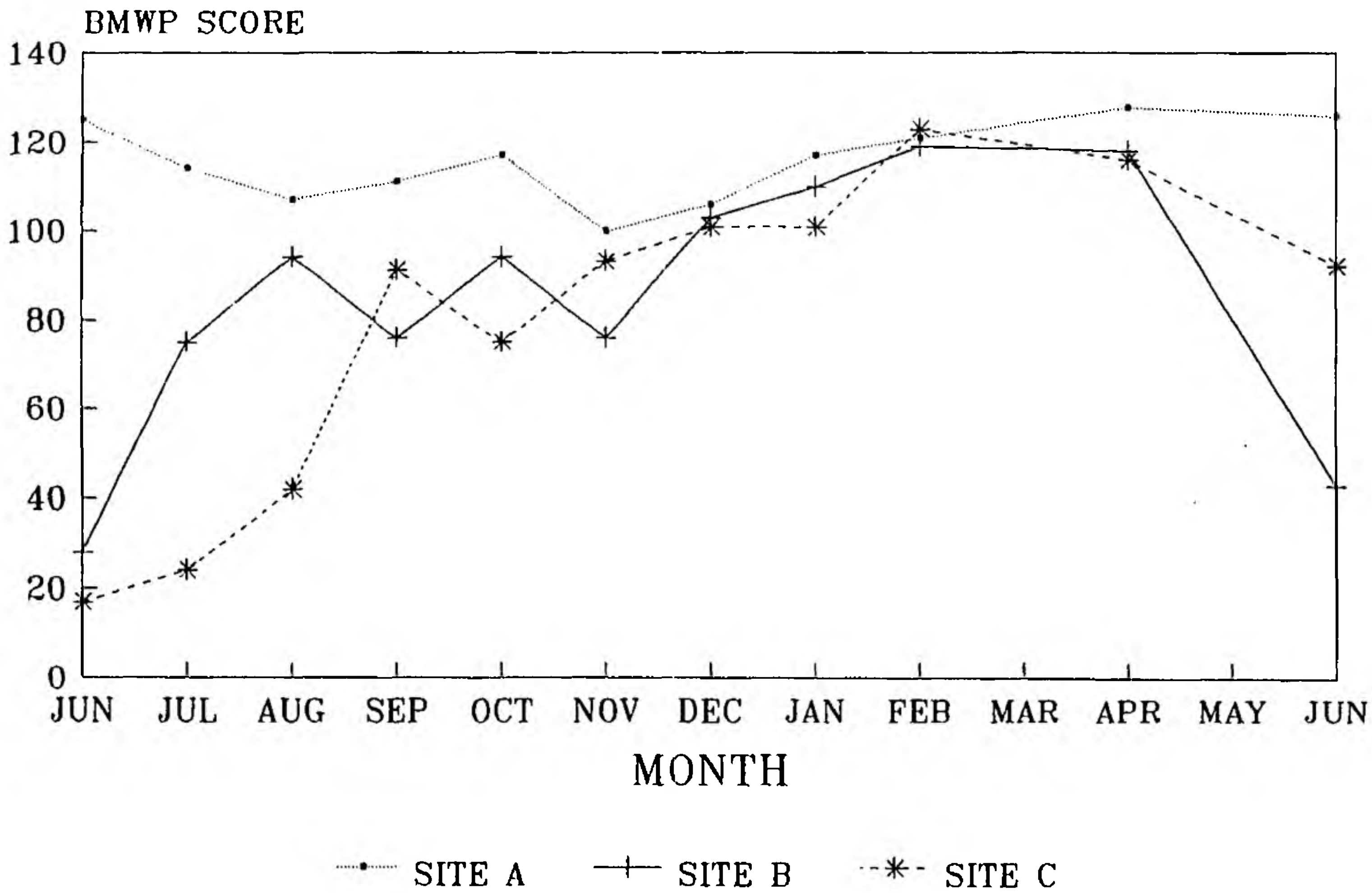
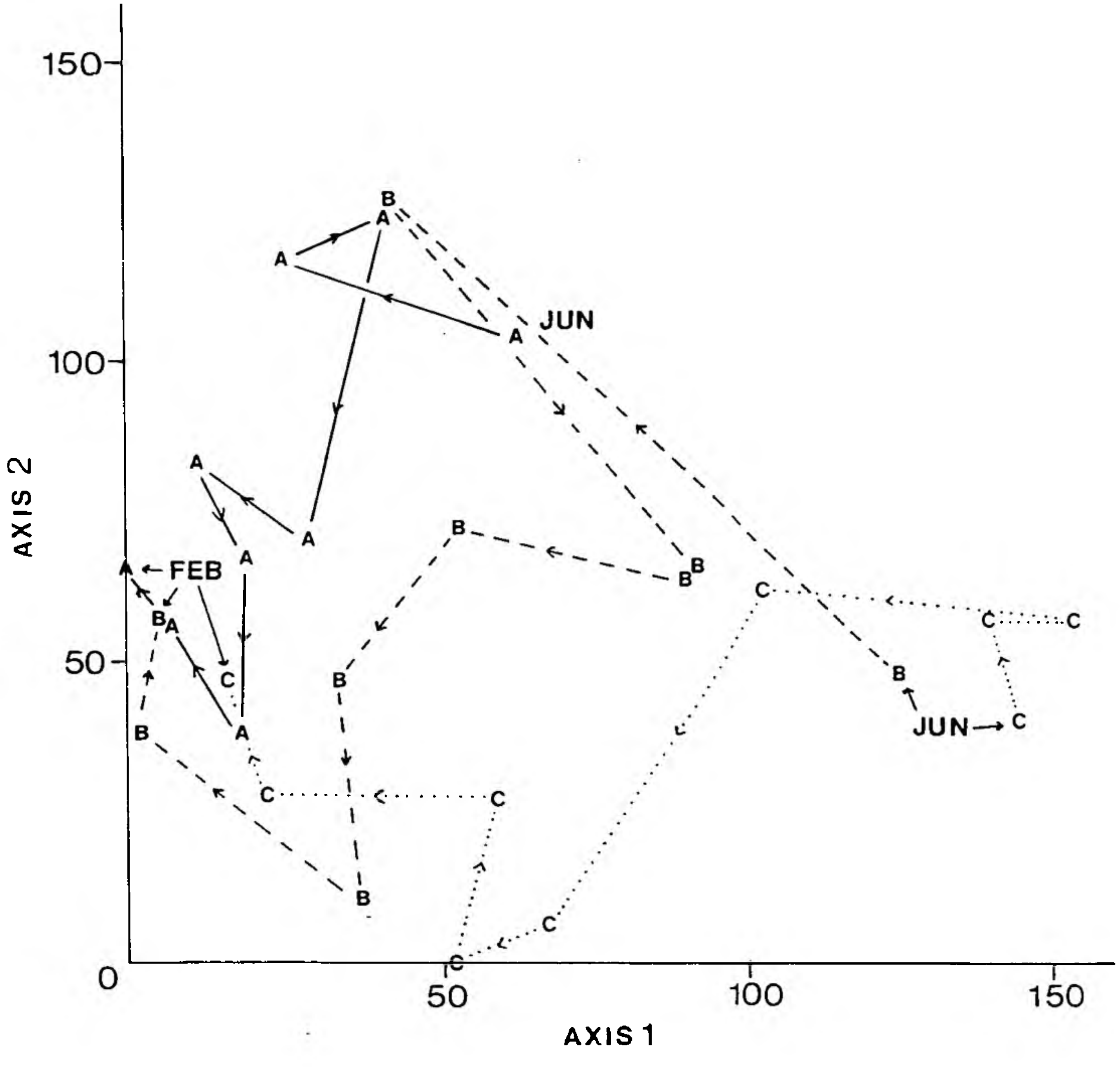


Figure 7.12 Recovery of the macroinvertebrate fauna at Pontfaen Brook.



Plot of axes 1 and 2 derived from DECORANA analysis of monthly macroinvertebrate data for sites A (upstream of ditch), B and C (both downstream of ditch), Jun 90 - Jan 91. Note how the sites converge towards the end of this period.

Table 7.2 Invertebrate community analysis of Pontfaen Brook

Taxa	Site A	Site B	Site C
Ancylidae	1		
Oligochaeta	2	4	3
Gammaridae	1	1	
Heptagenidae	2		
Ephemerellidae	2		
Baetidae	2	2	1
Leuctridae	3		
Perlodidae	2		
Chloroperlidae	1		
Veliidae	1		
Dytiscidae	2		
Hydrophilidae	1	1	
Elminthidae	2		
Rhyacophilidae	2		
Polycentropodidae	1		
Limnephilidae	2		
Hydropsychidae	2		
Tipulidae	1	1	1
Simuliidae	2	2	2
Chironomidae	2	3	3
BMWP SCORE	125	28	17
ASPT	6.25	4	3.4

Taxa scores are logarithmic abundance categories (i.e. 1=1-9, 2=10-99)

Fish populations

Pontfaen Brook is not typical of those sites used in the construction of the HABSCORE model. Altitude is high relative to the distance from the sea. To accommodate this, the model had to extrapolate and this may have resulted in an over estimation of the expected fish densities (Table 7.4). Nevertheless it would appear that all three sites had observed trout densities far below what might be expected. Fry were especially scarce with only one individual being caught at Site A, none elsewhere. It is suggested that all three sites have impacted trout populations but that impact is greater at Sites B and C which deviate markedly more from the predicted for the >0+ cohort (Table 7.4).

Table 7.3 Quantitative sampling of Pontfaen Brook, using a cylinder sampler

Taxa	Site A		Site B		Site C		P
Taxa positively affected by effluent from the ditch:							
Oligochaeta	76	(20-300)	17724	(1660-110880)	6998	(4080-16560)	0.01
Chironomidae	55	(20-140)	11727	(2140-24000)	6863	(3360-11040)	0.01
Taxa showing no significant change:							
Ancylus fluviatilis	6	(0-20)	--		--		*N.S.
Lumbricidae	10	(0-60)	--		--		N.S.
Simulidae	1017	(180-6280)	524	(0-5780)	297	(0 - 2400)	N.S.
Baetis rhodani	220	(80-540)	78	(0-880)	13	(0 - 240)	N.S.
Leuctra moselyi	6	(0-20)	--		--		*N.S.
Leuctra fusca	5	(0-40)	--		--		*N.S.
Isoperla grammatica	14	(0-80)	--		--		*N.S.
Limnebius truncatellus	3	(0-20)	--		--		*N.S.
Helophorus sp.	3	(0-20)	—		13	(0-240)	N.S.
Limnius volkmari	6	(0-20)	--		--		*N.S.
Silo pallipes	5	(0-40)	--		--		*N.S.
Plectrocnemia conspersa	3	(0-20)	--		--		*N.S.
Potamophylax sp.	5	(0.40)	--		--		*N.S.
Hydropsyche siltalai	6	(0-20)	--		--		*N.S.
Pedicia rivosa	3	(0-20)	--		--		*N.S.
Taxa negatively affected by effluent from ditch:							
Gammarus pulex	48	(0-140)	--		--		0.01
Rithrogena semicolorata	270	(60-740)	--		--		0.01
Leuctra inermis	23	(0-100)	--		--		0.05
Drusus annulatus	71	(0-480)	--		--		0.01
Chloroperla torrentium	287	(140-580)	--		--		0.01
Hydraena gracilis	13	(0.40)	--		--		0.05
Elmis aenea	67	(0-180)	--		--		0.05
Rhyacophila dorsalis	15	(0-20)	3	(0-20)	--		0.01
Dicranota spp.	48	(0-100)	12	(0-80)	--		0.05

p - level of significance of the difference between Sites A and B (Kruskal-Wallis test).
* Taxa only at control site but at low density so insufficient replicates for a significant result

Table 7.4 - HABSCORE analysis of Pontfaen Brook

	Site A Age class 0+	Site A Age class (>0+<20 cm)	Site B Age class 0+	Site B Age class (>0+<20 cm)	Site C Age class 0+	Site C Age class (>0+<20 cm)
Observed catch (No 100 m^{-2})	1	22	0	10	0	7
% of model sites with less than this	10	79	3	46	3	33
HABSCORE estimated population (No 100 m^{-2})	217	70	284	70	163	61
% of model sites with a lower estimate	100	100	100	100	99	100
Difference in terms of sd between observed and expected.*	-6.6	-3.5	-8.3	-5.0	-7.4	-5.1

* A value of >1.6 indicates a significant difference at the 90% confidence level.

7.1.5 Discussion

Edwards, *et al* (1991) have shown that various invertebrates sensitive to low dissolved oxygen exhibit downstream drifting behaviour in response to oxygen concentrations which fall below 7 mg l^{-1}. Spot sample data from Pontfaen Brook demonstrated that oxygen concentrations were continually above this value and consequently above the long term lower lethal limit of 5 mg l^{-1} for salmon and trout survival (Alabaster and Lloyd 1982). In addition the maximum unionised ammonia concentration calculated for Pontfaen Brook was 0.004 mg l^{-1} both from spot sample data and from samples taken automatically during rainfall events. This value is an order of magnitude lower than the EQS value for the protection of fresh water fish recommended by WRc (WRc 1988), or the concentration quoted in Alabaster and Lloyd (1980) of 0.025 mg l^{-1} below which no adverse effects to salmonid fisheries are thought to occur.

Caution should be exercised when interpreting water quality information derived mainly from discrete samples but the data suggest that the impact recorded on the stream fauna in the early months of the study was not due to general water quality but was more likely a result of localised stress due to the presence of dense mats of sewage fungus. Respiration of sewage fungus slimes has been estimated at 10 to 20 times greater than that of commonly occurring aquatic plants (WPRL 1969). This can lead to oxygen deficiency within the substratum preventing hatching of salmonid eggs and adversely influencing invertebrates sensitive to low dissolved oxygen concentration. These effects are in addition to the habitat disruption and loss of feeding niches resulting from thick growth of fungus (Curtis 1969).

As the study progressed, the ditch BOD load initially showed a steady decrease along with the downstream BOD concentration. As this dropped below 2 mg l^{-1} the sewage fungus cover also began to decrease and was absent during August. The apparent threshold of 2-3 mg l^{-1} BOD supports the findings of other workers who have observed significant growths of fungus at similar relatively low BODs, thresholds being related to the nature of the organic substrate responsible (WPRL 1969, Quinn and McFarlane 1988). The peak in ditch BOD load on 5 September was the result of a period of heavy rainfall during late August/early September (Figures 7.9, 7.10). Close inspection of Pontfaen Brook showed a small amount of sewage fungus in isolated patches downstream of the ditch but these were absent the following week.

Rainfall became heavier and more sustained after the beginning of October. This, in conjunction with the restoration of the lower section of the drainage ditch towards the middle of the month, appears to have increased the ditch BOD load and downstream BOD sufficiently to promote a short-lived recurrence of sewage fungus which reached an observed peak on 23 October. Rainfall continued throughout the autumn but the sewage fungus had disappeared by the beginning of November. This was possibly because most of the organic material which had built up during the summer in the ditch had been disturbed by the restoration work and flushed out of the system by the heavy rainfall. Also the silage clamp known to have been leaking effluent into the council drain was finally emptied of silage at this time.

Although this study demonstrates a direct relationship between the decline in load of BOD to the stream and an improvement in biological status the factors affecting the BOD loading remain unclear. Declining flows during the summer months, remedial work on the farm and progressive emptying of one of the silage clamps may all have interacted to produce the initial sharp reduction in BOD loading in the ditch and downstream BOD. When the site was visited on a final occasion in June 1991, the stream was found to be severely polluted once more. The cause of the problem was again leakage of silage effluent from a newly filled clamp which had been inadequately sealed despite warnings from the NRA. The case illustrates the reluctance of farmers to seek adequate professional advice over maintenance of storage facilities despite strong warnings from the NRA.

The results of this study contrast with the findings of earlier work describing the impact of farming practices on Clarbeston stream, a sub-tributary of the E. Cleddau (West Wales). In this case there were substantial episodic peaks in ammoniacal nitrogen concentration and depressions in dissolved oxygen in the water column. These resulted from a complex pattern of daily discharge of parlour washings to the stream and rain-generated inputs of yard water contaminated with slurry (Schofield and Bascombe 1990). At Pontfaen the input of silage effluent was more continuous, resulting in a chronic effect on the steam biota apparently due to the growth of sewage fungus rather than direct effects of water quality. Further work is required to investigate the effects of other discrete farm waste management problems on stream quality.

7.2 Study of the Nant Rhydw: silage effluent and parlour dairy washings

7.2.1 Introduction

The second intensive study was designed to compliment the earlier study of a chronic discharge of silage effluent at Pontfaen Brook.

7.2.2 Study area

The Nant Rhydw is a tributary of the Gwendraeth Fach which meets this river at SN 435129 about 9 km from the estuary which lies between Carmarthen and Llanelli in West Wales.

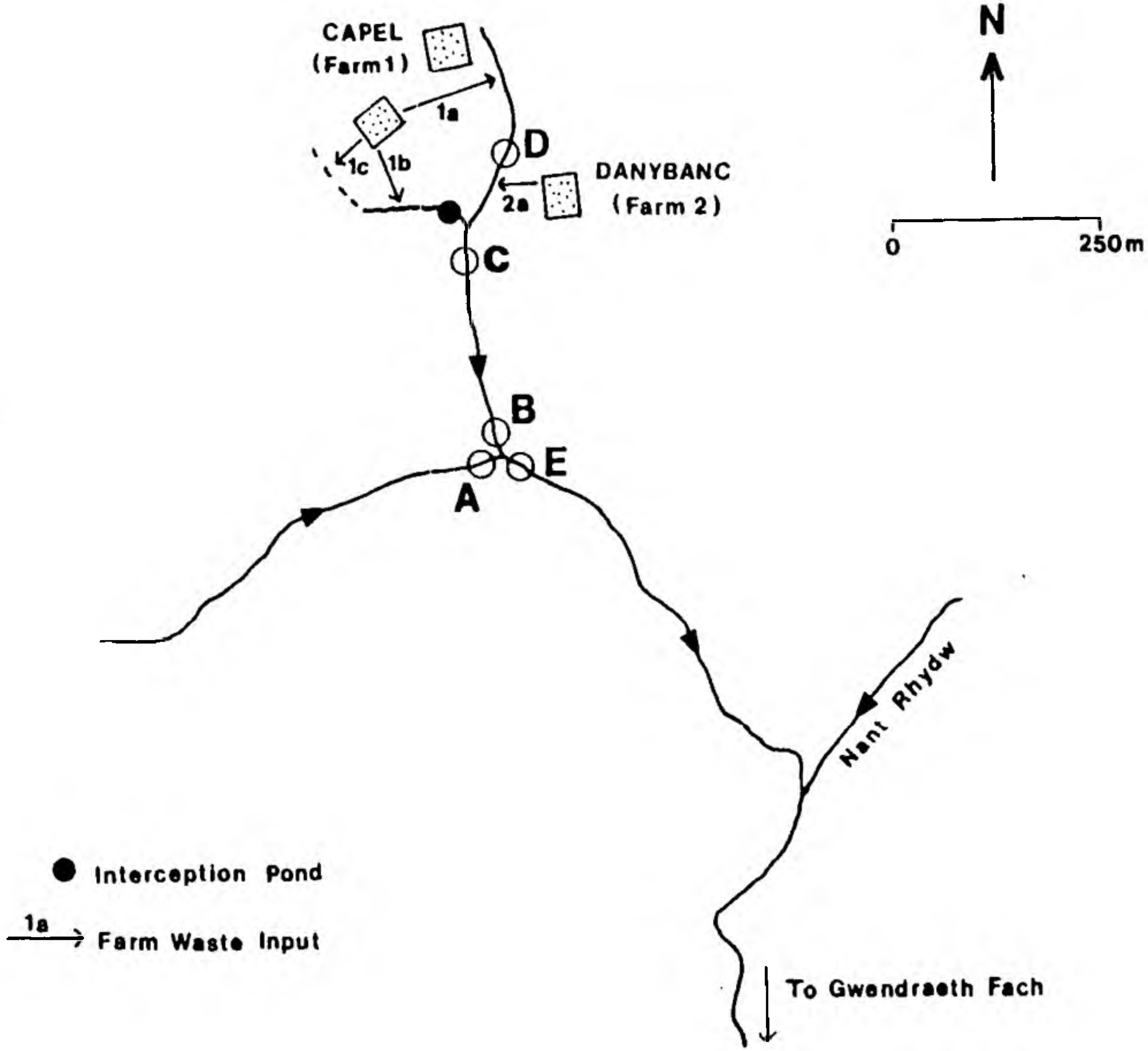
The Rhydw was one of the sub-catchments selected for survey during a large scale evaluation of the rapid appraisal system developed earlier in this project (Section 5.1). Survey work in mid-February confirmed the suspicions of pollution control staff that the catchment was seriously affected by farm pollution. Five farms were found to be causing a definite impact on watercourses whilst a further four could not be eliminated as sources of pollution (see Figure 6.5).

A first order stream polluted by two farms was chosen as a site for more intensive study because a very similar unpolluted tributary joining it some 600 m from its source provided an excellent control site (Figure 7.13).

On 26 March 1991, the NRA Agricultural Liaison Officer inspected both of the farms concerned as part of the programme of farm visits initiated as a follow-up to the findings of the rapid assessment exercise in West Wales (see Section 6.1). At Capel Farm (Farm 1) at the top of the catchment (SN 428149) yard and parlour washings were entering the very top of the stream (input 1a in Figure 7.13) via a road drain from an annex to the main farm buildings. In addition silage liquor was seeping into a roof water drain adjacent to a silage clamp. The drain opened into a field and the effluent entered a spring which ultimately entered the stream at SN 429148 (1c in Figure 7.13).

A small pond intended to intercept this effleunt was very inadequate. It was periodically emptied and the contents spread on land some distance away, but following heavy rain the pond frequently overflowed and polluted the stream. In addition to these two inputs on 9 July following heavy rainfall, a leakage of slurry from a cracked lagoon wall followed the same pathway as the silage effluent (1b in Figure 7.13).

Figure 7.13 Diagram of the Nant Rhydw study area showing farm inputs and study sites.



At Danybanc Farm (SN 431149) effluent from a soakaway for parlour and yard washings was found to be seeping into a highway drainage pipe which discharged into the stream (2a in Figure 7.13).

Site A, the control site, was situated on the unpolluted tributary which entered the main stream below all four inputs. Site B was situated on the stream just above this confluence and Site C was situated further up, just below the overflow from the interception pond. Inputs from a third polluting farm some 200 m downstream of the confluence prevented the study area from being extended further downstream than Site E immediately below the confluence.

Sites A, B, C and E were similar in physiographic terms such that the observed difference in biota could not readily be explained in these terms (See Table 7.5).

Table 7.5 Physiographical characteristics of Sites in the Nant Rhydw study

Site	Mean Width (m)	Mean Depth (cm)	Substratum Composition (%) Bedrock	Boulders	Cobbles	Gravel	Sand & Silt
A	1.2	7	1-5	1-5	21-50	> 50	1-5
B	1.3	7	6-20	1-5	21-50	21-50	6-20
C	1.4	12	6-20	1-5	21-50	21-50	6-20
E	2.0	12	1-5	1-5	> 50	21-50	6-20

Data were collected during HABSCORE assessment on 26 August 1991.
Particle sizes are as follows: Boulders >25.6 cm, Cobbles 6.4-25.6 cm, pebbles 0.2-6.4 cm, gravel 0.2-6.5 cm, sand and silt <0.2 cm.

7.2.3 Methods

Monitoring water chemistry

Monitoring began on 9 April 1991. Two pHOx 100 DPM continuous monitoring units were installed at Sites A and B on the unpolluted and polluted tributaries respectively (Figure 7.13). The meters were equipped to measure dissolved oxygen, temperature, pH, conductivity and ammonium (a close measure of ammoniacal Nitrogen at moderate temperature and circumneutral pH). Data was recorded using Technolog NEWLOG data loggers set to record every fifteen minutes (see Appendix H8). On 14 May a third unit was placed at Site D, on the polluted tributary between inputs 1a and 2a. A Casella rain gauge linked to a TINYLOG data recorder was situated adjacent to Sites A and B (data in Appendix H9). All the instruments were recalibrated and data retrieved at weekly intervals.

Weekly spot samples were taken at seven different points:

Sites A, B, and C, surface run-off contaminated by the interception pond, effluent from input 2a, surface run-off contaminated by input 1a and a sample from the stream source (Figure 7.13). Data are presented in Appendix H10.

Flow estimates were made at Site B by measuring cross-sectional area and estimating velocity.

Biological monitoring

Biological monitoring took place at the three sites; A, B and C and, in the case of kick sampling and electrofishing, at Site C below the confluence of the polluted and unpolluted tributaries. The following aspects of stream biology were studied:

- **Benthic Growth** - Detailed visual estimations of the percent cover of benthic growth (sewage fungus, algae etc.) were made weekly at Sites A, B and C, over ten contiguous 2 m sections of stream. Each week a sample of the growth was collected for microscopic examination so as to identify and quantify the constituent organisms.

- **Benthic Macroinvertebrates** - Three minute kick samples were taken at monthly intervals throughout the study and the invertebrates identified to family level in order that BMWP (Biological Monitoring Working Party) scores and ASPT (Average Score Per Taxon) values could be calculated

(Armitage *et al* 1983). In addition, on 30 April, seven replicate cylinder samples were taken from riffles at Sites A, B and C to allow the population densities of individual species to be compared between the sites.

- **Fish Populations** - On 26 August 1991, 50 m lengths of stream at Sites A, B, C and E (Figure 7.13) were electrofished quantitatively using stop nets. Details of the substratum and in-stream cover were recorded such that predicted salmonid densities could be calculated using the HABSCORE model (Milner *et al* 1985).

7.2.4 Results

Water chemistry

- **pH** - Data from spot samples and the continuous monitors indicated that values of pH at both Site A and Site B fluctuated between 6.3 and 8.3. No large or sudden changes in pH were observed.

- **Dissolved Oxygen** - Data from spot samples showed that the dissolved oxygen concentration at Site A never fell below 8 mg l^{-1} (Figure 7.14). Concentrations at Site C were lower than at Site B but followed a similar pattern. From the beginning of June onwards Sites B and C began to diverge more widely from the control Site A. Particularly low values were recorded on 9 July as a result of the sample being taken during a severe input of farm waste to the stream. The concentration for Site B was 3.6 mg l^{-1} and 0.9 mg l^{-1} at Site C as compared to 9.3 mg l^{-1} at Site A.

  In the first few weeks of the survey continuously monitored data from Site B showed a twice daily minimum in dissolved oxygen. No such pattern was exhibited by data from Site A (see Figure 7.15). The minima

Figure **7.14** Dissolved oxygen levels in the Ryhdw
from spot sampling.

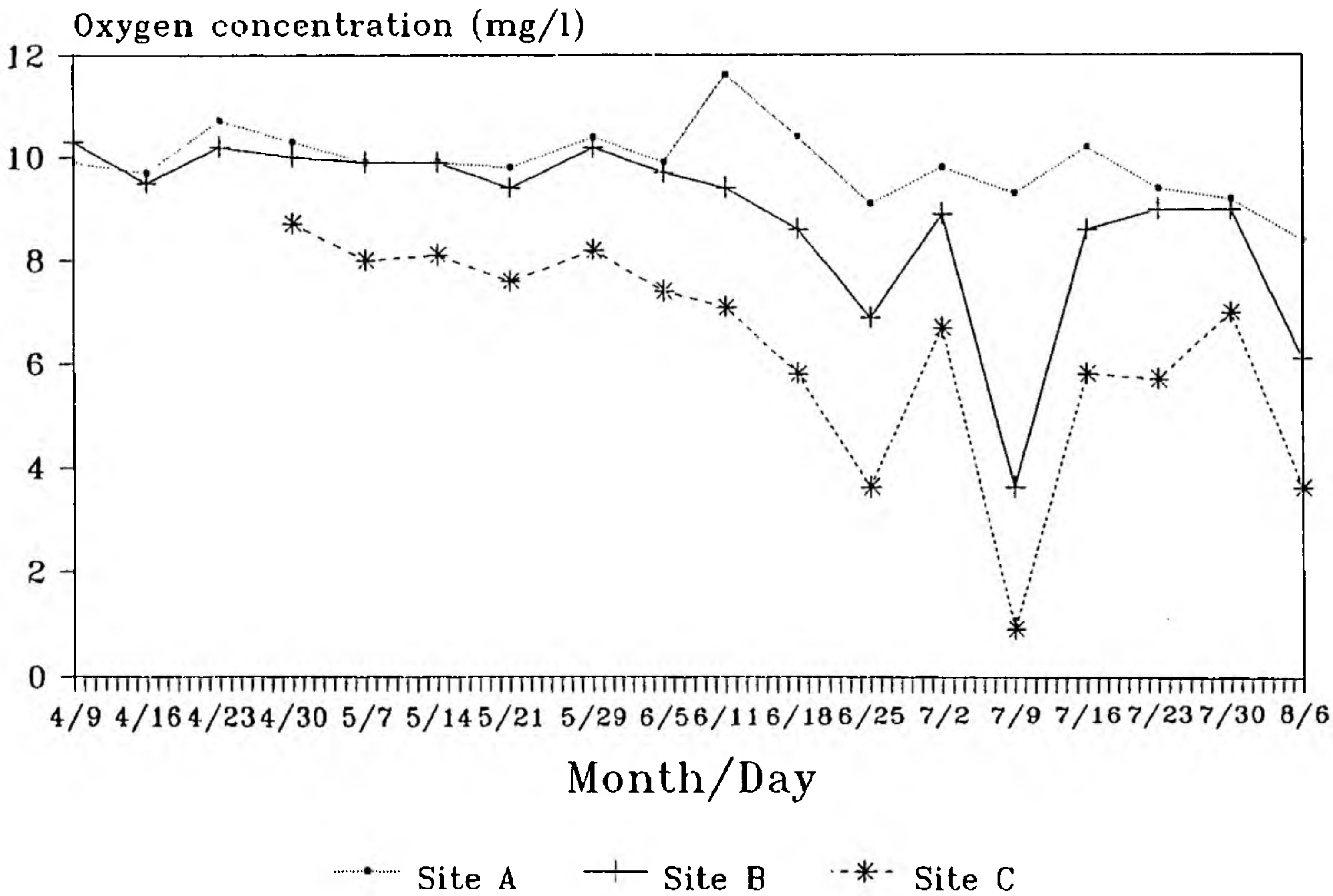
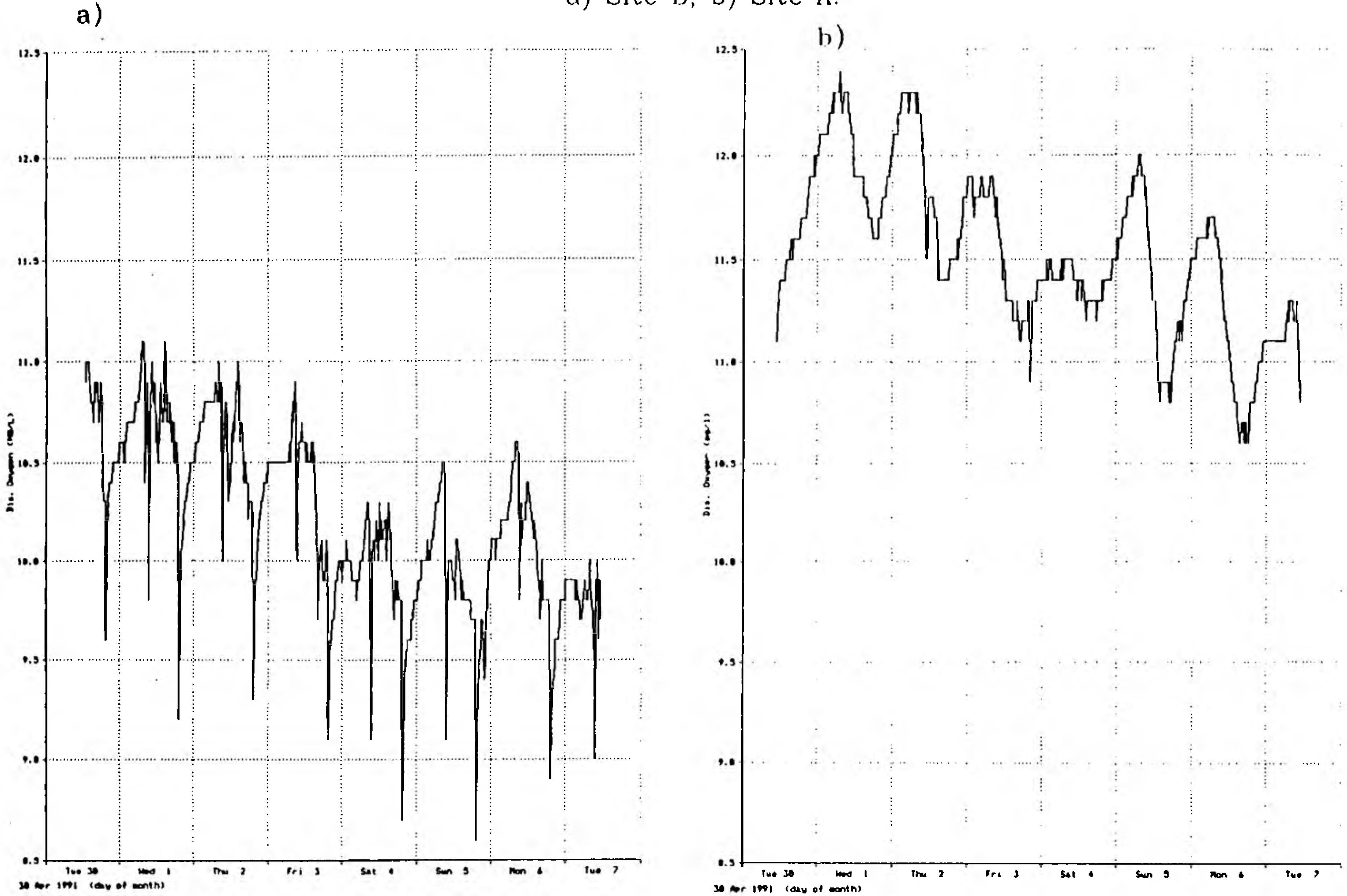


Figure **7.15** Dissolved oxygen levels in the Rhydw from continuous monitoring between 30 April and 7 May: a) Site B; b) Site A.



were about 0.75 mg l^{-1} lower then the background concentration, and lasted for about sixty minutes. They also showed a definite pattern, occurring at the same times each day; 09.30 and 19.30. By the middle of May, however, this phenomenon had ceased.

Later in the study, several severe decreases in dissolved oxygen were recorded at Site B with no corresponding decreases at Site A (Figures 7.16 and 7.17). Oxygen concentrations dropped to below 5.0 mg l^{-1} for periods of between eight and ten hours and on August 2nd (Figure 7.17) the oxygen concentration fell to a minimum of less than 1.0 mg l^{-1}. The sudden but sustained drop to below 2 mg l^{-1} during 6 August (Figure 7.17) was probably due to benthic growth fouling the probe and not as a direct result of the water quality. A spot sample taken on this day showed the dissolved oxygen concentration to be 6.2 mg l^{-1} at Site B.

- **Conductivity** - Spot sample data indicate that the base levels for sites A and B are comparable and lie between 300 and 400 µS cm^{-1}. During rainfall events, continuously monitored data tended to show a decrease in conductivity at the unpolluted Site A but a corresponding increase in conductivity at the polluted Site B.

- **Ammoniacal Nitrogen** - Spot sample data showed that ammoniacal nitrogen concentrations at Site A were generally below 0.1 mg l^{-1} (Figure 7.18). The continuous monitor, however, did pick up two small peaks, both reaching 1.2 mg l^{-1} and both occurring following heavy rainfall. Spot sampling recorded high values of ammoniacal nitrogen at Sites B and C on 9 July with correspondingly high values of the toxic component, un-ionized ammonia (Figures 7.18 and 7.19). There is a general increase in both ammoniacal nitrogen and un-ionized ammonia at Sites B and C to levels greatly in excess of Site A from the beginning of June onwards.

At Site B, at the beginning of the study, ammoniacal nitrogen concentrations exhibited twice daily peaks (see Figure 7.20). These occurred at the same time as the troughs recorded for dissolved oxygen.

Figure **7.16** Dissolved oxygen levels in the Rhydw from continuous monitoring and hourly rainfall between 11 June and 18 June at Site B.

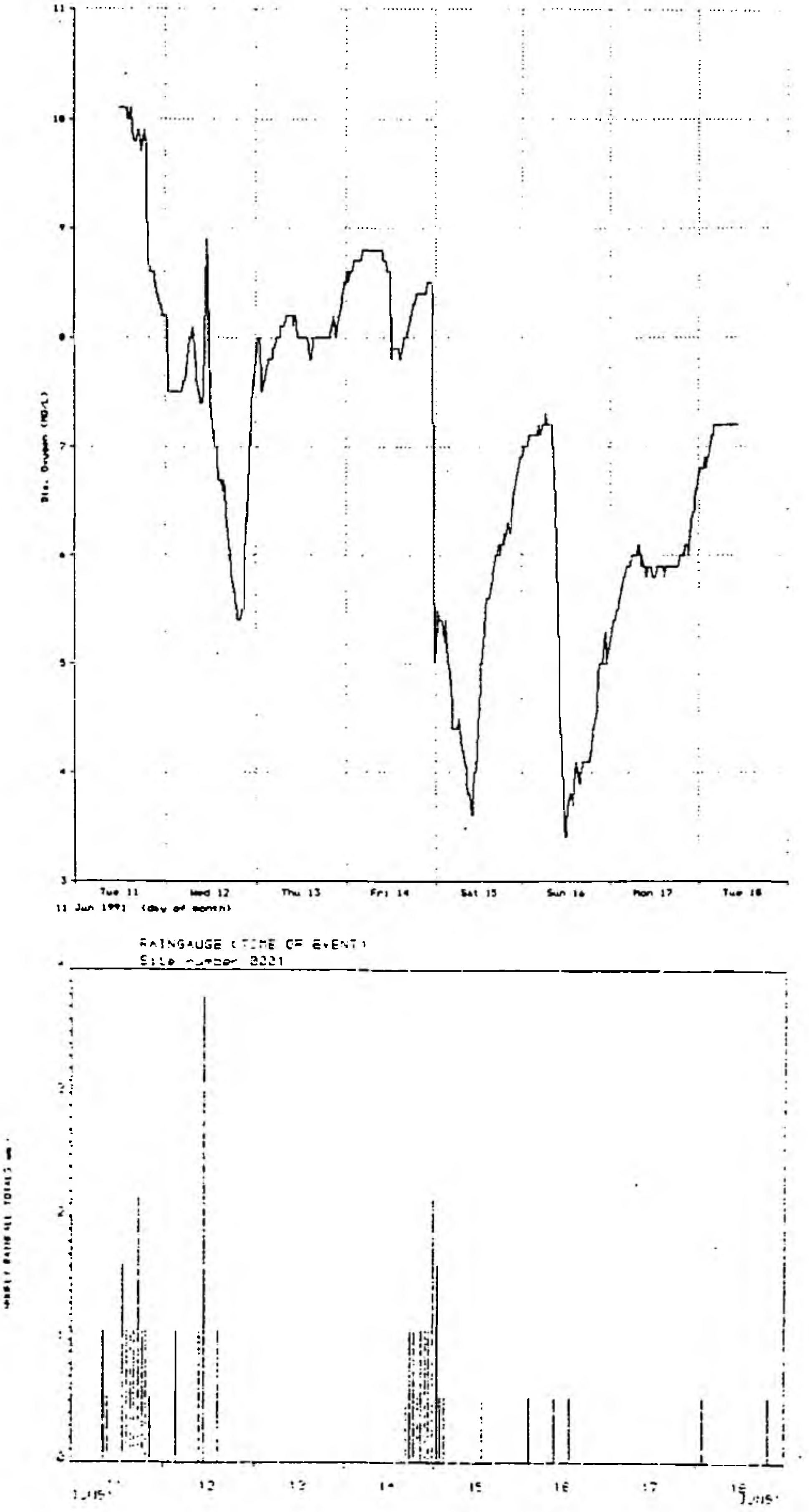


Figure 7.17 Dissolved oxygen levels in the Rhydw from continuous monitoring between 30 July and 6 August: a) Site B; b) Site A.

