SCHOOL OF CONSERVATION SCIENCES

HABITAT QUALITY IN CONSTRUCTED WETLANDS AS PART OF A SUSTAINABLE URBAN DRAINAGE SYSTEM (SUDS)

A DISSERTATION SUBMITTED AS PART OF THE REQUIREMENTS FOR THE MSc ENVIRONMENTAL QUALITY

By

R. AQUILINA

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List of Abbreviations

ASPT	Average Score Per Taxa
BMWP	Biological Monitoring Working Party
BOD	Biological Oxygen Demand
DO	Dissolved Oxygen
EQI	Ecological Quality Index
FBA	Freshwater Biological Association
IBI	Index of Biotic Integrity
IMS	Industrial methylated spirits
PCA	Principal Components Analysis
PSYM	Predictive System for Multimetrics
RIVPACS	River Invertebrate Prediction and Classification System
SSSI	Site of Special Scientific Interest
SUDS	Sustainable Urban Drainage Systems
TRS	Trophic Ranking Score

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Abstract

Artificial modifications to an urban stream, Bourne Stream, Dorset, as part of a Sustainable Urban Drainage System (SUDS), were investigated for their contribution to habitat quality as measured by macroinvertebrate and macrophyte diversity. The specific objectives were to investigate the environmental conditions prevailing in the three major freshwater habitats – stream, ponds and SUDS; to investigate macroinvertebrate and macrophyte species diversity within these habitats; to investigate the relationship between the environmental conditions and species diversity; to compare the stream quality over time in order to assess any changes and to assess the habitats in terms of conservation value. Diversity indices, multivariate statistics and two nationally recognised predictive methodologies (RIVPACS and PSYM) were used to categorise the habitats.

Species richness and diversity were not found to differ significantly between the stream itself, the existing ponds and the constructed wetlands. Habitat quality as measured by macroinvertebrate diversity was related to the pH of the water and the emergent vegetation. RIVPACS analysis showed the condition of the stream to be grade B, good, in the upper reaches. This shows an improvement in the uppermost sample point immediately below the constructed wetlands compared with previous published analysis. The lower sample point showed a slight degradation, factors which may account for this are discussed. It is concluded that the conservation value of constructed wetlands can be high, this site having several Notable water beetle species and uncommon dragonflies.

1.Introduction

Charles Elton records that he was attracted to ponds once because he thought that the communities in them must be determined by a limited set of factors, but repelled when he had found as many distinct communities as he had visited ponds (Macan, 1974). This recognition of the complexity of factors affecting the constitution of the freshwater community has, if anything, increased in the intervening years (Jeffries, 2003). There is now greater understanding of the role of biological determinants of community and a distinct move towards an holistic view which includes both environmental and biological determinants (Carpenter, 1998). Thus an integrated picture emerges of a complex of factors, chemical, physical and biological, that influence freshwater community structure, where even chance has a role (Jeffries, 1988). Overlaying this background of complex theoretical determinants is the need to extract simple practical knowledge in order to address the increasingly urgent needs of wetland conservation and remediation, and the desire to create sustainable urban drainage systems to tackle the problems of pollution and excessive variability in hydrological events.

The aim was to assess the artificial modifications (wetland and lagoons) built as part of a sustainable urban drainage system (SUDS) in terms of whether they are contributing to habitat quality within the Bourne Valley SSSI and whether they are degraded by their function in remediating pollution. The objectives of this study were to:

- investigate the environmental conditions prevailing in the three major freshwater habitats – stream, ponds and SUDS
- investigate macroinvertebrate species diversity in the three habitats and compare them

- investigate macrophyte species diversity within the ponds and SUDS and compare them
- investigate the relationship between the environmental conditions and species diversity
- compare the stream quality with a previously published survey (Armitage, *et al*., 1995) in order to assess any changes
- assess the habitats in terms of conservation value.

These assessments were based on biological indices as opposed to previous chemical investigation (Fleet, 2002). Habitat quality was assessed by measuring macroinvertebrate and macrophyte species richness and diversity. The hypothesis to be proven was that the role of the SUDS, in capturing polluted waters and ameliorating them before discharge downstream, would adversely affect the diversity of the SUDS in comparison to the existing ponds. The stream would be similarly affected at its upper reaches in comparison to downstream.

1.1 Sustainable Urban Drainage Systems

One of the consequences of the built environment is the effect that roads and buildings have upon the local hydrology. Impermeable roads and pavements, hard standing and roofing have traditionally been supplied with drainage systems designed to remove water from the site as quickly as possible. This typically means drainage pipes leading to watercourses. As a consequence of the high volume of water that then has to be handled by the watercourse, they in turn are often modified to transport water away quickly and prevent local flooding. This usually means straightening out bends, embanking the sides and culverting, all of which reduce habitat diversity within the stream (Scottish Executive, 2001). The result of such schemes can be to carry such a large volume of water at such speed that the receiving water courses downstream are overwhelmed and flooding occurs. Recognition of the problems of such an approach has been widespread and public in recent years as increasing rainfall and discharges have led to severe flooding in parts of the UK (Environment Agency, 2003).

Another consequence of the traditional drainage approach is that ground water levels can fall because they are not being replenished by sufficient volume. This in turn can lead to problems with reduced flow in water courses in dry weather and even problems with water supply in areas where ground water abstraction makes up a portion of the domestic water supply. Furthermore the quality of runoff water from roads and car parks and industrial estates in particular, can be severely degraded due to the presence of spilt hydrocarbons, heavy metals and other chemicals such as road salt (Sriyaraj and Shutes, 2001). This is especially severe after a dry period, when the first flush of heavy rain tends to wash off a high concentration of these chemicals. The Bourne Stream suffers in particular from this form of storm water runoff (Fleet, 2002).

As a result of these problems an alternative approach is increasingly being adopted by planners and designers, which is designated Sustainable Urban Drainage Systems (SUDS). These are structures designed to emulate the natural pattern of drainage by capturing and holding back water and releasing it more evenly. SUDS have three objectives:

- To control quantity of runoff from a development
- To improve the quality of the runoff
- To enhance the nature conservation, landscape and amenity value of the site and its surroundings (CIRIA, 2001).

SUDS should deal with runoff as close to its source as possible and balance the three objectives – they are not merely a flood prevention tool. SUDS may be very cost effective and even lead to cost savings by reducing the outlay on building traditional drainage connections. In recognition of the success of SUDS, many Local Plans insist that SUDS are incorporated into development proposals.

There are four families of drainage technique commonly in use: permeable surfaces and filter drains, filter strips and swales, infiltration devices and constructed wetlands. It is the last of these that features in this study. However it is usual to include combinations of techniques when a new development is designed to incorporate SUDS. Each contributes to the overall objective. Source control and prevention techniques such as permeable paving and infiltration trenches allow water to drain through and into subsoil, thereby reducing the amount requiring management. Permeable conveyance systems such as filter (sometimes known as French) drains and swales provide temporary storage for storm water, reducing peak flows and allowing reduction in volume through infiltration to subsoil and evapotranspiration. There are positive effects upon water quality through filtration, absorption and microbial degradation. Passive treatment systems such as constructed wetlands are primarily concerned with water quality but they also contribute to flow control and can have a high conservation, wildlife and amenity value (Environment Agency, 2003). Passive treatment systems are primarily designed to allow natural chemical and microbial interactions to remove pollutants from runoff and wastewater. They can range from filter strips to large-scale wetlands:

- Filter strips are vegetated strips of land which act as primary filters to runoff, slowing flow, depositing sediment and removing some pollutants, before the water reaches a water course. Removal rates for Biological Oxygen Demand (BOD) (18%), ammonia and ammonium (28%), nitrate (11%), suspended solids (22%), phosphate (5%), faecal coliform bacteria (28%) and *E.coli* (22%) are reported (Cameron, *et al*., 2003)
- Detention basins are similar in function with solids settlement being the main aim. They will be dry outside storm periods and provide temporary retention for storm waters thus reducing peak flows.
- Retention ponds are designed to contain permanent standing water and to remove nutrients, pollutants and even coliform bacteria from water prior to discharge downstream. It is important that such ponds are large enough as a retention time of 20 days is considered the minimum for effective pollutant removal (CIRIA, 1997). Larger ponds also provide more amenity opportunities and require a sufficient catchment area to ensure a minimum water level during dry periods.
- Constructed wetlands are more effective at filtering and removing nutrients and pollutants than retention ponds but are more problematic as amenity areas. Best practice incorporates pre-treatment with filter strips, detention basins and retention ponds with the wetland to prevent excessive sedimentation leading to drying up.

In a combination of constructed wetland and retention pond, removal rates achieved were BOD (34%), ammonia and ammonium (52%), nitrate (37%), suspended solids (93%), phosphate (90%), faecal coliform bacteria (52%) and *E. coli* (58%) (Cameron, *et al*., 2003). A review of the published literature on the effectiveness of SUDS is given by Pratt (2001). The SUDS implemented on the Bourne Stream are of three types, a constructed wetland which acts to capture and retain the first flush and downstream a detention (siltation) pond and a series of retention ponds. More details are given in the description of the study area.

1.2 Measuring Freshwater Habitat Quality

The freshwater environment is of increasing conservation importance as the amount of wetland being lost has increased dramatically in the last few decades (Oldham and Swan, 1997). Hand in hand with this loss, there has been degradation of the surviving environment from pollution and urban encroachment (Linton and Goulder, 2000). The environment is itself of considerable diversity, ranging from temporary pools and winterbournes (a winter stream) which dry out completely each year, permanent ponds and streams, and larger water bodies, lakes and rivers, to a range of wetlands from seeps and flushes to marsh, bog, fen and swamp. This diversity of environments is matched by the diversity of life in them. For example there are about 4,000 species of aquatic macroinvertebrates in the British Isles (Maitland, 1977).

There is a requirement in the water industry for a simple index that reflects the ecological quality of a site (Mason, 2002). Whilst diversity indices are widespread in ecological research, they reflect taxon richness and evenness, but without a context to place the values in, it is difficult to interpret them. The development of biotic indices for freshwater has been driven by the need to provide a measure of species diversity based on the relative importance of those species in reflecting environmental degradation through pollution. The Saprobic Index (and its derivatives) are widely used in Europe, whilst the standard used by the Environmental Protection Agency in the USA is the Index of Biotic Integrity. The most widely used system in the UK is the Biological

Monitoring Working Party (BMWP) which is simpler than the above mentioned and combines taxon richness with sensitivity to pollution (Mason, 2002). A further refinement of this approach has been to use the physical and chemical attributes of a site to predict the macroinvertebrates that would be expected to occur there in the absence of any pollution. A comparison of the observed values with the expected gives ratios that can be used to express Ecological Quality Indices (EQI) reflecting how degraded a site is relative to its expected pristine condition. This forms the basis for RIVPACS developed by the Centre for Ecology and Hydrology (and its pond equivalent, PSYM, developed by Oxford Brookes University). The parameters input to the systems are discussed in more detail later. These biotic indices are valuable for encapsulating the state of a site and comparing sites. Diversity indices are well-known ecological tools that can be used to reflect the complexity of a community but care is needed in comparing site results.

The identification of freshwater invertebrates is problematic because the number of species encountered is likely to be high, ranging across a large number of families, many of which are not well-studied taxonomically and identification keys may only exist in specialist literature, be based on microscopic dissection and examination, and may not exist at all. Identification to family level overcomes most of these difficulties and still provides a valuable means of assessing community diversity. In this study, specimens were identified down to species level where keys were available and life history stage allowed. Instars younger than stage III in the Trichoptera and younger than the final two stadia in Odonata are insufficiently differentiated to distinguish between species or even genus, in some cases. No key was available for Chironomidae.or Coleoptera larvae, although the latter were identified to genus based on Nilsson (1996).

The identification guides and keys used are listed in a separate bibliographic appendix (Appendix 8).

1.3 Environmental factors

The environmental factors that have been found to be important determinants of community composition were reviewed and a number selected for measurement in this study. A review of the published literature for factors which had been investigated (Jeffries, 1991) concluded that those found to have the most significant influence on plant community structure were pH, biogeographic factors (latitude, altitude and age) and area, whilst animal community structure was most influenced by that of the plants.

An investigation (Tucker, 1958) of a large number of components of the water chemistry of ponds found that the levels of calcium and magnesium (total hardness) had the most significant influence on Mollusca, Hirudinea, Turbellaria and Crustacea. At levels below 25 ppm of calcium, the species present were both less diverse and less abundant. Molluscan community diversity and composition in a study of Canadian ponds were found to be related to substrate and water chemistry (dissolved solids, pH and alkalinity being the most significant). These abiotic factors were more important than macrophyte diversity or distance between ponds (Pip, 1986). Locally, an investigation of ball-clay ponds on the Isle of Purbeck, Dorset, from the point of view of water chemistry and age concluded that invertebrate species composition was highly correlated with pH; many taxa being absent at pH lower than 5.5 (Friday, 1987). Components of water chemistry that were selected for measurement in the present study were alkalinity, pH and conductivity. These were chosen because of the relative simplicity of collecting the data and because they were felt to be strong candidates for influencing the communities of the Bourne Stream.

In a study of west Dorset ponds, Temple (1992) found that ponds with a stream outlet had twice as many non-insect species as ponds that did not have such connections. Godwin (1923), in a classic early study of pond flora found that ponds connected to streams had a much higher rate of colonisation than those not connected resulting in greater species richness. Area of pond and isolation have been found to exert an influence on species composition (although these may be indirect through an influence of immigration and colonisation success and are dealt with further in the discussion). Plant species richness was found to be greater in ponds closer together (Moller and Rordam, 1985) and the area of the pond perimeter plus low water was the best predictor of the number of plant species. However recently created or cleared ponds showed higher species number than predicted by area. In an assessment of the botanical conservation value of a range of pond types, Linton and Goulder (2000) also found that high species richness was associated with the number of adjacent water bodies and disturbance (trampling and vegetation clearance) through bankside angling. Highest conservation value tended to be associated with ponds that originated as gravel pits, clay pits or borrow pits, possibly because they tend to be created close together. Macrophyte assemblages have been found to be only weakly linked with environmental factors; species richness increased with area of deeper water and pH and emergent taxa richness increased with pond area, area of deeper water and the extent of drying down (Jeffries, 1998). Biogeographic factors (latitude and altitude) were the most significant variables. Site age has been reported as the greatest influence on water beetle assemblages (Fairchild, *et al.*, 2000) although other work has related water beetle assemblages to successional stage and area of pond (Nilsson, 1984) and has found completely the opposite effect as mean species number declined with the age of the pond as vegetated complexity was reduced. Hydroperiod features as a significant factor in a number of studies and there is great interest in the specialized fauna of temporary waters (Williams, 1997). The length of the hydroperiod as well as its season has been found to affect community structure (Eyre, *et al*., 1992) and to cause a significant reduction in diversity (Jeffries, 1994). In general, diversity is lower in temporary ponds than in permanent waters because they are subject to greater variation in physico-chemical parameters (Williams, 1996). This drying down period being associated with low pH, low dissolved oxygen and greater shading by emergent and surrounding vegetation (Fairchild, 2003). The length of the previous dry period was found to be the most common predictor of the following years species occurrence (Jeffries, 2003). In this present study data collected included area of the pond, age, time since last disturbance (vegetation clearance, silt removal or drying up) and distance from source. These factors were also believed to be likely important determinants of community composition.

At a geographic scale, lentic aquatic beetle species number was related to area and connectedness (Ribera, *et al*., 2003) whilst lotic species number was related solely to latitude. It is proposed that this difference is caused by the greater dispersive power of lentic species due to evolutionary pressure of high rates of habitat turnover and the need for strong dispersal mechanisms. Environmental determinants of community structure in streams at the gross scale are flow, substrate, temperature, dissolved oxygen and water chemistry (Jeffries and Mills, 1990). Invertebrate species richness and density in Irish streams within a single afforested catchment area increased with distance from source, as a result of increasing pH and water hardness caused by changes in climate, geology and land use, so-called ‘biogeographical factors’ (Clenaghan, *et al*., 1998).

Acidic streams were primarily determined by pH and water hardness whilst circum-neutral streams were affected more by substrate and shading. At the finer scale, microdistribution of species within a stream is affected by current velocity and substrate particle size (Rabeni and Minshall, 1977) and the physical characteristics of flow (‘stream hydraulics’) (Statzner and Higler, 1986). Statzner *et al* (1986) found species richness was related to environmental harshness and variability which appears to approximate to Grime’s stress and disturbance factors. Environmental factors which were therefore measured in the stream were flow, dissolved oxygen and temperature, alkalinity, pH and conductivity.

Thus the most important environmental criteria were selected for measurement in order to relate the biological components to the anticipated physical and chemical gradients across the sample sites.

1.4 A working hypothesis

Expectations resulting from the hypothesis that habitat quality is affected by intermittent pollution can be drawn up from the foregoing review. It would be expected that, even in the absence of pollution, the SUDS would be lower in diversity as they are clearly younger than the existing ponds. It might also be expected that area would have an influence on species richness as might grazing and management. Any effects of pollution would be expected to show themselves as a gradient that would lessen with the distance from source as the waters are progressively cleaner moving downstream. The residency time of the water in each pond would also reasonably be expected to influence habitat quality as a quicker passage through the pond would expose the inhabitants to pollutants for a shorter period. Thus the expectation from the working hypothesis is that gradients of diversity would be found relating to these environmental factors.

2.Materials and Methods

2.1 Study Area

Maps of the area of study (Figures 1 and 2) are presented together with photographs of some of the sample sites (Figures 3-6).

2.1.1 Geomorphology

The Bourne Valley is a shallow valley running southeast from Canford Heath through the urban boroughs of Poole and Bournemouth. The valley is drained by the Bourne Stream which rises at an elevation of approximately 50 metres above sea level and runs for 6.3 kilometres before discharging into the sea at Bournemouth Pier.

The geology of the catchment area consists of Eocene marine beds referred to as the Poole Formation (formerly known as Bagshot Beds); these are largely sands with seams of pebbles and clay overlain with gravels higher up the valley. These free-draining, nutrient-poor, acidic soils support dry heath on the valley sides whilst the valley bottom ('Bourne Bottom' in the upper half) remains wet all year round and supports wet heath.

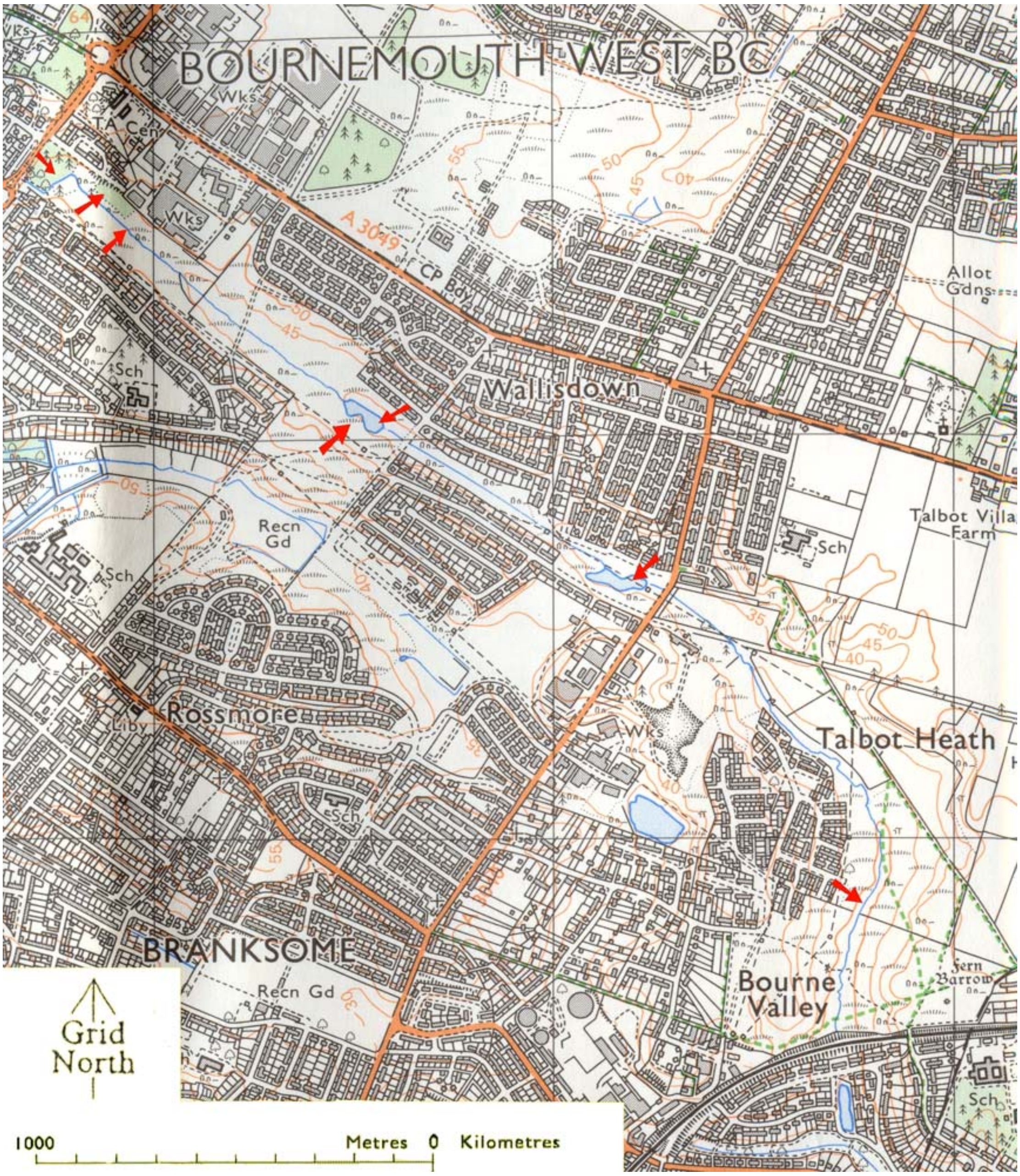

Figure 1. Map of Bourne Valley with sample sites indicated by arrows (source OS Pathfinder 1301).

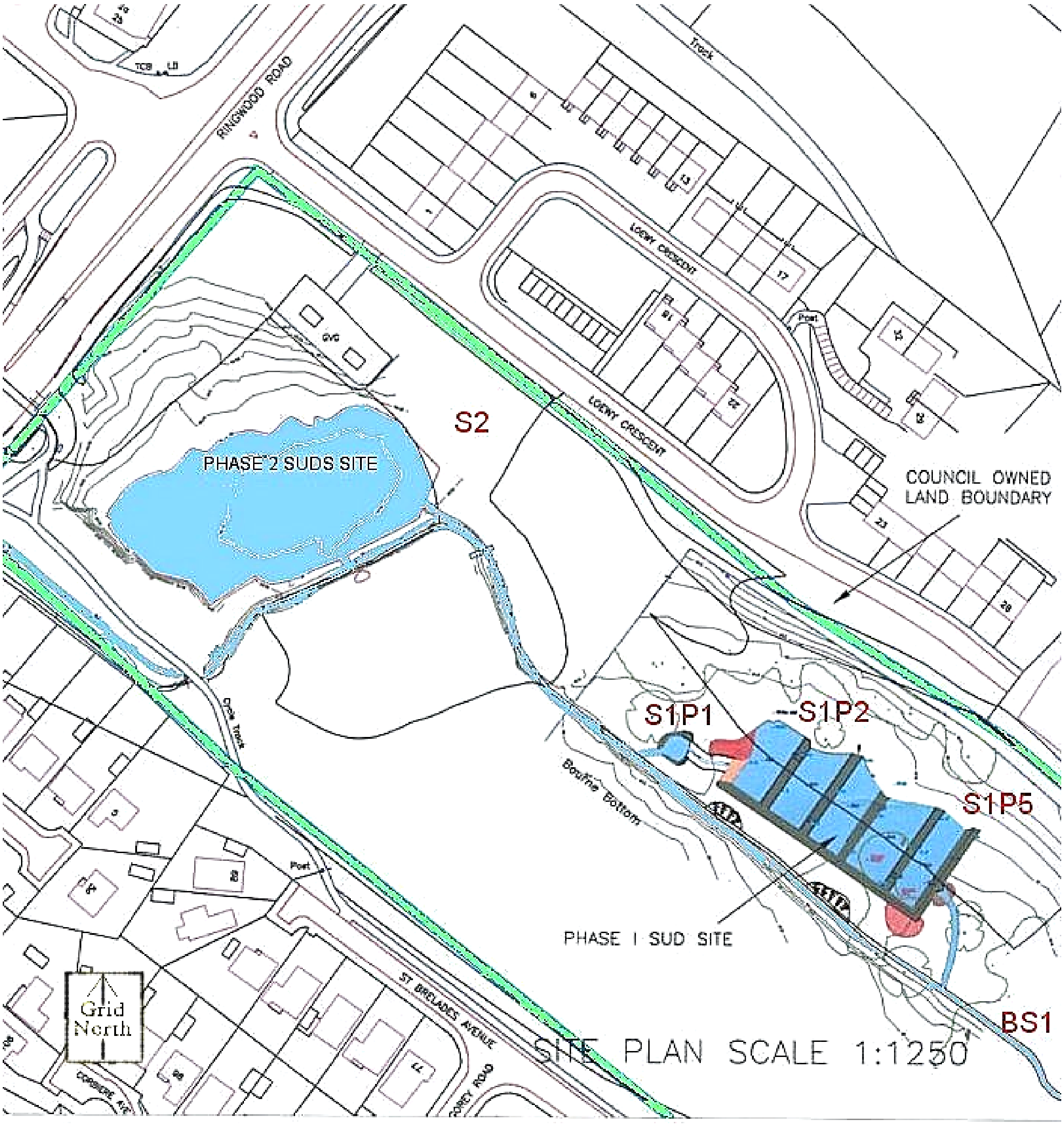

Figure 2. SUDS sample sites. Sample sites marked by site codes (source Poole Borough Council plans).

2.1.2 Conservation status

The Bourne Valley is a Site of Special Scientific Interest (SSSI), designated in 1985 and re-notified in March 1995 to include additional areas, such as Alder Hills which had separate SSSI designation. Additionally the Bourne Valley is a component of the Dorset

Figure 3. SUDS Phase 1 siltation pond (sample site S1P1) taken 19/6/03

Figure 4. SUDS phase 1 lagoon (sample site S1P2) taken 19/6/03

Figure 5. Bourne stream sample site 2 taken 11/8/03

Figure 6. Heathland pool (Bra) taken 29/7/03.

Heathlands Special Protection Area and a RAMSAR site (internationally important wetland site) on the basis of a number of birds associated with the area (Hen Harrier (*Circus cyaneus*), Dartford Warbler (*Sylvia undata*), Merlin (*Falco columbarius*), Nightjar (*Caprimulgus europaeus*) and Woodlark (*Lulluala arborea*)). It is also a candidate for Dorset Heaths Special Area of Conservation on the basis of its internationally important heathland flora and fauna (it supports populations of all six British reptiles, including the endangered Sand lizard (*Lacerta agilis*) and Smooth snake (*Coronopus austrica*)). The wet heath is important for a number of Red Data Book invertebrates such as the Small Red Damselfly (*Ceriagrion tenellum*) and Keeled Skimmer (*Orthetrum coerulescens*).