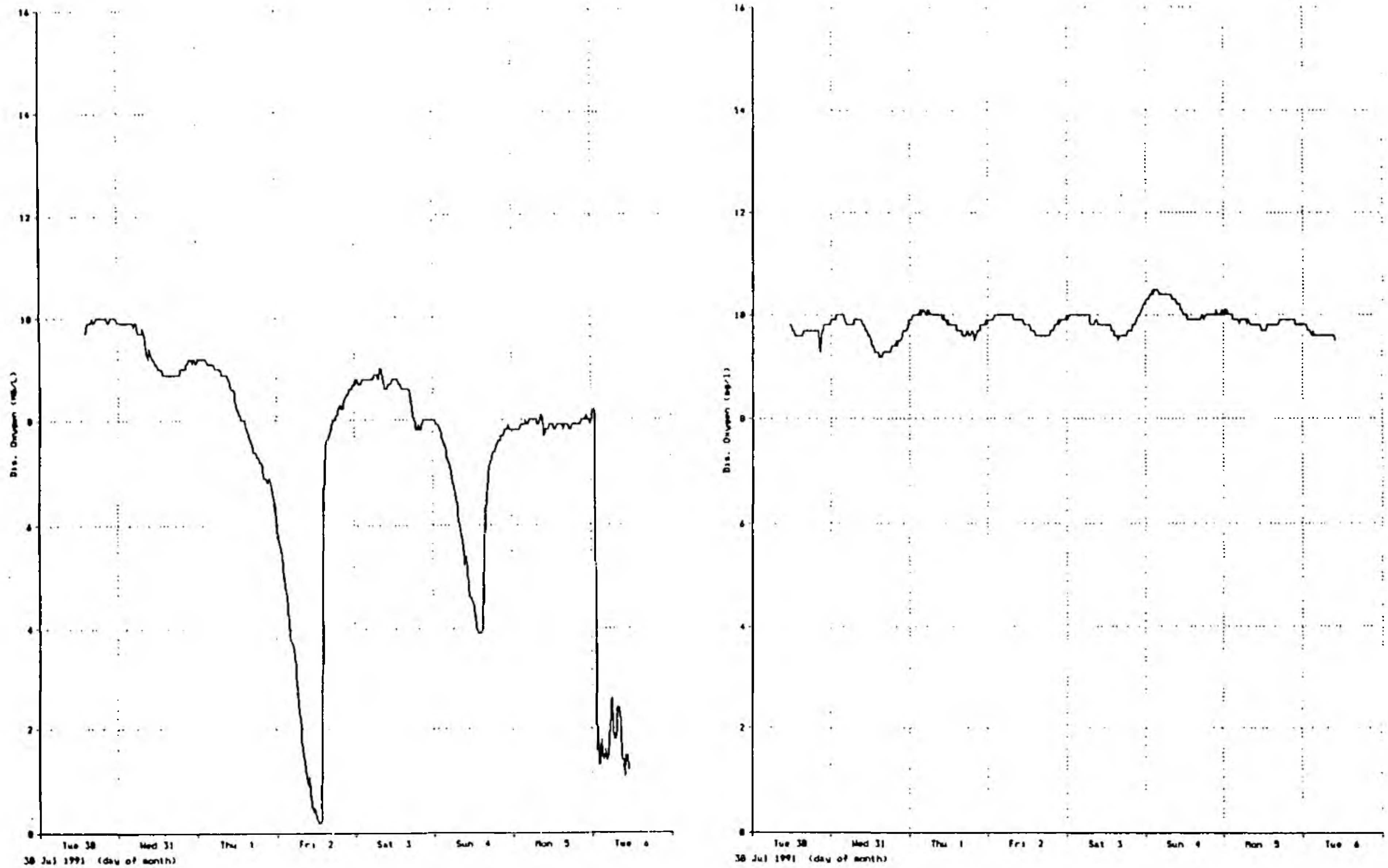


Figure **7.18** Ammoniacal nitrogen concentrations in the Rhydw from spot samples.

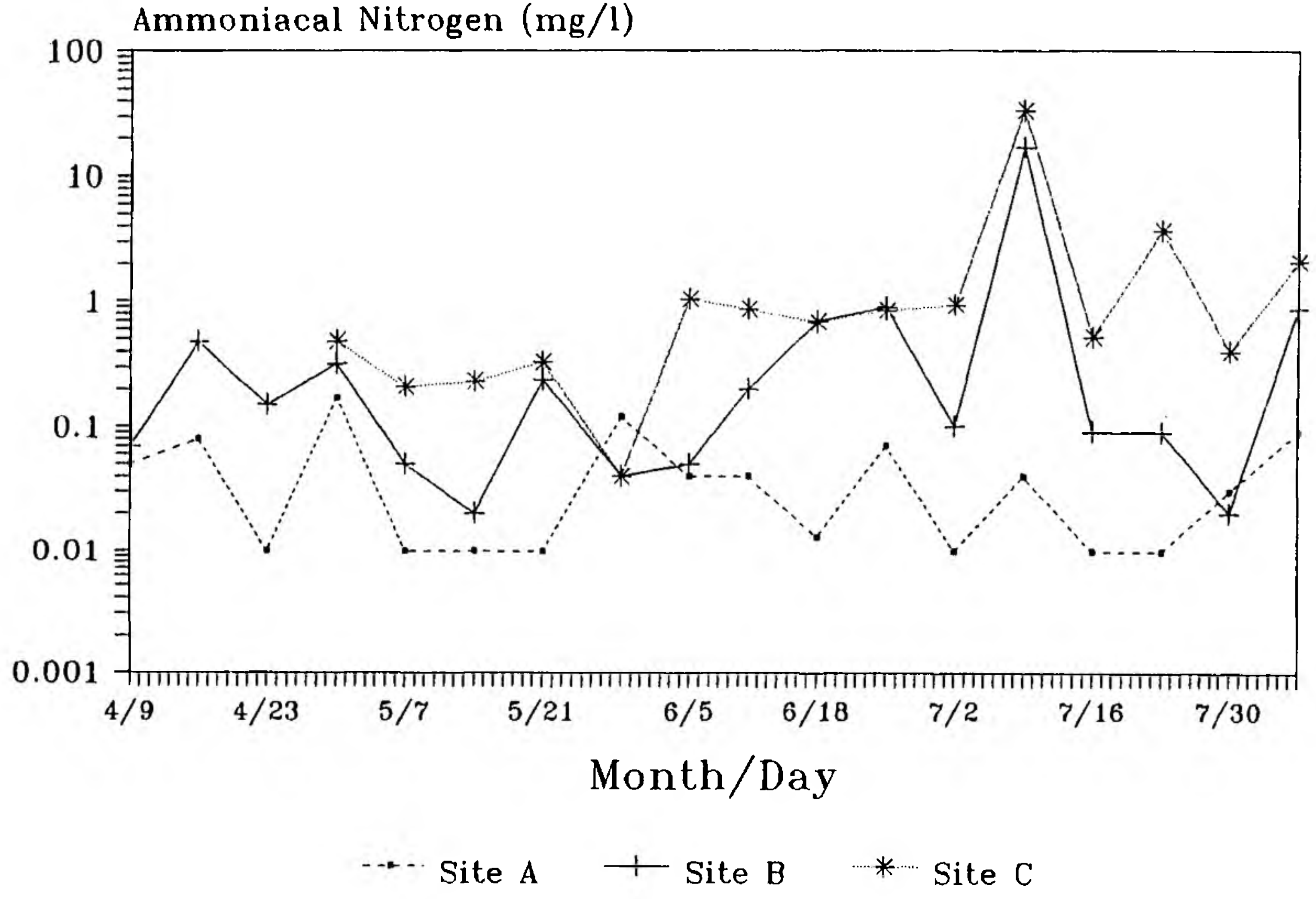


Figure 7.19 Concentrations of unionised ammonia at sites on the Nant Rhydw from spot sampling.

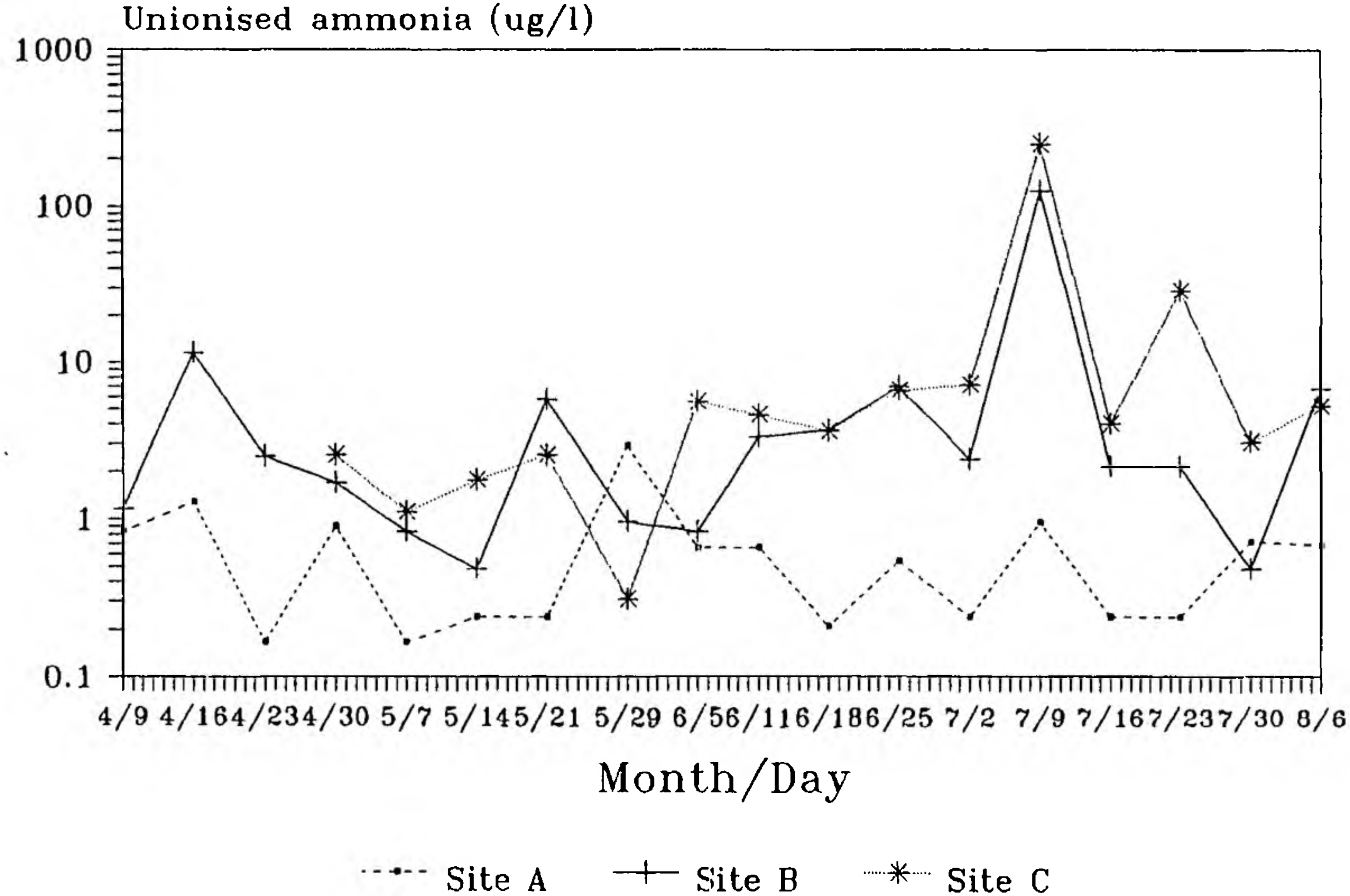
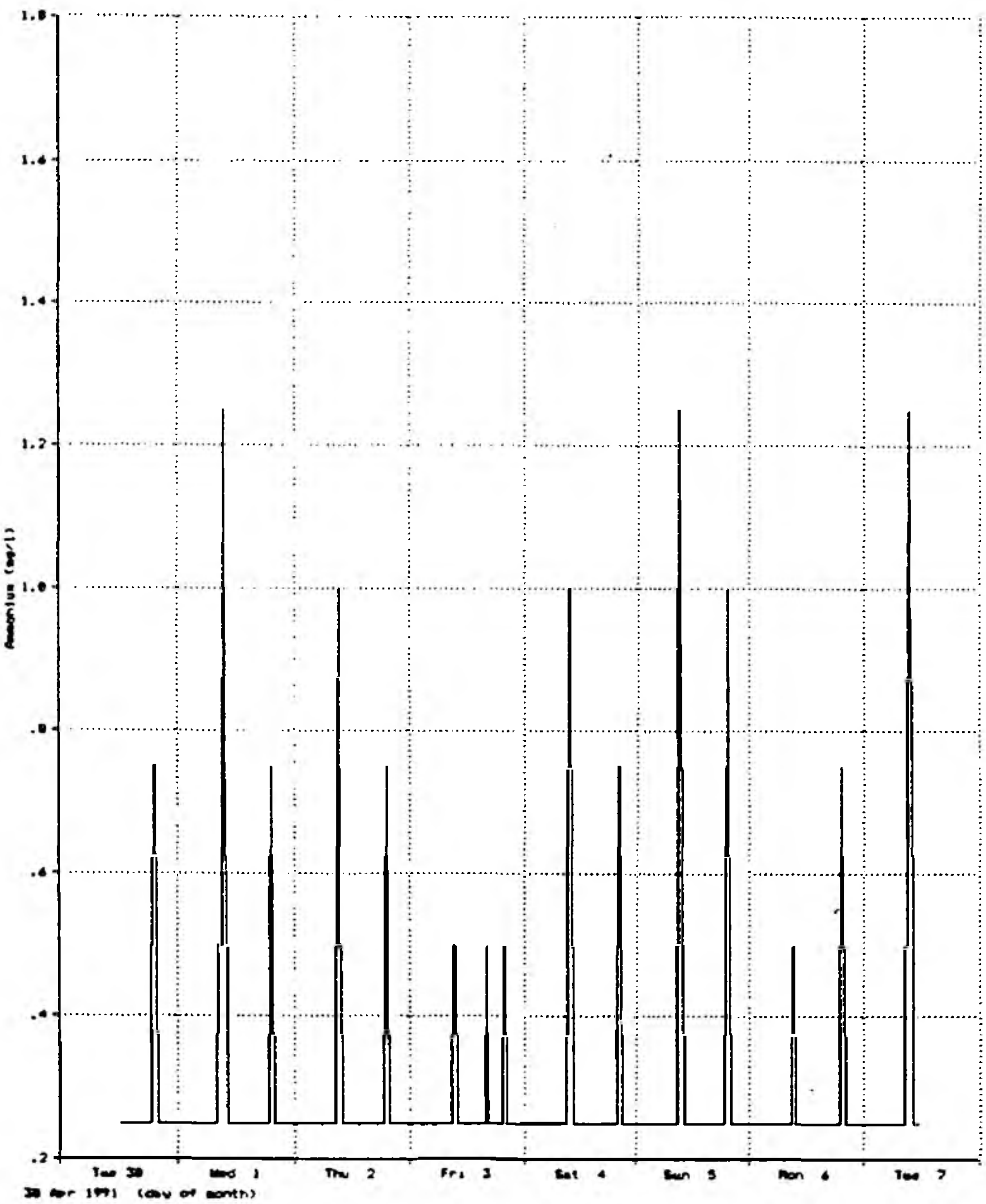


Figure 7.20 Ammonium ion concentrations in the Rhydw at Site B from continuous monitoring between 30 April and 7 May.



The peaks lasted for about an hour and ranged in amplitude from 0.45 mg l^{-1} to 1.22 mg l^{-1}. The un-ionized ammonia concentration corresponding to the maximum peak value was 0.021 mg l^{-1}. As with the dissolved oxygen pattern these peaks had also ceased by the middle of May.

Data from the monitor at Site D also showed twice daily peaks in ammonium concentration and these continued after those at Site B had ceased (Figure 7.21). These peaks were rather 'flatter' than those at Site B lasting for 5 and 7 hours and they also occurred at slightly later times suggesting that the source of ammonia was different for the two patterns. Since Site D was upstream of Farm 2 this effectively eliminated parlour/yard washings from Farm 1 from being the cause of the ammonia peaks at Site B.

Other peaks in ammonium appeared at Site B during heavy rainfall (Figure 7.22). A very high peak of 22 mg l^{-1} (Figure 7.23), occurred on 9 July, the magnitude being confirmed by a spot sample. The equivalent peak of un-ionized ammonia was calculated to be 0.16 mg l^{-1}. This event occurred shortly after, but not during, heavy rainfall. Ammonium concentration was elevated above background for about 20 hours.

- **BOD** - BOD at Site A generally remained below 1.8 mg l^{-1} (Figure 7.24). BOD for Sites B and C follow a similar pattern to that for the stream input (which was contaminated with silage effluent), especially after the beginning of June illustrating the importance of this source of organic loading. The maximum BOD recorded at Site B was >525 mg l^{-1} on 9 July. This compares to a BOD of 1.8 mg l^{-1} at Site A at this time.

- **General** - The most pronounced differences in water quality between the polluted and unpolluted tributaries corresponded to an increase in the frequency and intensity of rainfall and a slight increase in stream flow from early June onwards (Figure 7.25). The dramatic decreases in dissolved oxygen concentration recorded by the continuous monitors also began in early June.

Figure **7.21** Ammonium ion concentrations in the Rhydw at Site D from continuous monitoring between 11 and 18 June, 1991.

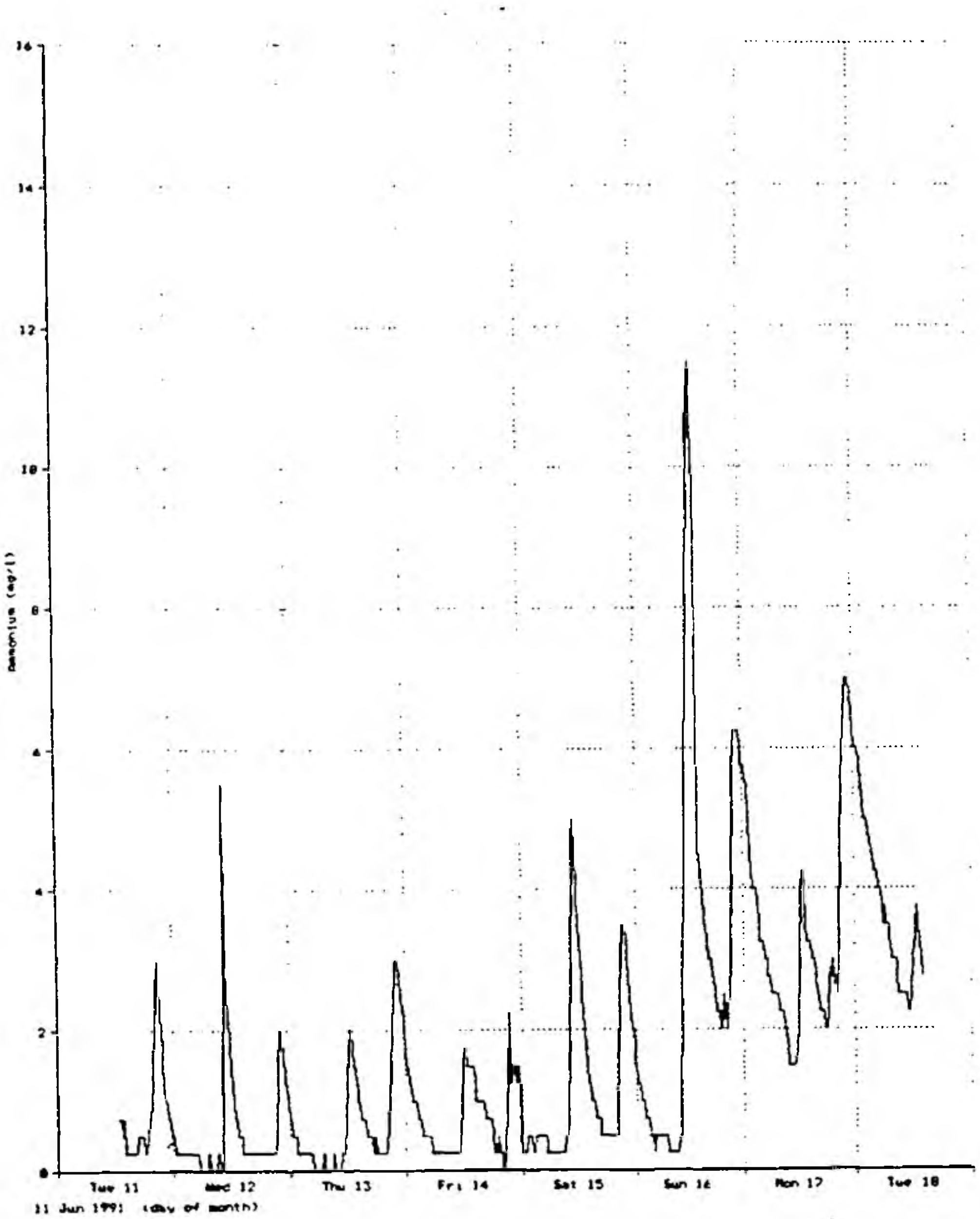
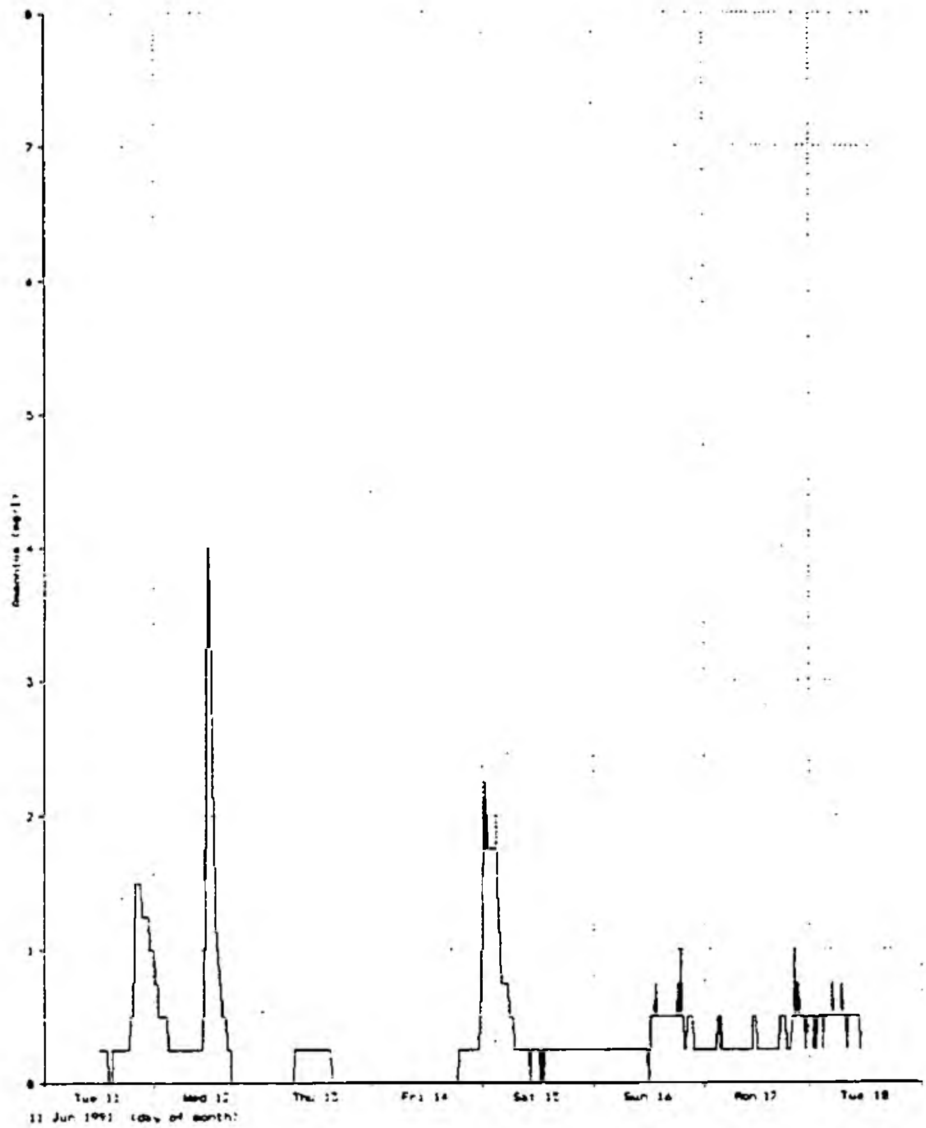


Figure 7.22 Ammonium ion concentrations in the Rhydw at Site B from continuous monitoring and corresponding hourly rainfall between 11 and 18 June, 1991.



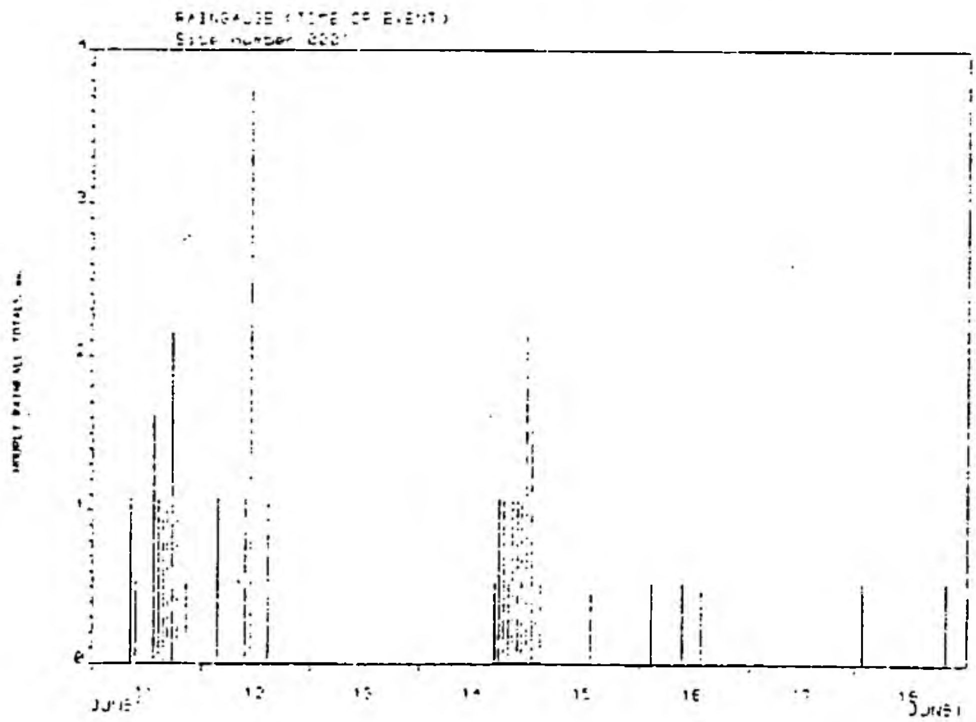


Figure 7.23 Ammonium ion concentrations in the Rhydw at Site B and corresponding hourly rainfall between 3 and 10 July, 1991.

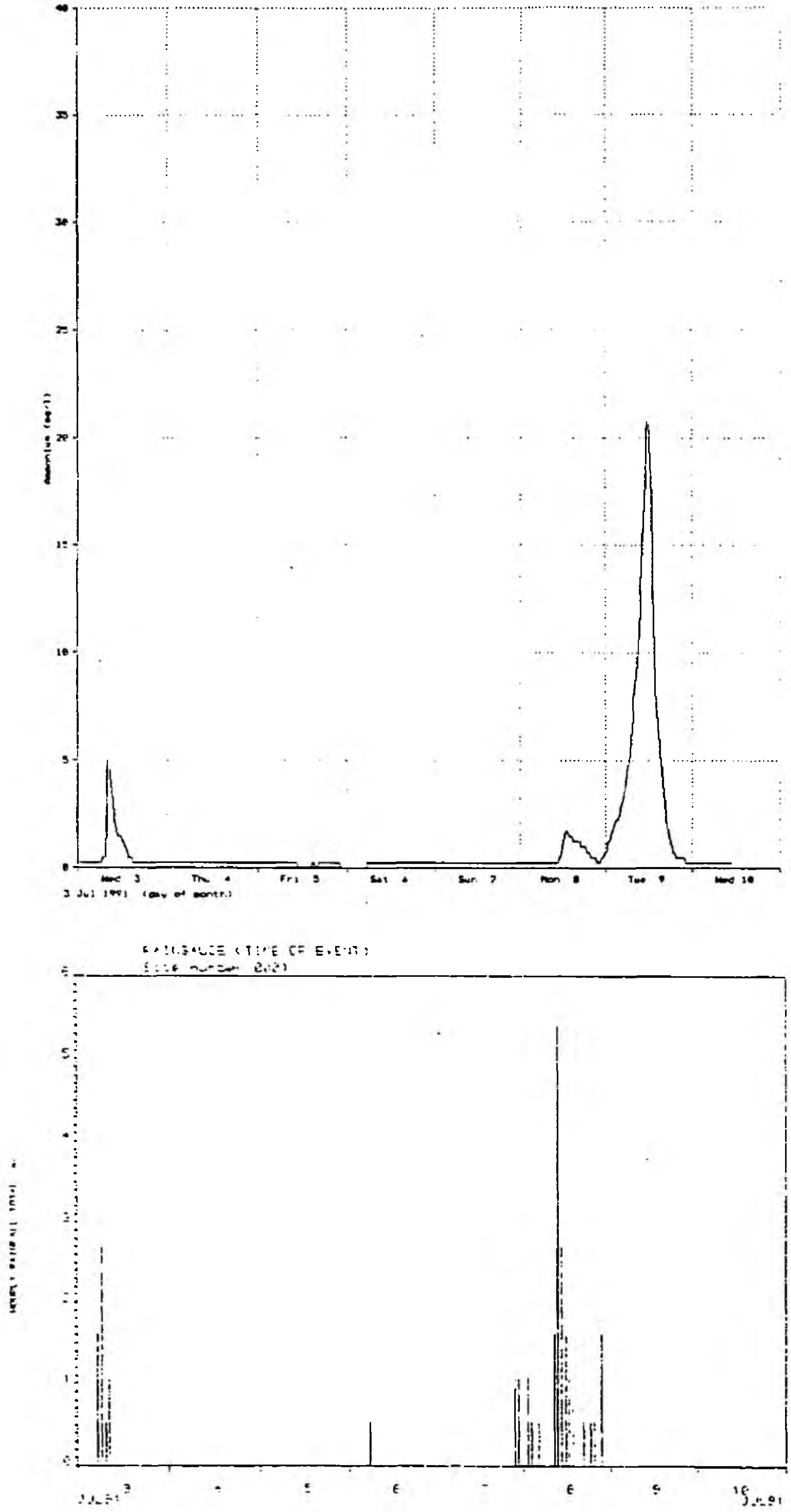


Figure 7.24 Biological Oxygen Demand (BOD) in the Rhydw catchment from spot sampling.

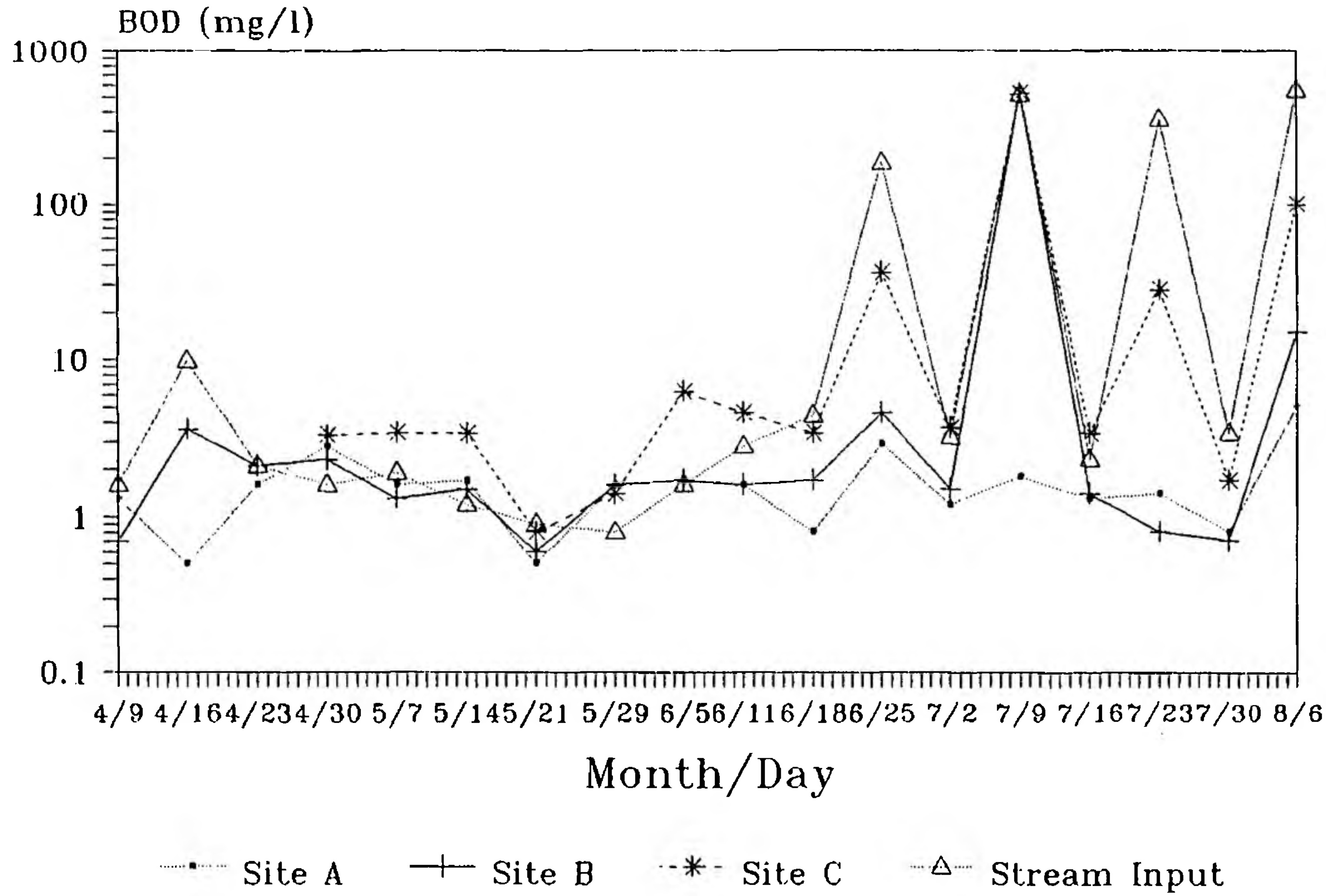
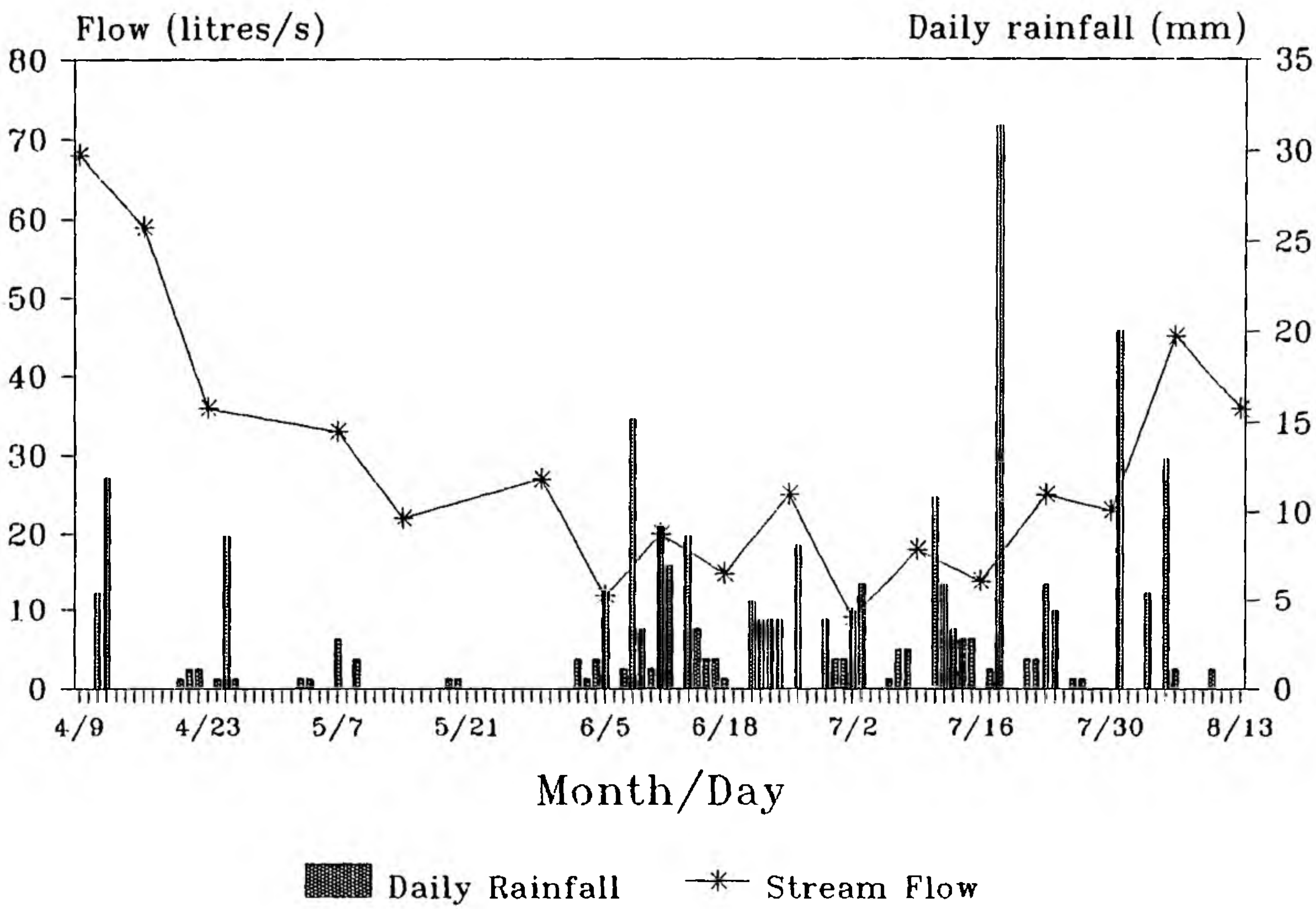


Figure **7.25** Stream flow in the Rhydw at Site B
and daily rainfall.



Benthic growth

Monitoring of benthic growth did not begin until the middle of April. No noticeable growth of any kind was recorded at Site A, the surface of the stones appearing clean and not slimy. By contrast there was a cover of benthic growth at Site C of variable composition (Figure 7.26). No estimate was made during the second week of July as the water was too discoloured for the stream bed to be visible.

Initially at Site C the filamentous bacterium *Sphaerotilus natans* was the dominant organism but protozoa, diatoms and other material (mainly decaying *Sphaerotilus*) took over from the second week in May to the second week in June. This coincided with a decrease in the percentage cover of the growth. From the end of June through to the end of the survey >60% growth returned to Site C and *Sphaerotilus natans* again dominated apart from the last two weeks when Zoogloeal growth became predominant.