The mosaic of wet and dry heath is interspersed with areas of willow carr, grassland and secondary oak/birch woodland. The wet heath is classified as Northern Atlantic wet heath with *Erica tetralix,* M16 *Erica tetralix* (cross-leaved heath) *Sphagnum compactum* wet heath and M14 *Schoenus nigricans* (black bog rush) *Narthecium ossifragum* (bog asphodel) mire (Rodwell, 1991). The dry heath consists of European dry heath, H4 *Ulex gallii* (western gorse) *Agrostis curtisii* heath and H8 *Calluna vulgaris Ulex gallii* heath (Rodwell, 1991).

2.1.3 Hydrology

The stream is fed from a number of sources, however there is little documentary evidence about some of the groundworks and the actual origins of some are unknown. The head of the stream consists of three culverts emerging from under Ringwood Road; it is believed that one drains Canford Heath, one is fed from the Bournemouth and West Hampshire Water works at Francis Avenue and the third is supplied by road runoff. The stream is joined at many points along its course by drainage pipes and culverts of unknown origins as well as being fed by natural seeps and flushes. There is an ongoing investigation of these connections as it is believed that they may be the occasional source of *E. coli* levels which cause the sea water at Bournemouth Pier to be below the required standards for bathing waters (EC Directive concerning Bathing Waters – 76/160/EEC). This is an intermittent problem and is assumed to occur through storm water overflows flushing illegal foul sewer connections.

Another aspect of water quality that is of concern is the level of road runoff that reaches the stream. The ‘first flush’ (first heavy rain after a dry period) can contain high levels of hydrocarbons, heavy metals and, in winter, road salt. In order to improve the quality of water within the stream a number of SUDS initiatives have been implemented at the top of the stream, with more planned for further downstream. These modifications to the stream consist of a constructed wetland (Alderney SUDS phase 2) and a series of lagoons downstream of this (SUDS phase 1) consisting of a siltation pond, sand filter and open water lagoon compartments planted with reeds. The purpose of the constructed wetland (2512 m^2 in area when full) is to capture the first flush runoff waters and retain them in the reedbeds so that the hydrocarbons, heavy metals and salts are reduced in the water returning to the stream. This only fills when the stream rises sufficiently to overflow the weir. The lagoon compartments are fed by the stream continuously, in fact 95% of stream water during normal flow passes through these ponds. Their function is again one of sediment and pollutant removal, but it was hoped that they would also have a role in the UV-mediated reduction in *E. coli* levels.

2.1.4 Sample sites

The Bourne Stream runs into a number of large ponds on its way down the valley. Two of these onstream ponds were selected for inclusion in this study, one at Alder Road and one at Bloxworth Road. There are also a number of offstream ponds in the heathland bordering the stream, especially around Bloxworth Road. These are much smaller and suffer from drying up in the summer months. They are typically heathland pools of low pH and are not affected by the high pH, nutrient content or potentially polluted waters of the stream. One of these was included in the study.

All these ponds appear to be artificial in origin and to have resulted from mineral extraction activities or possibly army training activities in the case of the heathland pools away from the stream. The Alder Road pond exists on the Ordnance Survey map of the area (SZ0693NW, 1:1250) dating from 1945. The previous edition of this sheet dated 1924 shows the area as a brick works without the pond. It is therefore assumed that the Alder Road pond is in excess of 60 years old.

Bloxworth Road pond is not shown on the relevant sheet (SZ0594SE, 1:1250) in 1963 but is on the 1988 revision of Pathfinder 1301 and therefore can be assumed to be at least 15 years old. The small heathland ponds not connected to the stream are not shown on either map of these maps but are assumed to be of a similar age.

The constructed wetland dates from March 2001 and the lagoons from March 2000 (D. Hurst, *pers. comm.*). The lagoons are regularly managed which has been quantified as time since last disturbance (or creation). The heathland pool dries annually so has been given a figure of 1 year since last disturbance.

The residency time was calculated by dividing the volume (area x depth) by the outflow rate. The physical characteristics of the pond sample sites are presented in Table 1 and that of the stream sample sites in Table 2.

The stream was sampled at two points, one below the confluence of the outflow from the lagoons and the stream. The second was on Talbot Heath, downstream of all other sample points and was included in order to compare the results with a previously published study using the same stream sites and methodology (Armitage, *et al*., 1995).

Table 1. Physical characteristics of the sampled ponds

Attribute\site	SUDS phase 2 Constructed wetland	SUDS phase 1 siltation pond	SUDS phase 1 lagoon 1	SUDS phase 1 lagoon 5	Alder Road pond	Bloxworth Road pond	Bloxworth Road heath pool
Site code	S2	S1P1	S1P2	S1P5	AH	BR	BRa
Grid reference	SZ047947	SZ048946	SZ048945	SZ049945	SZ062936	SZ055941	SZ054941
Area (m^2)	2512	154	261	576	3539	3818	20
Elevation (m asl)	48.0	47.2	46.7	46.4	31.9	37.6	37.1
Distance from source (m)	25	97	121	167	1900	1100	Not onstream
Age (yr)	2	3	3	3	60 – 80	15 – 40	15 – 40
Time since disturbance (yr)	2	1	2	3	?	?	1 dries annually
Residency time (hr)	Onstream only during high flow periods	1	1	4	48	53	Not onstream

Table 2. Physical characteristics of the stream sample sites.

Attribute\site	Bourne stream site 1 (spring)	Bourne stream site 1 (summer)	Bourne stream site 2 (spring)	Bourne stream site 2 (summer)
Site code	BS1a	BS1b	BS2a	BS2b
Grid reference	SZ050945		SZ069929	
Elevation (m asl)	45.7		27.5	
Distance from source (m)	225		2800	
Flow (m^3 / hr)	108		144	

2.2 Experimental design

Design of the sampling strategy was driven by the need for representative sampling of each of the habitat types, preferably replicated in some way to enable statistically valid testing of any differences between sample sites. However another approach was also included which was to assess the sampled habitats against national collections of data for similar habitats. Thus the stream was sampled and assessed in accordance with RIVPACS in order to quantify any degradation of the habitat through the impact of polluted waters. Because a similar survey had been carried out on the Bourne Stream in 1994 (Armitage, *et al*., 1995), any changes over this time period could also be assessed.

The ponds were similarly assessed against a national database using the PSYM system which provides a similar index of the degradation of the habitat. Thus the sampling strategy had to fulfil two roles:

- to provide input to RIVPACS and PSYM in an appropriate manner
- to provide data for statistical analysis.

2.3 Sampling protocol

The physical and environmental data collected were measured using a standard approach for both stream and pond samples. However not all the parameters were collected for both as the data were analysed differently for the stream and pond samples. The methods used are described and any differences between site approaches are noted.

2.3.1 Physical parameters

Flow was measured with an MJP Geopacks flowmeter in midwater until a constant rate was obtained. Sample sites were stream sites 1 and 2 as well as the in and outflows of the constructed wetland, lagoons and two onstream ponds. These flow measurements were used to calculate discharge categories for the stream sites and residency times for the ponds (see Table 1).

The width of the stream was measured using a calibrated metre rule. The depth of the stream was measured at one quarter, one half and three quarters the width using a calibrated metre rule. This allowed an average cross-sectional area to be calculated and used in the discharge calculations. The depth of the ponds was measured in the approximate centre using the calibrated metre rule where access was possible; otherwise depths were obtained from plans (S. Austin, *pers. comm*.). Substrate was sampled by net at the appropriate sampling sites although it should be noted that data were presented for analysis in slightly different formats for RIVPACS and PSYM. The altitude of the sample sites was obtained from contour lines on relevant Ordnance Survey maps and distance from source and slope calculated from the same maps. The perimeter of each pond was estimated in the field by pacing and at the same time the proportions of the pond shaded by trees and grazed was estimated. The area of the ponds was obtained from Ordnance Survey maps or plans.

2.3.2 Environmental parameters

pH was measured at each sample site using an IQ Scientific Instruments miniLab IQ120 pH meter accurate to 0.1. Conductivity was measured using a Hanna DiST WP conductivity meter. Both these were taken immediately on site from a water sample collected in a wide mouthed bottle. Dissolved oxygen was measured for the stream samples using a Hanna DO Spectrophotometer HI 93732N. Temperature was measured using a Hanna Minitherm HI 8751 thermometer placed directly in the water.

In addition a sample of stream water was collected for subsequent analysis of alkalinity in the laboratory. The 500 ml black plastic sample bottles were acid washed and rinsed with distilled water before use. This sample was returned to the laboratory and analyzed within two hours of collection. The analysis for alkalinity was an acid titration (0.01M hydrochloric acid) to pH 4.5 using BDH 4.5 indicator (Mackereth, *et al*., 1989). The results were expressed as meq / l and can be converted to mg $CaCO_3$ / l by multiplying by 50.

2.3.3 Biological sampling

The sampling protocol adopted was based on the Environment Agency standard sampling and analysis manual and refers to running water, but is broadly similar for ponds (Murray-Bligh, 1999). Differences in approach to the pond sampling are discussed at the end of the section. This had previously been taught as part of a unit of study at the Centre for Ecology and Hydrology and therefore was already familiar (National Rivers Authority, 1990).

2.3.3.1 Site selection

Site selection is a very important part of the procedure, the requirement being for a stretch of stream that is typical of the whole section being sampled. A section is usually 500 metres long and the stretch selected from within this area will be perhaps 10 metres. Artificial disturbance such as bridges or weirs should be avoided, although they can be useful for collecting the water chemistry data, because the stream will not be representative at such points. The sample site should be visited in a minimum of two seasons, three being preferable, with the results being totalled between all samples for macroinvertebrate counts, and averaged between all samples for water chemistry. This provides greater coverage of the invertebrates as it takes into account differences in life history and phenology.

In this study, the sample points were identified by grid reference from a previous investigation, which left approximately 100 metres of stream to be assessed for a suitable stretch.

2.3.3.2 Collecting macroinvertebrate samples

The general aim is to collect comparable samples rather than exhaustive samples, thus a strict protocol should be observed in order to standardise sampling. Each habitat in the sampling area must be sampled with an effort proportional to its cover. The sampling takes place in three stages:

1. A manual search for surface dwelling animals, such as water skaters and whirligig beetles which are easily disturbed and move away from the area. This search together with part 3 should last a total of one minute.

2. A three minute kick/sweep pond net sample from all the habitats in proportion to their cover. The technique depends on the habitat with riffles being kick sampled and weed being swept. The kick sample is taken with the operator using his foot to disturb the bottom sediment down to a depth of about 10 cms just above stream of the net, into which all disturbed animals should be carried.

3. A manual search of submerged stones, logs or vegetation taking up the remainder of the minute from part 1. Clinging animals are carefully washed or picked off the substrate into the net.

The net used was a standard FBA pattern long handled net with a 1mm mesh size and a frame width of 250mm and a bag depth of 0.3 m. It must be washed before and after each sample in order to ensure that nothing is carried from site to site and corrupts the next sample. It should also be washed during the sampling to free it of substrate, vegetation etc which can clog the net and makes handling more difficult. The site should be sampled by moving upstream and diagonally across the stream in order to cover as much of the habitat as possible even where it looks identical. The three minute and one minute periods only cover the time spent actively sampling or searching, it excludes the time taken emptying the net or moving around the site.

The net samples were placed into a polythene bag with a seal and transported live to the laboratory for sorting and identification. This took place within two hours of the sample being taken, usually less. Only sufficient water to keep the sample damp was retained as this reduces damage and the activities of carnivores. Fish and Amphibia were immediately returned to the water on site.

2.3.3.3 Processing samples

Samples were initially sorted live in the laboratory in a white flat-bottomed tray. This proved an effective technique as the movement of many invertebrate animals makes them highly visible. Soft-bodied animals – worms, leeches, and flatworms were transferred to a dilute solution of carbonic acid (‘soda water’) to narcotize them whilst examination and identification under the microscope took place. All other invertebrates were killed by immersion in 70 % Industrial Methylated Spirits (IMS) before examination. Attempts to identify them alive proved futile. Voucher specimens were retained for subsequent confirmation of identification.

After sorting through the sample in the tray until no more specimens were found, IMS was added and the sample washed and filtered through a 500 µm sieve to concentrate it. It was then returned to a white, flat-bottomed tray and resorted until no more specimens were found. This proved particularly effective for dealing with mayflies which were difficult to catch alive.

Although RIVPACS and PSYM only require the identification down to family level, the sample was identified, where life history stage and availability of identification keys allowed, to species and the abundance of each species recorded for further statistical analysis. Identification of certain groups proved problematic and these were only recorded down to the family level. For example the Chironomidae could not be identified to species level for want of an identification key. Moreover they were not taken to the subfamily level as originally intended because the process of clearing the head capsules with Berlese fluid and mounting them on microscope slides was found to be excessively time-consuming for the numbers being captured. Processing of each 3 minute sample took approximately two to three days to sort, identify and count.

2.3.3.4 Pond sampling protocol

In general, the principles and techniques used were identical; however there were minor differences which are noted here:

- Some of the environmental samples (alkalinity, DO, temperature) were not taken for the ponds. This was because they were unnecessary for the analysis proposed (PSYM) and because their variability throughout the year is such that no correlation was expected with the biological sampling.
- The biological sampling of the pond followed the PSYM protocol which broadly matched that described for the stream samples but differed in that the three minute net sampling was divided equally between habitats rather than by the proportion of habitat cover. This is because ponds are often dominated by areas of open water and bare silt which contain few invertebrates whereas the vegetation and margins contain the majority of species. For example Fairchild (2003) reports that 85% of aquatic beetles are found at the pond margins where access to atmospheric oxygen is energetically easier and less risky because there is a shorter distance to the surface.
- PSYM has some additional requirements for data that RIVPACS does not, because it includes macrophytes in its assessment. Therefore the sampling

protocol includes a survey of the vegetation within the boundaries of the pond at winter high water. PSYM sampling takes place in the summer months and includes vegetation growing in the draw down area of the pond that may be dry during the sampling period. Macrophytes were identified and recorded to the species level for PSYM but were additionally assessed for their percentage cover values for further statistical analysis.

2.4 RIVPACS

River **Inv**ertebrate **P**rediction **A**nd **C**lassification **S**ystem (RIVPACS) is a software system that provides a database of the macroinvertebrate communities of pristine sites across the UK. These represent the standard against which local samples collected following the standard methodology can be compared to evaluate river quality. The technique is used nationally by the Environment Agency and has informed and directed the drafting of the approach defined by the EU Water Framework Directive for water quality assessment (Clarke, *et al*., 2002).

Using the BMWP scoring system, in which taxa sensitive to pollution are given high scores and tolerant taxa are given low scores, the sample is totalled to give a BMWP score and the number of taxa present. The BMWP score gives an indication of the level of environmental stress that the site incurs. However the score can be biased by sample size therefore the Average Score Per Taxa (ASPT) is calculated by dividing BMWP by number of taxa. This represents an average pollution tolerance for the taxa collected and is independent of sample size and less influenced by season than BMWP. A list of the BMWP family scores is given in Table 3.

Table 3 BMWP family scores

Common name	Family	Score
Mayflies	Siphlonuridae, Heptageniidae, Leptophlebiidae, Ephemerellidae, Potamanthidae, Ephemeridae	10
Stoneflies	Taeniopterygidae, Leuctridae, Capniidae, Perlodidae, Perlidae, Chloroperlidae	10
River bug	Aphelocheiridae	10
Caddisflies	Phryganeidae, Molannidae, Beraeidae, Odontoceridae, Leptoceridae, Goeridae, Lepidostomatidae, Brachycentridae, Sericostomatidae	10
Crayfish	Astacidae	8
Dragonflies	Lestidae, Calopterygidae (=Agriidae), Gomphidae, Cordulegasteridae, Aeshnidae, Corduliidae, Libellulidae	8
Caddisflies	Psychomyiidae (incl. Ecnomidae), Philopotamiidae	8
Mayflies	Caenidae	7
Stoneflies	Nemouridae	7
Caddisflies	Rhyacophilidae (incl. Glossomatidae), Polycentropidae, Limnephilidae	7
Snails	Neritidae, Viviparidae, Ancylidae (incl. Acroloxidae)	6
Caddisflies	Hydroptilidae	6
Mussels	Unionidae	6
Shrimps	Coriphiidae, Gammaridae (incl. Crangonyctidae and Niphargidae)	6
Dragonflies	Platycnemidae, Coenagriidae	6
Water bugs	Mesoveliidae, Hydrometridae, Gerridae, Nepidae, Naucoridae, Notonectidae, Pleidae, Corixidae	5
Water beetles	Haliplidae, Hygrobiidae, Dytiscidae (incl. Noteridae), Gyrinidae, Hydrophilidae (incl. Hydraenidae), Clambidae, Scirtidae (=Helodidae), Dryopidae, Elmidae (=Elminthidae), Chrysomelidae, Curculionidae	5
Caddisflies	Hydropsychidae	5
Craneflies	Tipulidae	5
Blackflies	Simuliidae	5
Flatworms	Planariidae (incl. Dugesiidae), Dendrocoelidae	5
Mayflies	Baetidae	4
Alderflies	Sialidae	4
Leeches	Piscicolidae	4
Snails	Valvatidae, Hydrobiidae (incl. Bithyniidae), Lymnaeidae, Physidae, Planorbidae	3
Cockles	Sphaeriidae	3
Leeches	Glossiphoniidae, Hirudidae, Erpobdellidae	3
Hoglouse	Asellidae	3
Non-biting midges	Chironomidae	2
Worms	Oligochaeta (whole class)	1

The software generates expected values for BMWP, number of taxa and ASPT for the site based on environmental parameters. These are used to assign the site to a group of sites of similar environmental conditions and the macroinvertebrates from this unstressed group are extracted as a predictive list and used to generate the expected scores. By evaluating the observed / expected ratios an Ecological Quality Index is produced for each score. The EQI for number of taxa and ASPT are then assigned to quality bands (see Table 14) and the lower of the two bands used to categorise the river. The environmental parameters which are required by RIVPACS are included as Figure 7. Latitude and longitude are derived from grid reference, discharge category is derived from flow rate, mean substratum (phi) is derived from the proportions of substrate categories and the mean and annual air temperatures are derived from internal tables based on latitude and longitude.

Environmental data used:	
Latitude	50 degrees 44 minutes N
Longitude	1 degrees 55 minutes W
Altitude	46 m
Distance from source	.5 km
Discharge category	1
Mean width	.9 m
Mean depth	17.0 cm
Substratum composition:	
Boulders + cobbles	0 %
Pebbles + gravel	16 %
Sand	16 %
Silt	68 %
mean substratum (phi)	5.24
Slope	10.0 m/km
Alkalinity	168.0 mg/l CaCO3
Mean air temperature	10.63 C
Annual air temperature range	12.15 C

Figure 7. Environmental parameters used by RIVPACS for Bourne Stream site 1

2.5 PSYM

Predictive **SY**stem for **M**ultimetrics (PSYM) was developed to provide a method for assessing the biological quality of still waters in England and Wales. It essentially combines the predictive approach of RIVPACS with the multi-metric based methods adopted in the US. The PSYM methodology adheres to the approach defined in the EU Water Framework Directive.

PSYM operates in a similar way to RIVPACS, in that predictions are generated from a database of sites across England and Wales, based on environmental parameters. The assessment is based on macrophytes as well as macroinvertebrates and therefore involves a vegetation survey in addition to the net sampling (Pond Action, 2002). Invertebrate BMWP score and overall number of taxa are calculated together with the number of Odonata and Megaloptera families scored and the number of Coleoptera families scored. Macrophytes are identified to species level and given a Trophic Ranking Score (TRS) and a rarity score. The total number of species, the number of uncommon species and the average TRS (equivalent to ASPT for invertebrates) is input to the model.

The predictive nature of the analysis is based on environmental parameters (area, pH, shade, grazing, inflow and emergent plant cover) and geographical factors (grid reference, altitude and base geology). The analysis was run by Dr Anita Weatherby at the Ponds Conservation Trust (previously Pond Action), Oxford Brookes University as the software is not available publicly.

3.Results

Section 3.1 Environmental conditions presents the results of investigation of the environmental conditions prevailing in the three major freshwater habitats of stream, pond and SUDS (objective 1). Section 3.2 Species diversity presents the results of the investigations of the macroinvertebrate and macrophyte diversity (objectives 2 and 3 respectively). Section 3.3 presents the results of the investigations of the relationship between environmental conditions and species diversity (objective 4). Section 3.4 presents a comparison between the RIVPACS assessment of the stream in 1994 and the present study (objective 5). Section 3.5 presents the assessment of the conservation value of the habitats (objective 6).

3.1 Environmental conditions

The environmental data for the pond sample sites are presented in Table 4. The data were used as input to the PSYM analysis and were analyzed for correlations with biological data (see section 3.3).

Stream water chemistry data are presented in Table 5. These data were used as input to RIVPACS and were analyzed for correlations with biological data (see section 3.3). The dissolved oxygen levels measured in the field were converted to percentage saturation by dividing by the 100% saturation level for that temperature from tables given in Mackereth *et al* (1989). The particularly low level for the site 2 summer sample may be related to the long hot dry period preceeding the sampling and the position of the sample site which was downstream of a pond whose outflow was culverted for a short distance.

The data for each site were collected at the time of the biological sampling and therefore differed in date for each site. As a result it was felt that some of the chemical parameters were not necessarily comparable because their measurement was separated in time. Therefore a single set of reference pH and conductivity readings were made on the same day in order to demonstrate any gradients downstream that might otherwise have been obscured by changing water chemistry.

Table 4. Environmental characteristics of the ponds

Attribute\site	SUDS phase 2 Constructed wetland	SUDS phase 1 siltation pond	SUDS phase 1 lagoon 1	SUDS phase 1 lagoon 5	Alder Road pond	Bloxworth Road pond	Bloxworth Road heath pool
Date sampled	12/06/2003	21/05/2003	19/06/2003	06/07/2003	28/05/2003	29/07/2003	29/07/2003
Perimeter % shaded	15	5	0	0	55	65	0
Perimeter % grazed	0	30	30	30	0	0	0
% emergent vegetation	95	50	20	75	70	100	0
Substrate % mud/silt	100	100	90	100	100	100	100
Substrate %sand/gravel/cobbles			10				
pH	7.8	8.3	8.2	8.9	8.0	8.4	6.1
Conductivity	460	545	541	587	447	294	260
Alkalinity	25.3	30	29.6	32.4	26.9	19.5	[a]
Reference pH (8/8/03)	7.0	7.7	8.0	7.8	7.8	7.7	5.5
Reference Conductivity (8/8/03)	529	505	509	514	399	445	112

[a]Alkalinity was not measured due to the water level being very low and a clean water sample could not be achieved.

Table 5. Environmental parameters for the stream sample sites.

Parameter \ site	Bourne stream site 1 (spring)	Bourne stream site 1 (summer)	Bourne stream site 2 (spring)	Bourne stream site 2 (summer)
Date sampled	24/04/2003	18/08/2003	13/05/2003	11/08/2003
Substrate % mud/silt	67	67	10	10
Substrate %sand/gravel/cobbles	33	33	90	90
pH	7.6	7.7	8.1	7.4
Conductivity	654	445	325	364
Alkalinity	[a]	33.6	22.0	26.0
Dissolved oxygen	6.1	5.1	7.4	2.6
Temperature	6.6	18.4	11.3	19.3
Dissolved oxygen as % saturated	49.6	54.4	67.6	27.9
Reference pH (8/8/03)		7.9		7.1
Reference Conductivity (8/8/03)		495.0		377.0

[a] Alkalinity was not analyzed for this sample.

3.2 Species diversity

Biological sampling was carried out in two seasons (spring and summer) for the stream invertebrates and in the summer for pond invertebrates and macrophyte surveys. This is in accord with RIVPACS and PSYM protocols and reflects the period of maximum diversity. The BMWP scores for each site are presented in Table 6 together with the total number of individuals, the total number of species and the ASPT.

Table 6. Biotic scores for all sites.

Site \ score	BMWP	Total number of individuals	Total number of species	ASPT
BS1a	107	1263	40	4.46
BS1b	87	395	27	4.14
BS1 combined	110	1658	43	4.46
BS2a	69	29	19	4.31
BS2b	96	186	31	4.59
BS2 combined	125	215	36	4.81
S2	101	489	30	4.45
S1P1	92	467	30	4.38
S1P2	71	167	21	4.25
S1P5	68	262	18	4.25
AH	80	710	34	4.05
BR	89	170	27	4.48
Bra	58	231	24	4.83

The abundance data collected for all the sample sites allowed diversity to be measured (see Table 7). The concept of diversity encompasses two elements, species richness and the proportions in which species occur in the community. There are a number of diversity indices but the ones in commonest use are the Simpson Index, D, which measures species richness and the Shannon Index, H, which measures evenness. Simpsons Index relates to the probability that two consecutive random samples will pick the same species:

$$\mathbf{D = 1 - \Sigma(p_i \times p_i)}$$

where $\mathbf{p_i}$ is the proportion of the *i*th species which gives D a range from 0 (no diversity) to 1 (maximum diversity).

The Shannon Index relates to the information content of the data and is calculated as:

$$\mathbf{H = - \Sigma\, p_i \ln (p_i)}$$

In order to reduce this to a scale from 0 to 1 this is then converted to the Shannon Index of evenness, J, which is the ratio of the calculated diversity to the maximum possible diversity for the number of species found:

$$\mathbf{J = H / H_{max}}$$

J will range from 0 (community dominated by a single species) to 1 (community with equal proportions of all species). Strictly speaking, statistical comparisons of diversity indices are only valid if they are based on the same sample size (Fowler, *et al.*, 1998), however the information presented here is interpreted without further manipulation and so comparison is valid. The sites have broadly the same score for diversity as measured

Table 7. Invertebrate diversity indices for all sites.

Site \ Diversity index	Simpsons index, D	Shannon Index, J
BS1a	0.84	0.57
BS1b	0.89	0.77
BS1 combined	0.87	0.62
BS2a	0.90	0.91
BS2b	0.90	0.80
BS2 combined	0.91	0.80
S2	0.78	0.60
S1P1	0.83	0.67
S1P2	0.89	0.83
S1P5	0.83	0.80
AH	0.87	0.69
BR	0.87	0.74
Bra	0.43	0.40

by Simpsons Index, with the exception of the heathland pool. The Shannon Index of evenness shows a wider spread of results across the sites but still has the heathland pool as an outlier from all other sites.

The same diversity indices were calculated for the macrophyte data, which were only collected for the pond sites (see Table 8). However these data were not abundance but percentage cover which is also acceptable as it is ordinal data (Shaw, 2003). Once again the heathland pool stands out as an exception although the SUDS phase 1 lagoon 5 is even more species poor and also dominated by a single species (*Phragmites australis* in the case of the SUDS pond but *Potamogeton polygonifolius* in the heathland pool giving very different vegetation structures to the two sites).

Table 8. Macrophyte diversity indices for ponds

Site \ Diversity index	**Number of plant species**	**Number of uncommon species**	**Trophic Ranking Score**	**Simpsons index, D**	**Shannon Index, J**
S2	29	1	8.53	0.83	0.69
S1P1	19	0	8.96	0.64	0.60
S1P2	19	1	8.15	0.63	0.57
S1P5	6	1	8.60	0.28	0.36
AH	22	0	8.64	0.81	0.71
BR	17	0	8.18	0.82	0.72
Bra	10	0	3.70	0.17	0.22

3.3 Relationship between diversity and environment

In order to ascertain whether there are any correlations between the species present at each site and the environmental parameters, an analysis of correlation was carried out using SPSS 10.0 for Windows. Spearmans rank correlation coefficient, $\mathbf{r_s}$, was used because the data representing the sites consisted of indices (BMWP, ASPT, D and J) rather than directly measured parameters (Fowler, *et al*., 1998). The results are presented in full in Appendix 3 with the significant correlations extracted into Table 9.