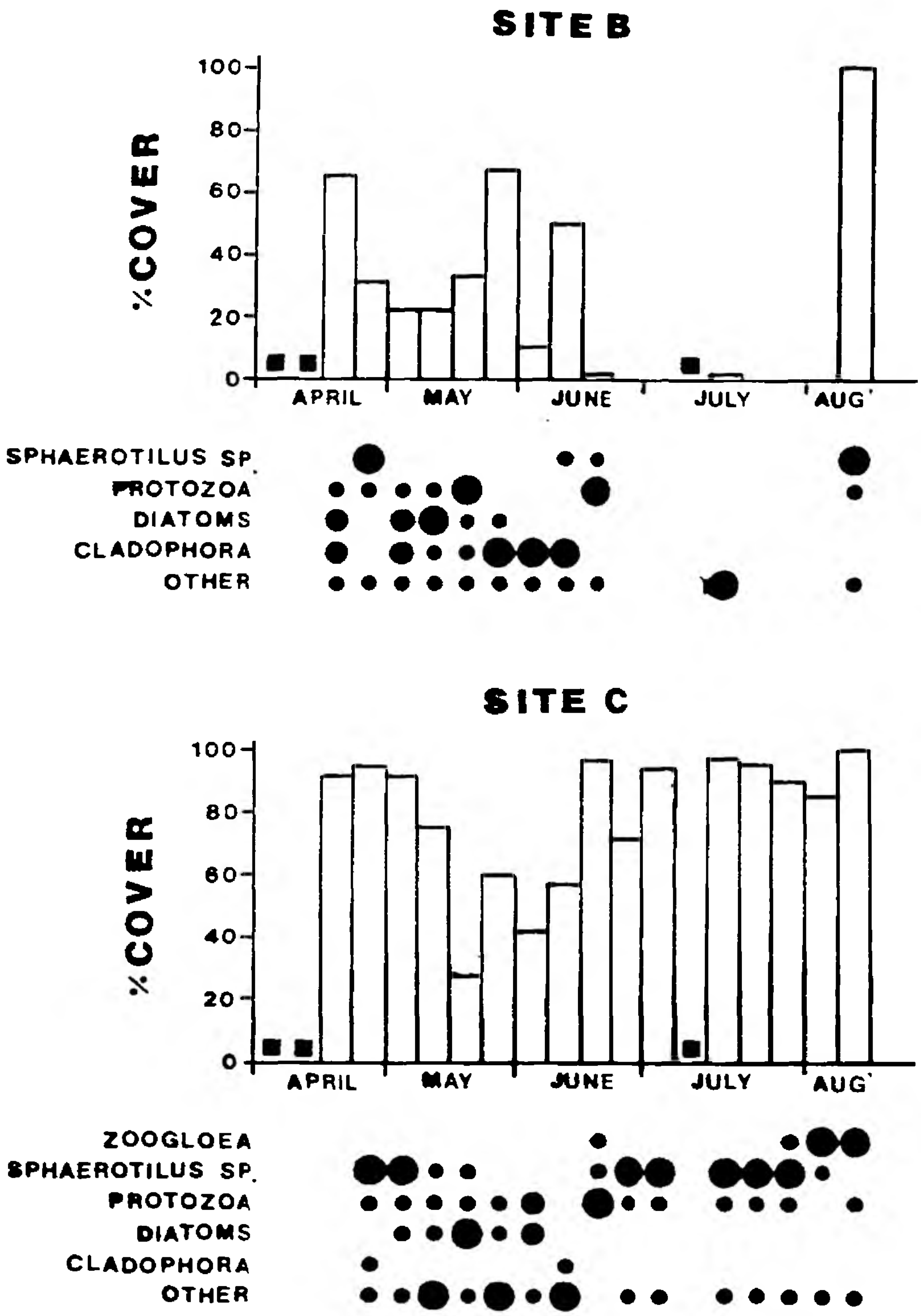
Cover at Site B began at 60% but by the end of April had fallen to below 40% where it remained until the end of May. *Sphaerotilus natans* was the dominant organism at the end of April but diatoms and protozoa and the filamentous alga *Cladophora* took over during May. Cover at Site B was practically absent from the end of June until a sudden regrowth during the second week of August when *Sphaerotilus natans* was again the dominant organism.

During the early weeks of the study decaying sewage fungus was found to be present above Site C and above the input from the interception pond. Thereafter benthic growth was seen to arise just below the input from the pond.

Macroinvertebrate populations

Only kick samples from the beginning and end of the survey period were processed. Initial sampling in April 1991 revealed a diverse invertebrate assemblage at Site A and impoverished assemblages at Sites B and C. An intermediate community was found downstream of the confluence. At Site A the BMWP score was 143 as compared with a score of 63 at Site B and 14 at Site C.

Figure 7.26 Benthic growth at sites in the Rhydw catchment.



The site downstream of the confluence scored 106 (Table 7.6).- At the end of the survey the scores show a similar pattern of 104, 67, 20 and 70 respectively. Two groups of organisms tolerant of organic pollution, oligochaetes and Chironomidae, showed high densities of 3960 m^{-2} and 3400 m^{-2} at Site B and 19 970 m^{-2} and 8840 m^2 at Site C respectively as compared with from 210 m^{-2} and 224 m^{-2} at Site A. Conversely the densities of several taxa intolerant of organic pollution showed markedly higher densities at the control Site A. For example the mayflies *Rhithrogena semicolorata* and *Baetis muticus* had densities of 240 m^{-2} and 100 m^{-2} at Site A as compared with total absence at Sites B and C (Table 7.7).

Table 7.6 Invertebrate community analysis of sites in the Nant Rhydw catchment

Taxa	Site A Apr	Site A Aug	Site B Apr	Site B Aug	Site C Apr	Site C Aug	Below Confluence Apr	Below Confluence Aug
Heptageniidae	2	1					1	
Leptophlebidae	1	1						
Taeniopterygidae							1	
Leuctridae	1	1						
Perlodidae	1						1	
Chloroperlidae	1							
Goeridae	2						1	
Psychomyiidae			1					
Philopotamidae	1	1						
Nemouridae	1		1	1			1	1
Rhyacophilidae	2	2		1			1	2
Polycentropodidae	1	1	1	1			1	1
Limnephlidae	2		1				1	
Ancylidae			2	2			1	1
Gammaridae	2	3	1	3		1	1	3
Dytiscidae				1				
Hydrophilidae		1		1				
Elminthidae	1	2						1
Hydropsychidae	1	1					1	
Tipulidae	1	1						1
Simulidae	1	1		4	1	4	1	4
Planariidae	1	1						
Baetidae	2	1	2	2			2	2
Hydrobidae	3	3	2	3	1	1	2	3
Planorbidae				1				1
Sphaeridae	2		1	1			1	1
Glossiphonidae	1	1	1			1		1
Erpobdellidae			1	2	1		1	
Asellidae		1					1	
Chironimidae	2	1	2	3	2	4	2	1
Oligochaeta	1	1	3	2	3	4	2	2
BMWP SCORE	143	104	63	67	14	20	106	70
ASPT	6.2	5.8	4.5	4.5	2.8	3.3	5.9	4.4

Taxa scores are logarithmic abundance categories (i.e. 1=1-9, 2=10-99 etc.)

Fish populations

The fish population at all four sites fished was very impoverished. The only species recorded were eels (*Anguilla anguilla*). Eleven were caught below the confluence, one each at Sites A and B and none at Site C. The HABSCORE information has not yet been analysed.

7.2.5 Discussion

Effluents from the two farms in the study area were found to be having a severe impact on the biology of the stream to which they discharged. The invertebrate community of this polluted stream was found to be comparable to that found below the discharge of silage effluent at Pontfaen Brook whilst pollution was occurring at this site (Section 7.1). No discernible improvement in biological quality occurred during the course of the study due to a combination of continuing inputs and lack of an unpolluted upstream source for drifting colonists.

The water quality observations can be interpreted in terms of the three types of input which were entering the polluted tributary: parlour washings, overflow from the interception pond and the input of slurry from the cracked lagoon on 9 July. The pattern of these inputs and their relative importance as sources of biological impact are discussed below.

Similar peaks in ammonium ion concentration to those detected at Site B at the beginning of the survey were found by Schofield (1988) who related them to daily inputs of parlour washings. From their timing and duration it is

Table 7.7 Quantitative macroinvertebrate sampling of tributaries of the Nant Rhydw using a cylinder sampler

Taxa	Density (No. m^{-2}) Site A	Site B	Site C	p(AvB)	p(AvC)	p(BvC)
Taxa preferential to polluted sites:						
Oligochaeta	210 (60-920)	3955 (1460-13920)	19971 (1440-46560)	***	***	*
Chironomidae	224 (80-500)	3398 (880-8400)	8841 (4320-14400)	***	***	*
Taxa, only found at unpolluted site but at low density:						
Ecdyonurus dispar	33 (0-40)	----	----			
Hydraena gracillis	2 (0-20)	----	----			
Helodidae	7 (0-20)	----	----			
Polycelis sp.	7 (0-20)	----	----			
Elmis aenea	13 (0-40	----	----			
Habrophlebia fusca	32 (0-400)	----	----			
Ceratopogonidae	17 (0-100)	----	----			
Wormaldia sp.	4 (0-20)	----	----			
Planorbis sp.	2 (0-20)	----	----			
Leuctra nigra	11 (0-60)	----	----			
Potamophylax cingulatus	4 (0-20)	----	----			
Micropterna sp.	2 (0-20)	----	----			
Dicranota sp.	7 (0-20)	----	----			
Ostracoda	5 (0-80)	----	----			
Rhyacophila dorsalis	2 (0-20)	----	----			
Thaumaleidae	2 (0-20)	----	----			
Silo pallipes	3 (0-40)	----	----			
Chelifera sp.	2 (0-20)	----	----			
Leuctra moselyi	2 (0-20)	----	----			
Taxa preferential to unpolluted sites:						
Baetis rhodani	124 (20-480)	70 (0-160)	2 (0-20)		***	*
Gammarus pulex	527 (40-2460)	23 (0-140)	7 (0-40)	**	***	
Potamopyrgus jenkinsi	835 (20-5340)	241 (120-560)	2 (0-20	*	***	***
Hydropsyche instablis	44 (0-100)	4 (0-40)	----	*	*	
Agapetus sp.	63 (0-580)	2 (0-20)	----	*	**	
Plectrocnemia conspersa	33 (0-100)	10 (0-60)	----		*	
Pisidium sp.	13 (0-60)	37 (0-80)	----		*	
Ancylus fluviatilus	9 (0-120)	14 (0-40)	----			
Isoperla grammatica	31 (0-100)	----	----		*	
Rhithrogena semicolorata	239 (80-460)	----	----	***	***	
Baetis muticus	102 (20-580)	----	----	***	***	
Elaeophila sp.	16 (0-60)	----	----		*	
Chloroperla torrentium	24 (0-80)	----	----		*	

Values quoted are mean densities calculated on logged data and then back-transformed. Figures in brackets are ranges. The final three columns give probabilities of there being a significant difference in density between the sites indicated as given by Mann-Whitney tests (Sokal and Rohlf 1981).

suggested that the peaks of ammonium detected at Site B and the corresponding oxygen troughs were a result of parlour washings from Danybanc farm. The peaks ceased following remedial action taken by the farmer. A further input of parlour washings from Capel farm was still being recorded at the end of the study. Edwards *et al* (1991) have shown that invertebrates begin to show enhanced drift when un-ionized ammonia concentrations rise to around 0.5 mg l^{-1}, but even when they are as high as 5.0 mg l^{-1} drift is still in the order of that occurring naturally during hours of darkness. Low dissolved oxygen is perhaps more likely to affect invertebrate communities as during summer conditions concentrations of below 4.0 mg l^{-1} are required to cause invertebrate drift to rise significantly above background levels. However in the case of the parlour washings the maximum un-ionized ammonia was 0.021 mg l^{-1} and the minimum dissolved oxygen concentration was 8.6 mg l^{-1}. Although this short-lived un-ionized ammonia concentration is equal to the maximum value recommended for the long-term protection of salmonid fisheries (Seager, *et al.*, 1988) it seems unlikely that parlour washings are responsible for the poor biological quality of the polluted tributary.

Overflow from the interception pond occurred following heavy rainfall but was also dependent on the speed with which the farmer was able to empty it. Hence there is no direct relationship between rainfall and overflow. The content of the interception pond was primarily dilute silage effluent and when it overflowed there were dramatic decreases in dissolved oxygen concentrations in the polluted tributary. Periodically the oxygen concentration at Site B fell below 4.0 mg l^{-1} for several hours. Decreases in dissolved oxygen of this magnitude are enough to cause an invertebrate drift greater than ten times background levels (Edwards *et al*, 1991), and are below the 5 percentile value of 5.0 mg l^{-1} recommended for the long-term survival of salmonids (Alabaster and Lloyd 1982). The combination of duration, frequency and intensity of depletion exceed the limits of the water quality criteria for the preservation of aquatic life proposed by Milne and Seager (1990). Oxygen levels at Site C which is much closer to the input are likely to have been lower still.

The input of slurry was an isolated event in which the resulting pulse of ammonia had a peak of 21 mg l^{-1} at Site B with a corresponding un-ionized component of 0.16 mg/l. A spot sample taken at Site C at this time recorded an ammoniacal nitrogen concentration of 33.7 mg l^{-1}. The corresponding un-ionized ammonia concentration for this was 0.25 mg l^{-1}. The pulse at Site B remained above 13 mg l^{-1} (0.1 mg l^{-1} un-ionized) for a period of about three hours, but these conditions do not exceed the criteria for the preservation of aquatic life proposed by Milne and Seager (1991). However, because the slurry was mixed with silage liquor in the interception pond the peak in ammonia was accompanied by a marked decrease in dissolved oxygen - a spot sample taken at the same time indicated this to be 3.6 mg l^{-1} and only 0.9 mg l^{-1} at Site C. It is known that reduction in dissolved oxygen concentration can increase the toxicity of ammonia to fish (Alabaster and Lloyd 1982) and it is therefore possible that conditions in the stream might have caused lethal or sublethal stress to salmonids had they been present at Sites B and C during the pollution episode.

The peaks of ammonia in direct association with rainfall were probably due to general farm run-off and on two occasions they appeared at Site A as well in muted form. However un-ionized ammonia concentrations were not high enough to cause adverse effects to the biology of the stream.

In addition to these intermittent episodes there was the probability of a more a chronic impact resulting from the recurrent growth of sewage fungus on the stream bed at Sites B and C due to low-level organic loading from the interception pond. These respiring growths can cause oxygen depletion within the substratum leading to impacts on fish eggs and oxygen sensitive invertebrates (Curtis 1969). In addition the growths tend to favour generalists feeders such as oligochaetes and chironomids at the expense of more specialist invertebrates. It was benthic growths of this kind which were the apparent cause of impoverished invertebrate communities and fish populations at Pontfaen Brook (Section 7.1).

Perhaps the major finding of the study was the identification of the interception pond below Capel Farm as the major pollution source affecting the polluted tributary. This illustrates the effectiveness of detailed work when the source of pollution problems is not obvious. The effects of the discharge became most evident from June onwards following silage cutting and corresponding to an increase in rainfall (Figure 7.25). The study, thus illustrates the importance of rainfall in facilitating the transport of polluting materials to watercourses. Effective remedial work is clearly required if the stream is to recover biologically. Should effective action be taken there could be considerable delay in recolonization due to a lack of a substantial unpolluted section upstream which could act as a source of invertebrate colonists. Turner has shown that this is the most important source of colonists and recovery was reasonably rapid following cessation of input both in his studies and at Pontfaen Brook (Section 9). Continued biological monitoring on the Rhydw will seek to study any recovery process.

7.3 Nant Pibwr: yard run-off and parlour/dairy washings

7.3.1 Introduction

Intensive studies carried out under the current contract have been undertaken at polluting farms identified by more extensive surveys. A study at Pontfaen Brook, a tributary of the Gwaun in West Wales, was initiated in June 1990 following an extensive survey during the development of the rapid appraisal technique (Section 7.1). A second study took place in April - August 1991 in the Rhydw catchment (near Carmarthen), after a severely polluted tributary was identified during testing of the rapid appraisal methods (Section 7.2). At Pontfaen Brook, the pollution was the result of a chronic leakage of silage effluent, whilst the tributary of the Rhydw was subject to daily inputs of parlour washings and to intermittent inputs of silage effluent whenever a reception pond overflowed following rainfall.

A third study was initiated to compliment the previous work, in an attempt to examine the effects of yard run-off on a water course. A suitable site was identified in the Pibwr catchment (south of Carmarthen) during the test of the rapid appraisal methods, and work began on 15 October 1991.

7.3.2 Study area

The Nant Pibwr rises at an altitude of 110 m, some 8 km east of Carmarthen in West Wales. It flows parallel to the Afon Tywi before discharging to the Tywi estuary 2 km south of Carmarthen (Figures 7.27).

The Pibwr was one of the catchments selected for Survey during a large scale evaluation of the rapid appraisal system developed to pin-point polluting farms using biological techniques (Section 6.1). On 13 February 1991, the pollution control officer responsible for sampling the Pibwr catchment observed discharges from Penbontbren farm (SN450189), which were probably responsible for a reduction in biological quality downstream of the premises: the BMWP score below the farm was 75, compared with 112 above, and sewage fungus cover (above stones) was 60% downstream compared with no growth upstream.

The farm was visited by both the NRA Agricultural Liaison Officer and Graham Rutt on 15 May 1991. It was found to be in a poor state of management and three possible polluting discharges were discovered (Figure 7.27). Immediately upstream of the farm was a small stream which could be contaminated with slurry washed off the road during rainfall (Discharge 1). further downstream was an input of parlour/diary washings and yard run-off contaminated with diesel oil (Discharge 2). The most downstream discharge was yard washings which leaked from a heavily ponded yard between an earth-banked slurry lagoon and solid manure store (Discharge 3).

7.3.3 Methods

Chemical monitoring began on 15 October 1991. Three pHOx 100 DPM continuous monitoring units were installed at sites A, B and C (Figure 7.27). Site A was a few metres upstream of Discharge 1, site B was 60 m downstream of Discharge 3 whilst site C was some 300 m below site B. The meters were equipped to measure dissolved oxygen, temperature, pH, conductivity and ammonium (a close measure of total ammonia at moderate temperature and circumneutral pH). Data was recorded using Technolog NEWLOG data loggers set to record every fifteen minutes and is presented as Appendix H11. A Casella rain gauge linked to a Technolog TINYLOG data recorder was situated adjacent to site C. This however,

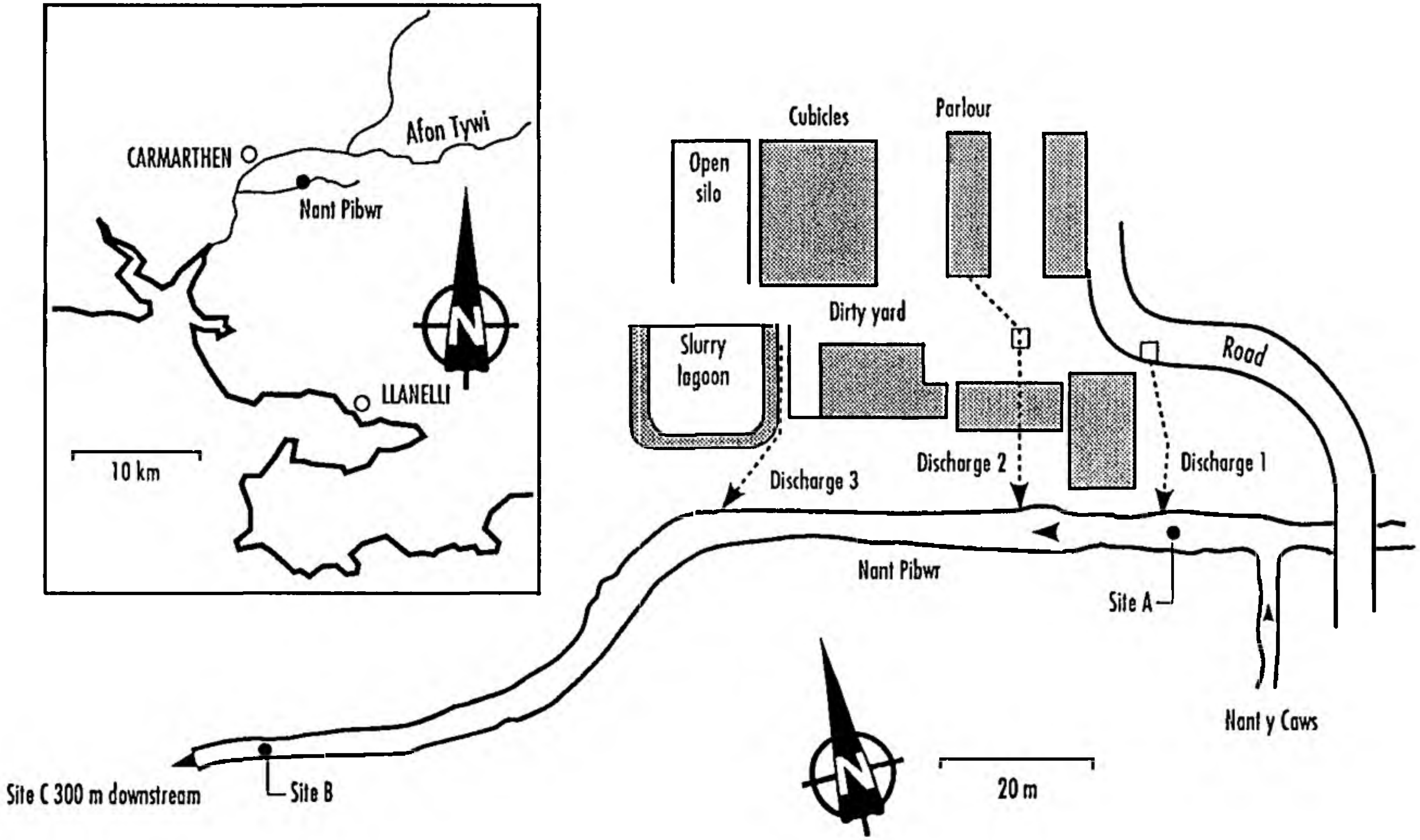


Figure7.27 Diagram of the Nant Pibwr study area showing farm waste discharges and study sites. Inset shows the location of the Nant Pibwr in Southwest Wales

proved unreliable and rainfall data was obtained from an NRA rain gauge at Towy Castle (SN406142) some 6.5 km to the south-west. All the instruments were visited at weekly intervals for calibration and data retrieval. Monitoring ceased on 18 December 1991.

Weekly spot samples were taken a five different points: sites A, B and C, Discharge 1 and Discharge 2 (Figures 7.27). Discharge 3 was not monitored because it had ceased following remedial work carried out prior to the study. The samples were analysed at the NRA laboratory at LLanelli for a range of sanitary determinands such as BOD, dissolved oxygen, inorganic nutrients and Dissolved Organic Carbon (DOC). Data are presented as Appendix H12.

Flow estimates were made below site A by measuring cross-sectional area and estimating current velocity.

Detailed visual estimations of the precentage cover of benthic growth (sewage fungus, algae, etc.) were made weekly at sites A, B and C, over ten contiguous 2 m sections of stream. In addition, five large stones at each site were examined for growth both on top and bottom surfaces. The character and extent of benthic growth immediately downstream of Discharge 2 was also noted. Each week a sample of growth from each site was collected for microscopic examination so as to identify and quantify the constituent organisms.

Three minute kick samples of the macroinvertebrate community were taken from riffles at sites A, B and C at the beginning of the study on 15 October 1991, on 12 November 1991 and at the end of the study on 18 December 1991. Samples were fixed at the laboratory in 4% formaldehyde solution and the invertebrates present were later identified to family level enabling the calculation of BMWP (Biological Monitoring Working Party) Scores and ASPT (Average Score Per Taxon).

The RIVPACS (River Invertebrate Prediction and Classification System) program was used to predict presence/absence of invertebrate families and BMWP Score and ASPT from environmental data (Cox *et al.* 1991). The predictor variables used were distance from source, altitude, slope, width, depth, discharge category, chloride, hardness and substratum composition, and an autumn

(September - January) only prediction option was employed. Only one prediction was performed as the three sites were close together and very homogeneous, having essentially the same environmental characteristics. Predicted families and biotic indices were compared with those actually observed. This technique was not used in two previous intensive studies because the steams were too small, falling outside of the environmental range of RIVPACS program.

7.3.4 Results

Water Chemistry

Data from spot samples and continuous monitors indicated that pH was generally in the range 7.0 -8.0 at sites A, B and C. There were occasional depressions in pH fown to 6.5 following prolonged rainfall but these levels are far above those known to adversely affect aquatic life.

Dissolved Oxygen levels in spot samples from the three steam sites A, B and C showed a minimum recorded value of 9.8 mg l^{-1}, at site A (Figure 7.28). Discharge 2 showed much lower dissolved oxygen concentrations rarely exceeding 1.0 mg l^{-1}. Continuously monitored data showed consistently high dissolved oxygen concentrations at the three stream sites until the last two weeks of the study, when declines in oxygen concentration to below 6.0 mg l^{-1} were recorded at site A (Figure 7.29). Oxygen electrodes at sites B and C were not working correctly at this time so no comparative data was available.

Spot samples and continuous monitoring data indicated a baseline level of conductivity of 200 - 220 $uScm^{-1}$ at all three stream sites. On one occasion conductivity decreased to around 120 $uScm^{-1}$ corresponding to heavy rainfall and high flow. Peaks in conductivity were more common: on several occasions, all three stream sites showed conductivity peaks of 400 - 800 $uScm^{-1}$, and on one occasion conductivity reached 1700 $uScm^{-1}$ at all three sites.

Spot sample data indicated that ammoniacal nitrogen concentration rarely exceeded 0.1 mgl^{-1} N at any of the three stream sites (Figure 7.30). The highest value recorded was 0.17 mg l^{-1} N at site C, with a corresponding

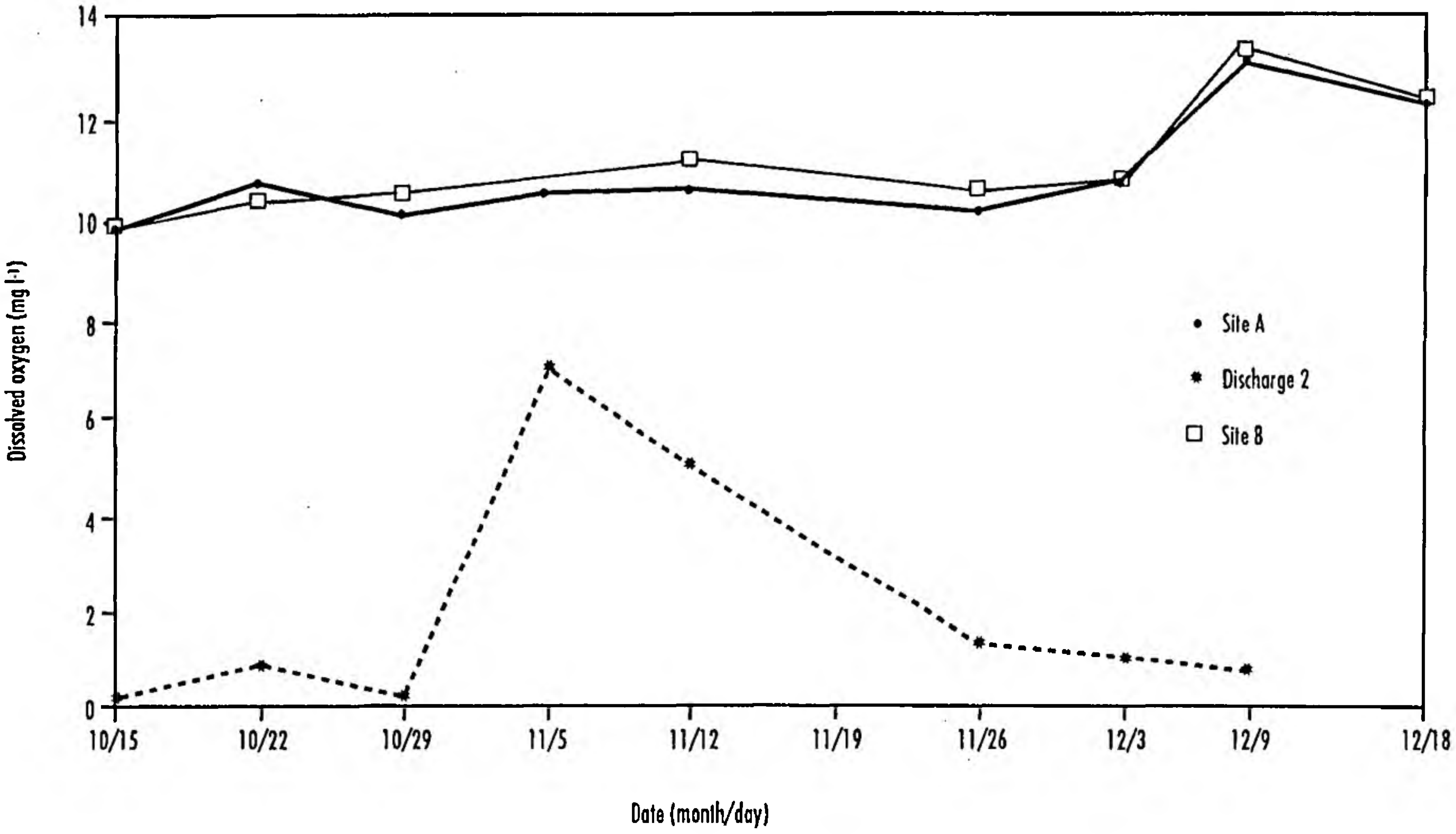


Figure7.28 Dissolved oxygen levels in the Nant Pibwr from spot sampling between October 15 and December 18, 1991

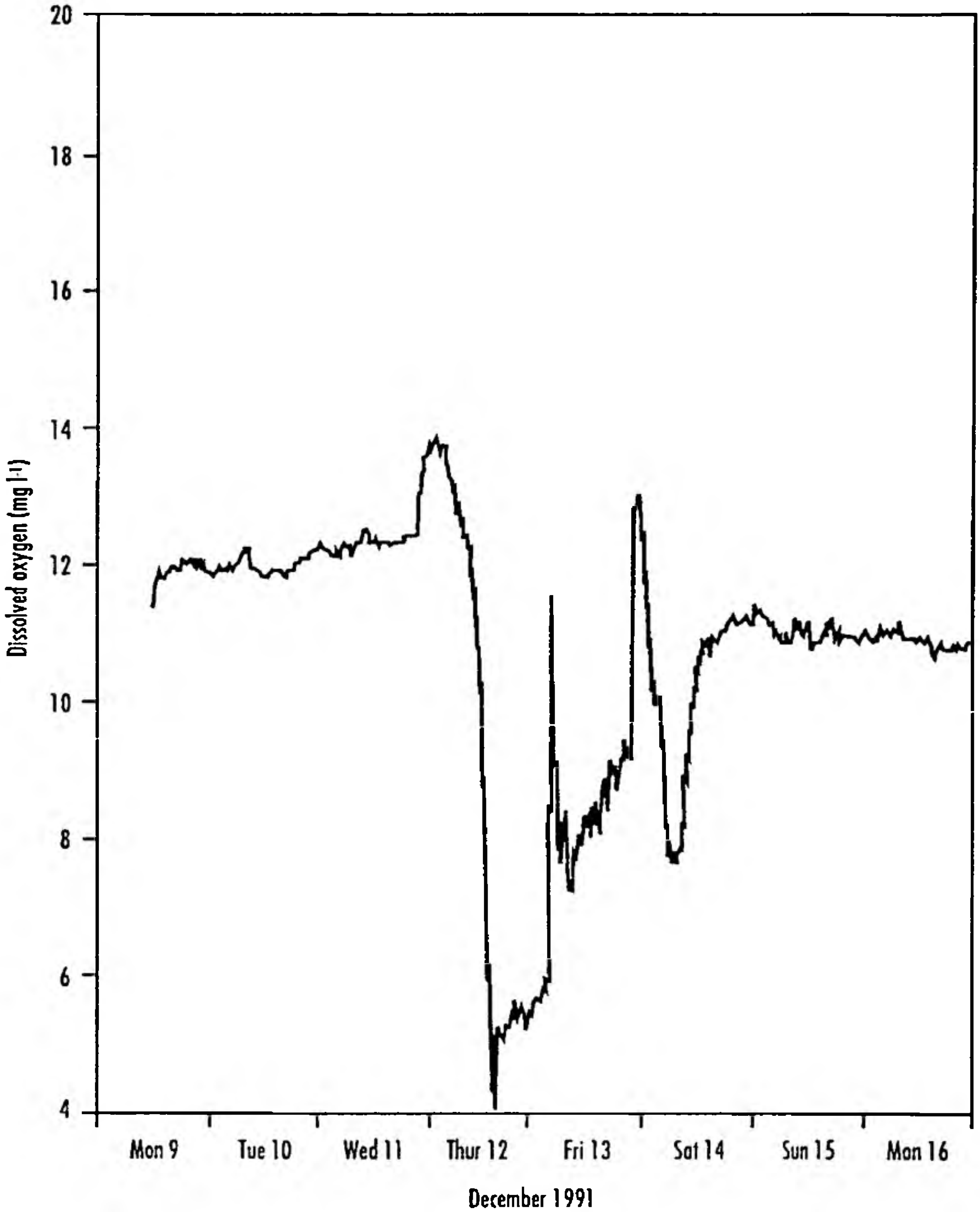


Figure 7.29 Dissolved oxygen levels in the Nant Pibwr from continuous monitoring between December 10 and December 17, 1991

un-ionized ammonia concentration of 1.0µg l^{-1}. Concentrations in Discharge 2 were much higher and generally in the range 10 - 20 mg l^{-1} N, with a maximum of 43 mg l^{-1} N of which the un-ionized component was 0.08 mg l^{-1} (Figure 7.30). Continuously monitored levels of ammonium were below detection limits for all three streams sites for the majority of the study. An exception occurred at sites B and C on 15, 16 and 17 December 1991, when ammonium peaks were recorded corresponding to peaks in conductivity and rainfall (Figure 7.31). The largest peak occurred on 15 December with a maximum of 0.75 mg l^{-1} ammonium (0.004 mg l^{-1} un-ionizes ammonia) and lasted for 10 hours (Figure 7.31). There was no ammonium recorded at site A at this time.

Spot sampling revealed that BOD at the three stream sites rarely exceeded 2.0 mg l^{-1} with BOD generally in the range of 1 - 2 mg l^{-1} (Figure 7.32). The BOD of Discharge 1 was generally slightly higher than the stream sites but never exceeded 3.0 mg l^{-1}. The BOD of Discharge 2 was considerably higher than the other sites, being generally in the range of 200 - 400 mg l^{-1} with a maximum in excess of 559 mg l^{-1} (Figure 7.32). Despite the high BOD of this discharge the input was relatively small compared to the size of the stream. The input was estimated to have a flow of 0.062 litres sec^{-1}, as compared with the 118 - 860 litres sec^{-1} recorded for the Pibwr (Figure 7.33), yielding a dilution factor of 1990 - 13,900. Taking the worst case scenario of the lowest dilution factor (1900) and the highest recorded BOD (559 mg l^{-1}), the BOD contribution of Discharge 2 to the Pibwr would be only 0.3 mg l^{-1}.

Benthic Growth

At site A, upstream of the farm discharges, there was no evidence of heterotrophic benthic growth at any time during the study. At sites B and C there was never any significant growth above stones but small patches of growth were often present below large stones. These growths were variable in composition but the major constituents were the colonial bacterium *Sphaerotilus nations*, iron bacteria and the sessile protozoan *Charchesium*.