Table 9. Significant correlations between factors as measured by Spearmans Rank correlation coefficient with percentage level of significance in parentheses.

Factor	Reference pH	Macrophyte diversity (D)	Macrophyte diversity (J)	%emergent vegetation
BMWP		.893 (1%)		
# macrophyte species		.757 (5%)		
Alkalinity		-.886 (5%)	-.886 (5%)	
% shaded		.852 (5%)	.964 (1%)	
Residency time		.812 (5%)		.870 (5%)
Invertebrate diversity (J)	.909 (1%)			
Invertebrate diversity (D)	.852 (5%)			
ASPT	-.853 (5%)			
Pond EQI invertebrates	-.853 (5%)			
Pond area			.857 (5%)	.857 (5%)
# Coleoptera species				-.775 (5%)

The most significant correlation for plants was between macrophyte diversity (Shannon index, J) and the percentage of the pond shaded by trees, which was significant at the 1% level. Plant diversity (J) was weakly correlated with pond area (at the 5% level). Macrophyte diversity as measured by the Simpson index (D) was also correlated with shade but only at the 5% level. This index was also correlated with the number of plant species, residency time and the BMWP score for invertebrates, all at the 5% level. It was also negatively correlated with alkalinity. Percentage cover of emergent vegetation was correlated with residency time and pond area.

The most significant invertebrate correlation was between the invertebrate diversity (Shannon Index, J) and pH as measured on 8/8/03, which is significant at the 0.01 level. This reference pH was also correlated with three other measures of invertebrate diversity but only at the 0.05 level. It was positively correlated with Simpsons Index, D, and negatively correlated with ASPT and the pond EQI score (ratio of observed to expected pond species as derived from PSYM). High ASPT and pond EQI scores reflect a less pollution tolerant invertebrate community, so a negative correlation with pH suggests that the more polluted sites have a higher pH.

The number of Coleoptera species was negatively correlated with % emergent vegetation at the 0.05 level. This is explicable in terms of the habitat available to the beetles, 100% emergent vegetation represents an overgrown pond with relatively fewer architectural components (submerged and floating leaved vegetation, etc) that a less overgrown pond would offer.

Correlations were primarily between environmental factors and measures of diversity (plants and alkalinity, area, shade and residency time; invertebrates and pH) but there were correlations between invertebrate richness and diversity and vegetation factors. There were also a number of weak correlations between environmental factors which are not explored here.

It is instructive to examine the lack of correlation exposed in this analysis as well. Expectation of a relationship between measures of diversity and environmental factors such as age, area, distance from source and residency times was not fulfilled for invertebrates but was partially for plants. Reasons for this are suggested in the discussion.

3.3.1 Multivariate analyses

Principal Components Analysis (PCA) in SPSS 10.0 for Windows was used to investigate any trends in the invertebrate abundance data for each pond site. The abundance data, as is often the case in ecological data, contains many zero values and is therefore not normally distributed and is highly skewed (heteroskedastic). In order to compensate for this, the data was log transformed using

$$\mathbf{Log_{10}\,(x+1)}$$

The eigenvalue resulting from PCA represents the importance of an axis in representing an underlying trend in the data. The criteria for assessing the significance of the eigenvalue are largely subjective as there are no formal statistical tests for doing so. The Kaiser-Guttmann criterion, commonly used as default option in many software packages, is to reject any eigenvalue less than one. However this tends to overestimate the number of non-trivial axis and therefore the broken stick model was preferred. This sets the threshold for accepting an axis as worthy of investigation as an eigenvalue of

$$\boldsymbol{\lambda_p = {}^{i=p}\Sigma_{\,i=n}\,(1/i)}$$

for the *p*th axis in a matrix of *n* variables (Shaw, 2003). The proportion of variation accounted for by axis 1 was only 23% which, with 9 degrees of freedom (the sample sites), is not greater than that expected from random data. The broken stick model sets a minimum level of 31% for the first axis before the results should be considered to contain any meaningful trend.

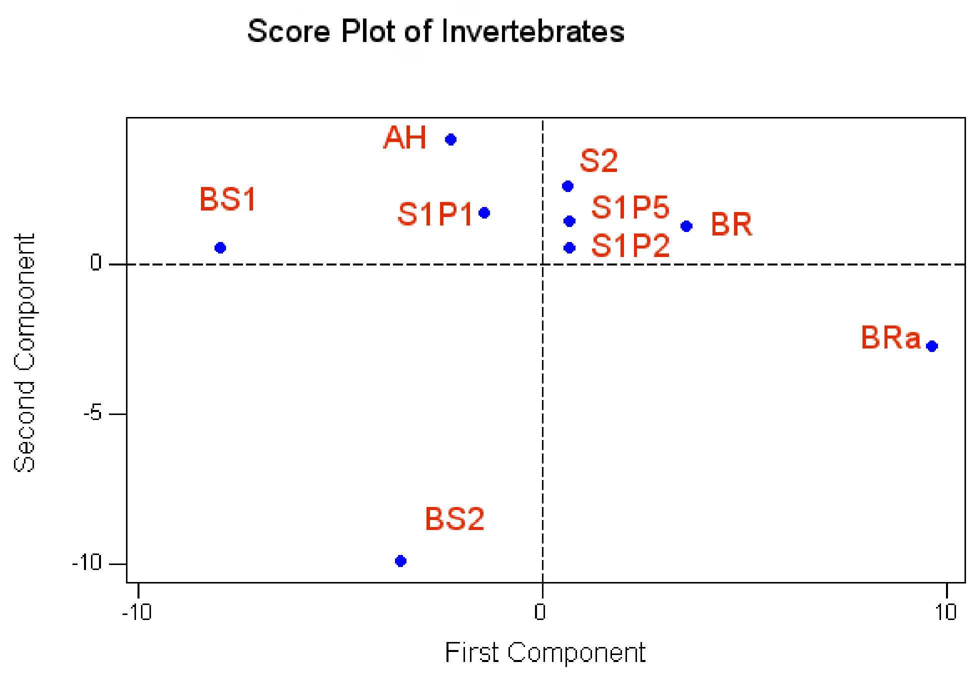

Figure 8. Ordination diagram of sample sites by invertebrate diversity.

Although invertebrates are judged to successfully represent the diversity of the whole ecosystem, recent work (Briers and Biggs, 2003, Fairchild, *et al*., 2000) suggests that certain groups reflect this diversity and could be used in place of all macroinvertebrates. Coleoptera, Odonata and Trichoptera have been successfully used to measure habitat quality whilst Amphibia are often used to assess the conservation value of a freshwater environment (Wong *et al*., 1998). Therefore PCA was repeated using various subsets of the invertebrate data which have been identified as reflecting the diversity of the whole invertebrate community. The eigenvalues and proportion of variation accounted for in the subsets were

- Coleoptera, Odonata and Trichoptera 2.14 (24%)
- Coleoptera 2.20 (25%)

and therefore did not improve the analysis.

PCA was also used to investigate the plant data, again log transformed although this time representing percentage cover values rather than abundance. The results (eigenvalue = 14.0 and proportion accounted for = 25%) again are not significant enough to suggest that there is trend in the occurrence and cover of macrophytes across the pond sites sampled. However inspection of the score plot (Figure 9) indicates that the ordination appears to have ordered the sites along an axis relating to distance from source for the first component and area for the second. These gradients would have been expected from the hypothesis being tested but are not supported by the correlation analysis which has a statistically calculated degree of confidence associated with the result which ordination does not.

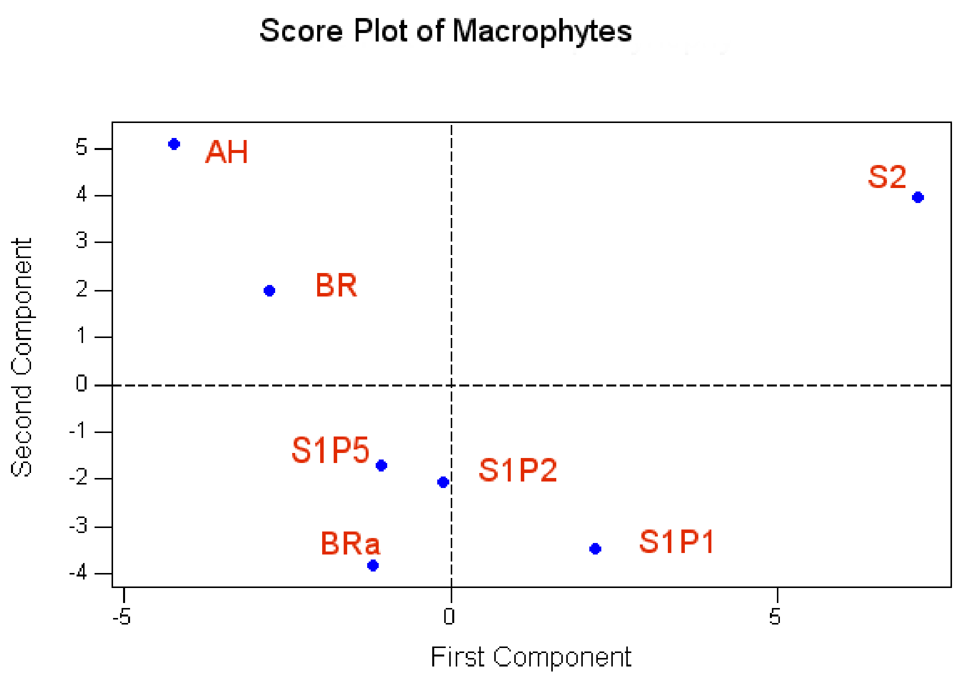

Figure 9. Ordination diagram of sample sites by macrophyte diversity.

The conclusion to be drawn from this preliminary investigation of the data using PCA is that there are no, or only very weak, gradients or trends. This suggests that the sites are all similar to one another in terms of invertebrate and macrophyte species composition.

Cluster Analysis in Minitab 13 for Windows was used to investigate the similarity of sites and the results were in agreement with subjective opinion formed during sampling. The same log transformed data were used for the cluster analysis as was used for Principal Components Analysis. Cluster analysis was based on correlation coefficient distance as a measure of the similarity between sites. Similarity and distance levels are presented in Appendix 4.

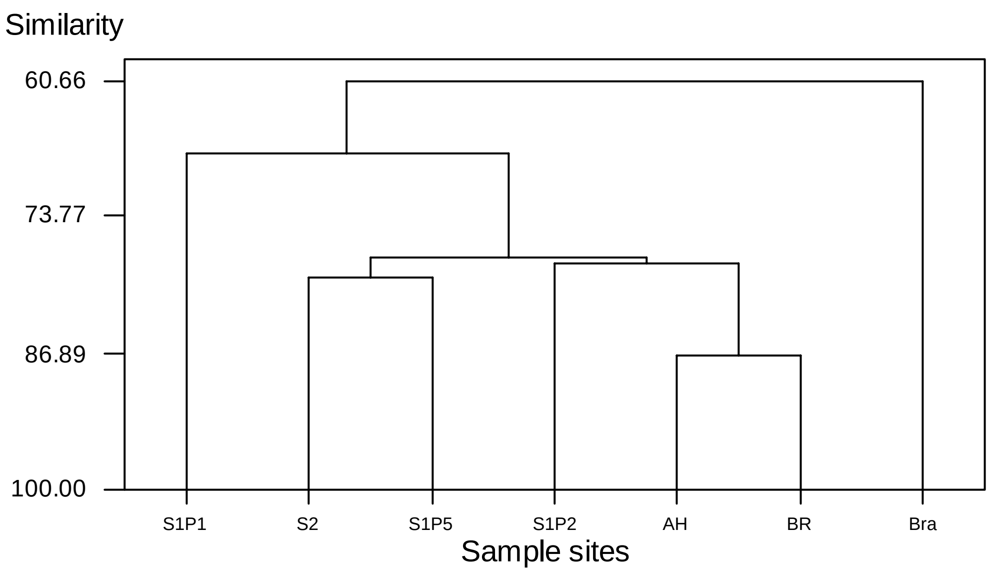

Figure 10. Dendrogram of pond sites based on cluster analysis of plant data

The dendrogram (Figure 10) produced for the pond sites based on similarity of plant cover data clearly identifies the heathland pool (Bra) as an outlier. It separates the SUDS siltation pond (S1P1) from the other sites which is not unexpected as this was the most recently cleared and has most in common with the stream. The remaining sites are dominated by reeds or reedmace with little open water (S1P2 having the most). The older ponds (AH and BR) are clearly closely allied whilst the SUDS wetland (S2) and lagoon (S1P5) are clustered together in spite of the great differences in overall diversity.

The cluster analysis of invertebrate data for the pond sites (Figure 11) shows a higher level of similarity between all sites than does the plant data analysis. Once again the heathland pool is the first to be separated. The SUDS sites S2 and S1P5 are closely linked as was the case for vegetation, but they are joined with AH which is considerably separated from BR (AH and BR were in the same vegetation cluster). The two most recently cleared lagoons (S1P1 and S1P2) are clustered together.