The only significant growths of 'sewage fungus' were found in the mixing zone associated with Discharge 2. Here there were thick growths of *Sphaerotilus* or Zoogloea in a tapering strip extending at most 0.5 - 1.0 m across the stream

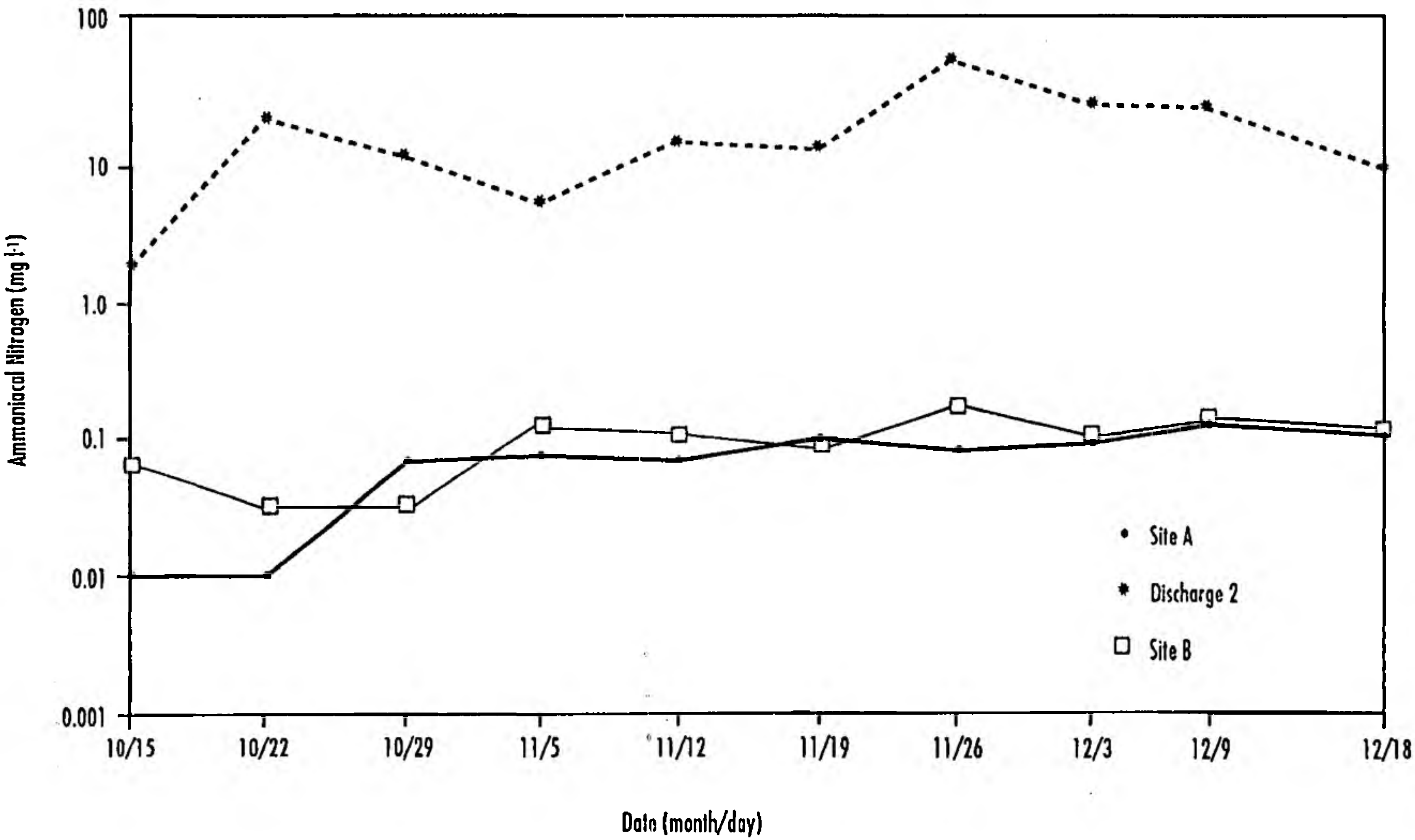


Figure7.30 Ammoniacal Nitrogen levels in the Nant Pibwr from spot sampling between October 15 and December 18, 1991

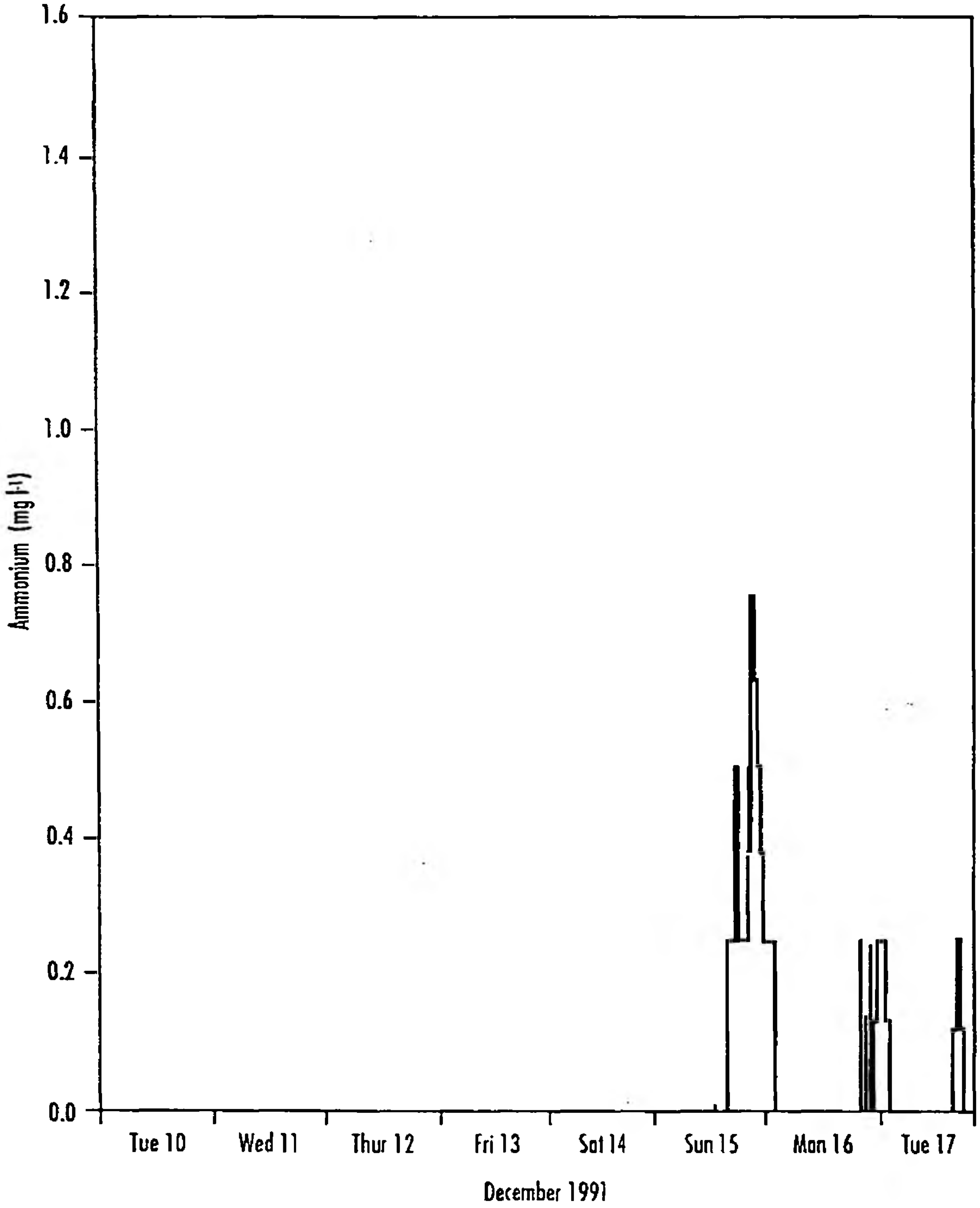


Figure 7.31 Ammonium ion concentrations in the Nant Pibwr at site B from continuous monitoring between December 10 and December 17, 1991

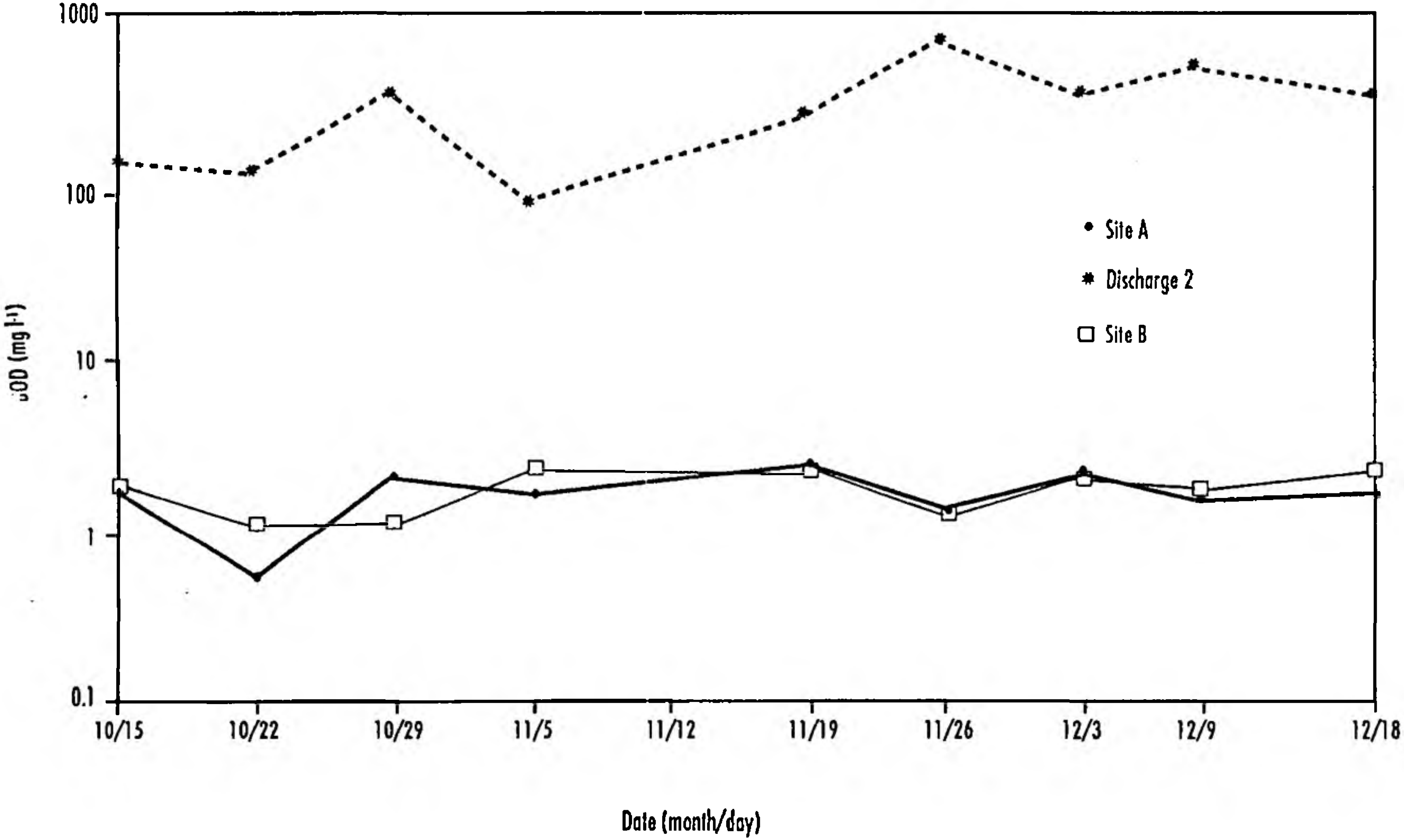


Figure7.32 Biological Oxygen Demand (BOD) in the Nant Pibwr from spot sampling between October 15 and December 18, 1991

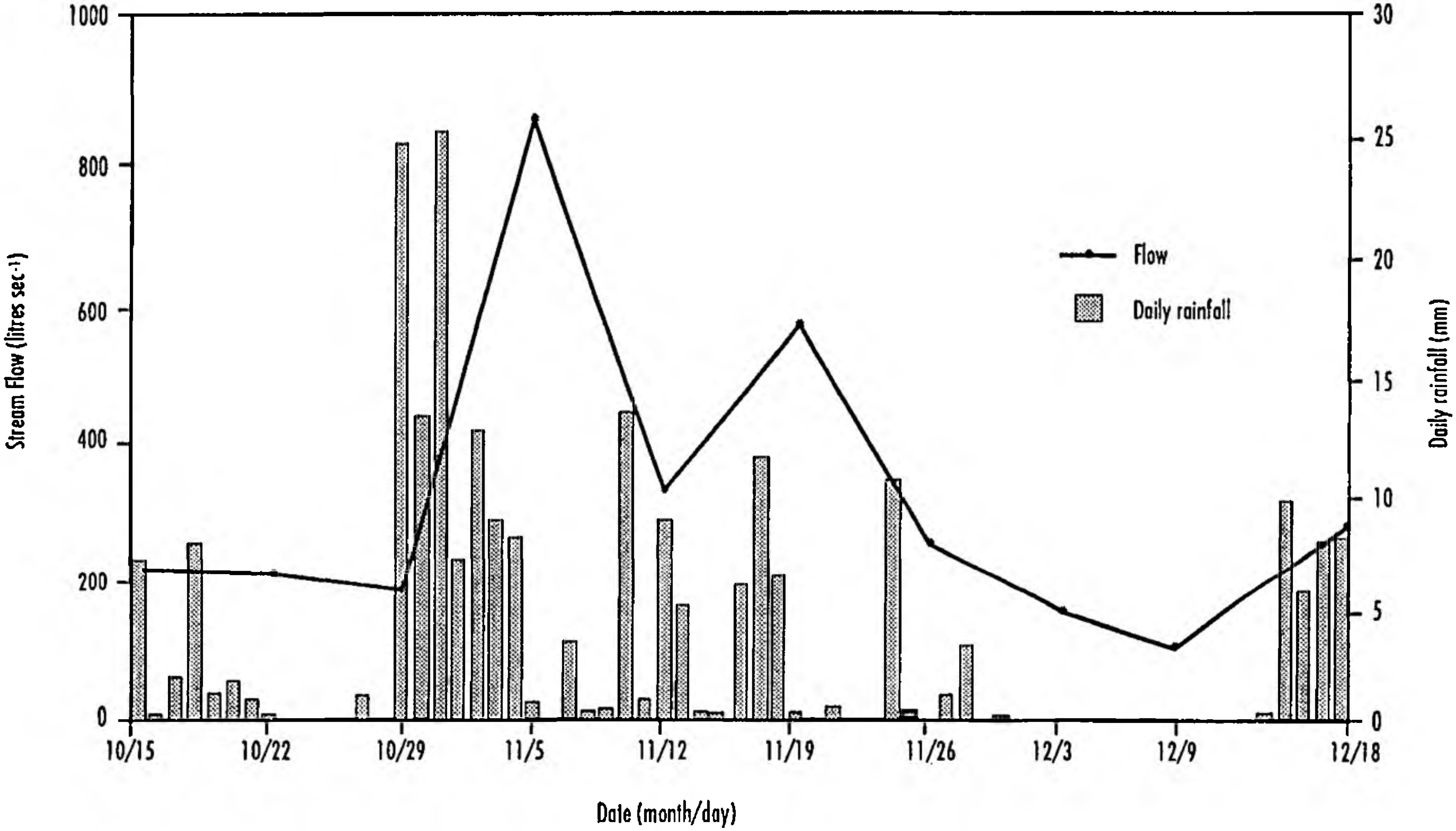


Figure7.33 Stream Flow of the Pibwr below site A and daily rainfall between October 15 and December 18, 1991

(stream width 3 -4 m) and 3 - 12 m downstream. Growths of fungus were generally present below stones for a few metres beyond the limit of visible above stones.

Macroinvertebrate Populations

Examination of the macroinvertebrate data derived from kick sampling, including comparison of BMWP scores, ASPT values and abundances of the different families, revealed no clear pattern of differences between the three stream sites, A, B and C (Table 7.8).

The observed taxa agreed quite closely with those predicted by RIVPACS. Of twenty families with greater than 40% predicted probability of occurrence, only six families were absent from all sites. Of these, three (*Ephemeridae, Dytiscidae* and *Odontoceridae*) are principally margin dwelling taxa and are less likely to occur in samples taken exclusively from riffles. Of the remaining three families (*Planariidae, Goeridae* and *Chloroperlidae*), the absence of *Chloroperlidae* is perhaps the most interesting as these were found to be present at three sites in the Pibwr catchment in February 1991, including site A. Predicted BMWP score and ASPT were 158 (confidence limits, 117 - 198) and 6.3 (confidence limits 5.7 - 6.9). Thus observed BMWP scores (Table 7.8) were rather low by comparison, but generally lay within the confidence limits about the RIVPACS prediction. ASPT values were also relatively low and tended to lie below the lower confidence limit. Although differences in sampling methodology may account for some of the difference between observed and predicted biotic indices, there is perhaps some suggestion of low level perturbation of the fauna at all three sites.

7.3.5 Discussion

Data from both discrete and continuous monitoring suggested that during the course of the study, there was no marked reduction in the water quality of the Pibwr as a result of discharge of parlour/dairy washings and yard water from Penbontbren Farm. There was correspondingly little evidence of a biological impact attributable to the farm, except for localised growth of 'sewage fungus' in the mixing zone below the most polluting discharge.

Table 7.8 Invertebrate community analysis of sites on the Nant Pibwr

Taxa	Site A			Site B			Site C		
	Oct	Nov	Dec	Oct	Nov	Dec	Oct	Nov	Dec
Heptageniidae	2	2	1	2	2	2	1	2	1
Leptophlebiidae		1	1					1	
Taeniopterygidae						2			1
Leuctridae	1	2	1	1	1	1	1	1	1
Perlodidae	1	2	1	2	1	2	1	1	1
Sericostomatidae		1	1				1	1	
Caenidae		1							
Nemouridae	1	1	1	1	1	1	1		
Rhyacophilidae	1	1	1	1	1	1	1	1	1
Polycentropodidae	1	1	1	1			1	1	
Limnephilidae				1	1		1	1	
Ancylidae	1	1	1	2	1	2	2	1	1
Gammaridae	2	1	1	2	1	2	3	2	1
Gyrinidae	1	1	1	2	1	2	1	1	1
Hydrophilidae	2	1	1	2	1		1	1	1
Elminthidae	3	1	1	3	1	1	1	1	1
Hydropsychidae	1	1	1	1			1	1	1
Tipulidae	2	1	2	2	1	2	1	1	1
Simuliidae	2	2	1	2	1	1	1	1	1
Baetidae	2	2	3	2	2	3	2	2	1
Hydrobidae	2	1	2	3	1	2	4	4	3
Lymnaeidae				1	1				
Planorbidae		1					1	1	
Sphaeridae	1	1			1	2	2		1
Glossiphonidae	1	1		1	1		2	1	1
Erpobdellidae	1			1		1		1	1
Asellidae	1								1
Chironomidae		2	1	1	1	1	1	1	1
Oligochaeta	2	2	2	3	2	3	3	2	1
BMWP SCORE	113	129	123	129	107	102	122	132	118
ASPT	5.4	5.6	6.2	5.6	5.4	5.7	5.5	5.7	5.4

These results contrast with the severe impacts associated with farm effluent found in intensive studies carried out earlier in the project. A chronic discharge of silage effluent to Pontfaen Brook resulted in damaging growths of sewage fungus and severe disruption of the stream fauna (Section 7.1), whilst more intermittent discharges to a tributary of the Nant Rhydw produced a similar reduction in biological quality (Section 7.2). The contrast may be partially explained by the much higher dilution capacity of the Pibwr, where measured flows during the study period were in the range 118 - 860 litres sec^{-1}, in contrast to 20 - 240 litres sec^{-1} for Pontfaen Brook and 10 - 70 litres sec^{-1} for the tributary of the Rhydw. Thus, although Discharge 2 had a consistently high BOD (200 - 400 mg l^{-1}), the BOD downstream did not reach the level of 2 - 3 mg l^{-1} required to promote significant growth of sewage fungus beyond the mixing zone (Quinn and McFarlane, 1988; Mainstone *et al*, 1991). Similarly, although the ammoniacal nitrogen concentration was relatively high (10 - 20 mg l^{-1}) in this discharge, the maximum recorded level of ammonium below the farm was only 0.75 mg l^{-1} (0.004 mg l^{-1} unionised ammonia). The value of unionised ammonia is six times lower than the EQS value of 0.025 mg l^{-1} recommended by WRc for the protection of freshwater salmonids (Seager *et al* 1988). This figure is based on the value quoted by Alabaster and Lloyd (1982), below which no adverse effects on salmonid fisheries are thought to occur.

The lack of observable impact can also be partly explained by improvements in the management of the farm between the initial visit to the site in February 1991 and the start of the intensive study. It is likely that the biological impact observed in February 1991 (Section 7.3.2.) was caused by a leakage of yard water which formerly by-passed the lagoon but has now been sealed following a visit by the NRA Agricultural Liaison Officer.

Despite a lack of impact attributable to the farm, there is some evidence to suggest that the Pibwr in the vicinity of Penbontbren is affected by organic pollution from unknown sources further upstream. The reductions in dissolved oxygen observed in mid December at site A were quite marked and the occurrence of intermittent episodes of this type may account for the disparity between observed and predicted biotic indices and the absence of certain riffle dwelling taxa which had a reasonable probability of occurrence. The peaks in conductivity recorded at all sites appeared to be rainfall-generated and may be the result of salt-contaminated run-off from the adjacent A40.

7.4 Slurry transport studies

7.4.1 Introduction

The development of farm waste management plans requires a knowledge of the suitability of fields for disposal of slurry. Earlier work by WRc and the soil survey had produced slurry acceptance maps for the Eastern Cleddau catchment (Schofield 1988).

Pollution risk was determined by slope, soil type and proximity to watercourse. A study of slurry transport processes, initiated in an attempt to validate these criteria was completed as part of the present contract.

7.4.2 Methods

Monitoring equipment was installed on three fields of differing soil type and slope characteristics in the Clarbeston stream catchment (SN 051202):

Field 1 (Cegin soil type, 1 degree slope) - 4 sets of two 40-cm and two 80-cm access tubes for collecting soil solution samples arranged at intervals upslope (Sites 1-4), together with 4 corresponding overland flow samplers. Two field drains discharging into the stream were also monitored.

Field 2 (Denbigh soil type, 4 degree slope) - as for Field 1 but no field drains (Sites 6-9 numbered upslope).

Field 3 (Denbigh soil type, 9 degree slope) - 4 overland flow samplers arranged at intervals downslope (Sites 10-13).

Samples were collected at weekly intervals between 19 January and 19 April 1990 to ensure collection of data before slurry spreading up to the first cut of silage. Access tubes were evacuated with a hand-held pump such that soil water was forced in over a 2-3 hour period. Volumes of overland flow were recorded on each sampling occasion. Samples were analysed for pH, BOD, ammoniacal nitrogen, Total Oxidised Nitrogen (TON), nitrite, orthophosphate and Dissolved Organic Carbon (DOC). Data are presented as Appendix H13. Hourly rainfall was recorded between 14 February and 19 April (Appendix H14).

7.4.3 Results

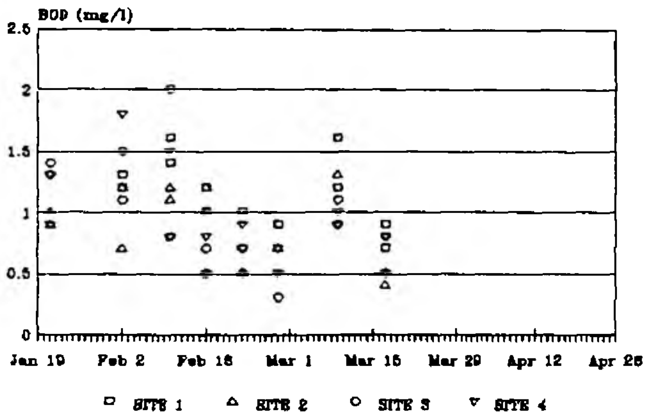
Due to delays in installation, problems with flooding and lack of sufficient rainfall, data for overland flow was not obtained on all sampling occasions but the soil solution data was comprehensive. Results presented are restricted to BOD and ammoniacal nitrogen levels which are the most important determinands from the point of view of impact upon receiving water courses.

1. Field 1 - Due to water-logging, slurry was not applied to this field during the sampling period and therefore BOD was not measured after 15 March to relieve pressure on the laboratory. BOD never exceeded 2.5 mg l^{-1} in the soil solution samplers and ammoniacal nitrogen rarely exceeded 0.1 mg l^{-1} (Figures 7.34 and 7.35). Values of both determinands in the land drains were similarly low. Very little data were obtained from the overland flow equipment due to water logging. Values obtained were in the ranges 3.8-9.4 mg l^{-1} for BOD and 0.44-1.24 mg l^{-1} for NH_3-N.

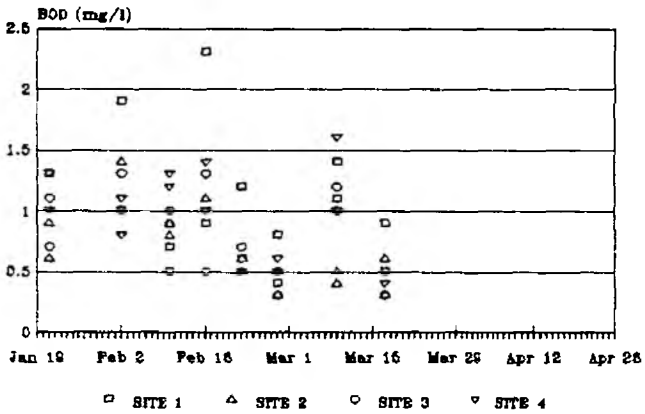
2. Field 2 - Slurry was applied to the top and bottom parts of this field on 25 February covering two sets of samplers only (6 & 9), as spreading was curtailed by the resumption of heavy rain (see Figure 7.36). The overland flow sample from Site 6 taken the day after spreading had a BOD in excess of 205 mg l^{-1}, and ammoniacal nitrogen was recorded as 30.6 mg l^{-1} (Figures 7.37 and 7.38). A week later after slurry had been spread on the rest of the field the maximum BOD recorded for overland flow was only 25.0 mg l^{-1} and the maximum NH_3-N value was 26.4 mg l^{-1}.

Figure 7.34 Slurry transport studies at Clarbeston – BOD in field 1 (Cegin soil type, 1 degree slope) 19 January – 26 April 1990

(a) 40 cm depth access tubes



(b) 80 cm depth access tubes



(c) field drain

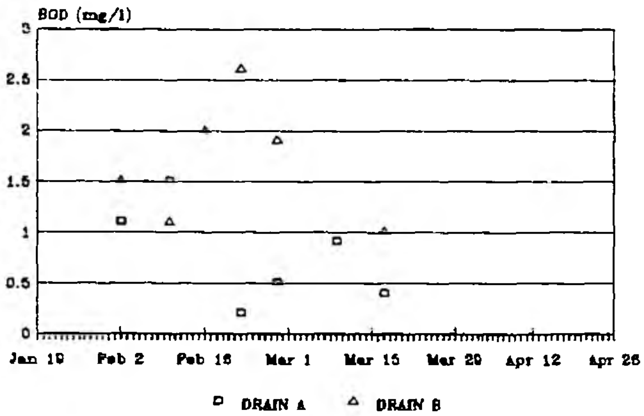
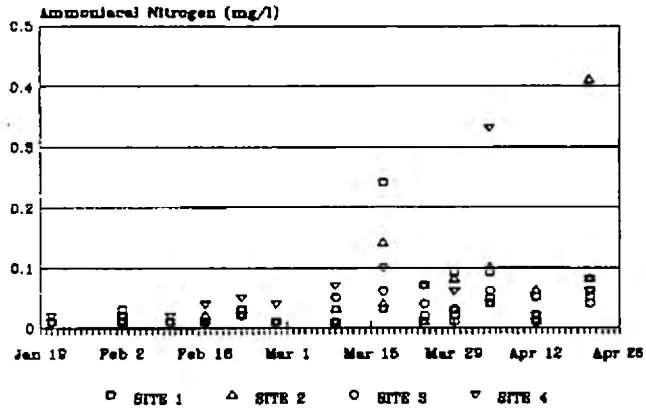
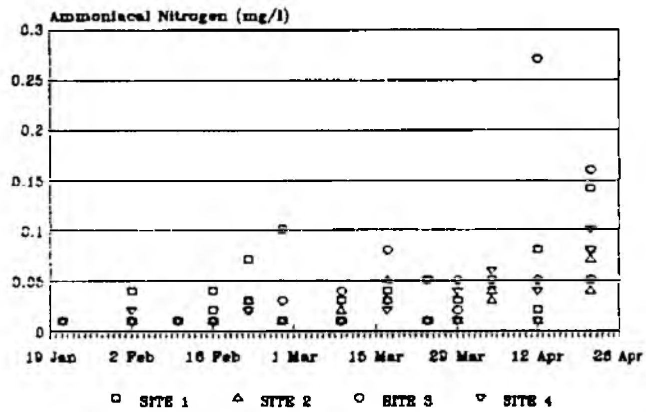


Figure 7.35 Slurry transport studies at Clarbeston – ammoniacal nitrogen in field 1 (Cegin soil type, 1 degree slope) 19 January – 26 April 1990

(a) 40cm depth access tubes



(b) 80cm depth access tubes



(c) field drains

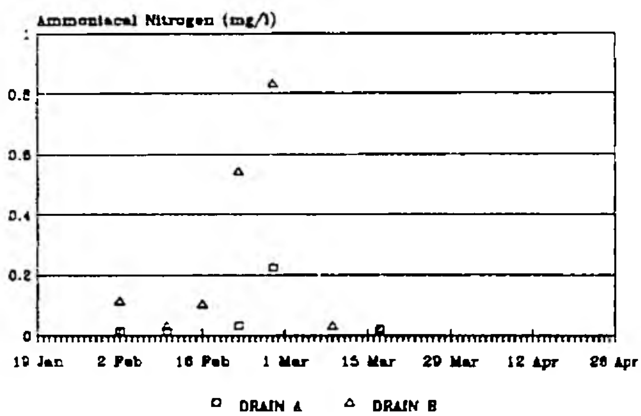


Figure 7.36 Daily rainfall at Clarbeston
14 February – 24 April 1990

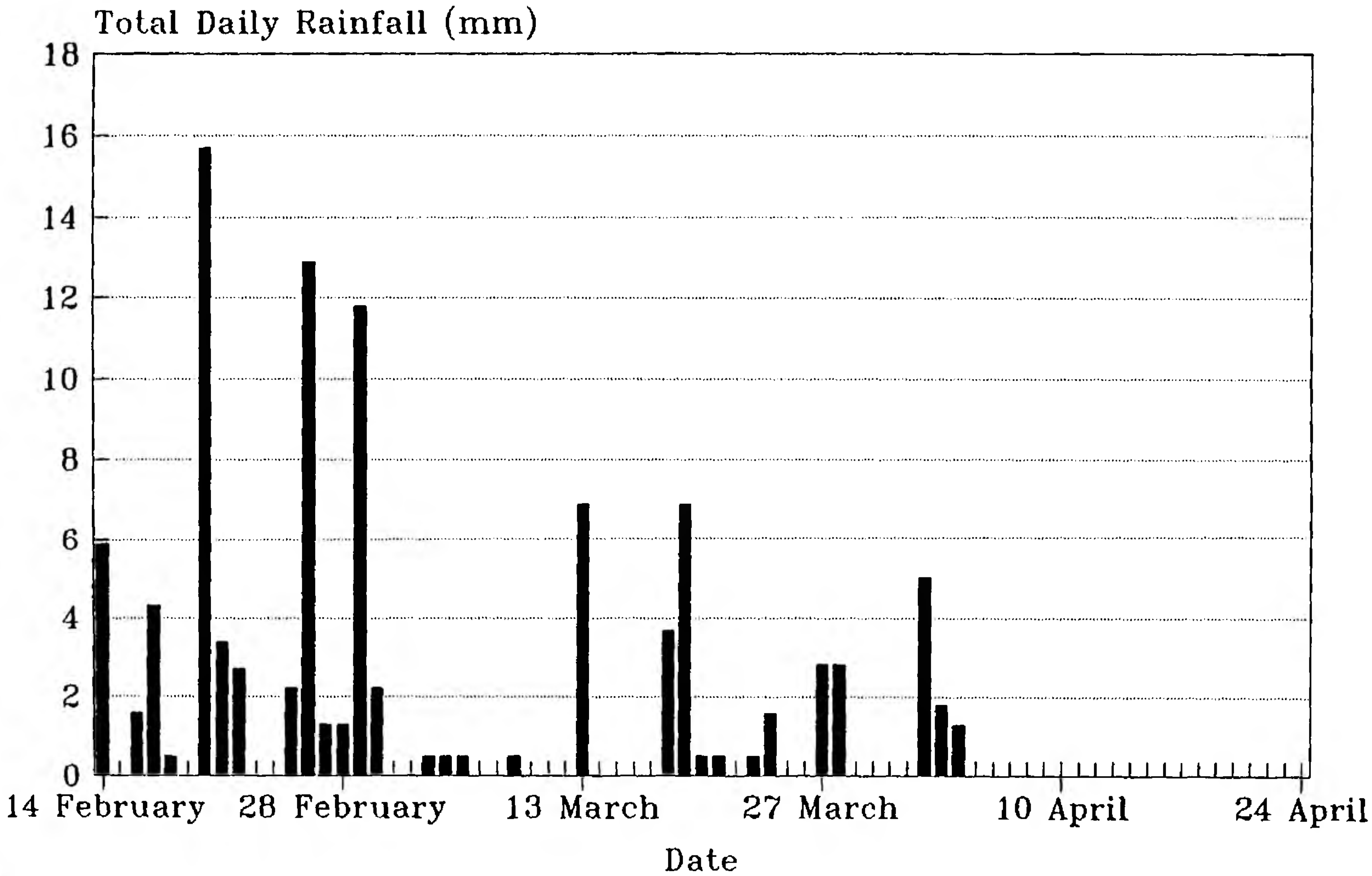
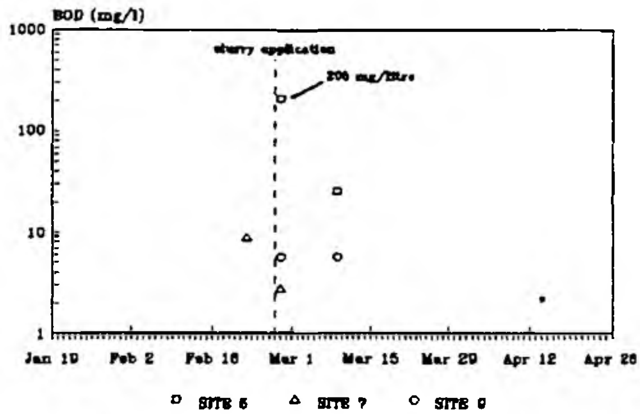
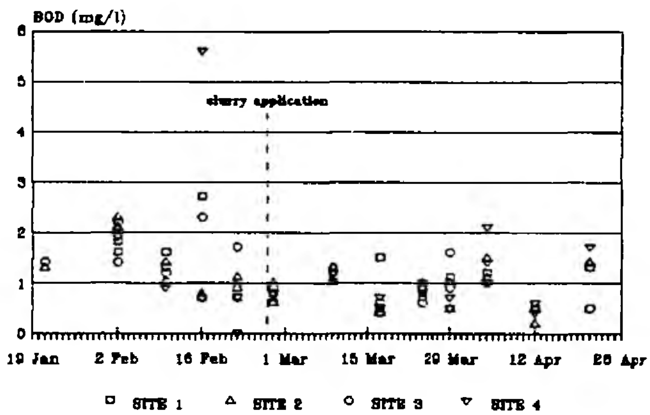


Figure 7.37 Slurry transport studies at Clarbeston – BOD in field 2 (Denbigh soil type, 4 degree slope) 19 January – 26 April 1990

(a) overland flow samplers



(b) 40cm depth access tubes



(c) 80cm depth access tubes

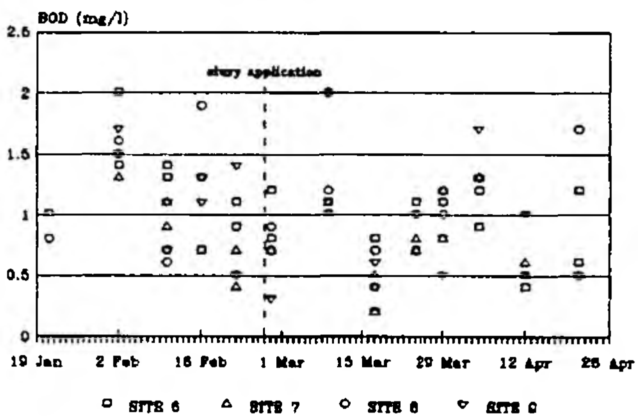
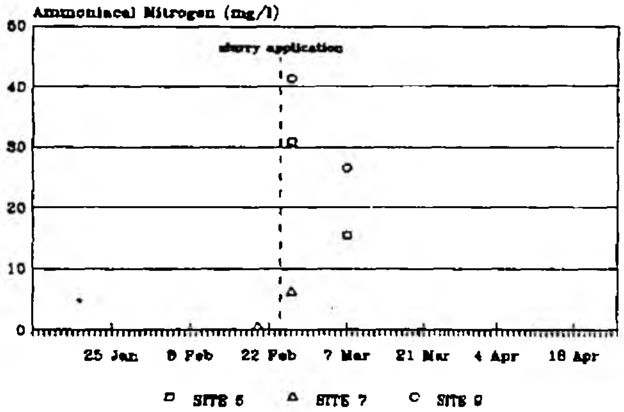
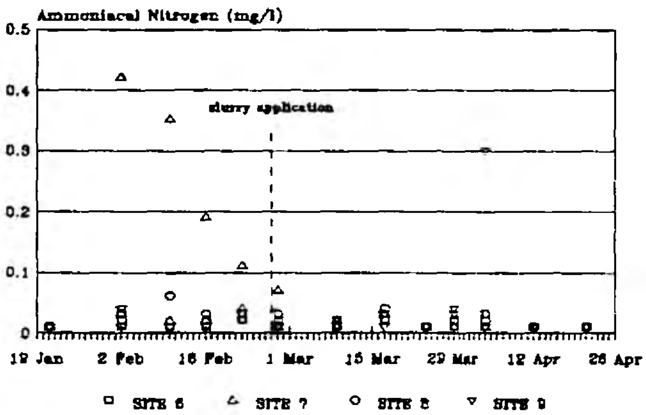


Figure 7.38 Slurry transport studies at Clarbeston – ammoniacal nitrogen in field 2 (Denbigh soil type, 4 degree slope) 19 January – 26 April 1990

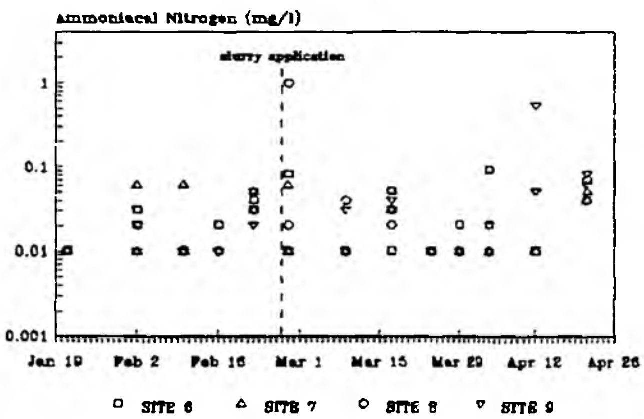
(a) overland flow samplers



(b) 40cm depth access tubes



(c) 80cm depth access tubes



Throughout the sampling period BOD rarely exceeded 2.0 mg l^{-1} in the sub-surface samples (max 3.0 mg l^{-1}) and NH_3-N generally remained below 0.1 mg l^{-1} (Figures 7.37 and 7.38).

3. Field 3 - Slurry was spread on this field on 24 February. When sampled two days later after heavy rain (Figure 7.36), all but the bottom-most overland flow sampler contained water with a BOD in excess of 250 mg l^{-1} (Figure 7.39). The four sites had levels of ammoniacal nitrogen between 16.5 and 113 mg l^{-1} (Figure 7.39). A week later the BOD of the run-off had fallen to 3.2-29.0 mg l^{-1} with ammoniacal nitrogen in the range 8.2-49.6 mg l^{-1}. Prior to spreading BOD never exceeded 10.0 mg l^{-1} and ammoniacal nitrogen remained below 1.0 mg l^{-1}.

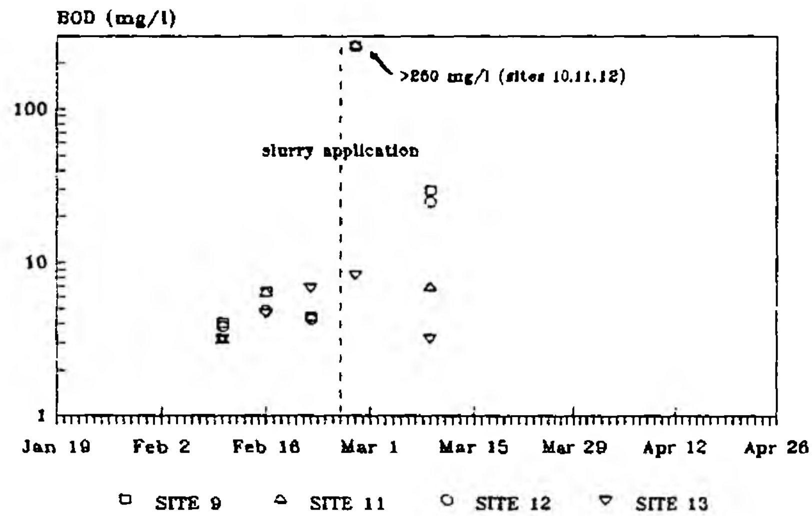
7.4.4 Discussion

The results would appear to indicate that neither BOD, nor ammoniacal nitrogen displayed significant increase in soil water at 40 and 80 cm depth on the well-drained Denbigh soil (Field 2) whereas overland flow exhibited high BOD and high ammoniacal nitrogen when slurry spreading was followed by a period of intense rainfall. Overland flow on the steeper Denbigh field (Field 3) also showed significant slurry contamination with BOD in excess of 250 mg l^{-1} and ammoniacal nitrogen up to 113 mg l^{-1}. These values compare with 11 401 mg l^{-1} BOD and 1032 mg l^{-1} ammoniacal N in the applied slurry.

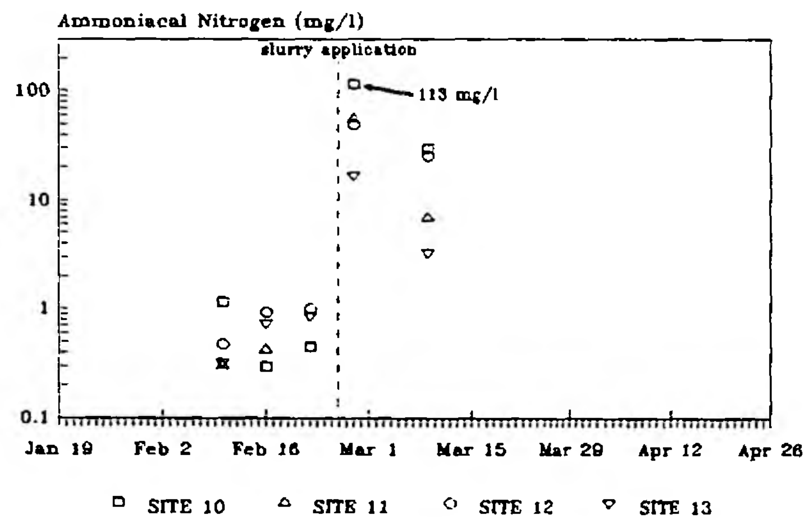
However, drainage through the soil profile cannot be ruled out as a pathway for transfer of polluting material to the adjacent stream. This is because the hydraulic conductivities estimated for the Denbigh soil type are very high: 200, 840 and 450 cm/day for topsoil, immediate subsoil and lower subsoil respectively. Thus with a weekly sampling regime it might be possible to miss a pulse of polluting material descending rapidly through the profile following heavy rain. This low sampling frequency has restricted the usefulness of the data collected in validating the slurry acceptance mapping technique.

Figure 7.39 Slurry transport studies at Clarbeston – overland flow quality in field 3 (Denbigh soil type, 9 degree slope), 19 January – 26 April 1990.

(a) BOD



(b) ammoniacal nitrogen



8. SALMONID RECOLONIZATION STUDIES

8.1 Introduction

Following acute pollution events resulting in large scale fish mortalities, a series of electrofishing surveys were carried out on two streams in South West Wales. Previous results from Deepford Brook have been reported in Schofield (1988) and Schofield and Bascombe (1990). The objectives of the work are to assess the extent of damage to fish stocks and to monitor the rate of natural recovery. Such information will be of value to the NRA in relation to the restoration strategies adopted following fish mortality.

The two study streams are the Afon Fenni, a tributary of the River Taf, and the Deepford Brook, within a sub-catchment of the Easter Cleddau. A slurry spill occurred on the Afon Fenni on 21 August 1989. It originated from Trehoose Farm (SN245249) and entered the stream in its upper reaches less than 1 km from source (Figure 8.1). Based on the recovery of corpses and historical data from previous surveys, it was estimated that a total of 9040 trout fry (*Salmo trutta* L.) and 5310 trout parr had been killed. 120 salmon parr (*Salmo salar* L.) were also thought to have been lost. A previous survey (NRA 1990) had found no salmon fry in the stream and so these were assumed absent at the time of the kill. As the polluting farm was located close to the stream source, and the total kill zone extended almost to the confluence with the River Taf (a distance of 10 km), the catchment's fish stock suffered a severe loss.

The Deepford Brook was subject to an input of cattle slurry on 28 April 1988. The spill originated from a collapsed slurry lagoon at North Lamborough Farm (SN165236) and entered the brook some 2 km from its source, via a drainage ditch (Figure 8.2). It was estimated that 2700 trout fry and 2780 trout parr were killed along with 480 salmon fry and 67 salmon parr. Stocks in the two main tributaries, the Churchill stream and the Cotland Brook, were unaffected by the pollution.

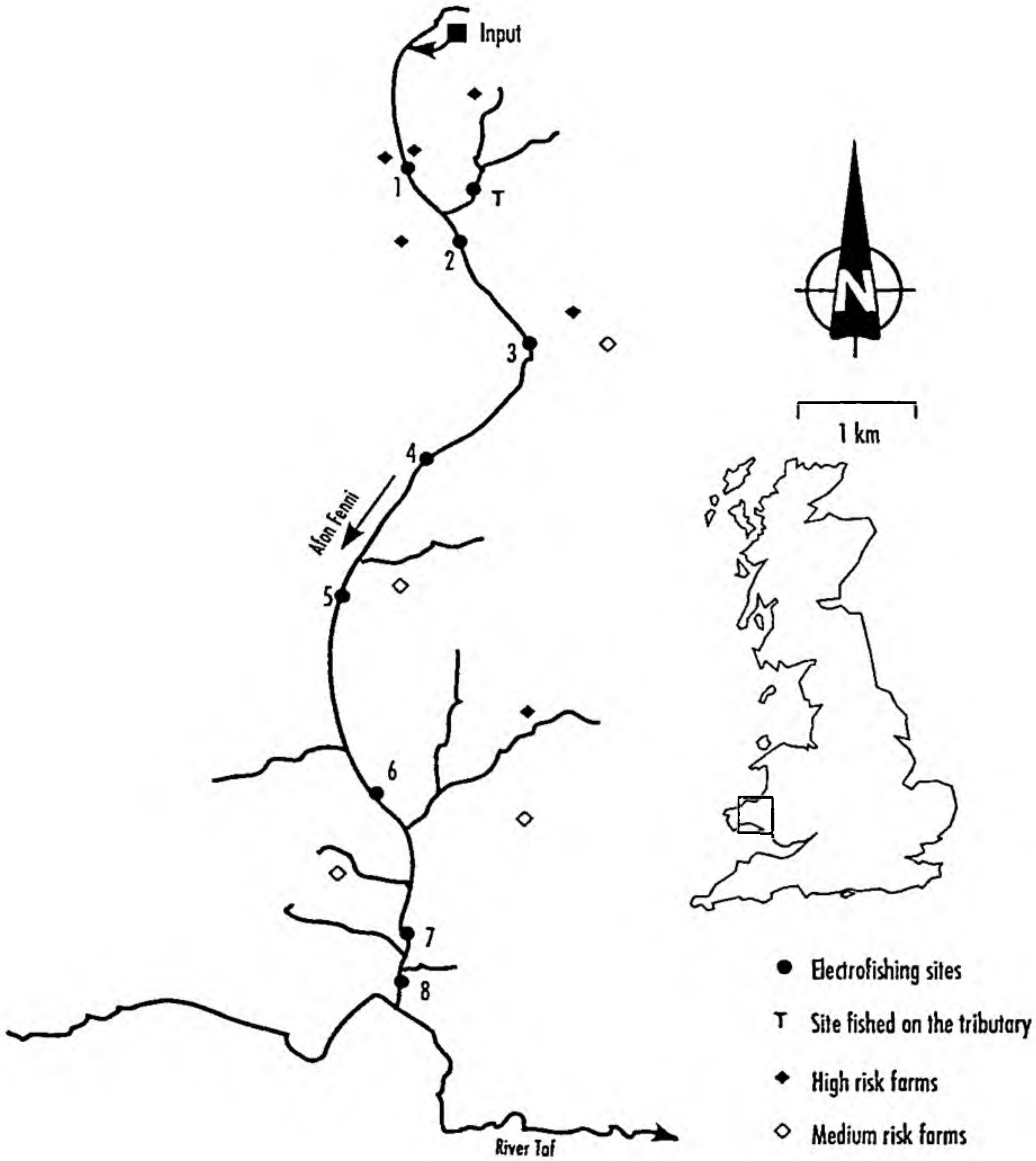


Figure 8.1 Location of electrofishing sites and farms at risk of polluting, on the Afon Fenni

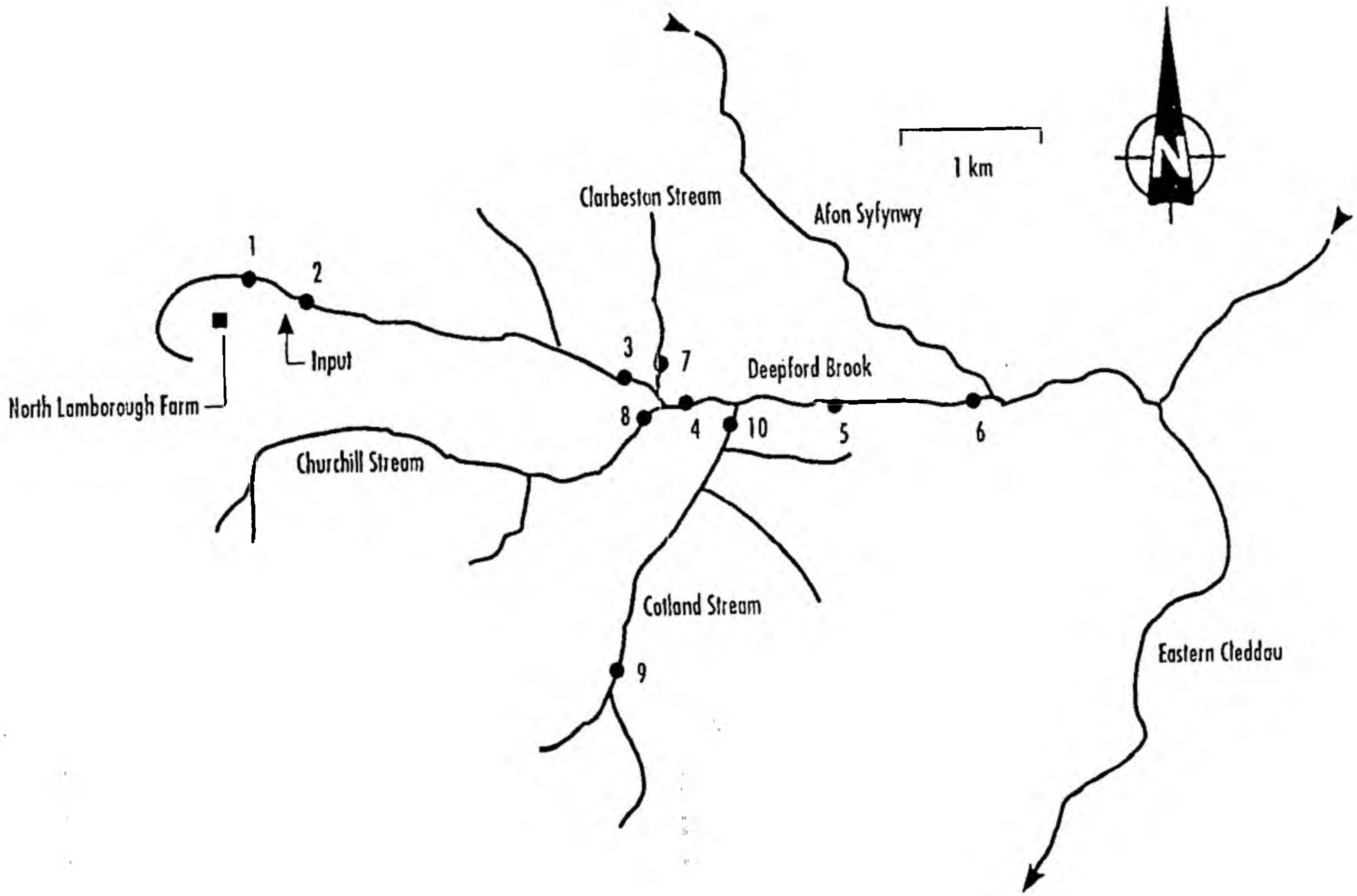


Figure 8.2 Electrofishing sites on the Deepford Brook and its tributaries

Restocking was withheld on the Deepford Brook and its tributaries and on the Afon Fenni, in order that salmonid reclonisation of the two streams could be studied under natural conditions.

8.2 Methods

Eight, easily accessible sites on the Afon Fenni, each of approximately 50 m length, were selected for electrofishing. They constituted 2.5% of the total stream length from its source to its confluence with the river Taf (Figure 8.1, Table 8.1). Following the mortality, an initial fishing of the stream took place on 25/26 August 1989. A second was carried out on 17/18 October 1990 and a third between 16 and 20 September 1991. In the last two surveys, a site of 85 m^2 in area was semi-quantitatively fished (i.e. a single electrofishing run with no stop nets) on a tributary near the top of the stream, in order to assess its potential as a source of colonists (T in Figure 8.1).

Ten electrofishing sites were selected on the Deepford Brook and its tributaries. These were again each of approximately 50 m in length and also constituted 2.5% of the total length of the streams from their sources to the confluence with the Syfynwy (Figure 8.2, Table 8.2). The sites were initially fished on 5/6 May 1988. Successive surveys were undertaken on 19/20 September 1988, 22/23 August 1989, between 26 September and 4 October 1990 and between the 23 and 27 September 1991.

The first post-pollution survey on the Afon Fenni was carried out by NRA staff in rapid response to the pollution event, and was semi-quantitative. The results were adjusted so as to be comparable with quantitative data by dividing them by estimates of catch efficiency for each site, obtained from subsequent surveys. The remaining surveys on the Afon Fenni, and all those on the Deepford Brook, were quantitative; sites were enclosed by 1 cm mesh stop nets and two or three electrofishing runs performed to ensure catch depletion. Fork lengths, to the nearest 1 mm, of all salmonids caught were recorded and fry (0+) and parr (>0+) of both species were distinguished on the basis of length frequency data. Population estimates for the different age classes were calculated using the methods of Zippin (1956), or for two electrofishing runs, Seber and Le Cran (1967). All raw data are presented in Appendix I.

Table 8.1 Electrofishing sites on the Afon Fenni

Site	OSGR	Location	Width (m)	Gradient (m km^{-1})
1	SN241239	Afon Fenni 100m downstream of Liechclawdd	1.9	20.0
2	SN245230	Afon Fenni at Maenoch	2.2	20.0
3	SN251220	Afon Fenni at at Nantyreglwys Mill	2.9	13.3
4	SN241211	Afon Fenni At Rhydycaeshyd	3.5	10.0
5	SN234199	Afon Fenni at Pistyll Gwyn	3.6	8.0
6	SN234184	Afon Fenni at Llwyncrwn	4.1	6.6
7	SN238169	Afon Fewnni at Pont y Fenni	4.2	5.0
8	SN236163	Afon Fenni just above the confluence with the Taf	3.8	5.0

Historical data for sites 4 and 7 on the Afon Fenni, and site 3 on the Deepford Brook, were obtained from earlier work undertaken as part of NRA Welsh Region's salmonid monitoring programme (Welsh Water Authority 1988; NRA 1990).

HABSCORE, developed by NRA Welsh region, is a multiple regression model designed to predict salmonid densities under given habitat conditions, assuming pristine water conditions (Milner *et al* 1985; Milner and Wyatt 1991). Habitat assessment was undertaken during electrofishing in 1991. This involved making a detailed assessment of habitat characteristics at each site on the two streams and combining this with general information on the catchments obtained from 1:50000 Ordnance Survey maps. HABSCORE predicted salmonid densities for each site and compared them to those observed, indicating when the observed densities were significantly lower. The same sites were fished each year so the predictions could be compared with the observed densities from previous years, including those obtained from the regional salmonid monitoring programme (Welsh Water Authority 1988; NRA 1990).

8.3 Results

8.3.1 Afon Fenni

Following the pollution in 1989, trout fry were absent from all sites downstream of the input (Figure 8.3). In 1990 they were present at six of the eight sites. HABSCORE revealed that the densities at sites 2, 4, 5 and 6 were not significantly lower than would be expected for those sites under clean water conditions. In 1991 trout fry densities remained within predicted values at sites 4, 5 and 6; however, the density at site 2 fell from 14.8 100 m^{-2} in 1990 to 0.7 100 m^{-2} in 1991, and was significantly lower than predicted ($p < 0.01$). In 1990, nine trout fry were caught at the site on the tributary, but fry were absent in 1991.

Historical data for the Afon Fenni were limited but were available for site 4 in 1986 and 1987 (Figure 8.4), and for site 7 in 1987 and in the summer of 1989, immediately before the pollution (Figure 8.5). At both sites, trout fry densities fluctuated widely in the years before the fish kill: from 4.1 100 m^{-2} in 1986 to 77.8 100 m^{-2} in 1987 at site 4, and from 11.2 100 m^{-2} in 1987 to 0 100 m^{-2} in 1989 at site 7. Fry densities are generally highly variable between years, and there is a large die-off through the first year of life that makes observed densities very dependent upon the timing of the survey. Nevertheless, fry densities appeared to have returned to pre-pollution levels at site 4 by 1990. At site 7, however, densities remained depressed and significantly lower than the HABSCORE prediction ($p < 0.01$) in both 1990 and 1991.

Immediately following the pollution, trout parr were present at sites 6, 7 and 8, with densities increasing with distance downstream (Figure 8.6). HABSCORE revealed that only the density at site 8 (9 100 m^{-2}) was not significantly lower than predicted. In 1990, trout parr were present at all sites apart from

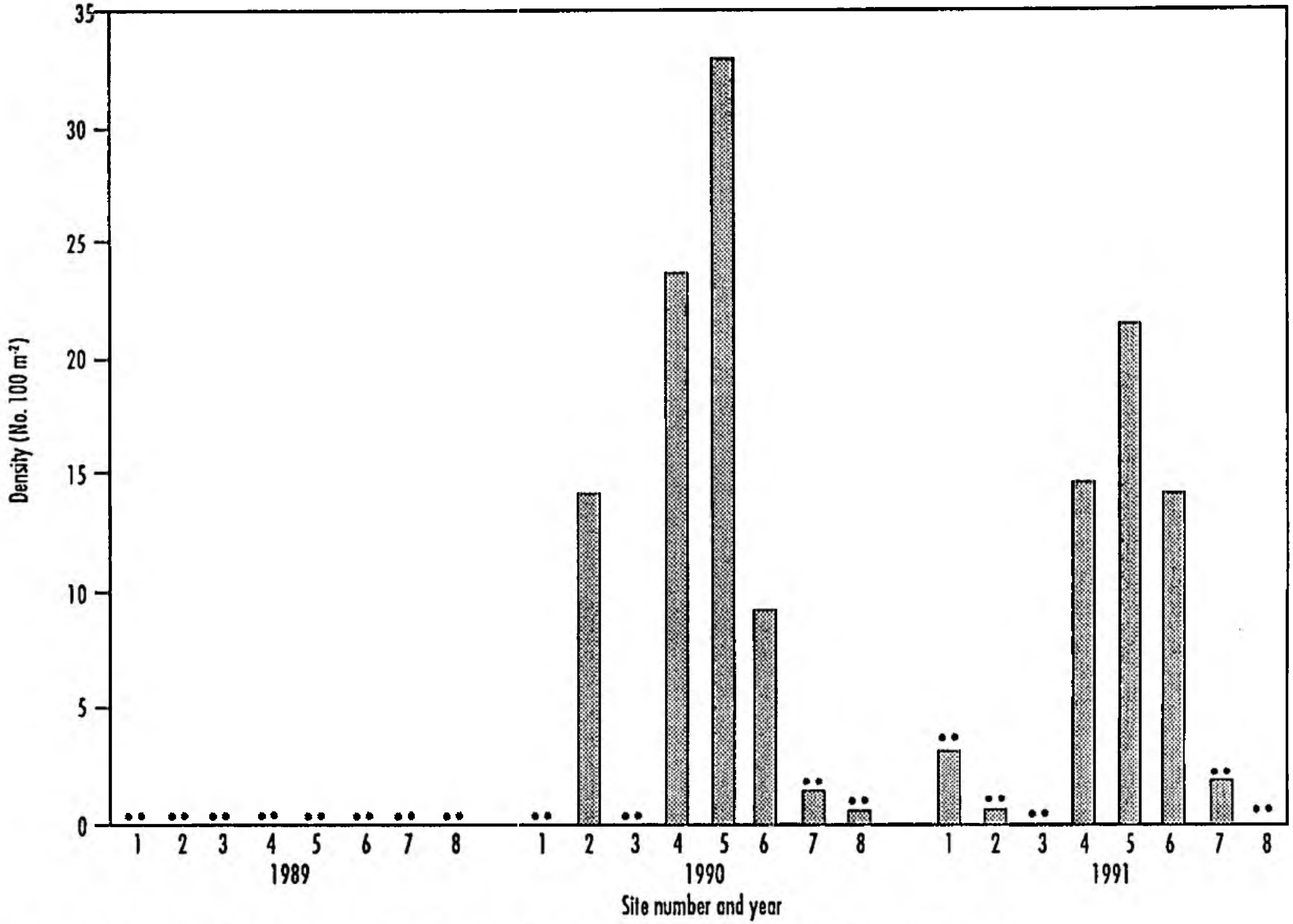


Figure 8.3 Density of trout fry (0+) at sites on the Afon Fenni. **: Significantly lower than predicted by HABSCORE (P< 0.01)

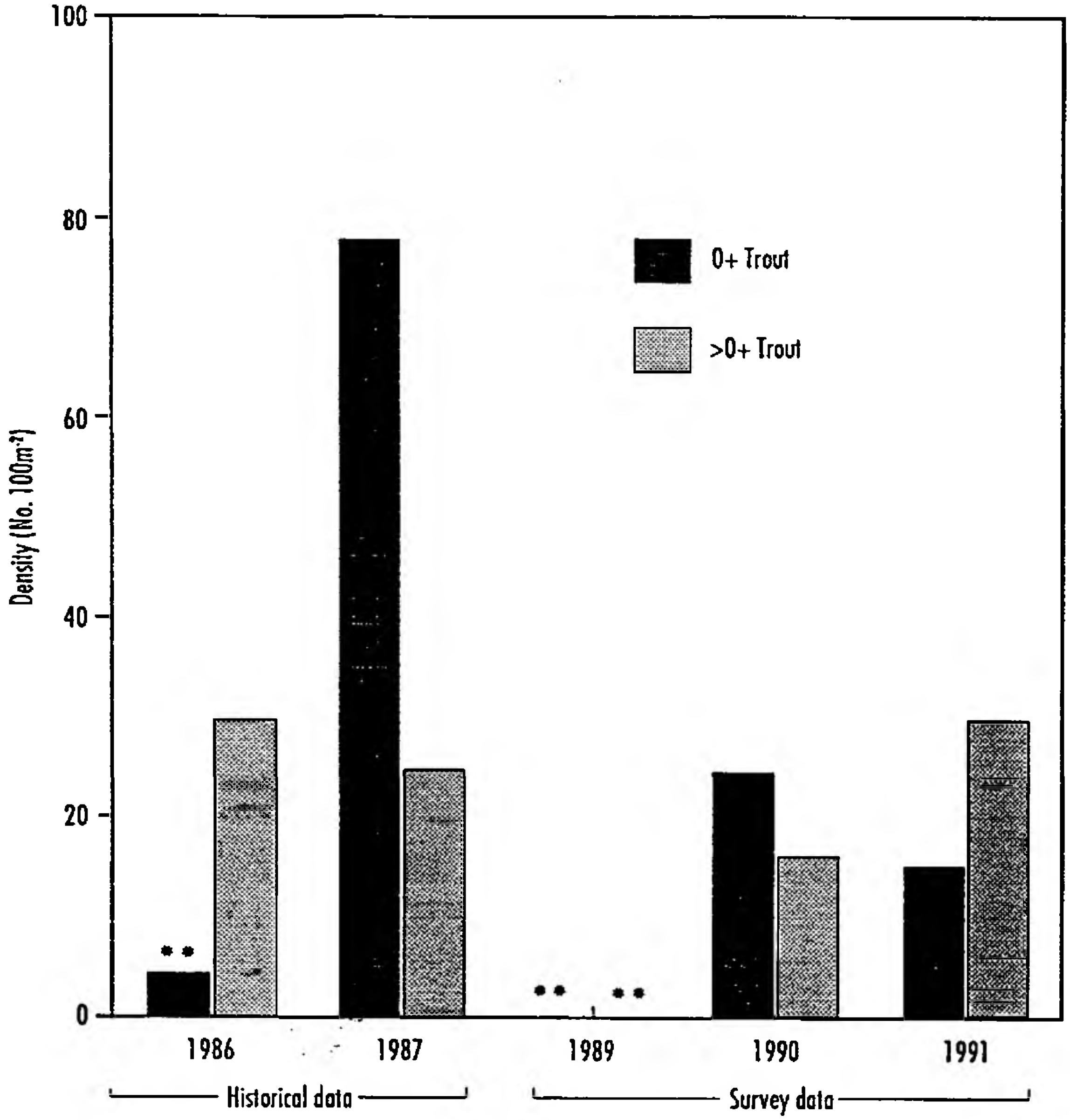


Figure 8.4 **Comparison of historical data with survey data, for trout, at site 4 on the Afon Fenni.**
****: Significantly lower than predicted by HABSCORE (P< 0.01)**

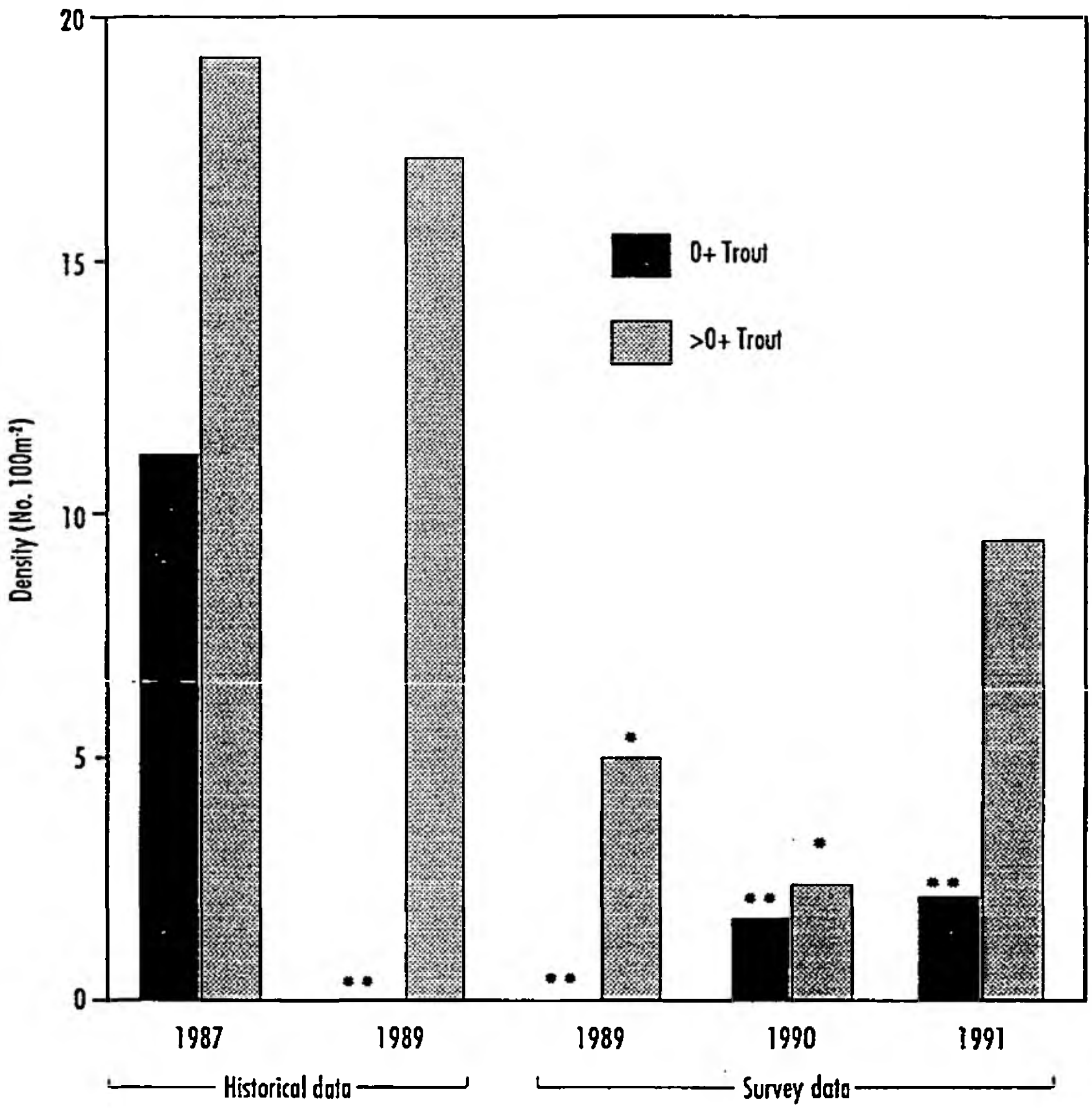


Figure 8.5 **Comparison of historical data with survey data, for trout at site 7 on the Afon Fenni. Asterisks indicate the significance of differences from HABCSORE predictions** ** $p<0.01$; * $p<0.05$**

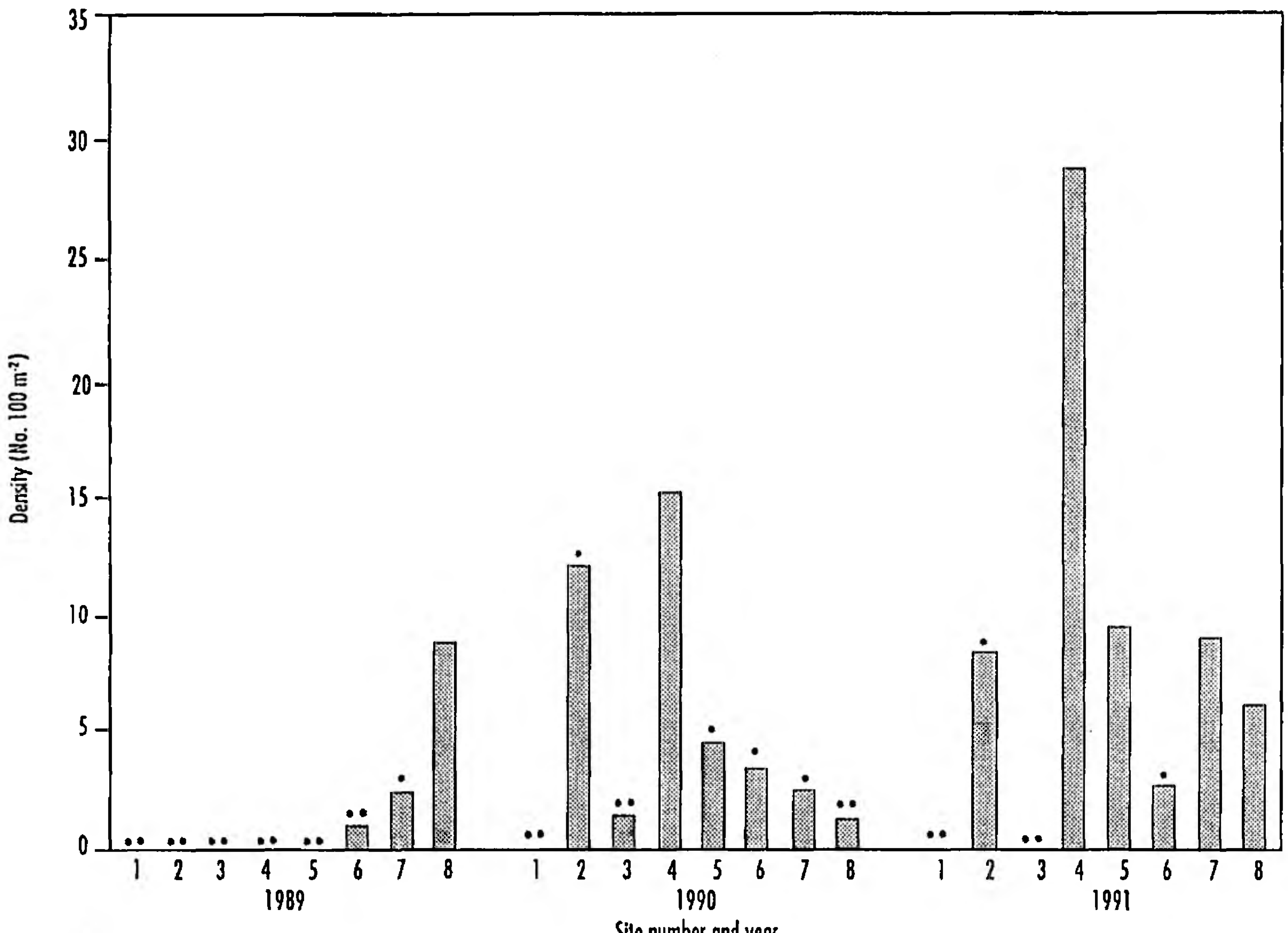


Figure 8.6 Density of trout parr (>0+) at sites on the Afon Fenni.
Asterisks indicate the significance of differences from HABSCORE predictions ** $p<0.01$; * $p<0.05$

site 1, but only the density at site 4 (15.5 100 m^{-2}) was not significantly lower than predicted. In 1991, densities had increased at sites 4, 5, 7 and 8 such that none of these were significantly lower than predicted. Two trout parr were caught at the site on the tributary in 1990 but none in 1991.

Table 8.2 Electrofishing sites on the Deepford Brook and its tributaries

Site	OSGR	Location	Width (m)	Gradient (m km^{-1})
1	SM027206	Deepford Brook 50 m above effluent input	1.0	10.0
2	SM029205	Deepford Brook 50 m below effluent input	1.4	6.7
3	SM049200	Deepford Brook above road bridge	2.4	4.4
4	SM043198	Deepford Brook between Cotland Brook and Churchill Stream	2.5	4.4
5	SM060198	Deepford Brook at Drim Farm, below confluence of Cotland Brook	3.2	4.4
6	SM071199	Deepford Brook 50 m upstream of Syfynwy confluence	3.6	3.3
7*	SM050200	Clarbeston Stream at confluence with Deepford Brook	-	-
8	SM050198	Churchill Stream at Lower Lamborough	2.4	6.7
9	SM048184	Cotland Brook at Duckspool Farm	1.8	4.4
10	SM054197	Cotland Brook 200 m upstream of Deepford Brook	2.8	6.7

Historical data for site 4 (Figure 8.4) showed trout parr density back to pre-pollution levels by 1990. Historical data for site 7 (Figure 8.5) showed a slower recovery with only the 1991 density comparable with pre-pollution densities.

With respect to salmon, fry were absent from all sites downstream of the input following the pollution in 1989 (Table 8.3, Figure 8.6). In 1990 they were present at sites 5, 6, 7 and 8, whilst in 1991 they were present at sites 7 and 8. At no time, even when fish were absent, did HABSCORE indicate that the observed densities were significantly lower than expected.

Table 8.3 Densities of salmon fry and part (No. 100 m^{-2}) at sites on the Afon Fenni

	1989		1990		1991	
SITE	0+	>0+	0+	>0+	0+	>0+
1	0	0	0	0	0	0
2	0	0	0	0	0	0
3	0	0	0	0	0	0
4	0	0	0	0	0	0
5	0	0	1.0	0	0	0
6	0	0	1.1	0	0	0.5
7	0	0	19.8	0.8	0.9	3.7
8	0	0	4.6	0.63	3.5	3.6

In 1989, salmon parr were absent from all sites, whilst in 1990 they were present at sites 7 and 8 and in 1991 at sites 6, 7 and 8. HABSCORE revealed that only the absence of parr from site 6 was significantly lower than predicted ($p<0.05$), in 1989 and 1990.

There were no historical data for salmon at site 4. Historical data for site 7 showed salmon fry to be absent in 1989, prior to the pollution, with salmon parr at a density of 3.3 100 m^{-2}. Fry and parr were present in both 1990 (20 and 1 100 m^{-2} respectively) and 1991 (4 and 1 100 m^{-2} respectively) at densities that were not significantly lower than HABSCORE predicted.

8.3.2 Deepford Brook

Following the pollution in May 1988, trout fry (0+) were absent from all sites below the pollution input (Figure 8.7). HABSCORE revealed that the absence of fry from these sites, and the small population that was present at site 1

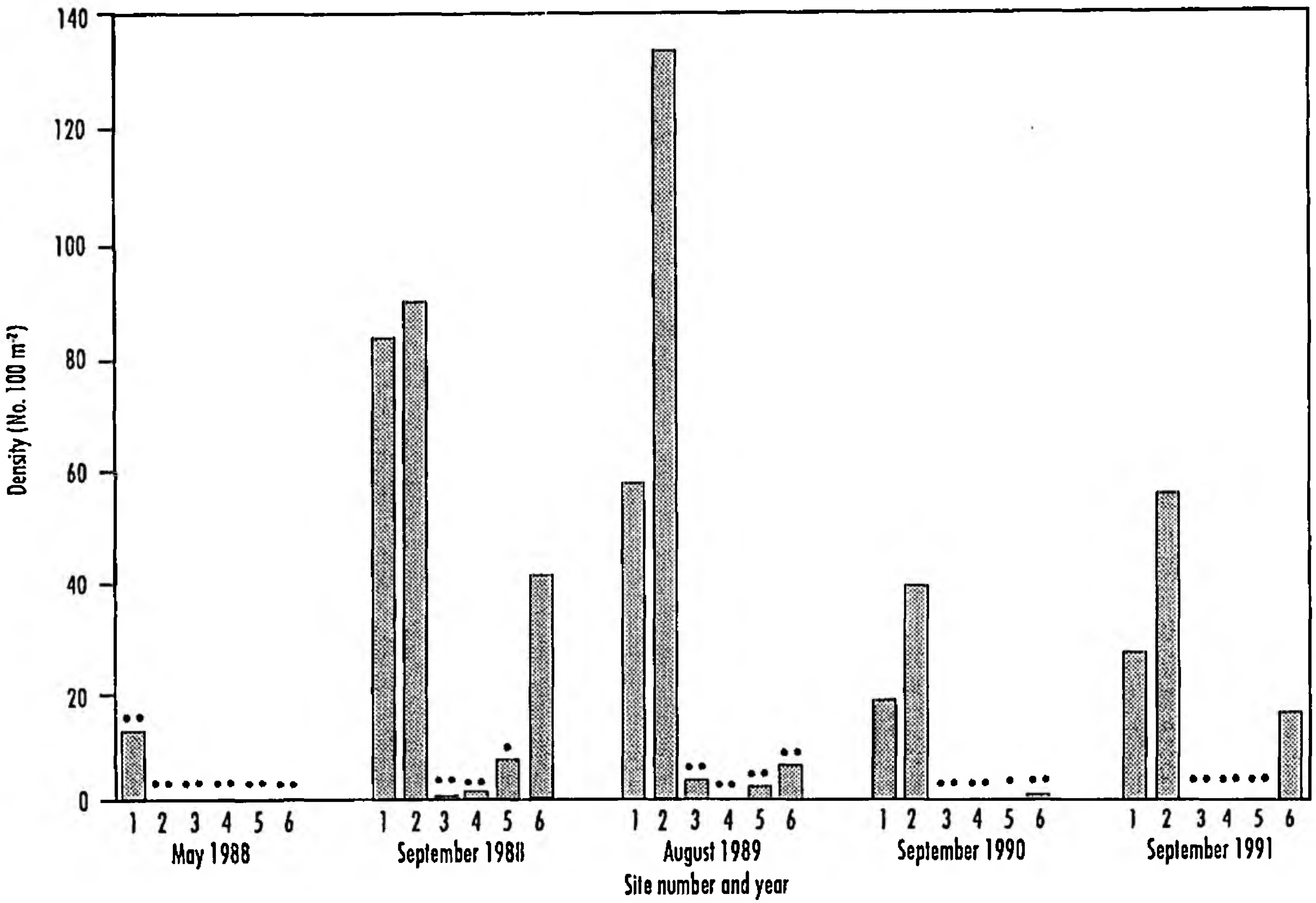


Figure 8.7 Density of trout fry (0+) at sites on the Deepford Brook.
Asterisks indicate the significance of differences from HABSCORE predictions ** $p<0.01$; * $p<0.05$; + indicates data unavailable

(above the pollution input) were significantly lower than would be expected under clean water conditions. By September of the same year fry were present at all the sites, such that only the densities at those sites in the middle reach of the stream (3, 4 and 5) were significantly lower than HABSCORE predicted. Throughout the rest of the study trout fry densities remained significantly lower than expected at these middle sites, with the density at the bottom site (site 6) significantly lower in August 1989 and September 1990.

Historical data for site 3 (Figure 8.8) indicated that even before the pollution, trout fry density at this site was significantly lower than predicted.

Trout parr populations exhibited a more varied response than fry. They were absent from site 3 following the pollution in May 1988 and the densities at sites 2, 3 and 4 were significantly lower than HABSCORE predicted (Figure 8.9). The density at site 1, above the input, was also lower than predicted. The following year, trout parr had returned to site 3 but the densities at this and site 2 were still significantly lower than expected. In August 1989, the same was true at sites 1 and 4, and in September 1990 at sites 2, 3, 4, 5 and 6. In the final year of the study the densities at sites 1, 2, 4 and 5 were significantly lower than predicted. Historical data (Figure 8.8) suggested that trout parr populations at site 3 had returned to pre-pollution levels in August 1989, within 15 months of the pollution.

Data from the sites fished on the tributaries (Figures 8.10 and 8.11) showed that trout fry were absent from site 8, at the bottom of the Churchill Stream, during the final three years of the study. Parr, however were present at this site during all fishings and only at a density significantly lower than predicted in September 1991. At the bottom of the Cotland Brook (site 10), fry densities were significantly lower than predicted in May 1988 and September 1990 and 1991. Fry densities at site 9, near the top of Cotland Brook, were only significantly lower than predicted in September 1990. Trout parr density at site 9 was significantly lower than predicted during all fishings except September 1990.