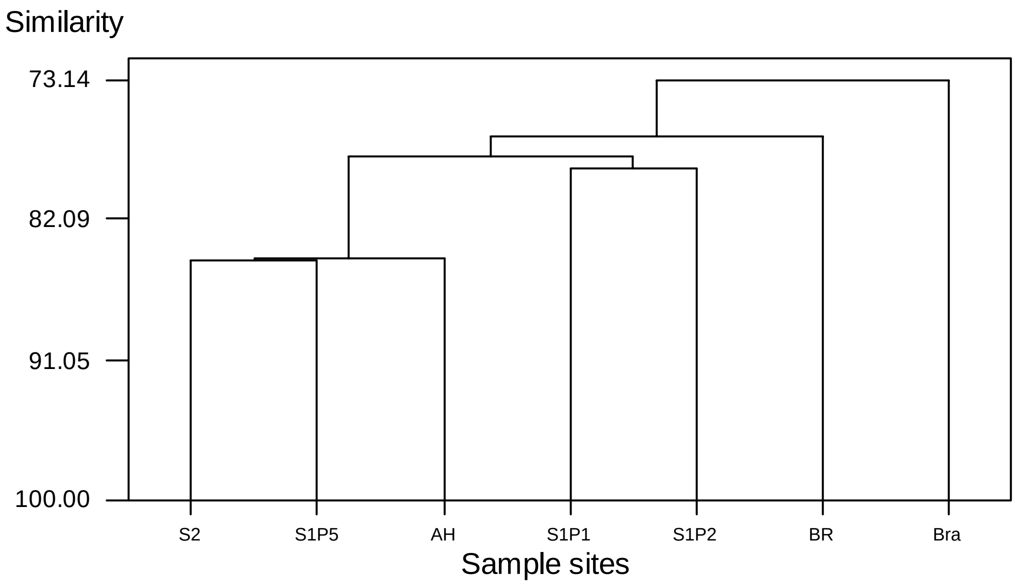

Figure 11. Dendrogram of pond sites based on cluster analysis of invertebrate data

If the stream sites are included (Figure 12), then the S2, S1P5 and AH cluster remains the same as do the outliers Bra and BR. However the SUDS siltation pond (S1P1) is placed in a cluster with the upper stream site and the second lagoon (S1P2) remains separated. This emphasises the dissimilarity between S1P1 and the other ponds as noted in the plant data cluster analysis.

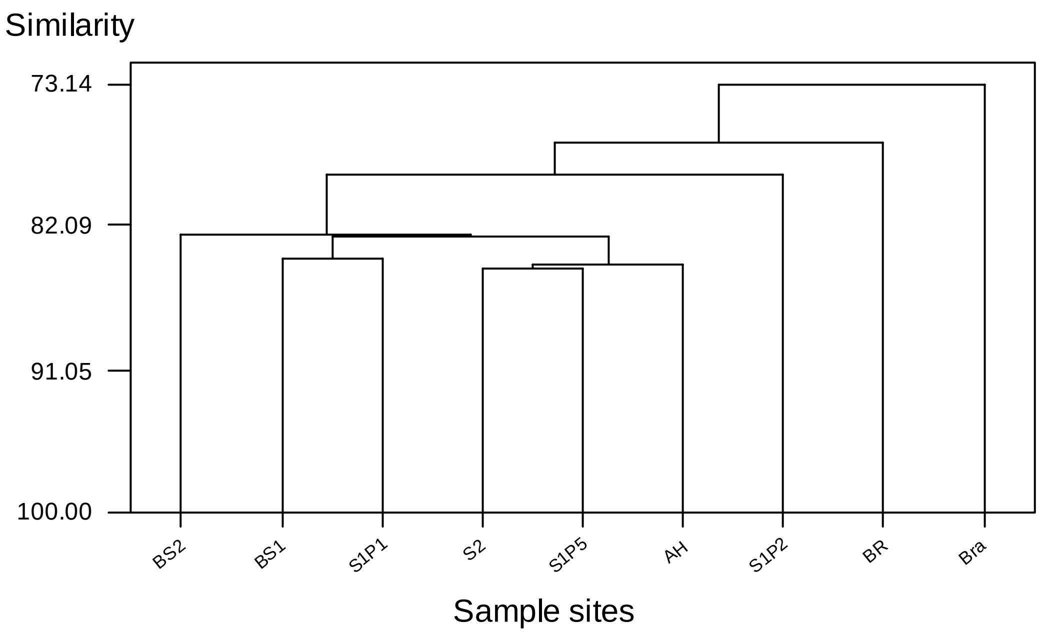

Figure 12. Dendrogram of all sites based on cluster analysis of invertebrate data

3.3.2 RIVPACS

The results from the RIVPACS analysis are presented in Appendix 7, but because the predictive nature of RIVPACS is very sophisticated and detailed, output relating to expected species is omitted for brevity. The indices used to summarise the quality of the site are presented in the Table 10.

Table 10. RIVPACS summary scores

site	**BS1**		**BS2**	
Metric	**Expected score**	**Observed score**	**Expected score**	**Observed score**
BMWP	137.1	110	168.3	125
Number of taxa	27.2	25	27.9	26
ASPT	5.03	4.40	6.02	4.81
EQI band		B		C
Probability		50.8%		91.2%

Although the second site has greater BMWP, number of taxa and ASPT scores, the expected scores were proportionally higher because the site offers a greater range of habitats. The ratio of observed to expected is higher at site one and therefore receives a higher banding (see section 3.4 for further details). Site one is grouped by RIVPACS on the basis of the environmental parameters in group 31 which is primarily made up of small Dorset streams whilst site 2 is placed in group 8 which are primarily New Forest streams.

3.3.3 PSYM

PSYM generates expected values for the number of plant and invertebrate species based on the environmental factors input and location of the ponds. The ratio of observed to expected gives an EQI which is then transformed to a 4 point scale called the Index of Biotic Integrity (0-3, where 0 represents poor quality and 3 good). Finally the individual scores of the Index of Biotic Integrity are summed to give a total for the pond and an overall percentage of the maximum (18) is used as the indicator of habitat quality. An example of the output is presented in Table 11 for one site whilst the scores are presented in Table 12. The full set of results is presented in Appendix 7.

The results (Table 12) show that four of the ponds are of good quality (above 70% IBI) and the others can be categorised as fair. However there is a warning that the data on which the heathland pool comparison is based are more limited than for other sites and therefore should be treated with caution.

Table 11. PSYM output for constructed wetland S2

Metric	**Observed**	**Predicted**	**EQI**	**IBI**
Plants				
No. of submerged + marginal plant species	23	25.15	0.91	3
Number of uncommon plant species	1	4.29	0.23	0
Trophic Ranking Score	8.53	8.57	1.00	3
Invertebrates				
ASPT	4.45	5.14	0.87	3
Odonata+Megaloptera (OM) families	4	3.30	1.21	3
Coleoptera families	2	3.80	0.53	2
Sum of individual metrics				14
Index of Biotic Integrity (%)				0.78

Table 12. PSYM scores

Metric \ site	**S2**	**S1P1**	**S1P2**	**S1P5**	**AH**	**BR**	**Bra**
Plants							
No. of submerged + marginal plant species	3	3	3	0	2	2	2
Number of uncommon plant species	0	0	1	1	0	0	0
Trophic Ranking Score	3	3	2	3	3	2	0
Invertebrates							
ASPT	3	3	2	2	2	3	3
Odonata+Megaloptera (OM) families	3	2	3	2	1	3	3
Coleoptera families	2	3	3	1	2	3	2
Sum of individual metrics	14	14	14	9	10	13	10
Index of Biotic Integrity (%)	0.78	0.78	0.78	0.50	0.56	0.72	0.56

The significant factor affecting these scores is the absence of uncommon plant species which could be explained by the recent creation date for the SUDS, leaving little time for colonisation. For the more established ponds it is difficult to explain, as a nearby pond of similar appearance (Alderney recreation ground) held at least three species that had uncommon status (*Nymphaea* sp., *Polygonum mitis* and *Ranunculus lingua*).

3.4 Comparisons with previous survey

A previous study (Armitage, *et al*., 1995) carried out in 1994 using an earlier version of RIVPACS provides data (Table 12) that can be used for comparison although some provisos are necessary. Although the sites sampled in this study matched the top two sites of the 1994 study, that study was carried out over three seasons which has the effect of increasing the sample size. It is not known whether this difference in sample season was accounted for by the earlier version of RIVPACS but it is assumed that it was because the expected scores from the 1994 study (Table 13) are higher than those generated for this study, although this may be due to an improvement in the database used to generate predictive family and species lists.

Table 13. RIVPACS summary scores from 1994 (Armitage, et al., 1995).

site	**Site 1**		**Site 2**	
Metric	**Expected score**	**Observed score**	**Expected score**	**Observed score**
BMWP	151.7	87	201.3	153
Number of taxa	26.3	18	33.6	31
ASPT	5.7	4.83	6.0	4.94

The banding system used to summarise the general condition of the stream changed between the versions of RIVPACS and so the scores generated in 1994 were used to recalculate the bands assigned using the currently accepted limits (see Tables 14 and 15).

Table 14. Banding scores for RIVPACS.

Lower limit of Taxa score	Lower limit of ASPT	Band	Grade
0.85	1.00	A	Very good
0.70	0.90	B	Good
0.55	0.77	C	Fairly good
0.45	0.65	D	Fair
0.30	0.50	E	Poor
0.00	0.00	F	bad

The lower banding values overrides any higher values and therefore sites 1 and 2 from 1994 would be both graded as C. This compares with this present study which gave a score of B to site 1 and C to site 2.

Table 15. Recalculated bands for 1994 scores.

Metric \ site	Site 1	Band	Site 2	Band
# taxa index (o/e)	0.68	C	0.92	A
ASPT index (o/e)	0.85	C	0.82	C
Overall band		C		C

3.5 Conservation value

In order to examine the differences between the stream, ponds and SUDS from the conservation value point of view, an analysis of the number of invertebrate species found within each sample was carried out using a chi square test to distinguish between two alternative hypotheses:

H_0 : that the observed frequencies are homogenous

H_1 : that the observed frequencies depart from an homogenous distribution by a factor that is greater than sampling error.

The sampling statistic was 16.11 with 10 degrees of freedom and is less than 18.31 (5% level of significance), therefore H_1 is rejected in favour of H_0 and it is concluded that there is no significant difference in the frequency of invertebrate species taken in each sample. Therefore, it can be concluded that the conservation value of the different freshwater habitats is not significantly different in terms of overall invertebrate species number.

The data collected were frequency data and the experimental design did not include replicates for each site due to time constraints, therefore statistical comparisons such as ANOVA are not appropriate as they are analysis of variance of means.

Table 16. Comparison of the number of species in major habitat groups.

Group	stream	SUDS	old ponds
Total inverts	62	52	62
Coleoptera	11	12	17
Odonata	3	8	4
Macrophytes		42	35

The total number of invertebrate species, Coleoptera species, Odonata species and macrophytes in the major habitat types are presented in Table 16 and graphed in Figure 13.

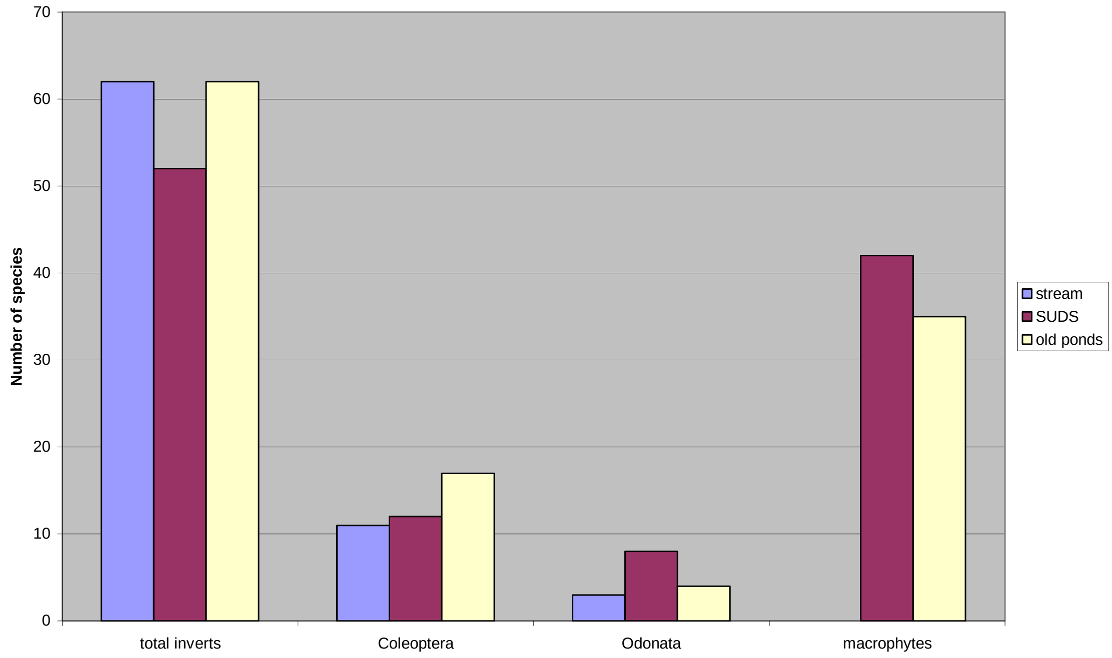

Figure 13. Histogram of the number of species in the major habitat groups.

The conservation value of the freshwater sites within the Bourne Valley are comparable to one another. From the conservation perspective the Bourne Valley supports a number of invertebrate species that are nationally rare or endangered. It is a particularly good site for dragonflies although the range of species encountered as stadia was more limited than the range reported as adults. Of the species of particular conservation interest, the White-legged Damselfly (*Platycnemis pennipes*) was found to be breeding in two of the sample sites (Bloxworth Road pond and the constructed wetland). This is an encouraging reflection of the conservation value of the newly created SUDS as this species had not been reported as breeding in the Bourne Valley previously.

The lack of uncommon plant species as noted in the PSYM analysis does not detract from the overall value of the SSSI, as there are other sites within it that have a larger range of uncommon species (for example Alderney Recreation Ground pond) and species such as the Bog Pimpernel (*Anagallis tenella*) occurring very close to some of the sample sites (BS1 and S1P5).