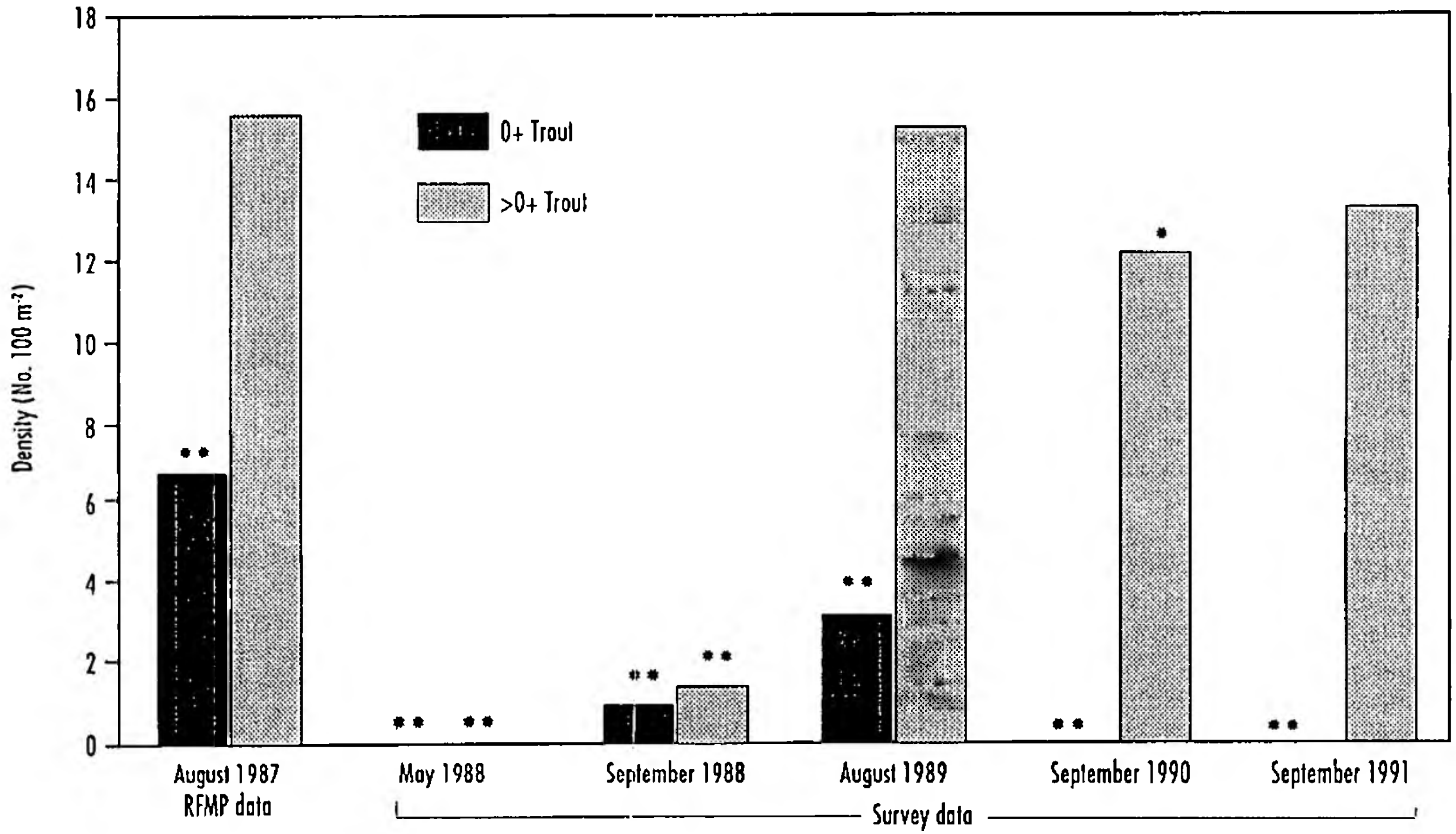


Figure 8.8 Comparison of historical data with survey data for trout, at site 3 on the Deepford Brook.
Asterisks indicate the significance of differences from HABSCORE predictions ** $p<0.01$; * $p<0.05$

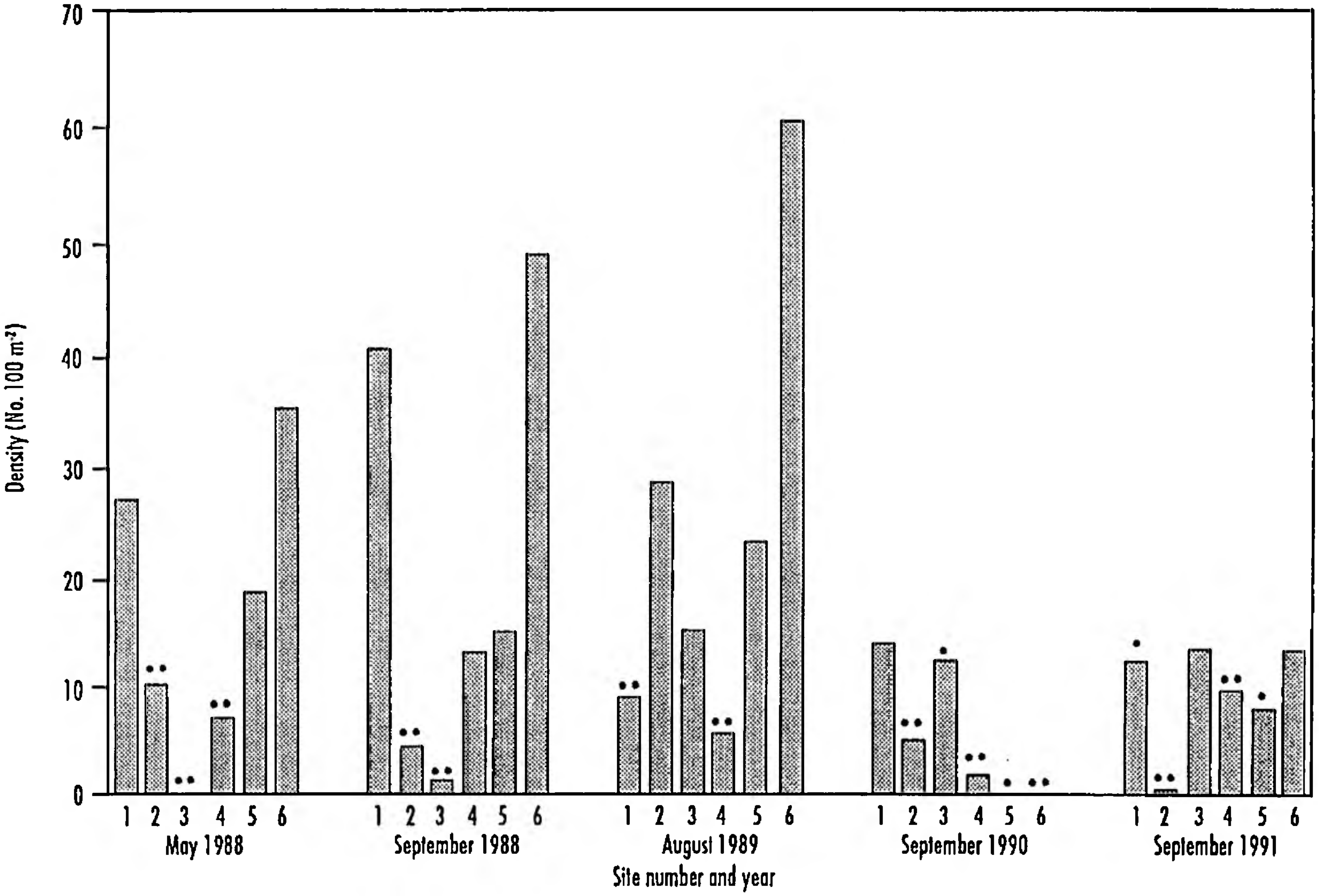


Figure 8.9 Density of trout parr (>0+) at sites on the Deepford Brook.
Asterisks indicate the significance of differences from HABSCORE predictions ** $p<0.01$; * $p<0.05$; + indicates data unavailable

Salmon fry were absent from all sites on the Deepford Brook in May 1988, immediately following the pollution (Table 8.4). In the remaining four surveys they were present only at site 6. At no time was the density of salmon fry significantly lower than that predicted by HABSCORE.

Salmon parr were present at site 6 in all years, apart from September 1990 (Table 8.4 Figure 8.12). They were also present at site 4 in September 1988 and site 5 in September 1991. As with salmon fry, at no time were the densities significantly lower than those predicted by HABSCORE.

In the tributaries, fry were present at site 10 on the Cotland street in September of 1990 and 1991. Salmon parr were present only at the very beginning of the study at site 8 on the Churchill Stream.

Table 8.4 Densities of salmon fry and parr (No. 100 m^{-2}) at sites on the Deepford Brook and its tributaries

Site	May 1988		Sept 1988		Aug 1989		Sept 1990		Sept 1991	
	0+	>0+	0+	>0+	0+	>0+	0+	>0+	0+	>0+
1	0	0	0	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0	0	0	0
3	0	0	0	0	0	0	0	0	0	0
4	0	0	0	3.3	0	0	0	0	0	0
5	0	0	0	0	0	0	+	+	0	1.6
6	0	34	30.8	44.5	5.7	19.1	5.35	0	5.94	2.3
7	-	-	-	-	-	-	-	-	-	-
8	0	3	0	0	0	0	0	0	0	0
9	0	0	0	0	0	0	0	0	0	0
10	0	0	0	0	0	0	0.7	0	0.8	0

+Data unavailable

8.4 Discussion

8.4.1 Trout

Both the Afon Fenni and the Deepford Brook showed the ability to recover to a certain degree following their fish mortalities. Recovery was much more marked in the Afon Fenni whose fish stocks had originally been the more seriously depleted.

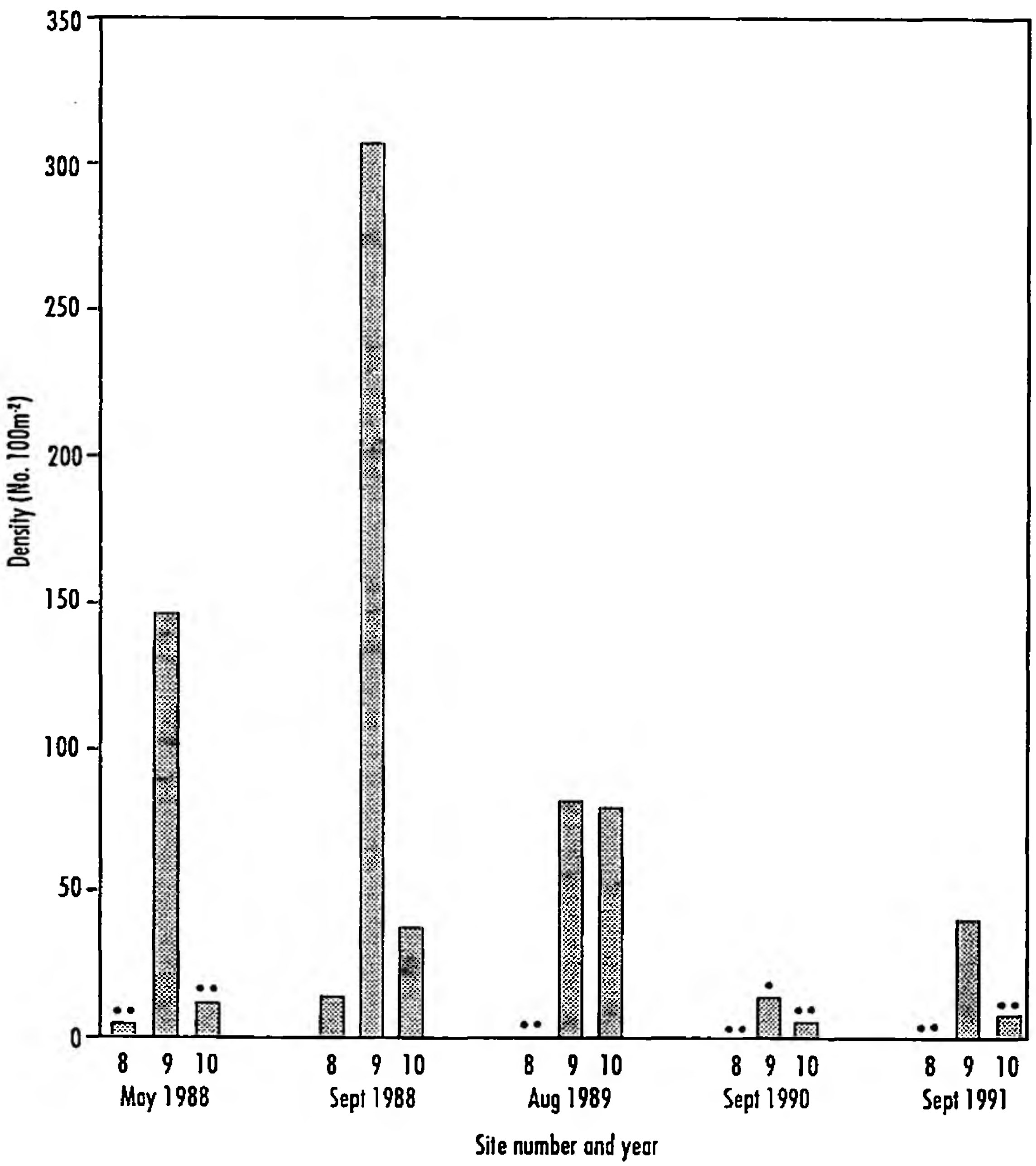


Figure 8.10 **Density of trout fry (0+) at sites on the tributaries of Deepford Brook. Asterisks indicate the significance of differences from HABSCORE predictions ** $p<0.01$; * $p<0.05$**

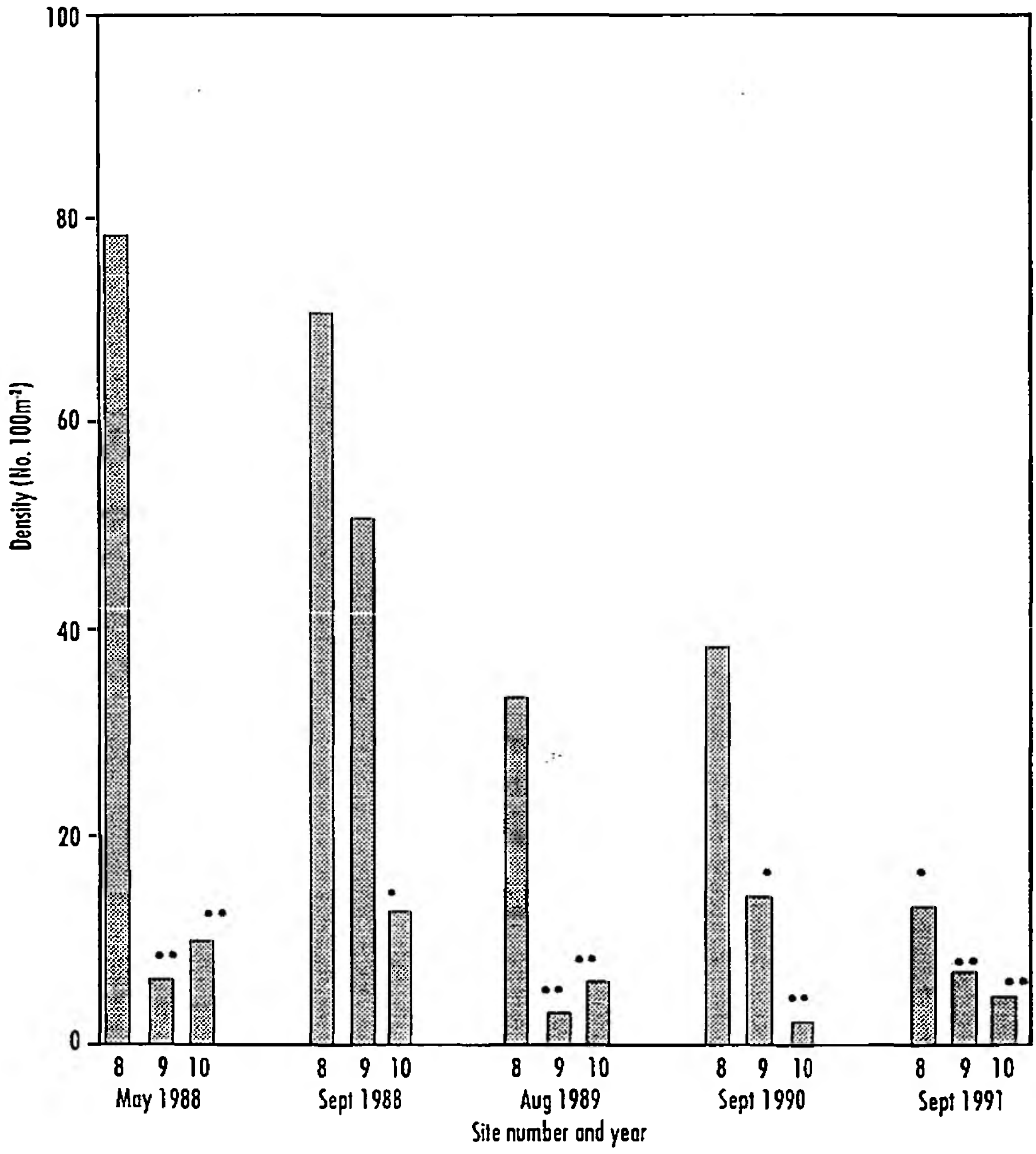


Figure 8.11 Density of trout parr (>0+) at sites on the tributaries of Deepford Brook. Asterisks indicate the significance of differences from HABSCORE predictions ** $p<0.01$; * $p<0.05$

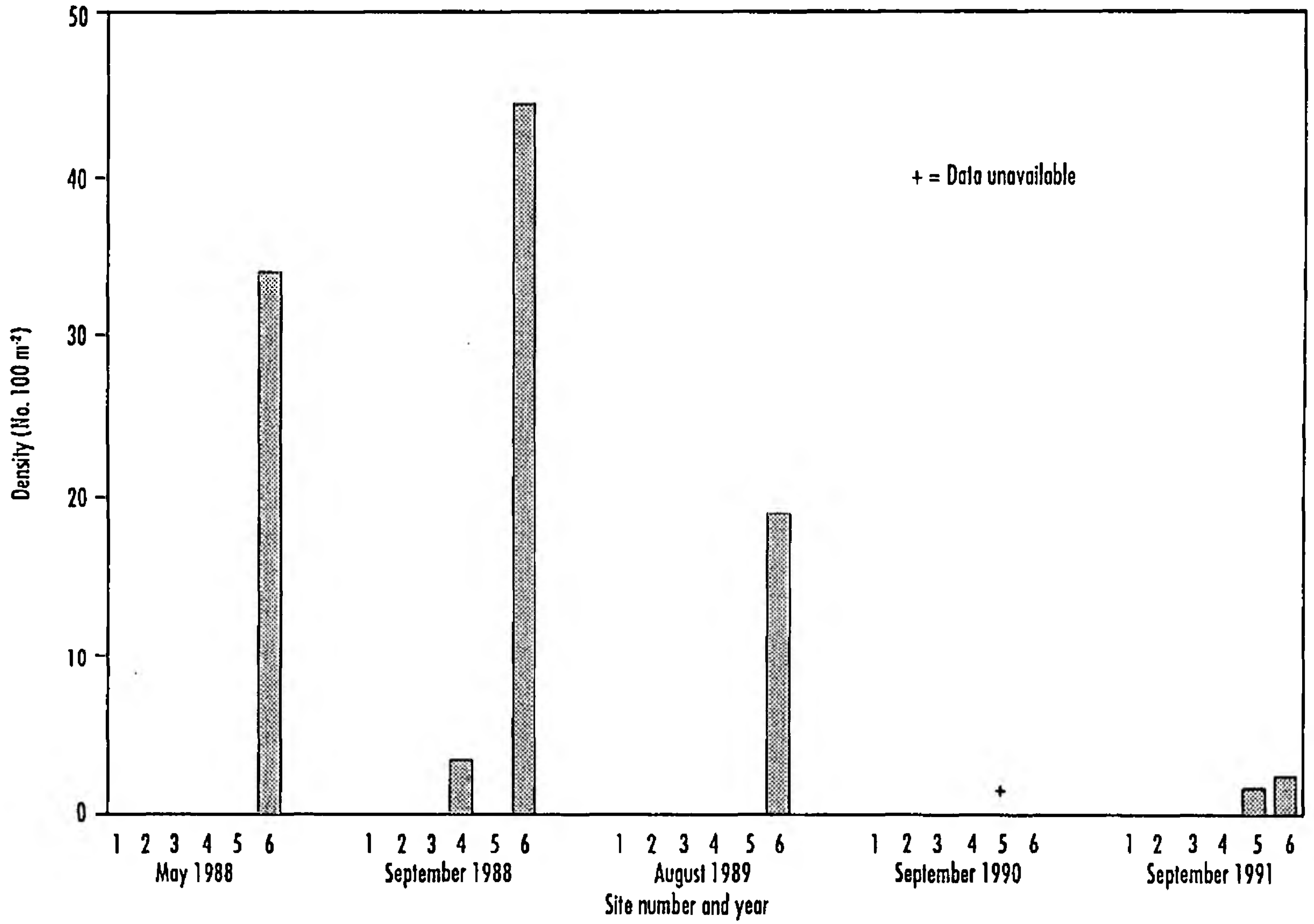


Figure 8.12 Density of salmon parr (0+) at sites on the Deepford Brook

Trout populations on the Afon Fenni had shown a substantial recovery within two years of the fish mortality. Comparison with HABSCORE predictions and historical data indicated that in the middle region of the stream, trout fry populations had recovered within a year of the fish mortality. Trout parr were slower to recover but numbers had reached predicted levels at most of the middle and lower sites within two years of the mortality. Redistribution within the stream and its tributaries would have been responsible for the parr populations in 1990, with recruitment from the previous years fry supplementing this in 1991.

In Deepford Brook, partial recovery of fry populations had occurred by September 1988. This had to be attributed to the redistribution of fry within the stream, as no restocking had been undertaken. Jorgensen and Berg (1991) showed that trout fry can move up and downstream up to a maximum distance of 600 m. The Deepford mortality occurred in May, before density-dependent mortality mechanisms would have had time to operate fully, and hence in the unaffected reaches of the catchment there would have been a high density of newly emerged fish available to recolonise (the significantly low density of fry at site 1 may have been due to difficulties in electrofishing the small, recently emerged fish, causing an underestimation of their numbers). Following this initial recolonisation, recovery appeared to cease and fry populations remained depressed at the lower sites throughout the rest of the study.

Trout parr on the Deepford Brook were thought to have recolonised site 2 immediately after the pollution, in the period before the first survey. Parr had returned to site 3 by September 1988. Further assessment of recovery was difficult as the densities fluctuated so much.

Previous work on salmonid recovery includes a study by Workman (1981) who looked at the recovery of trout in a Montana stream (U.S), following chemical poisoning. The results suggested that it took 4 years and 3 spawning periods before the carrying capacity of the habitat had been attained over the affected reach of 35.6 km. More recent, unpublished, work includes a study by NRA Yorkshire Region who have been examining the recovery of trout stocks following a release of chlorine into Fossdale Beck, a tributary of the River Ure (NRA 1991a). Results showed that trout populations had largely re-established themselves within a normal life span (three to four years) in the study reach of 300 m. Both these studies differed from the Afon Fenni and the Deepford Brook in that they each had large unaffected upstream sections which provided a source of colonists.

Recovery of trout on the Afon Fenni may have been aided by the presence of sea trout, which were absent from the studies of Workman (1981) and NRA Yorkshire Region (1991a). They were also known to be present on the Deepford brook but only at the lowest site, and are therefore likely to have contributed little to the overall recovery of the stream. Work by NRA Welsh Region on the River Teifi (NRA 1991b) showed that salmon populations recovered quicker than those of non-migratory trout. It was suggested that the migratory salmon were less susceptible to sudden pollution events in the streams where they spawned, than were the resident, native trout populations. Historically, redds excavated by migratory sea trout have been present on the Afon Fenni as far upstream as site 4. If migratory trout share the same ability to recover from this type of sudden pollution event as do salmon, it may help to explain the rapid recovery of juvenile trout in the middle and lower reaches of the Afon Fenni. It is also possible that a fall in trout density within the stream may occur in the future when fish that were killed in the mortality should have returned to spawn.

The low observed (compared to predicted) trout fry and parr densities in 1990 and 1991 at the upper sites on the Afon Fenni were most probably the result of further inputs of agricultural waste. From time to time during the survey the effects of chronic pollution were observed in the upper reaches of the stream. Sewage fungus cover of up to 30% was observed within the stream: at site 3 in spring 1990 (Section 5.1) and during electrofishing in 1991, and in the top tributary (T), considered to be a potential source of colonists. This may account for the absence of fish from this tributary and the reduction in the density of trout fry at site 2 in the final year of the survey. Indeed there were 6 farms graded by pollution control as having either a high or medium pollution risk in the vicinity of these top sites (Figure 8.12). In contrast, there was only one such farm in the middle region of the stream, where sites 4, 5 and 6 were situated.

Recruitment of trout fry at the bottom sites on the Afon Fenni and at the middle and lower sites on the Deepford Brook was poor. Historical data from site 7 on the Afon Fenni indicated that there was an adverse influence on populations at this site even before the fish mortality occurred. This may be a result of further inputs of agricultural pollution in the lower reaches. In Spring 1990 an overflowing slurry lagoon was observed discharging into the Churchill Stream (site 8), causing the growth of sewage fungus (Section 5.1). A ditch was observed during electrofishing in 1990, discharging slurry into Deepford Brook just below site 5.

Trout populations in the lower reaches of both streams may also be affected by siltation. Naismith and Wills (1991), working on the Torridge catchment in Devon, noted an elevated silt content in the spawning gravels of intensively farmed catchments with poor salmonid populations, and suggested that this was affecting egg and alevin survival. The lower reaches of the Deepford Brook were of a gentler gradient than the Afon Fenni and silting would have been more pronounced over a greater proportion of its length (see Tables 8.1 and 8.2). This may help to explain why the recovery of trout populations on the Deepford Brook was not a good as on the Afon Fenni.

8.4.2 Salmon

The distribution of salmon in West Wales tends to be restricted to streams with a width greater than 3 - 4 m (Wightman 1989) and so the absence of salmon from the upper sites on both streams was not considered unusual (see Tables 8.1 and 8.2). On the Afon Fenni, fry and parr both returned to the bottom sites within a year of the pollution, indicating that some recovery had taken place. In 1991, results indicated further improvement in parr populations but the fry densities at the bottom sites had declined. Comparison with historical data for site 7 suggested that both fry and parr populations had recovered by 1990. On the Deepford Brook, salmon fry had returned to site 6 in the September following the mortality, again suggesting that some recovery had taken place. Accurate assessment of recovery of salmon on both streams was difficult as they were at the limit of their distribution and prone to large variations in density.

9. RECOVERY OF STREAM COMMUNITIES FOLLOWING SIMULATED INCIDENTS

9.1 Introduction

9.1.1 Background

Pollution control measures taken in recent years have decreased pollution loads from chronic discharges to receiving waters. However, sporadic pollution from farm wastes, storm sewage overflows and chemical spills is now being recorded more commonly, posing significant problems of water quality management. The number of episodic pollution incidents recorded in England and Wales has almost doubled, from 12 500 in 1981 to 24 153 in 1988 (NRA/MAFF, 1989). In 1988, industrial incidents, including the release of oil and a large variety of chemicals accounted for 34% of these events. Discharge of sewage from storm overflows etc. and the release of farm wastes were also major problems, contributing 20% and 17% of total discharges respectively.

Pollution episodes involve a toxicant input of short duration and high concentration, which passes downstream as a wave of increasing length and reducing amplitude (Edwards 1987, Pascoe 1988). The physico-chemical processes in aquatic systems (e.g. dispersion, sorption) cause attenuation (weakening) of pollutant activity, and there is also evidence that dispersion in the stream bed is related to depth of substrate penetrated by the pollutant (Bencala 1984, Edwards *et al.*, 1991). After an episode it is difficult to establish the time-concentration profile of exposure and the previous ecological status of downstream reaches unless, of course, an incident can be anticipated or arranged. After the event, recovery of populations and communities occurs through the processes of recolonisation and succession. This is in contrast to the classical descriptions of spatial recovery (e.g. Kolkwitz and Marsson 1909, Streeter and Phelps 1925, Hynes 1960) characteristic of continuous discharges, which involve selective processes based on species sensitivity and physiological and genetic adaptation to a quasi-steady-state.

9.1.2 Objectives

Since there is rarely an opportunity to record the impact of, and recovery from, these episodes, their effects on stream biology are difficult to interpret. In view of this, a programme of field experiments, simulating pollution incidents, was undertaken to provide a fuller understanding of episodic pollution and the processes of recovery. Whilst experiments of this type are subject to statistical strictures described by Hurlbert (1984), the effects are frequently dramatic and unequivocal in nature and conclusions drawn from them may be supported by other evidence (Edwards *et al.*, 1991).

Farm waste was used as a model pollutant type for the investigation. Agricultural pollution incidents are of increasing concern, particularly those resulting from accidental or deliberate discharges of animal slurry and silage. These effluents are highly polluting with a typical biological oxygen demand (BOD) for undiluted slurry of 20 000 mg l^{-1} and for silage liquor of 80 000 mg l^{-1}. Total ammonia concentrations are approximately 1200 mg l^{-1} and 200 mg l^{-1} for slurry and silage respectively. Concentrations recorded in small watercourses following incidents are highly variable, with concentrations of total ammonia up to 700 mg l^{-1}, DO down to 0.7 mg l^{-1} and pH as low as four (in association with discharges of silage). Cattle slurries also contain significant concentrations of sulphides.

The objectives of the research programme were:

- o To provide a fuller understanding of organic pollution episodes through field-simulations. This was achieved by dosing streams to produce low DO, increased ammonia and increased sulphide concentrations relevant to farm wastes.

- o Investigate the recovery of streams following these disturbances.

- o Identify some of the processes involved in the changes detected above.

The emphasis of the study was on the response of macroinvertebrates; however, some data were gathered on the impact upon fish. The results of the three year research programme has been reported fully in a thesis submitted in candidature for the higher degree of Philosophiae Doctor. This section summarises the main findings.

9.1.3 Research programme

Seven experiments simulating pollution episodes were undertaken between July 1988 and June 1990 (Table 9.1) consisting of: reduced dissolved oxygen (four); increased ammonia (two); increased sulphide (one). A further field experiment, using sodium chloride as a tracer, was carried out to study dispersion. In addition, laboratory experiments examining migration responses of macroinvertebrates (drift and vertical movements) were undertaken in 1989, using a small-scale artificial stream.

9.2 Methods

9.2.1 Field experiments

Site selection

Sites offered by NRA Welsh Region were chosen on the following criteria:

- Streams were of low order (1st - 3rd, Strahler, 1957) and of good quality (equivalent to NWC class 1A or 1B), supporting diverse fish and invertebrate faunas.
- The impact of field experiments was spatially limited by dilution from downstream sources.
- Access was relatively easy, in view of the large volume of equipment and chemicals transported to the sites.
- Landowners were sympathetic.

Figure 9.1 refers to the locations of field sites. National grid references are: Mounton Brook, ST485945; Nant Dowlais with tributary, ST074840 and Yazor Brook, SO455427.

Pollution episode simulation

Although the experiments were designed to investigate the biological effects of low DO, increased ammonia and increased sulphide episodes relevant to farm wastes, the results have wider application to other forms of organic episodic discharge (e.g. storm-sewer overflows), since the component effects could originate from a wide range of sources of organic waste.

A brief description of each simulated episode is given below and in Table 9.1. In each simulation, the downstream effects of the pollutant release were compared to an untreated reference zone upstream of the dosing point.

a) Reduced dissolved oxygen

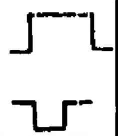
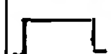
Episode 1, on Mounton Brook (July 1988), was designed to investigate the impact of a short episode (6 hr) of very low (1 mg l^{-1}) DO.

Episode 3, on Yazor Brook (May 1989), examined the effect of a brief episode (6 hr) of intermediate dissolved oxygen concentration (3-4 mg l^{-1}). The number of dead animals in the drift was determined.

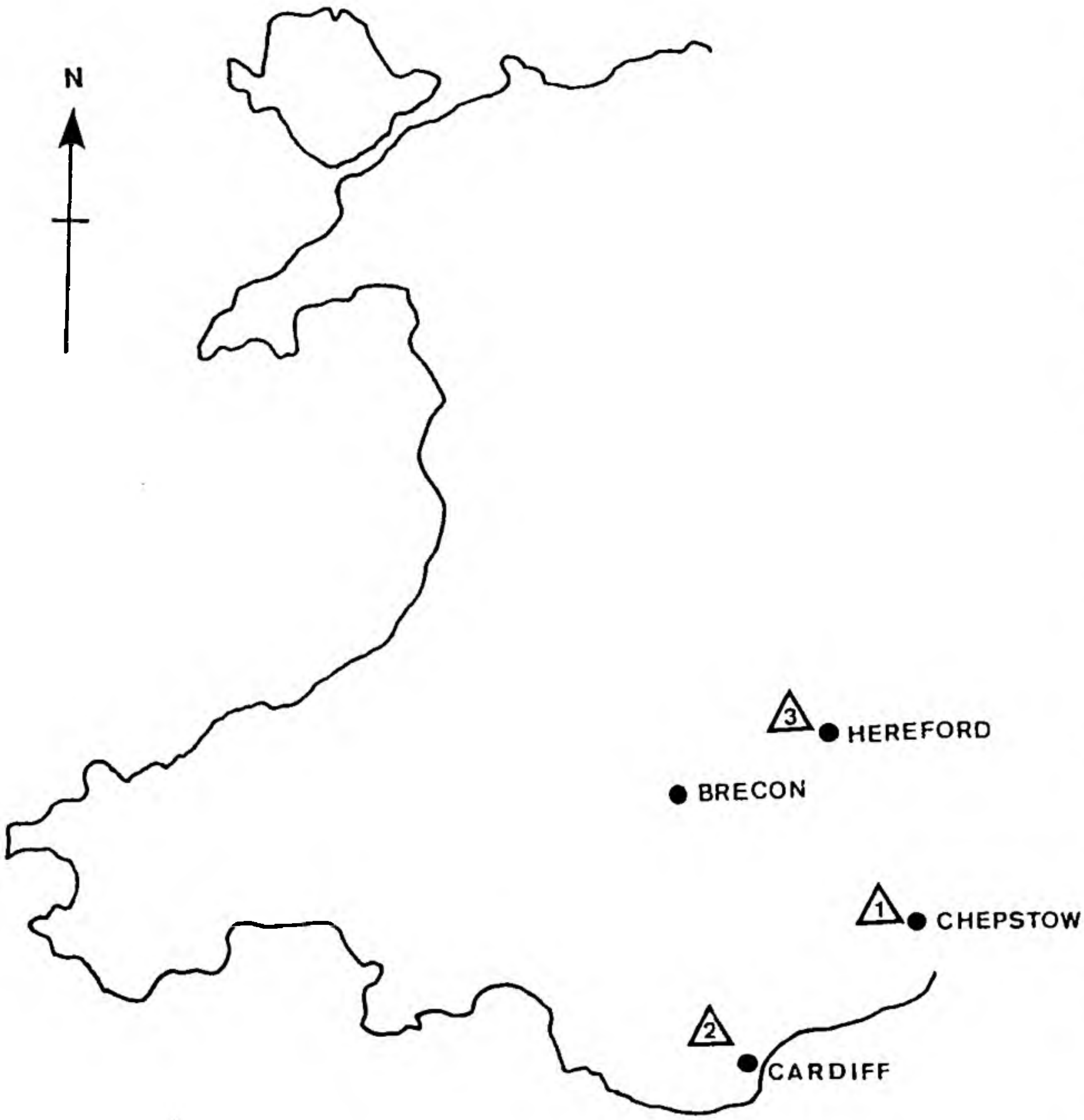
Episode 4, on Mounton Brook (July 1989), sought to determine the DO concentration of at which invertebrate drift increased during short exposure (1 hr). This was achieved by progressively reducing the concentration in a stepwise fashion, with intervening periods of recovery (1 hr), when the oxygen concentrations were returned to normal.

Episode 6, on Mounton Brook (November 1989), examined the influence of season by comparing responses with those in July (Episode 4).

Table 9 . 1 Summary of simulated episodes.

EPISODE	TYPE	LOCATION	DURATION (hr)	NOMINAL Concentrations (mgl^{-1})	SHAPE	DATE	INVESTIGATIONS	
							Chemistry	Biological
1	Reduced Dissolved Oxygen	Mounton Brook	6	1		July 1988	Surface	Macroinvertebrate drift, Precopula Gammarus pulex, Mortality of fish and macroinvertebrates. Impact and recovery of Macrobenthos.
2	Increased Ammonia*	Nant Dowlais	6	5		September 1988	Surface	Macroinvertebrate drift, Precopula G.pulex, Invertebrate behaviour, Mortality of Fish and Invertebrates. Impact on Macrobenthos.
3	Reduced Dissolved Oxygen	Yazor Brook	6	2		May 1989	Surface and interstitial	Macroinverterate drift, impact and recovery of Macrobenthos.
4	Reduced Dissolved Oxygen	Mounton Brook	12	12-1		July 1989	Surface	Drift vs. Oxygen concentration, Precopula G.pulex, impact and recovery of Macrobenthos, recovery pathways.
5	Increased Ammonia* with Reduced Dissolved Oxygen	Tributary Nant Dowlais	24	*Ammonia 6 D.O. 5		September 1989	Surface and interstitial	Macroinvertebrate Drift, Precopula G.pulex Mortality G.pulex, Impact and recovery of Macrobenthos.
6	Reduced Dissolved Oxygen	Mounton Brook	12	12-1		November 1989	Surface	Time of year comparison with July (No. 4). Drift vs. oxygen concentration, Impact and recovery of Macrobenthos.
7	Sodium Chloride	Tributary Nant Dowlais	4	65		April 1990	Surface and interstitial	None.
8	Increased Sulphide	Tributary Nant Dowlais	24	5		May 1990	Surface and interstitial	Macroinvertebrate drift, Precopula G.pulex, Mortality G.pulex, Impact on Macrobenthos.

* Unionised



1. Mounton Brook.

2. Nant Dowlais.

3. Yazor Brook.

Figure 9.1 Site locations for simulation of pollution episodes

b) Ammonia

Episode 2, on Nant Dowlais (September 1988), examined the impact of a short episode (6 hr) of increased unionised ammonia (5 mg l^{-1}, pH 8.5) mimicking the shape of an attenuated pulse (sinusoidal).

Episode 5, on a tributary of Nant Dowlais (September 1989), was designed to study the impact of prolonged (24 hr) high concentrations of unionised ammonia (6 mg l^{-1}, pH 9.0). The ability of semi-natural substrate to protect *G. pulex* from the full effect of the toxicant was also investigated. In an additional downstream zone where unionised ammonia concentrations had decreased to ~3 mg l^{-1}, the DO concentration was artificially lowered to 5 mg l^{-1} for 12 hours.

c) Sulphide

Episode 8, again on a tributary of Nant Dowlais (May 1990), investigated the impact of a prolonged episode (24 hr) of increased sulphide (nominal 6 mg l^{-1} as total sulphide. The influence of water velocity upon the toxicity of sulphide to *G. pulex* (mortality) was also examined.

Dosing of streams

Dissolved oxygen concentrations were reduced by the injection into the stream of sodium sulphite (Na_2SO_3 solution) the rapid oxidation of which was catalysed by cobalt maintained in the stream at trace concentrations (0.2-1.0 mg l^{-1}). Stream concentrations of sulphate (from oxidation of sulphite) and cobalt occurring during dosing were non-toxic.

Ammonia concentrations were raised by the addition of ammonium chloride (NH_4Cl) solution (150-200 g l^{-1}). Ammonia is most toxic in the un-ionised state (Wurhmann and Woker 1948, Alabaster and Lloyd 1980, Williams *et al* 1986) and therefore at high pH. In designing experiments with ammonia, a compromise is required to ensure the increased pH is not directly damaging (Edwards *et al* in press) and was increased to between 8.5 and 9.0 using sodium hydroxide (NaOH) solution.