3.5.1 Coleoptera species and their conservation status

Bloxworth Road pond was known to have a rare reed beetle (*Donacia*), but there is no general survey information available for the Bourne Valley. As a number of the beetles encountered in this study had high conservation interest, they are reported here in detail.

All the species found have been recorded for Dorset except where indicated in Figure 14. Sources are Armitage (1994, 1995, 1996, 1999, 2001) and Dorset Environmental Records Centre. Status is from (Foster, in press). Note that those species not confirmed for Dorset are awaiting confirmation of identity. ■ = common species, ● = notable species.

Family	*Species/Genus*	*Status*	BS1a	BS2a	S2	S1P1	S1P2	S1P5	AH	BR	Bra
Haliplidae	*Haliplus lineatocollis* (Marsham)	.	■	■	■						
Dytiscidae	*Agabus didymus* (Olivier)	.		■							
	Agabus nebulosus (Forster)	.									■
	Agabus sturmii (Gyllenhal)	.									■
	Colymbetes fuscus (L.) larva	.	■			■					
	Dytiscus sp larva	.	■		■	■	■		■		
	Hydaticus sp larva	*Na/Nb (not confirmed for Dorset)*							●		
	Hydroporus pubescens (Gyllenhal)	.							■		
	Hydroporus gyllenhalii (Schiodte)	.									■
	Hyphydrus ovatus (L.) larva	.					■				
	Ilybius ater (Degeer)	.	■								■
	Ilybius fuliginosus (Fab.)	.									■
	Ilybius subaeneus (Erichson)	*Nb (not confirmed for Dorset)*	●		●						●
Gyrinidae	*Gyrinus substriatus* (Stephens)	.	■	■					■	■	
	Gyrinus urinator Illiger	*Nb*	●	●							
Hydrophilidae	*Helochares punctatus* (Sharp)	*Nb*									●
	Helophorus brevipalpis (Bedel)	.				■					
	Hydrobius fuscipes (L.)	.									■
	Hydrochus angustatus (Germar)	*Nb*									●
	Laccobius atratus (Rottenberg)	*Nb*									●
	Paracymus scutellaris (Rosenhauer)	*Nb (not confirmed for Dorset)*									●
Hygrobiidae	*Hygrobia hermanni* (Fab.)	.	■			■	■	■			
Elmidae (Elminthidae)	*Elmis aenea* (Muller)	.		■							
	Limnius volckmari (Panzer)	.		■							

Figure 14. Coleoptera species recorded and their conservation status

4.Discussion

The sampling protocols have been strictly adhered to in order to satisfy the requirements of the methodologies used and to ensure that data can be validly treated by statistical means because they were collected in the same manner. However the point has been made that these samples are not expected to be accurate censuses, they are merely representative samples and as such may have missed components of the communities. Indeed it is estimated that the standard three minute net sample used here captures only 62% of invertebrate families and 50% of species, but rises to 78% of families and 68 % of species upon a second sample (Furse, *et al*., 1981). However, as the approach was consistent across all sites, it is assumed that capture rates were similar across all sites. This assumption may be flawed as there are differences between the stream and pond sampling techniques in as much as the stream relied more on kick sampling and allowing disturbed organisms to be washed downstream into the net, whilst the pond has no flow and therefore relied on a more vigorous sweeping technique. It was apparent that some beetles were strong swimmers and evaded the net at times. Whether this had any significant effect on the samples is questionable and a literature search found no other research on the subject. Alternative means of capturing samples were considered during the experimental design phase (baited traps, ultra-violet lamps and colonisation trays) but all were rejected because of the risk of losing equipment in the field and because these methods were not a part of the standard methodologies for RIVPACS and PSYM.

4.1 Environmental determinants of habitat quality

The results of the various analyses undertaken all support the view that there is little significant difference between the sites surveyed with the one exception of the heathland pool which is consistently an outlier in the ordinations and in the diversity measures.

This is not unexpected as this pool has no direct connection with the stream unlike all the other ponds. It is therefore influenced by the chemistry of the surrounding heath and its acidic groundwater. Its small size relative to the other ponds investigated and its temporary nature (drying up in the summer) will also be factors in its dissimilarity. Hydroperiod has been shown in many studies to be a dominant factor controlling community structure (see for example, Williams (1997)).

pH was the only factor found to be significantly influencing the invertebrate community composition. It was found to be rather variable over the course of sampling and it is suspected that the different origins of the waters feeding the stream cause this variation through changes in their relative proportions. The ‘natural’ waters for a heathland such as the Bourne Valley would be more acidic.

Some similarity between the ponds and the stream samples was expected as they are all fed by the stream and clearly this would have influenced their colonisation when first created. Cluster analysis suggests a closer similarity between the stream and the SUDS phase 1 siltation pond and first lagoon than between the stream and the other ponds. This is probably due to the very high level of stream flow that passes through these ponds (95%). This is higher than originally planned and as a result the stream alongside these ponds dries to a trickle too shallow to measure flow. The fifth lagoon in the SUDS phase 1, whilst part of the series is rather different in nature and closer to the older ponds in terms of vegetation structure. It is heavily overgrown with common reed

(*Phragmites australis*) with no open water. These lagoons were originally planted with *P. australis* which established quickly but now are being invaded by local native grasses (*Glyceria maxima* and *Phalaris arundinacea*) and *Typha latifolia.*

4.2 Biological determinants of habitat quality

Discussions of the biological determinants of the community structure of ponds involve consideration of three main themes:

- Colonization, dispersal and other island biogeographical factors
- Competition, predator-prey relations and other population dynamics
- Habitat structure

4.2.1 Island biogeography

An age-series of ball-clay ponds on the Isle of Purbeck, Dorset, from 6 months to 15 years old was investigated (Barnes, 1983) as the basis for mapping colonization and successional changes. Ponds were of two types, acid and neutral, which differed in the rate and total number of species accumulated. Acid ponds exhibited slower succession and fewer species, whilst neutral ponds showed rapid colonization by macroinvertebrates relating to local sources of dispersal with variations being related to dispersal strategy and ability. The age-series demonstrated communities dominated by algivores and their predators in the early stages progressing to communities dominated by epiphyton grazers and detritivores in the later stages. Macroinvertebrates showed a colonization curve that rises rapidly during the first two years and then levels out to approximately one new taxon per year thereafter. Dipteran taxa were the most rapid colonizers, although Chironomidae were slower than other dipteran taxa. Hemiptera and Coleoptera were among the earliest colonizers. Non-dipteran taxa accumulated species much more gradually and some early colonists (from the Diptera, Odonata and

Trichoptera) became extinct in later phases. Macrophyte succession in these ball-clay ponds followed a rapid and predictable sequence of species arrival with close agreement between the number and identity of species in individual ponds in consecutive years. However some species potentially available as sources in nearby ponds did not colonize any new ponds. In acid ponds macrophytes were fewer and dominated by a single species. In relating the results to MacArthur and Wilson's model of island biogeography, Barnes concludes that non-interactive models of insular colonization, with constant immigration and extinction rates are of limited applicability in this case.

In a study of colonization of artificial ponds (Layton and Voshell, 1991), Diptera were again found to be the first macroinvertebrates to appear, presumably related to dispersal ability and reproductive capacity of adults, although unlike the previous study, Chironomidae and Ceratopogonidae were among the earliest Dipteran families. Coleoptera, Ephemeroptera and Odonata followed, as expected, and non-insect taxa came later (Oligochaeta, Nematoda and Amphipoda) and in low numbers. As a result, the artificial ponds were dominated by Chironomidae and their predators Ephemeroptera. A pattern in early colonization events can be seen from these, and other, studies which has highly vagile species appearing early on in the succession, and leaving as competitive species arrive, whereas species with low dispersal ability rely on vectors and consequently take longer to arrive. Dispersal mechanisms in freshwater invertebrates are manifold (Bilton, *et al*., 2001).

It would appear that extinction or emigration drives early colonists out quickly as they are marked differences in species richness, species rarity and community type between temporary and permanent ponds (Collinson, *et al*., 1995). This is supported by studies of water beetle assemblages (Fairchild, 2003) which demonstrate that differences in community structure are due to both habitat preferences and temporal changes in the pool of available species dispersing among ponds – landscape connectivity factors being found to be important to dispersal between ponds. Such aquatic beetle assemblages often have higher taxa richness relative to abundance than other invertebrate groups. In the absence of fish and Odonata (usually because of an unsuitable hydroperiod) aquatic beetles are usually the top predators. But they are highly vagile and the adults return to permanent water as temporary ponds dry. In a study of temporary ponds (Eyre, *et al*., 1992), two distinct assemblages specific to temporary ponds were found, separated in their ordination on the basis of their relationship to vegetation, and Eyre *et al* relates these to Southwood's habitat templet model. This model predicts that axes of decreasing durational stability and resource levels led to higher turnover and lower diversity, as has been observed. There is a resonance with Grime's disturbance and stress factors that lead to the dominance of r-selected plant species in temporary (disturbed) habitats. Other studies of temporary ponds (Williams, 1996) have shown that seasonal succession is commonplace in these habitats.

Faunal turnover has been found to be considerable even in permanent ponds (Jeffries, 1994) and a picture emerges of constant dispersal, colonization and emigration making up the dynamics of a metapopulation distributed across locally adjacent habitats (Briers, 1997). Although the importance of sources of colonisation is a consistent theme in the studies reviewed, the speed at which it occurs in relation to isolation or connectivity is less well documented. Therefore the expectation was that recently created or cleared SUDS would exhibit a community structure consistent with an early succession (dominated by Diptera). However this was clearly not the case from the invertebrate species diversity results. The reasons for this are that there are plenty of adjacent sources of colonisation for vagile species such as Coleoptera and Hemiptera whilst the stream connectivity provides a route for the species that rely on external means of transport (flood events, animal vectors etc) between ponds such as Oligochaeta, Amphipoda and Mollusca. The similarity in distribution between the three habitats suggests that such movement is rapid and commonplace.

4.2.2 Population dynamics

A number of studies have investigated predator - prey relationships in ponds. Species richness and the proportion of predators:prey was found to be related to permanence rather than pond area (Bilton, *et al*., 2001). Community structure reflects the predator:prey ratios. In another study, site age was found to have the greatest effect on aquatic beetle assemblages (Fairchild, *et al*., 2000), with predatory Dytiscidae being early colonists at younger sites and herbivorous Curculionidae and Chrysomelidae associated with particular vegetation types at older sites. The role that competitive exclusion plays in Dytiscid community structure was investigated in Swedish ponds by looking at guild size (Nilsson, 1986). Interspecific competition should theoretically limit this to two species per guild, but Nilsson found 4 or 5 species per guild coexisting and concluded that prey was sufficiently abundant to prevent interspecific competition among Dytiscid predators. Other studies (Fairchild, 2003) have commented on the high taxa richness of aquatic beetle assemblages. Experimental manipulation of predator:prey ratios in ponds (Jeffries, 2002) demonstrated that ratios converged towards the same level across all the ponds although the mechanism was through the spread of taxa into adjacent ponds rather than loss within ponds. This is another example of species across a set of ponds acting as metapopulations.

The role of predation in shaping community structure is seen as a highly significant one (Jeffries, 1996) and small ponds are seen as spatial refugia for some species seeking to escape predators in larger ponds. This lack of predation is cited as an advantage driving the exploitation of temporary waters. The presence of fish in numerous studies has a dramatic effect on macroinvertebrate community structure, as might be expected from such a major predator. Fairchild (2000) found that they reduced beetle biomass by 3 fold. In the Bourne Valley, the lack of isolation between the habitats is likely to result in the same or similar community structure across all the habitats. However there maybe opportunities to exploit spatial refugia within SUDS and the ponds through differences in vegetation architecture and the hydroperiod. Fish are present in the stream (Three-spined Stickleback, *Gasterosteus aculeatus*, and Minnow, *Phoxinus phoxinus*) and therefore predation will be a factor affecting community structure. Within the ponds and SUDS there are opportunities for fish to exploit the invertebrate resources at times, however low water levels at other times are likely to exclude them and allow other predators (Odonata and Coleoptera) to dominate and modify community structure. The exception to this highly connected environment is the heathland pools, represented by Bra in this study, which is a consistent outlier in all the analyses and demonstrates the effects of relative isolation in reduced diversity. However, this will also be strongly influenced by the size of the pool and the water chemistry of low pH derived from the acid soils. This study has not attempted to separate the effects of these factors, merely to confirm that they exist.

4.2.3 Habitat structure

The relationship between invertebrate community structure and macrophytes is a consistent theme in the literature on freshwaters (Jeffries, 1991). In a study of eutrophic ponds, pond area, macrophyte diversity and the number of species in nearby ponds (isolation) were all found to affect gastropod diversity (Bronmark, 1985). Linkage between ponds via flooding (connectivity) and macrophyte density are significant in many models (Jeffries, 2003). The architecture of submerged macrophytes is a key component of habitat structure. Increased abundance of invertebrates was found to be associated with increased fractal complexity of artificial macrophytes (Jeffries, 1993).

This present study similarly found a correlation between invertebrate diversity and macrophytes in that the BMWP scores were positively correlated with plant diversity at the 0.01 significance level. Coleopteran species diversity was negatively correlated at the 0.05 level with emergent plant cover. This suggests that as emergent plant cover increases so the diversity of habitats available decreases because the ponds tend to become dominated by a single species. For example, the constructed wetland (S2), first lagoon (S1P2) and Alder Road pond (AH) are dominated by *Typha latifolia* and the fifth lagoon (S1P5) is dominated by *Phragmites australis.* The vegetation dynamics are, however far from static and the Bloxworth Road pond (BR) shows a much greater range of species with high cover values (*T. latifolia, P. australis* and *Phalaris arundinacea).* The first species to colonise (or be planted in) a newly created habitat has the advantage of lack of competition, that subsequent invaders must overcome.