During Episode 7 (tracer experiment), sodium concentrations were increased using a 75 g l^{-1} solution of sodium chloride (NaCl). During Episode 8 sulphide concentrations were increased using sodium sulphide (Na_2S) solution (150 g l^{-1}). Chemicals were delivered into the stream using peristaltic pumps or by gravity feed. Dosing rates were dependent upon stream discharge, estimated before the period of dosing from the mean flow velocity (ms^{-1}) multiplied by the cross-sectional area (m^2) (BS 3680: Part 3: 1964). Mixing zones and travel times were gauged using fluoroscene dye.

Water quality monitoring

Dissolved oxygen, pH and temperature were monitored using pHOX 100 DPM multiparameter instruments and surface water samples were taken for metals analysis by ICP. Ammonia samples were preserved with sulphuric acid to <pH 2. Sulphide samples were preserved with zinc acetate (2N) after settling of solids. Total ammonia and sulphides were determined colorimetrically (APHA 1985, parts 417 and 427) and the undissociated species (NH_3 and H2S) determined from pH and temperature readings (Seager, Wolff and Cooper 1988, Mance, O'Donnell and Campbell 1988).

Attenuation studies

During Episodes 3, 5, 7 and 8 interstitial water from the sediment was sampled using stainless steel stand-pipes (1 cm x 1 m) at 10-15 cm and 30-35 cm depth in the stream bed, to assess the 'within-sediment' attenuation of pollutant episodes. In Episodes 5, 7 and 8 samples were withdrawn using 50 ml plastic syringes and preserved in plastic universal bottles (30 ml). Dissolved oxygen was measured *in situ* during Episode 3 using a micro-medical oxygen electrode inserted into the stand-pipes.

Biological studies

The following were investigated during toxicant episodes.

- o Invertebrate drift - this was measured using 0.5 mm mesh nets held inside 0.3 m diameter, 1 m long steel tubes in which the flow of water was measured. Drift was expressed as animals/m^3 (after Elliott 1970) and the total drift could be estimated using stream flow data.

- o Benthic invertebrates - kick samples (Hynes 1961) were taken before and after the treatments in riffles (1 min) and margins (1 min) to assess impact and recovery. Quantitative cylinder sampling (Niell 1938) was used at the Yazor Brook (Episode 3).

- o *In situ* toxicity tests - opportunities were taken by colleagues to use the simulated episodes for toxicological studies. They conducted *in situ* tests with a range of caged invertebrates and fish. Lethal and sub-lethal responses were measured (McCahon *et al* in press). The influences of semi-natural substrates (sand and stones) and water velocity upon the toxicity of ammonia (Thomas *et al* in press) and sulphide to *G. pulex* were also investigated.

Recovery mechanisms

Williams and Hynes (1976) described four principal pathways or mechanisms by which stream benthos colonise small areas of implanted substrate: downstream, upstream, aerial and within-substrate movements (see also Townsend and Hildrew 1977, Williams 1977). However, these earlier studies did not address the relative importance of each pathway following a pollution episode because only small patches of substrate were initially cleared of invertebrates and migration from adjacent areas could occur.

A study of recovery mechanisms was undertaken following deoxygenation of the Mounton Brook in July 1989 (Episode 4). Four traps were designed, each to limit the recolonisation to a single route into the stream substrate, and a fifth trap was included which allowed colonisation by all pathways. Four replicates of each trap type, containing a semi-natural substrate of limestone chips, were used. All twenty traps were constructed to the specifications of Williams and Hynes (1976).

9.2.2 Laboratory investigations

Episodic pollution has been shown to have an impact upon various facets of invertebrate behaviour, particularly drift (Edwards *et al.*, 1992) and precopular disruption in *G. pulex* (McCahon *et al* in press). The vertical migrational behaviour of invertebrates within stream beds has been observed during spate conditions (Williams and Hynes 1974). However, field observations made during simulated episodes of reduced pH were equivocal (Ormerod *et al* 1987). In the current study, a small scale laboratory stream (1 m length) was designed to examine vertical migration (Figure 9.2). An artificial substrate of glass beads, bounded by sheets of perspex, simulated a section of gravel bed 20 cm deep and 4 cm thick. This allowed animals to be observed and counted so that migration into and out of the substrate could be assessed. A small drift net at the downstream end was used to monitor downstream migration. In a series of experiments of reduced dissolved oxygen and increased ammonia, the stream was seeded with animals (*E. venosus* or *G. pulex*) and dosed in a similar manner to the field experiments described earlier. The substrate was covered by opaque screens between observations of the number of animals within the glass beads before, during and after dosing.

9.3 Results and discussion

9.3.1 Field investigations

Biological responses in the field were determined from invertebrate drift densities, *in situ* toxicity tests and comparisons of the benthic macroinvertebrate fauna before and after the experiments.

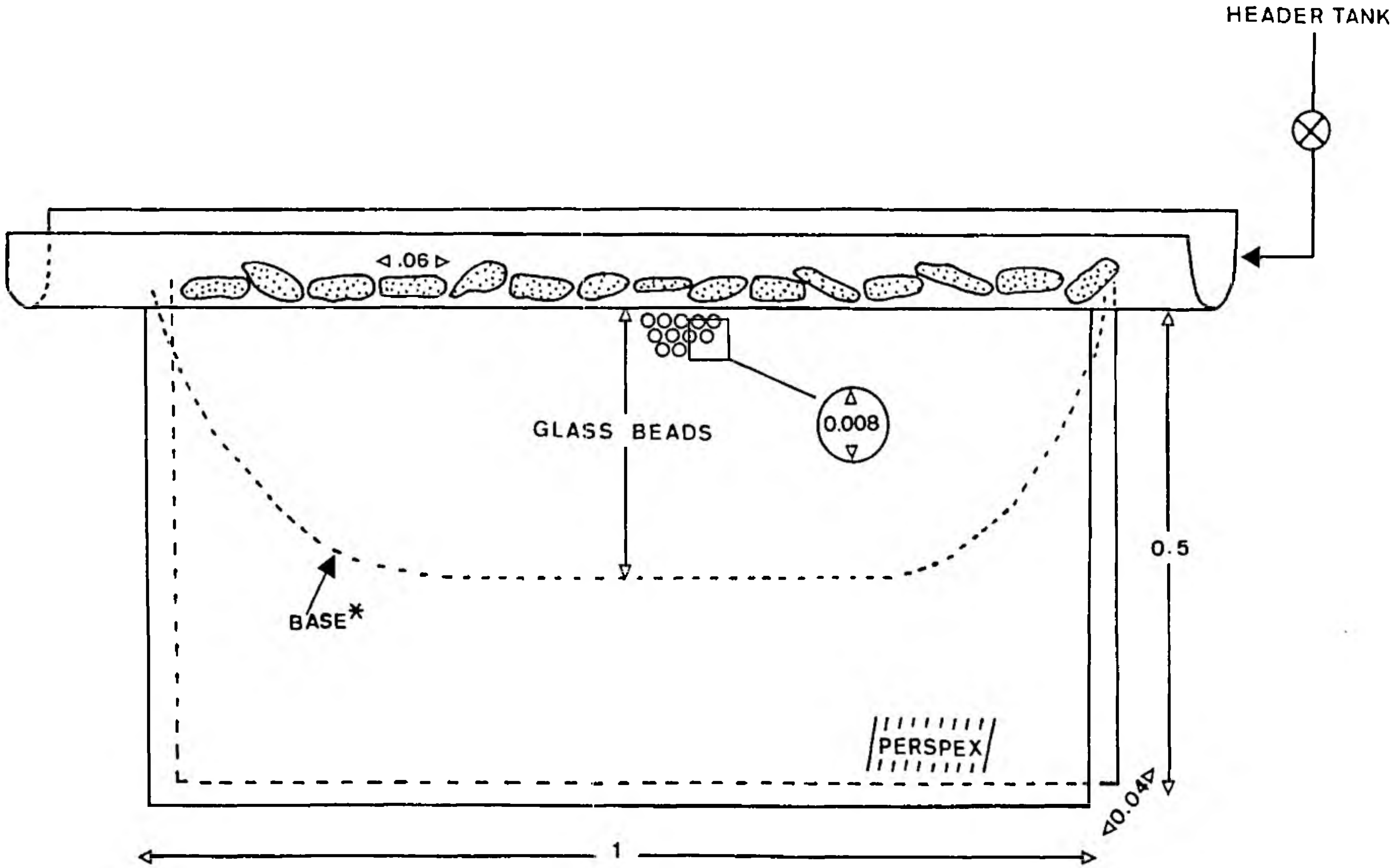


Figure 9.2 Artificial stream section for investigating vertical migration. Dimensions are indicated in metres. *Base is a plastic divider which supports beads and separates perspex sheets. The whole structure is suppprted in a steel frame.

Dissolved oxygen

a) Invertebrate drift and *in situ* toxicity tests

The drift density at the Mounton Brook was normally very low during the day (<0.5 animals m^{-3}). This increased in response to the reduced DO during Episode 1 (Figures 9.3a and b) to a peak of 43 animals m^{-3}, consisting mainly of *Ephemerella ignita*, *Ecdyonurus venosus*, *Rhithrogena semicolorata*, *Baetis rhodani* and *Isoperla grammatica*. Another peak occurred later associated with *Gammarus pulex*, *Ephemera danica*, *Limmius volckmari*, *Simuliidae* and *Chironomidae*. Densities in the upstream reference zone remained low during daylight and there was the normal night-time peak; this night-time peak did not occur in the treatment zone. There were also substantial mortalities in caged fish and invertebrates and a range of sensitivities was observed (Table 9.2). Studies with *G. pulex* demonstrated that the low-oxygen regime disrupted precopular behaviour and that the presence of a parasite (*Pomphorynchus laevis*, Ancanthocephala) increased mortality.

During Episode 3, dissolved oxygen concentrations of 3-4 mg l^{-1} (Episode 3) also caused substantial increases in drift over a 6 hour period. The proportions of dead *Baetis* and *Gammarus* in the drift were normally very low during daylight (<1%), but the proportion of dead *Baetis* increased during the night to between 5 and 10%. At the end of the 6 h exposure to 3-4 mg l^{-1} DO the proportions of dead *Baetis* and *Gammarus* were 52% and 43% of the total number caught for each taxa, respectively. During Episode 4, the DO concentration was lowered in a stepwise fashion to determine the concentration at which invertebrate drift increased. Normal drift densities were low in both the treatment and reference zones (<3 and <1 animals m^{-3}). The first significant increase in total drift occurred at 4 mg l^{-1} dissolved oxygen (Figures 9.4a and 9.5a) and recovered to background levels when the DO returned to normal. Exposure to 2 mg l^{-1} DO also increased drift densities (>10 fold); however, the density remained above background in the recovery period following this treatment (Figures 9.4a and 9.5a).

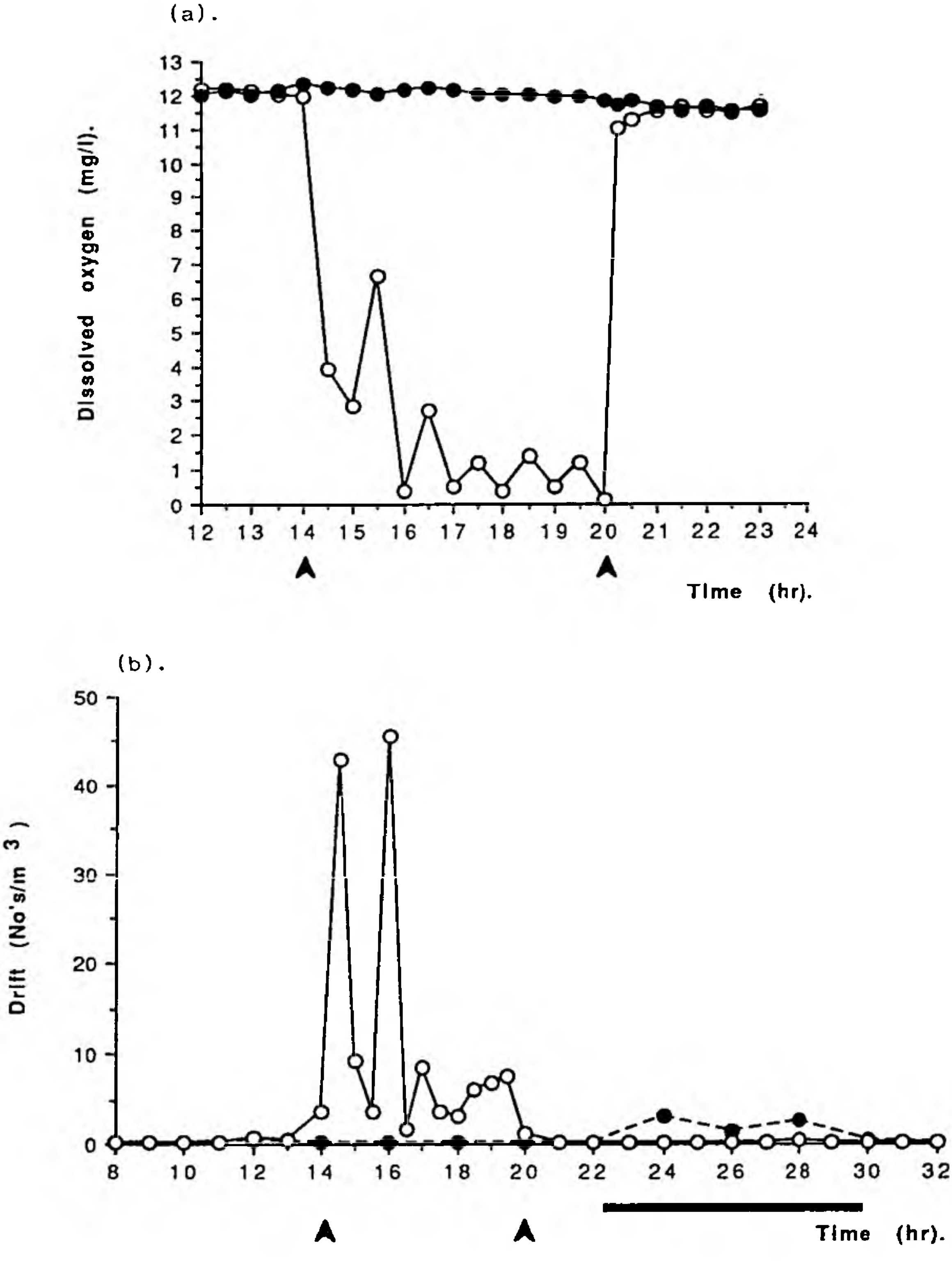


Figure 9.3 Dissolved oxygen regimes during episode 1 (a) and drift response (b) in reference (●) and treatment (O) zones.

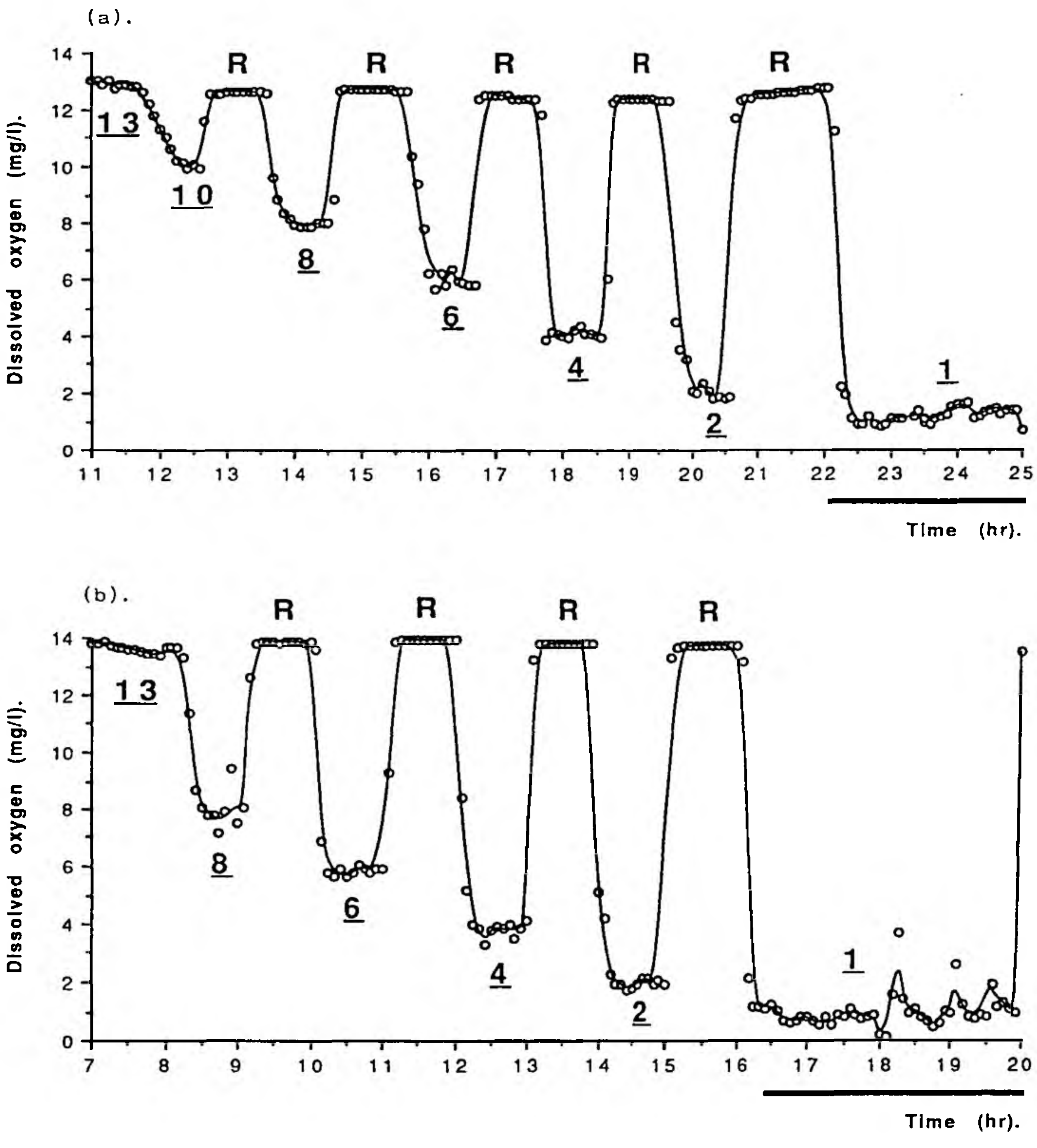


Figure 9.4 Dissolved oxygen regimes during episode 4 (a; July) and episode 6 (b; November). Underlined figures refer to nominal oxygen concentrations and R=recovery period.

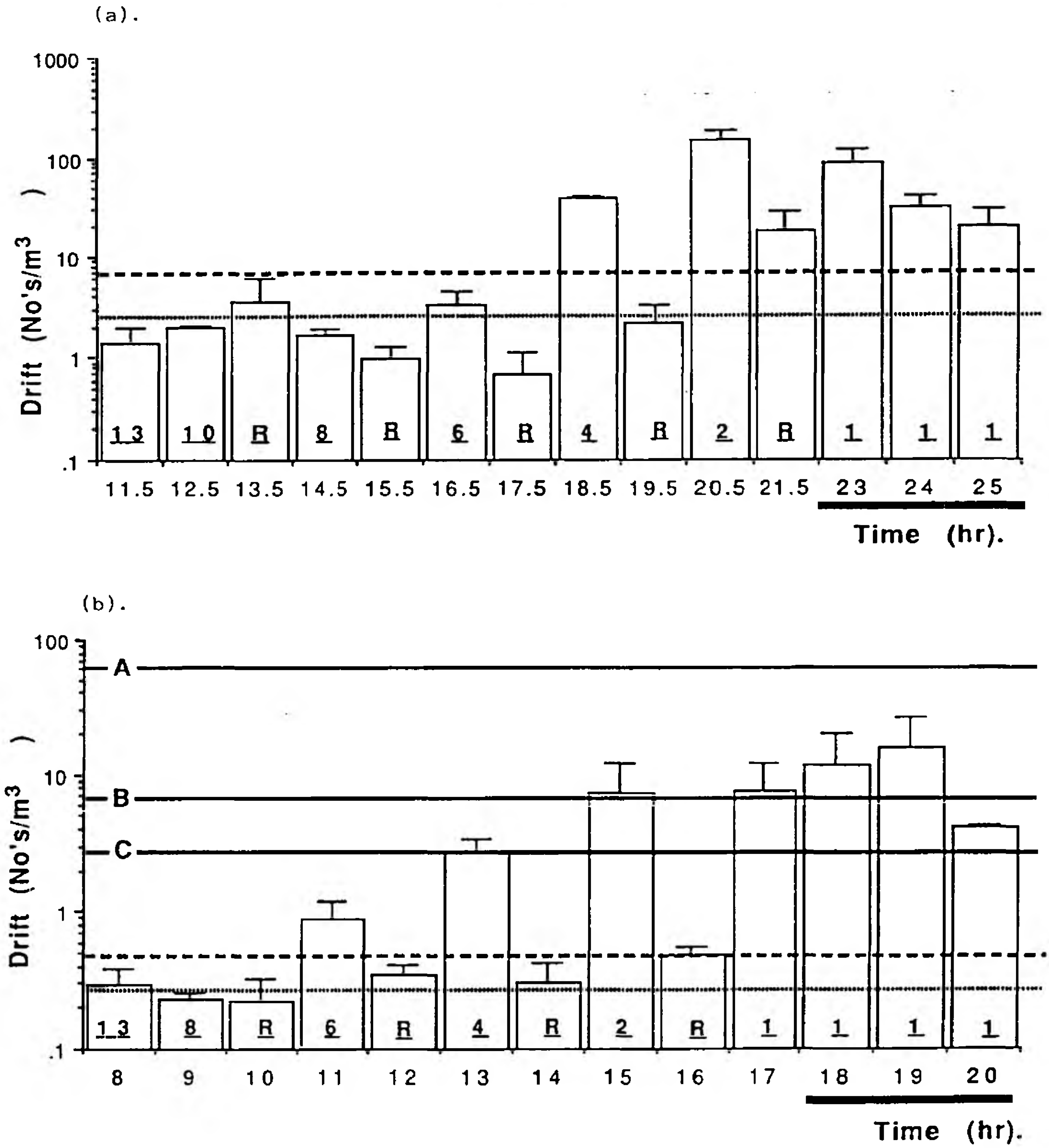


Figure 9.5 Invertebrate drift response to stepwise reductions in dissolved oxygen in July (a) and November (b) showing mean background drift (......) and mean + 3sd*(– –). (b) also gives the July drift density at 4mg/l.D.O. (—A—) and the July mean background density with mean + 3sd (—C— and —B—).

*Note: 3sd= three standard deviations (equivalent to probabilty of 0.001 that deviation from background is by chance).

b)   Natural benthic macroinvertebrate communities

Low dissolved oxygen episodes simulated in the Mounton Brook in July of 1988 (Episode 1) and 1989 (Episode 4) resulted in severe disturbances to the benthic fauna. However, recovery was rapid and pre-treatment conditions were recorded within two months (Figures 9.6a-c). A similar pattern was also recorded by quantitative sampling following Episode 3.

Recolonisation of benthic invertebrates following Episode 4 (DO depletion) was principally by downstream movement (44.9%). Within substrate migration, upstream movements and aerial sources contributed 31.5%, 15.6% and 7.9%, respectively. Differences between pathways were also found for major groups of invertebrates (Table 9.3, Figure 9.7), however, all four sources were equally significant for the Chironomidae.

c)   Seasonal differences in impact

A comparison of benthic invertebrate responses recorded during Episodes 4 and 6 (July and November) in the same stream (Mounton Brook), showed clear seasonal differences. Invertebrate drift was normally extremely low in November (<0.3 animals m^{-3}) compared with July (3 animals m^{-3}). Drift at this time increased above background when the DO, was reduced to 6 mg l^{-1}; however the density was approximately 2% of the July figure at 4 mg l^{-1} DO (Figure 9.5b). The high drift density at 4 mg l^{-1} DO, which occurred in July, was not achieved in the November experiment even after 3 hours exposure to 1 mg l^{-1} DO despite comparable invertebrate abundances at both times of year. Consequently, the impact on the benthos in November was minimal.

Ammonia

a)   Invertebrate drift and *in situ* toxicity tests

The drift response of macroinvertebrates to an attenuated pulse of ammonia (Episode 2, Figure 9.8a) was similar in scale to the nocturnal peak (Figure 9.8b). Caged fish were killed rapidly and, as expected, salmonids were

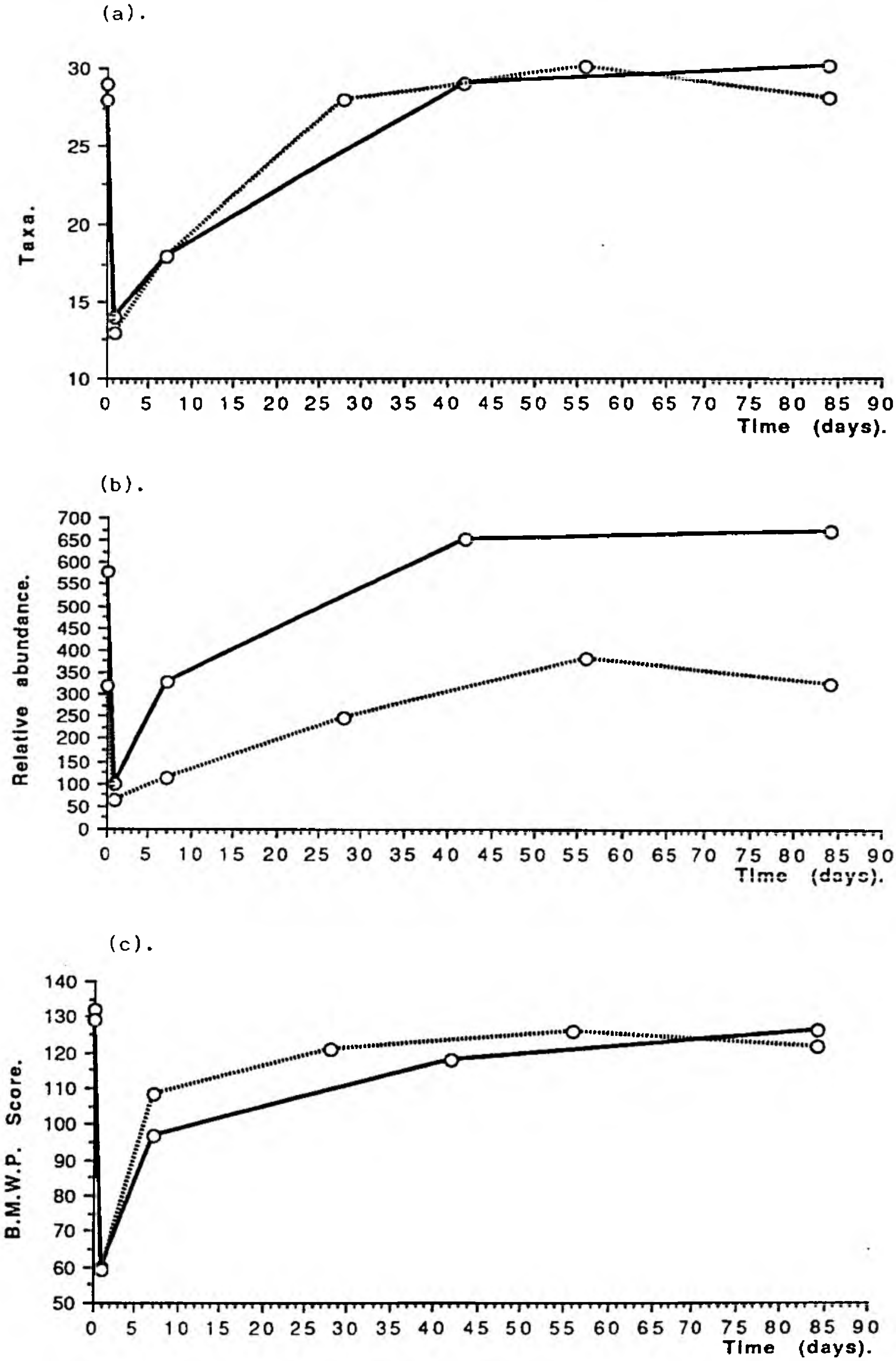


Figure 9.6 Recovery of taxa (a), abundance (b) and biotic score (c) at the Mounton Brook, July 1988 (—O—) and 1989 (··O··) Episodes 1 and 4, respectively.

Table 9.2 - Spectrum of sensitivities of caged fish and invertebrates exposed during episodes 1 and 2.

Episode 1: Oxygen		Episode 2: Ammonia	
Fish			
0+ _Salmo trutta_	I ^	0+ _Salmo trutta_	
1+ _Salmo salar_	N	0+ _Salmo salar_	
Cottus gobio	C	_Neomacheilus barbatulus_	
Neomacheilus barbatulus	R	_Cottus gobio_	
	E	_Gasterosteus aculeatus_ (parasitised	
	A	with _Hydrodactylus_ sp.)	
	S	_Gasterosteus aculeatus_ (unparasited)	
	I	_Cyprinus carpio_	
	N		
	G		
Invertebrates			
	S		
Dinocras cephalotes	E	_Polycelis tenuis_	
Gammarus pulex	N	_Gammarus pulex_	
(parasitised with	S		
Pomphorhnychus laevis)	I	_Physa fontinalis_	
Chironomus riparius**	T	_Ephemera danica_	**
Rhyacophila dorsalis	I	_Dinocras cephalotes_	
_Gammarus pulex_*	V	_Chironomus riparius_	
(unparasitised)	I		
_Asellus aquaticus_*	T	_Hydropsyche angustipennis_	
Chironomus riparius**	Y	_Asellus aquaticus_	

* no difference in sensitivity
** no mortalities

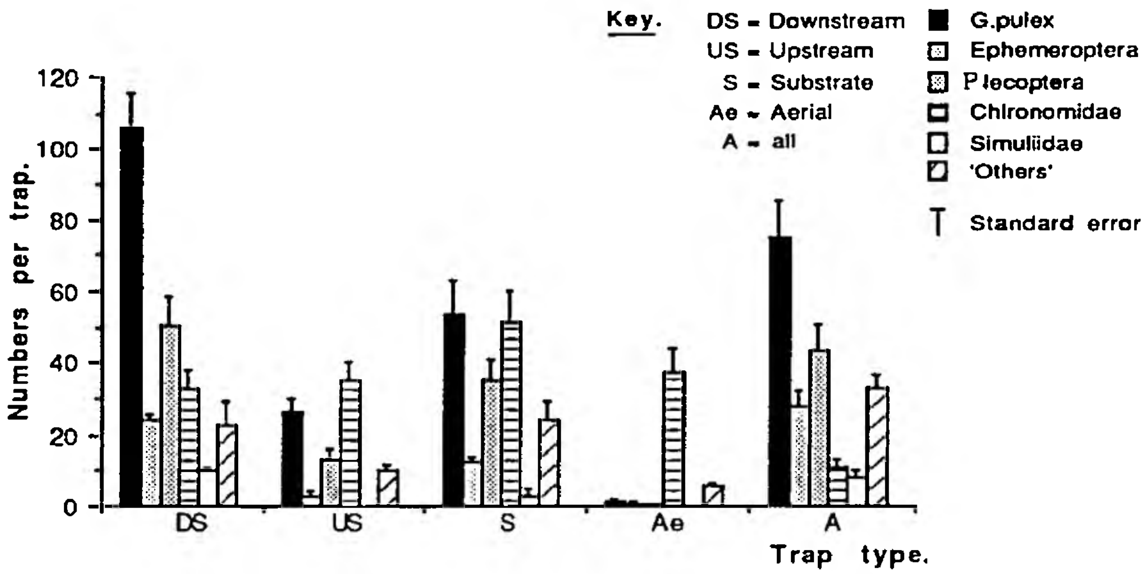


Figure 9.7 Relative importance of recovery mechanisms/pathways for major groups of invertebrates.

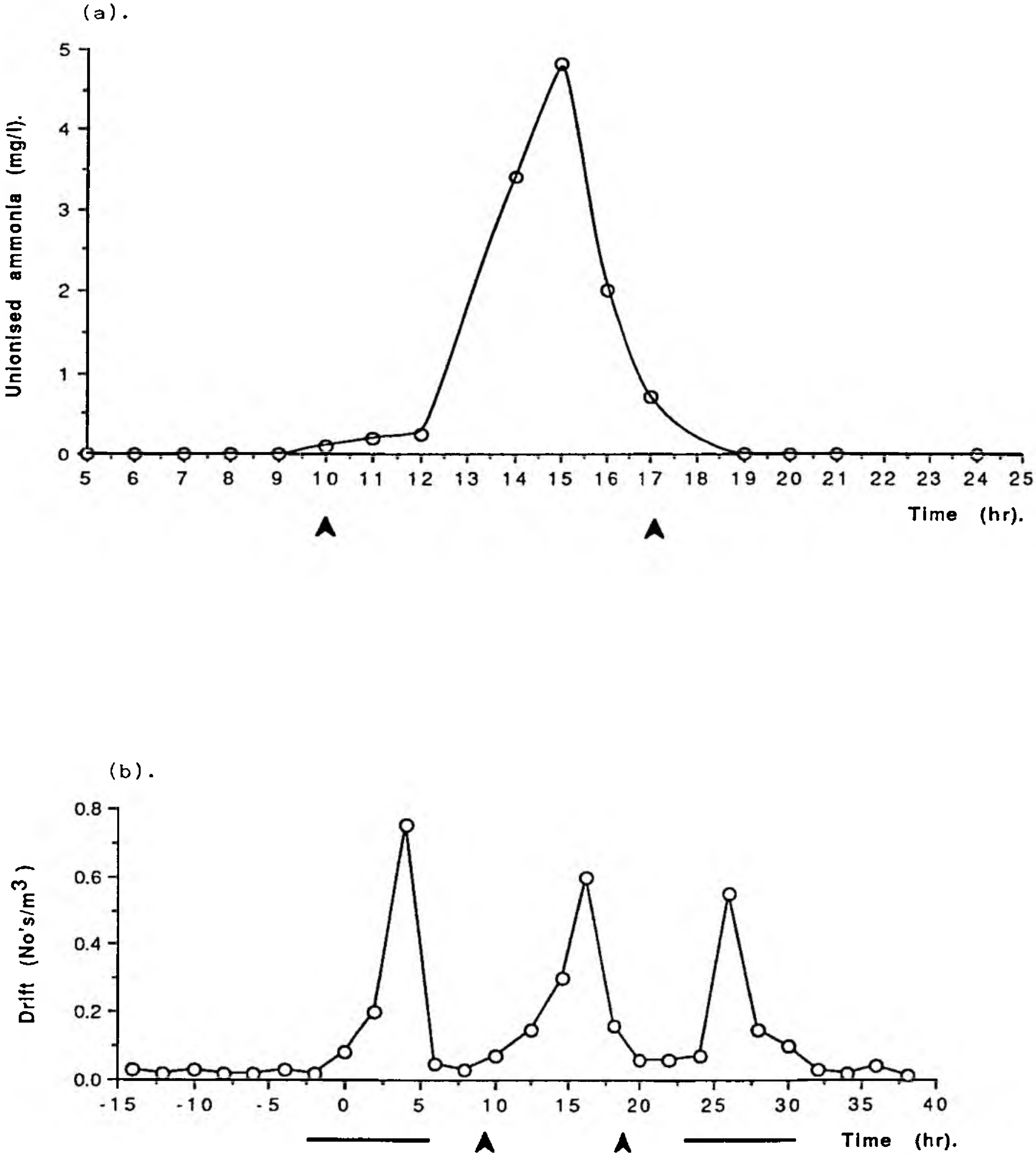


Figure 9.8 Simulation of an attenuated pulse of unionised ammonia (a) and drift response (b).

the most sensitive (Table 9.2). There were no mortalities in caged invertebrates, but behavioural changes were observed in: *Polycelis tenuis* (contraction), *Physa fontinalis* (vertical migration from water) and *G. pulex* (precopula disruption). There were no perceptible changes in the benthos (Figures 9.9).

As expected, a prolonged pulse of ammonia (Episode 5), in the form of a square wave (Figures 9.10a and b), had a greater impact on the drift (Figures 9.11a and b), also causing precopular disruption and mortality in *G. pulex.* The presence of semi-natural substrate (sand and stones) in the tanks holding *G. pulex* significantly reduced mortality ($p<0.05$, ANCOVA, Sokal and Rohlf 1981). This was later shown to be affected by water velocity in laboratory experiments (Thomas *et al* in press). In a second treatment zone (100 m downstream), attenuation of pH and unionised ammonia concentrations occurred (Figures 9a and b). Here, the dissolved oxygen concentration was artificially lowered to 5 mg l^{-1} for 12 hours (Figure 9.10c). Invertebrate drift in this zone also increased (Figure 9.11c) but was significantly lower than in the first treatment zone. Disruption of precopula in *G. pulex* was also observed to be significantly less than in the first treatment zone ($p<0.05$ Litchfield 1949).

b) Natural benthic macroinvertebrate communities

Although increases in drift and some mortalities were recorded, the ammonia episodes had a relatively low impact on benthic invertebrates. Reductions in taxonomic diversity and biotic score (BMWP) were not observed (Figures 9.9a and c) however, a decrease in relative abundance was apparent (Figure 9.9b) following Episode 5.

Sulphide

In Episode 8 the total sulphide concentration was raised to 5 mg l^{-1} (Figure 9.12) and drift increased to an order of magnitude above background (Figure 9.13). Disruption of precopula in *G. pulex* occurred and males were observed guarding dead females. In a downstream zone (50 m) in which total sulphide concentrations had reduced to 2 mg l^{-1} groups of *G. pulex* were exposed

Table 9.3 - Relative importance of biological recovery mechanisms.

GROUP	NUMERICAL ORDER OF PATHWAYS
G. pulex	DS > S > U
Ephemeroptera	DS > S > U > Ae
Plecoptera	DS > S > U > Ae
Chironomidae	DS // S // U // Ae
Simuliidae	DS > S*
All groups	DS > S > U > Ae

* = No other mechanisms involved
DS = downstream
S = within substrate
U = upstream
Ae = aerial
// = not significant
> = significantly greater than ($p < 0.05$), two sample t test

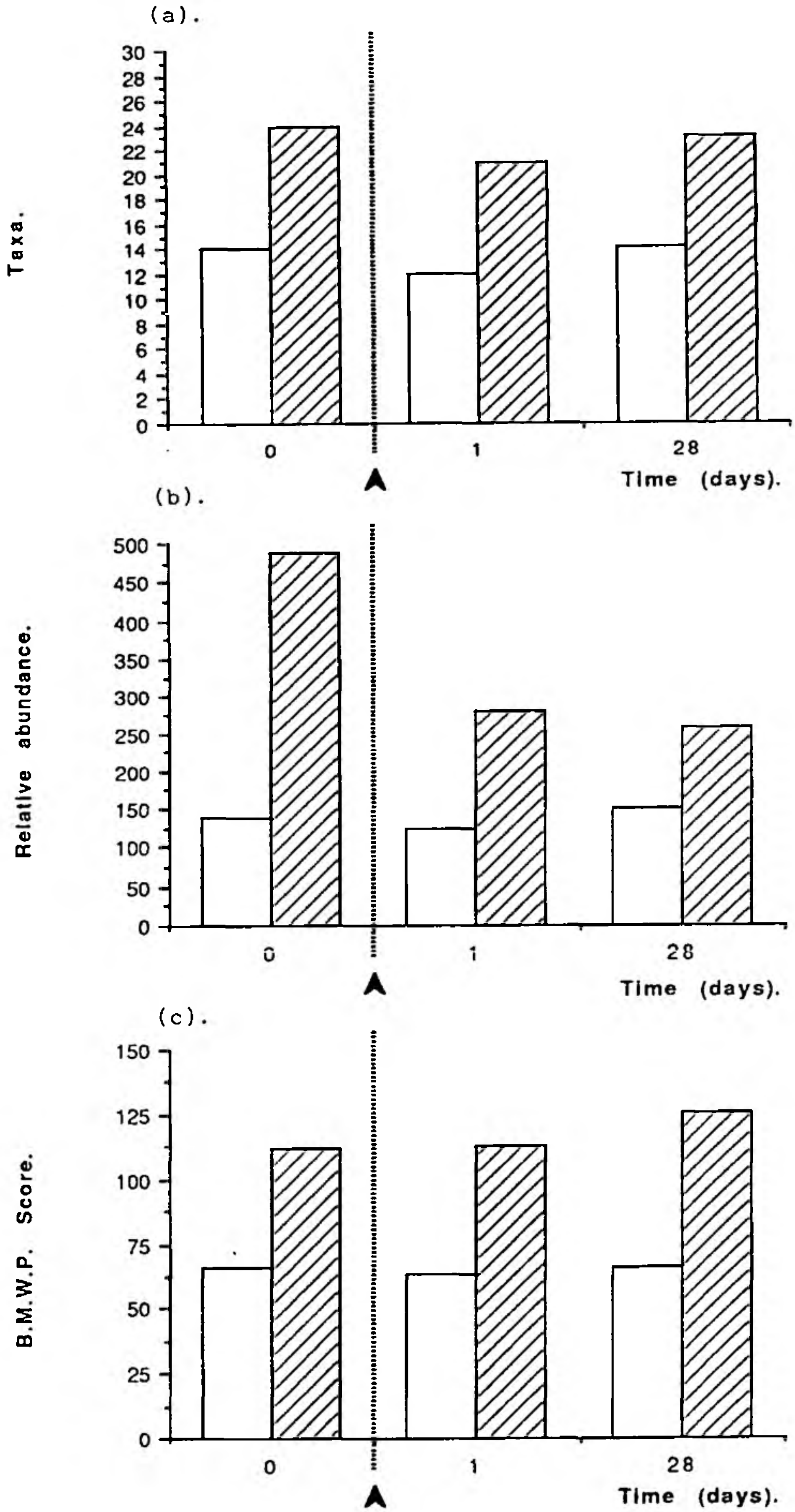
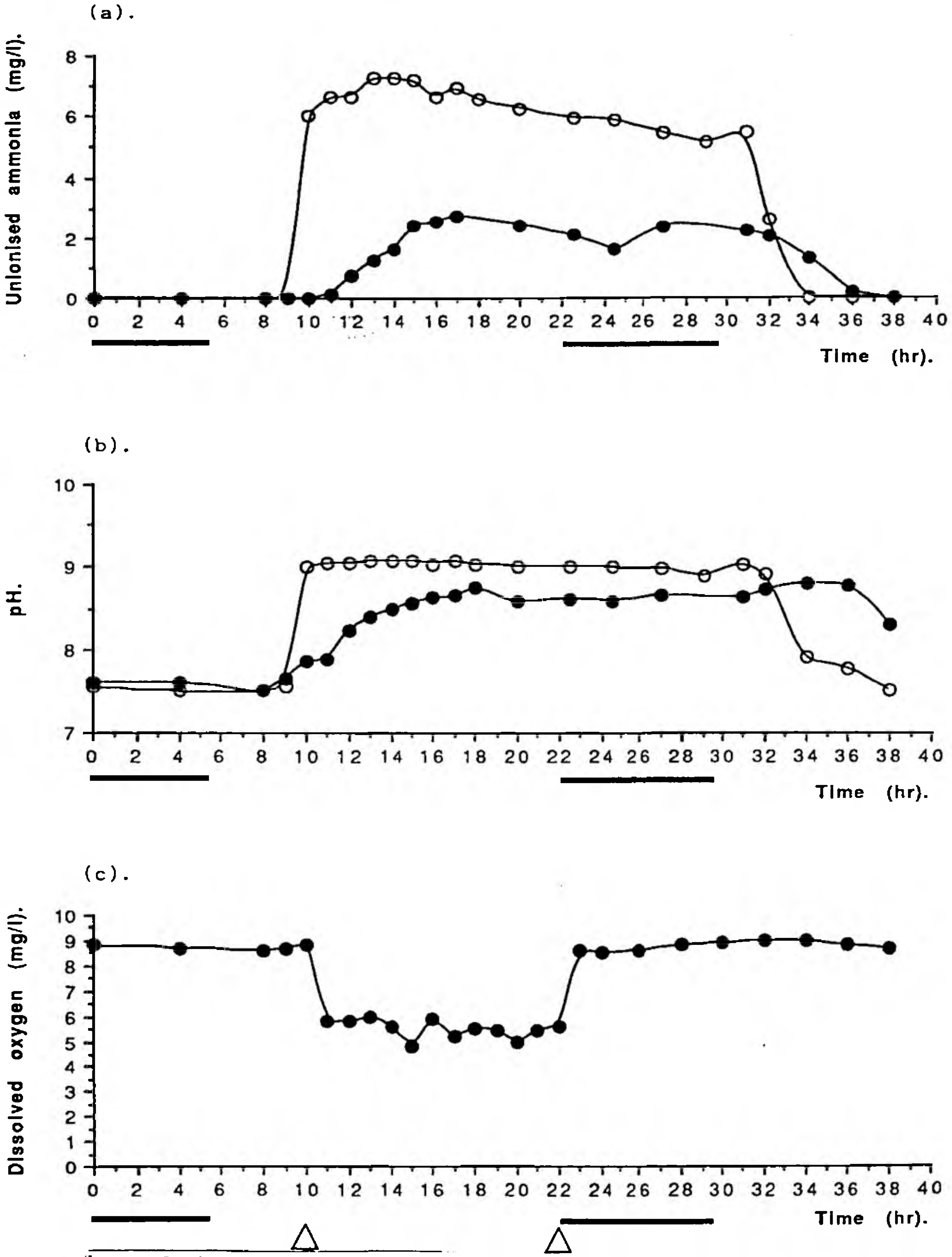


Figure 9.9 Impact of ammonia episodes upon benthic invertebrates, Nant Dowlais 1988 (□) and 1989 (▨). ▲ denotes the episode.



istry during Episode 5, Tributary Nant Dowlais, 1989.

Figure 9.10 Unionised ammonia (a), pH (b) and dissolved oxygen (c) in the first treatment zone (O) and second treatment zone (●).

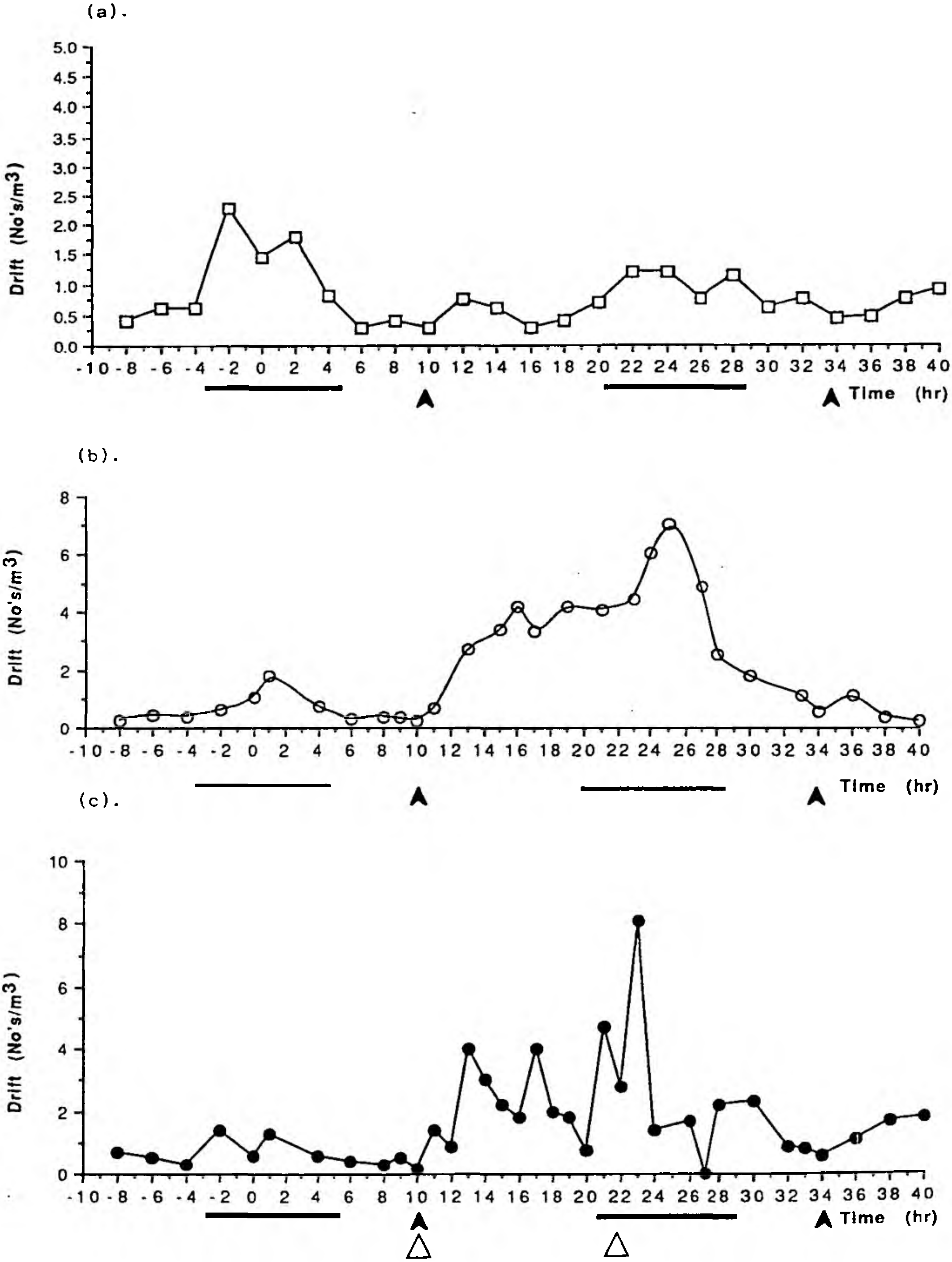


Figure 9.11 Invertebrate drift response during episode 5 in the reference (□) 1st treatment (O) and second treatment (●) zones. △ △ denotes duration of reduced D.O.

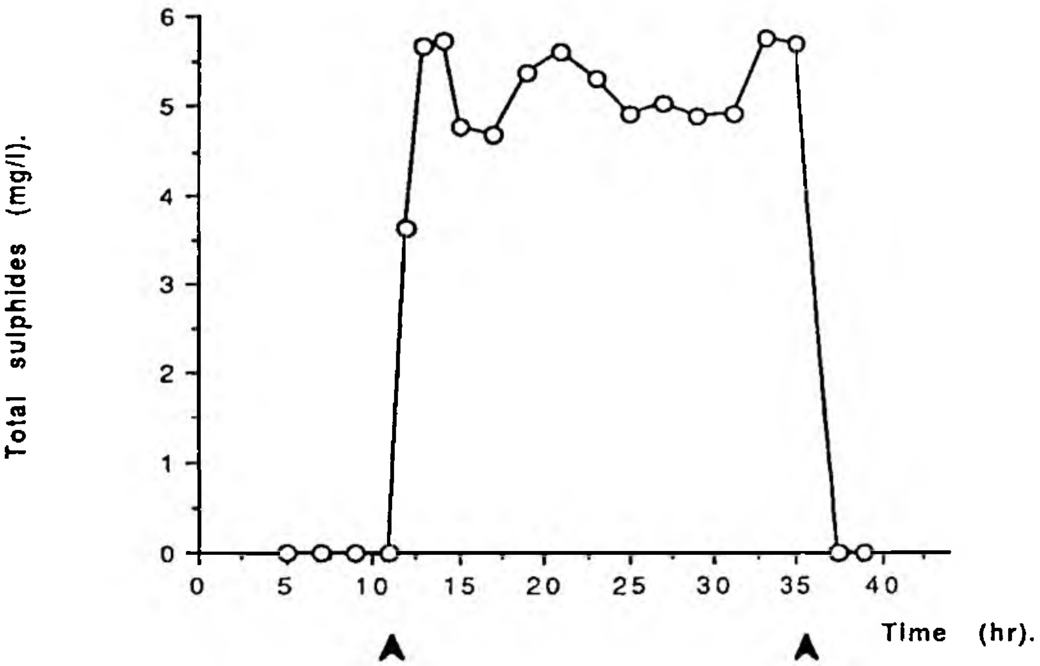


Figure 9.12 Surface water concentrations of total sulphides in the 1st treatment zone, episode 8, Tributary Nant Dowlais.

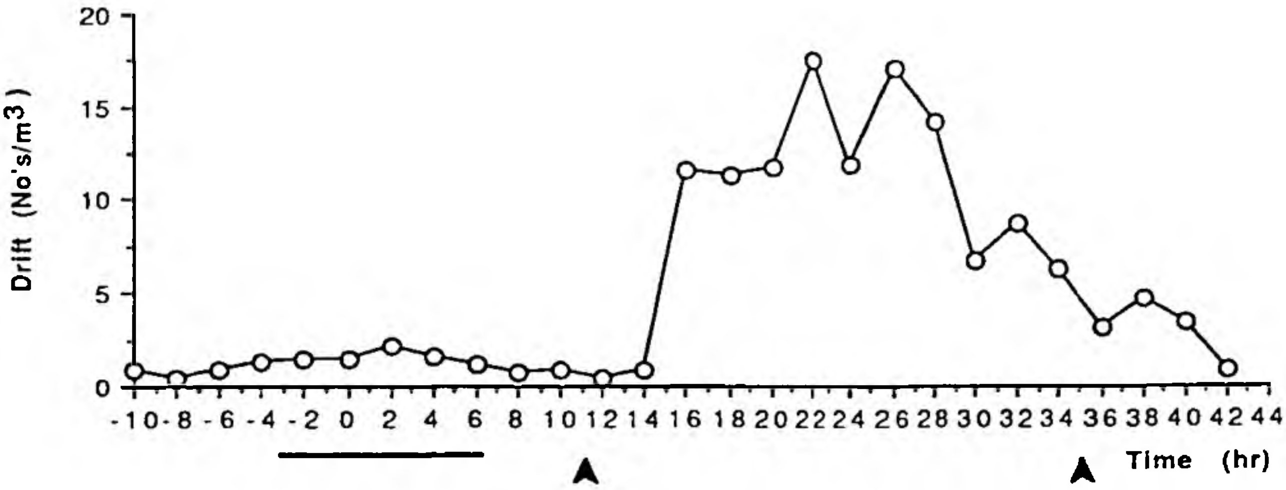


Figure 9.13 Drift response to increased sulphides.

to different water velocities, in standardised cages containing substrate (sand and stones). A significant relationship between ambient water velocity and mortality due to sulphide toxicity was found (Figure 9.14). No mortality was recorded in *G. pulex* held under similar flow conditions in the reference zone.

Interactions of pollutant episodes with the stream bed

Physico-chemical processes (dispersion, sorption) in aquatic systems may cause attenuation (weakening) of pollutant activity. The longitudinal dispersion of tracers and pollutants has been extensively studied (McBride and Rutherford 1984, Whitehead *et al* 1986). Mass transfer of solutes between surface and interstitial water in sediments has been studied in the laboratory (Thidbodeaux and Boyle 1987, Nagaoka and Ohgaki 1990) and in the field (Bencala 1983, Bencala *et al* 1984, Bencala 1984). However, few studies have described the distribution of a pollutant or solute in relation to depth and duration within the substrate (e.g. Munn and Meyer 1988, Edwards *et al.,* 1991).

In the current study the attenuation of pollutant episodes and the penetration of a solute tracer (sodium chloride) into the bed of a small stream were investigated. Observations of the penetration of the tracer were related to interstitial seepage rates.

The locations of sampling sites for surface and interstitial water (10 and 30 cm) at the Tributary Nant Dowlais are indicated in Figure 9.15. During the solute tracer experiment only Site 1 (riffle) showed significant penetration of sodium into the bed to a depth of 10 cm (Sites 1-6, Figure 9.16). Site 2 was in the margin, one metre from Site 1; however, there was only a slight increase in concentration in the substrate at 10 cm. The solute did not penetrate to 30 cm at any of the sites investigated. Sediments in the stream are sandy and very compacted, and as expected the seepage rates (Carling and Boole 1986) were extremely low at all the sampling points. Notably, Site 1 showed the highest seepage velocity at 10 cm (Figure 9.16).

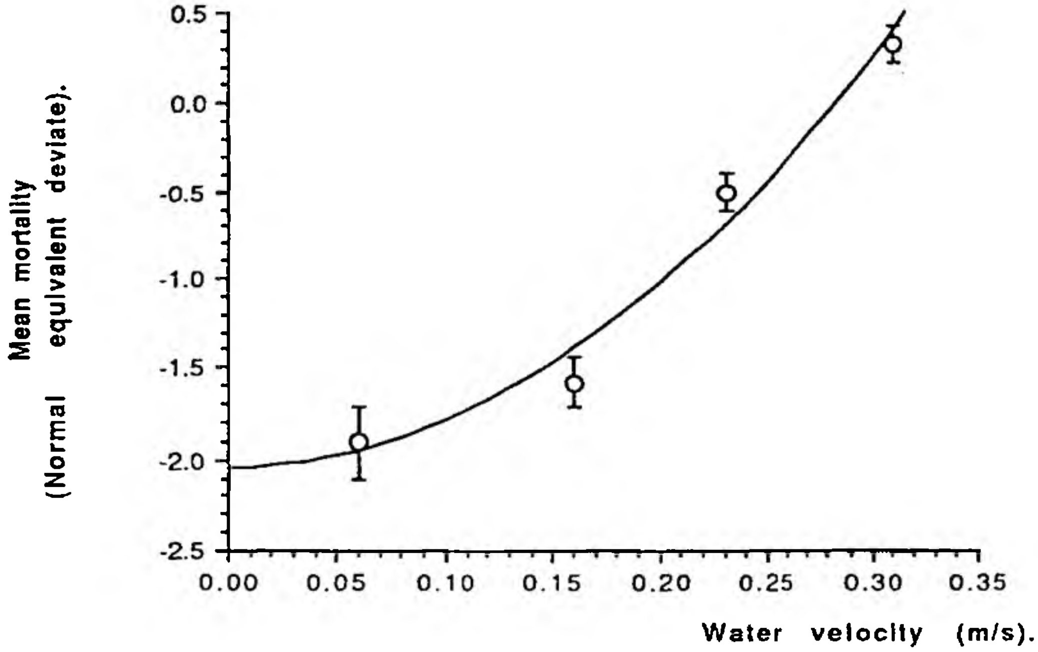


Figure 9.14 . Mortality response of G.pulex to sulphides in relation to water velocity ($r^2 = 0.974$; $p<0.001$). I = least significant differences between means of transformed (normal equivalent deviate) mortality curves (ANCOVA, Sokal and Rholf, 1981).

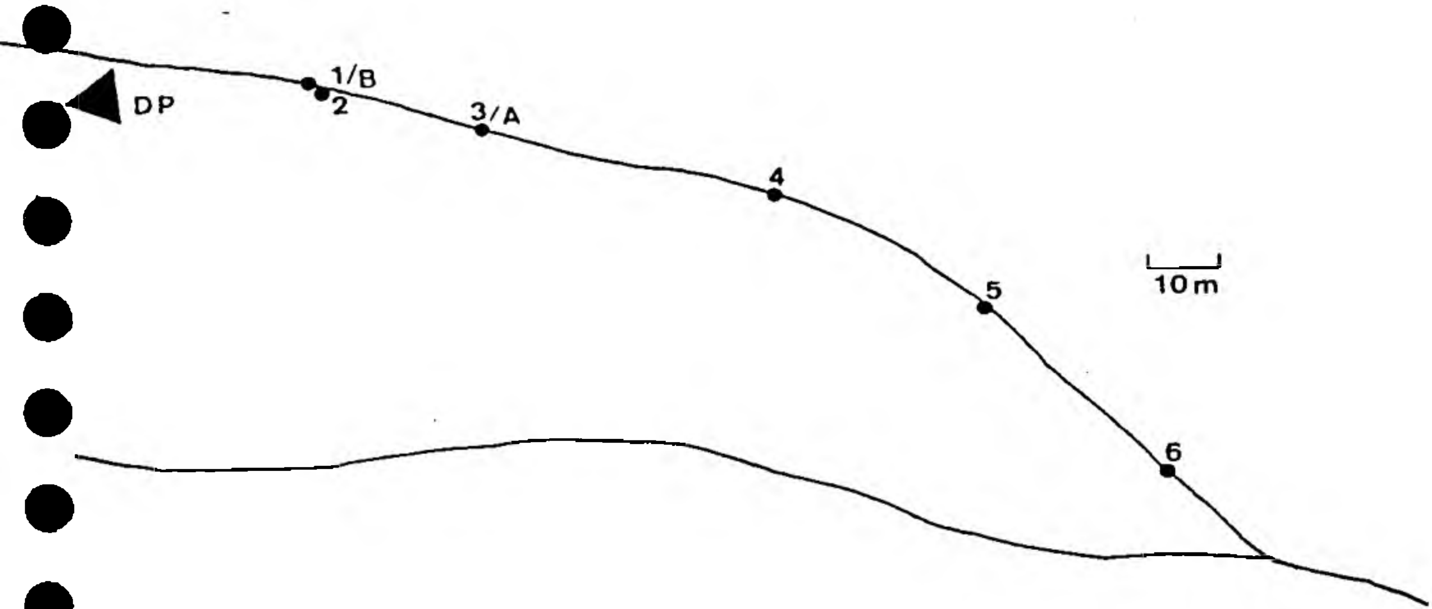
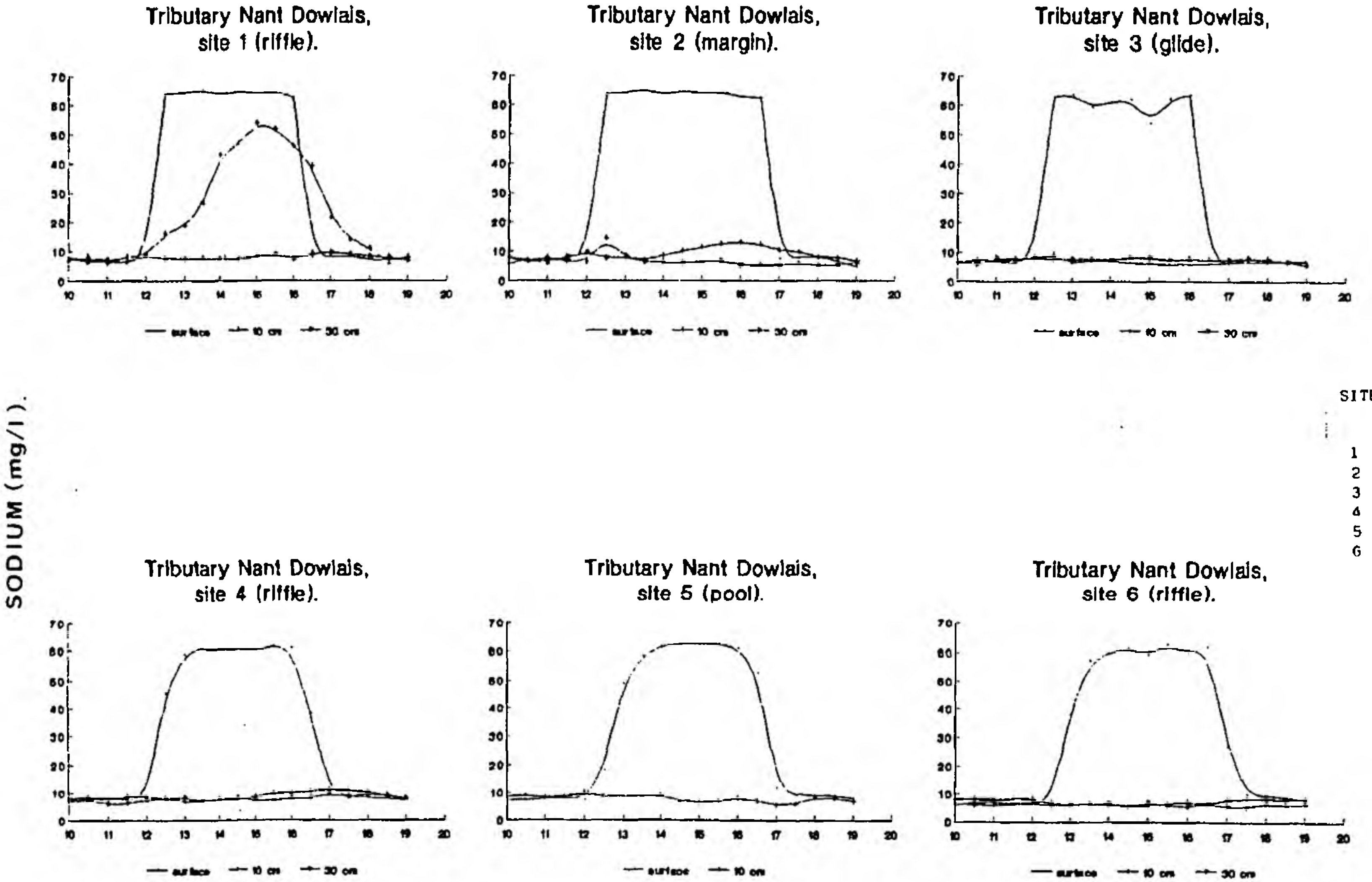


Figure 9.15 Locations of sampling points for attenuation studies, Tributary Nant Dowlais. Sites 1-6 refer to sodium tracer experiment, A to episode 5 (ammonia) and B to episode 8 (sulphide). DP= dosing point.



SEEPAGE VELOCITIES (cm.hr^{-1}).

SITE	DEPTH IN SUBSTRATE (cm).	
	10	30
1	2.56	0.22
2	0.25	0.11
3	0.11	0.07
4	0.16	0.04
5	0.08	0.001
6	0.13	0.09

Figure 9.16 Sodium time-concentration profiles during the sodium chloride tracer experiment.

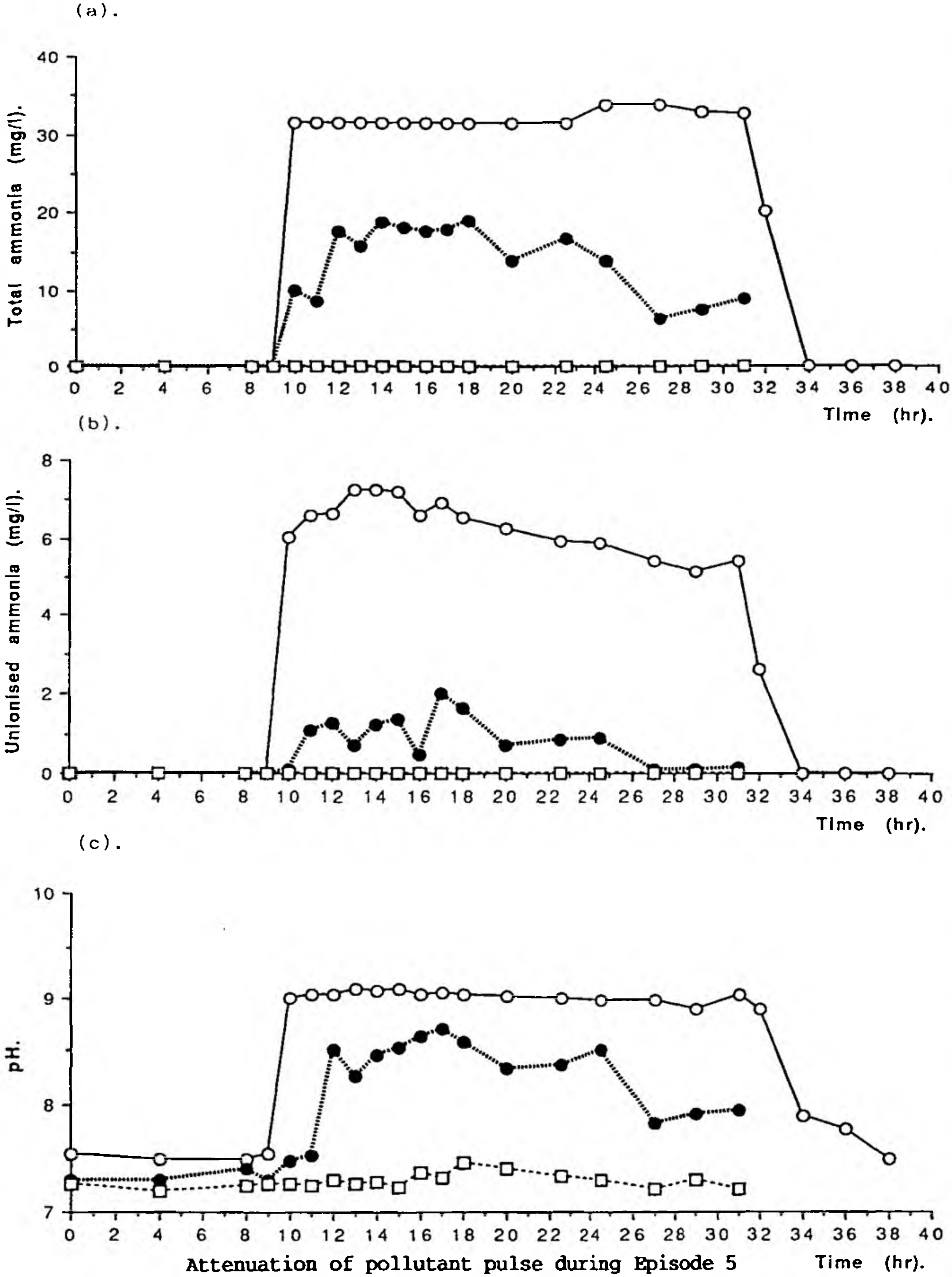
The stream bed at the Tributary Nant Dowlais also attenuated unionised ammonia concentrations (at Site A in Figure 9.17) and total sulphides (at Site B in Figure 9.18). Apparent anomalies between sodium and ammonia penetration at Site 3/A were probably due to dramatic changes in sediment and flow patterns (riffle -> 'glide') at this point, brought about by flooding in the winter of 1989. During Episode 3 (Yazor Brook), dissolved oxygen concentrations at 10 cm and 30 cm in the substrate (riffle) closely followed the surface profile (Figure 9.19).

9.3.2 Laboratory investigations

The laboratory stream confirmed downstream movement as a response to reduced dissolved oxygen and increased ammonia. Migration of animals into the substrate was observed soon after the onset of dosing. Figure 9.20 shows the migration pattern for *G. pulex* during an ammonia episode. There was clearly an increase in the total number of *G. pulex* in the substrate in response to increased ammonia. Similar patterns occurred with *G. pulex* and *E. venosus* with dissolved oxygen reductions.

9.4 Conclusions

Field simulations of pollution episodes were a useful tool to provide data on the biological effects of intermittent low DO, ammonia and sulphide, which are major components of most organic wastes such as those emanating from farms. 'Real' slurry or silage pollution events often result in fish mortalities and cases have been recorded where angling clubs have suspended fishing following serious pollution (Howells and Merriman, 1986). As expected, caged fish were highly sensitive to reduced DO and elevated ammonia concentrations during brief exposure.



Attenuation of pollutant pulse during Episode 5

Figure 9.17 Total ammonia (a), unionised ammonia (b) and pH (c) in surface (O) and interstitial water; 10cm= ● ,30cm= □ .

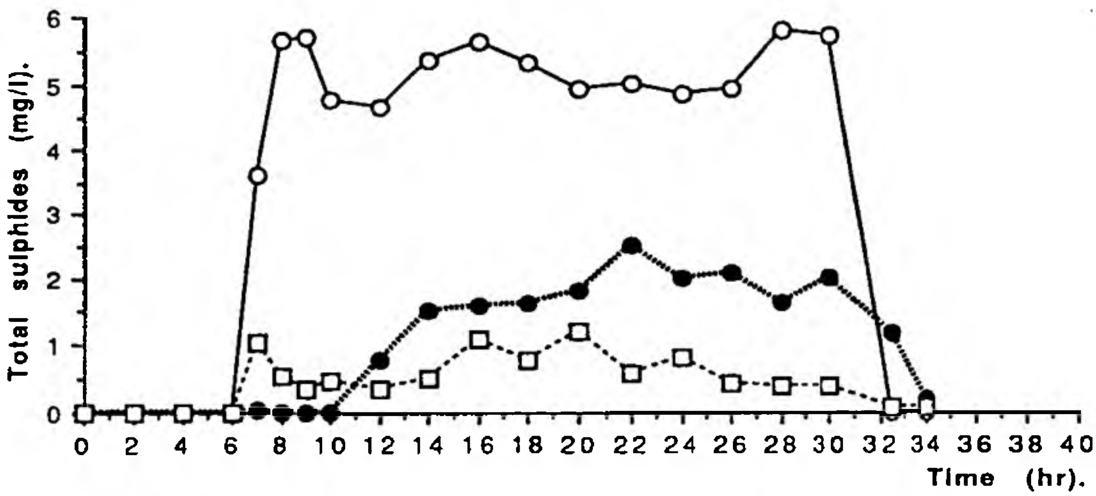


Figure 9.18 Attenuation of total sulphides during episode 8; surface water (O), 10cm= ● ,30cm= □ .

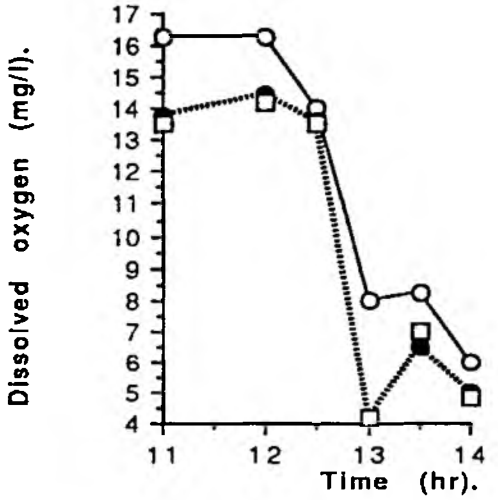


Figure 9.19 Penetration of reduced dissolved oxygen into the bed of the Yazor Brook (episode 3). Surface water (—O—) ; 10 and 30cm (●·······□).

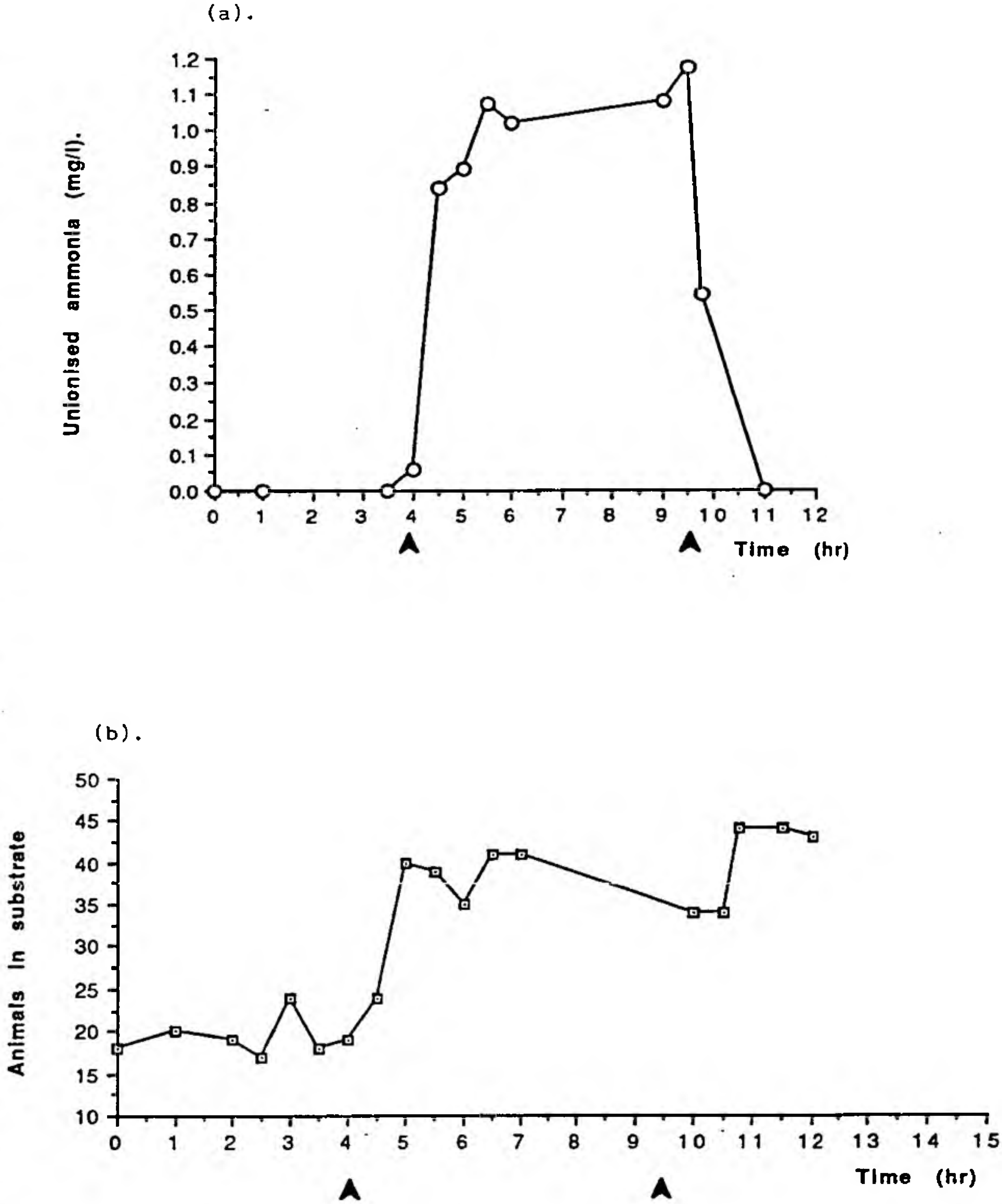


Figure 9.20 Simulated episode of ammonia (a) and downward vertical migrational response of G.pulex (b).

Studies with caged invertebrates and natural populations demonstrated a wide spectrum of responses, with reduced DO causing the most severe disturbances even when compared with prolonged episodes of sulphide or ammonia. Brief exposures at or below 4 mg l^{-1} DO in the summer caused partial denudation of benthic invertebrates in contrast with the winter study when the response was considerably reduced. However, recovery occurred rapidly (<2 months) and followed a clear 'exponential' pattern despite differences in sites, sampling techniques and year of study (1988, 1989). The pattern is consistent with the MacArthur - Wilson model (1963) for island fauna and similar curves have been found for aquatic animals colonising implanted substrates (Dickson and Cairns 1972, Sheldon 1977). Clearly, recovery following short-term disturbances can be rapid providing there are sources of organisms available to recolonise. In the current study, downstream movement was the principal pathway for recovery, a similar result to that described in previous studies (e.g. Williams and Hynes 1976); however, other pathways were important, particularly for certain groups such as the Chironomidae, and should not be ignored.

Sources of animals from within the substrate also played a major role in the recolonisation process, furthermore, the penetration of pollutants into stream beds demonstrated extensive spatial patchiness. Large areas of the stream bed could provide a refuge from episodes, especially since some animals demonstrate an ability to move deeper in response to events. These animals could survive to recolonise surface layers of the stream bed. However, care should be taken when extrapolating the laboratory observation of vertical migration to real streams and further work is needed to satisfactorily investigate this phenomenon of invertebrate behaviour under field conditions.

Pollution incidents, particularly those due to farm wastes and storm-sewer overflows, rarely occur in isolation and repeated events may not allow complete recovery of the system. Further work is required to increase our understanding of how animals, in terms of both populations and species assemblages, respond to repeated challenges by short-term episodes.

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