4.2.4 Chance

Chance is cited in a number of freshwater studies as a determinant of community structure (Jeffries, 1991, 1997). In an attempt to quantify the effects of chance, identical artificial ponds of the same age were repeatedly sampled (Jeffries, 1988). It was found that 80% of species occurred in likely ponds and 10% occurred in unlikely (based on proximity) ponds. In another study of chance, the genetic structure of pond zooplankton was investigated (Boileau and Taylor, 1994) and it was concluded that the element of chance due to founder effects was more persistent than that due to taxonomic composition and that it was more difficult for a new genotype of an existing species to colonize than it was for a new species altogether.

4.3 Management

The management of plant cover is of concern to conservation, especially in freshwater ecosystems where successional changes can occur very quickly. Macroinvertebrate numbers and biomass were found to be generally higher in cut areas (to generate vegetation with differing densities but also simulates grazing) of seasonally inundated wetland. Mosquitoes (Culicidae), brine flies (Ephydridae) and hoverflies (Syrphidae) were positively correlated with plant cover whereas Corixidae, Chironomidae and Hydrophilidae were negatively correlated (de Szalay and Resh, 2000). Dense emergent plant cover promoted greater diversity but open water had greater abundance of macroinvertebrates but less diversity. In an Oregon stream system, grazing by cattle was found to result in increased Chironomid (prey) and decreased Ephemeroptera (predator) abundance, presumably through the effects of organic enrichment rather than changes in habitat structure (Reed, 2003).

These ponds are subject to a regular cycle of clearing in order to provide the variety of habitat that encourages diversity – it is after all an SSSI. They are also subject to grazing by two ponies for 4 months of the year. This regime appears successful in maintaining some variety of vegetation structure and providing some disturbance to the pond edge. The fifth lagoon is the least accessible to the ponies, having the steepest banks. Grazing pressure is rather lighter than required to limit the domination of reed and the period has been increased this year. Increasing the number of grazing animals is likely to be more problematic and may cause nutrient enrichment and loss of diversity

(Reed, 2003). The larger ponds downstream of the SUDS phase 1 and the constructed wetland of SUDS phase 2 are not grazed. They have similar vegetation structure; dominated by reed beds but with great diversity of plants. The similarity between the recently constructed wetland and ponds established more than 60 years ago is remarkable and demonstrates the ease with which suitable habitats are colonised when a source is close at hand. Even uncommon species have established themselves, for example breeding White-legged damselfly, *Platycnemis pennipes*. It is hoped that the rare reed beetle (*Donacia)* known from Bloxworth Road pond, but not found in this survey, will be encouraged to spread upstream to new reed beds.

4.4 Habitat quality

The SUDS would appear to be having a positive effect on water quality of the stream judging from comparisons with RIVPACS results from 1994, but without chemical data any comparison is limited. The two stream sites were assessed as band b and c (good and fairly good) which compares with the 1994 result which fall into the c band for both. The accuracy of the assessment is also subject to a probability analysis by RIVPACS which places site 2 firmly in band c (91%) whilst site 1 only had 51% probability of being band b (otherwise dropping to band c). No such probabilities were available for the 1994 assessments so no further comparison can be made.

The upper site is of better quality than further downstream which goes against the prediction of the working hypothesis, as any expected gradients of quality should improve downstream. However this expectation is confounded by the numerous unmapped and unlicensed connections that join the stream at various points. It is highly likely that some of these are the cause of intermittently poor water quality and at present are bypassing the SUDS. Future SUDS initiatives will be based further downstream and will tackle these inputs. However the site description for site 2 from 1994 suggests that this site has become overgrown in the intervening years and this may possibly contribute to a poorer score through shading the stream. Alternatively site 1 may have been improved by the grazing that has been imposed over the last few years.

In general, these bands compare favourably with other streams in East Dorset that have been similarly assessed, for example the Win Stream (Armitage, *et al*., 1994), the Furzebrook Stream (Armitage, *et al*., 1996), Luckford Lake (Blackburn, *et al*., 1997) and the Holy Stream (Armitage and Blackburn, 2001).

5.Conclusion

The environmental conditions prevailing in the stream, ponds and SUDS (objective 1) were all comparable in terms of water chemistry, with the exception of the heathland pool, which is not surprising as they are all connected bar the pool. Alkalinity, conductivity and pH were all higher than expected for a heathland stream and may be accounted for by the mixture of water sources. The invertebrate species diversity was comparable across the three habitats (objective 2). The macrophyte diversity was comparable across the ponds and SUDS (objective 3), although plant diversity showed a greater range between sites than did the invertebrate diversity. No rare species were encountered and all sites had a number of pollution tolerant species suggesting some nutrient enrichment.

A number of relationships between environmental conditions and species diversity (objective 4) were found through correlation analysis. The most significant (0.01level) were between invertebrate diversity and water pH and between invertebrate and macrophyte diversity. A number of less significant correlations (at 0.05 level) were found, again between different measures of invertebrate diversity and water pH and between macrophyte diversity and environmental factors such as shade, although the expected correlations between diversity and age, distance from source and residency time were not found. Number of Coleoptera species was correlated with emergent vegetation which in turn was correlated with pond area and residency time. This is important from the conservation point of view as a number of notable beetles were found and successful breeding by the uncommon White-legged Damselfly (*Platycnemis* *pennipes*) was confirmed. The conservation value of the SUDS has been demonstrated by the comparable species diversity with the ponds (objective 6). The stream appears to have slightly improved in quality in the upper reaches compared with 1994 (objective 5), which supports the view that the SUDS are having a beneficial impact on the Bourne Valley freshwater habitats.

The aim of this research was to assess the SUDS in terms of their conservation value and any evidence for degradation due to pollution. No such evidence from the species diversity data could be found and the conservation value is regarded as comparable across the freshwater habitats. Although in terms of notable beetle species as many were found in the single heathland pool investigated as in all the SUDS and ponds.

There were no correlations between some factors which were expected to support the hypothesis that pollution was causing gradients of environmental quality and PCA also failed to show any discernible patterns in the data. Therefore the hypothesis was rejected in favour of the view that the stream and its onstream ponds, whether newly created as part of SUDS, or well-established old ponds, are so connected that communities show significant commonality in their makeup. This is good news for conservation attempts as it demonstrates how quickly opportunities for colonisation created will be filled.

5.1 Recommendations

The SUDS initiatives which are ongoing are expected to continue to benefit the water quality and conservation value of the Bourne Valley. Some observations follow which should be considered in this ongoing work. In planting new areas, existing vegetation should be exploited rather than importing species, as these appear to be ousted eventually by local species. Clearing small areas of the larger ponds would be beneficial to invertebrate and plant diversity by allowing greater variety of plant architecture. Grazing pressure should be maintained and if possible extended to new areas. Clearance of trees along the stream is planned in order to open up corridors for invertebrate dispersal, however some shade should be retained as it appears to be beneficial from the correlation analyses. The removal of some tree shade will also encourage macrophyte growth. Presently there are few submerged macrophytes in the stream as a whole and these should be encouraged to promote diversity. Improving connectivity between sites can also be achieved through the creation of new SUDS, which is recommended especially if they are of the intermittent flow type with high residency times. The low residency times in some of the existing SUDS may be reducing their efficacy as 20 days is the minimum recommended for effective pollutant removal (CIRIA, 1997).

5.2 Future research

This investigation has provoked many questions that would provide interesting lines of future enquiry. The relationship between habitat and the chemical parameters is one that has been approached but clearly needs more work. The ongoing work in creating more SUDS would be worthy of impact assessment, especially the two sites modified in September 2003 after this study was completed. These involved the diversion of the Bourne Stream to enter Alder Road pond higher up and to introduce a greater amount of water flow through the whole length of the pond. The second involved clearing a significant area of the Bloxworth Road pond of vegetation, which will offer opportunities for colonisation by both invertebrates and plants which would be interesting to track. The stream is also subject to change with planned channel modifications and onstream ponds being built at Coy Pond Gardens which are also expected to improve diversity.

One curious observation that would be worthy of follow up was that the common Jenkins spire shell (*Potamogyrus jenkinsi*), which occurred in the majority of samples, was very different at the second stream sample site, consisting almost completely of a population of keeled individuals which were not found at any other site. These keeled individuals are of a known variant which has readily identifiable spikes and ridges around the shell. What factors determine their occurrence and their relations with their non-keeled varieties?

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Appendix 1 Invertebrate species abundance across sample sites

Group	Family	*Species/Genus*	BS1 24/04/2003	BS1 18/08/2003	BS2 13/05/2003	BS2 11/08/2003	S2 12/06/2003	S1P1 21/05/2003	S1P2 19/06/2003	S1P5 06/07/2003	AH 28/05/2003	BR 29/07/2003	Bra 29/07/2003
TRICLADIDA	Planariidae	*Polycelis nigra group*	1	17		2	9	1		1	6		
		Polycelis felina			1								
		Dugesia polychroa	1				1				1		
		Dugesia lugubris	1	4									
GASTROPODA	Hydrobiidae	*Potamogyrus jenkinsi (Smith)*	117	28	3	38	7	10	3		52		
		Bithynia leachii	1										
	Valvatidae	*Valvata macrostoma*				1							
	Lymnaeidae	*Lymnaea peregra (Muller)*	311	71	1	12	7	31	27	13	3		
		Lymnaea stagnalis (L.)	284			6	4	67	9		59	1	
	Physidae	*Physa acuta group*	160	29	1	5	6	29	21	11	1		
	Planorbidae	*Planorbis carinatus*					1						
		Planorbarius corneus									2	2	
BIVALVIA	Sphaeriidae	*Pisidium sp.*	161	6			6	66	7	19	112	1	
OLIGOCHAETA	Tubificidae	*Tubifex sp.*	1	1				1			4		
	Lumbriculidae	*Lumbriculus variegatus (Muller)*	1	19			109	10	8	19	51		
	Naididae	*Nais sp.*			1							3	2
		Stylaria lacustris				1						8	
	Lumbricidae	*Eiseniella tetraeda*	1									1	
HIRUDINEA	Glossiphoniidae	*Helobdella stagnalis (L.)*	2	2				1	7	13	9		

		Theromyzon tessulatum (Muller)									1	1	
	Erpobdellidae	*Erpobdella octoculata*				8							
		Erpobdella testacea (Savigny)		1							3	1	
		Trocheta subviridis	1										
CRUSTACEA	Asellidae	*Asellus aquaticus (L.)*	13	16	1	10	30	2	6	77	151	25	10
	Gammaridae	*Crangonyx pseudogracilis Bousfield*	23	56	2	5	191	1	8	51	129	35	174
EPHEMEROPTERA	Baetidae	*Baetis rhodani*	5	6	1	9					9		
		Cloeon dipterum		2		3	2	22	10		18	15	2
ODONATA	Coenagriidae	*Coenagrion puella group*	1		1		9	1	7		1	14	
		Enallagma cyathigerum (Charpentier)	3							1			
		Ishnura elegans (Vander Linden)											6
		Pyrrhosoma nymphula (Sulzer)					4		1				
	Aeshnidae	*Aeshna cyanea (Muller)*											3
	Cordulegasteridae	*Cordulegaster boltonii (Donovan)*	2	1					1				
	Lestidae	*Lestes sponsa (Hansemann)*					1						
	Libellulidae	*Libellula depressa (L.)*							1	1			1
		Sympetrum striolatum (Charpentier)					6	6				1	
	Platycnemididae	*Platycnemis pennipes (Pallas)*					2					4	
HEMIPTERA	Hydrometridae	*Hydrometra stagnorum (L.)*	1	1			1	3		2		2	

	Veliidae	*Velia caprai (Tamanini)*	1		1	1			1		1		
	Gerridae	*Gerris lacustris*	1	1						1	1		2
		Gerris najas	1										
		Gerris nymph	1	1		1	1		1	2	8	3	
	Nepidae	*Nepa cinerea (L.)*	1	1						3			
	Notonectidae	*Notonecta glauca (L.)*		2		1	16	6			1		
		Notonecta maculata		4							7		
		Notonecta marmorea viridis										5	
		Notonecta obliqua		1		1							
		Notonecta nymph	6										
	Corixidae	*Callicorixa praeusta*									3		
		Sigara dorsalis			1	1	6				21		
		Corixid nymphs									25		
COLEOPTERA	Haliplidae	*Haliplus lineatocollis (Marsham)*	3	6		7	1						
		Haliplus sp larva	3			5		5	1				
	Dytiscidae	*Agabus didymus (Olivier)*				1							
		Agabus nebulosus (Forster)											1
		Agabus sturmii (Gyllenhal)											3
		Colymbetes fuscus (L.) larva	1					9					
		Dytiscus sp A	3				1	1			1		
		Dytiscus sp B						2	36				
		Hydaticus sp larva									1		
		Hydroporus pubescens (Gyllenhal)									1		

	Hydroporus gyllenhalii (Schiodte)											1
	Hyphydrus ovatus (L.) larva							2				
	Ilybius ater (Degeer)	1										2
	Ilybius fuliginosus (Fab.)											1
	Ilybius subaeneus (Erichson)	1				1						1
	Ilybius sp larva				1						2	4
	Laccophilus sp larva						16					
	Rhantus sp larva						7					
Gyrinidae	*Gyrinus substriatus (Stephens)*	2			1					1	1	
	Gyrinus urinator Illiger		4		2							
Hydrophilidae	*Helochares punctatus (Sharp)*											2
	Helochares sp larva											4
	Helophorus brevipalpis (Bedel)						1					
	Hydrobius fuscipes (L.)											1
	Hydrochus angustatus (Germar)											1
	Hydrophilid sp larva						1					
	Laccobius atratus (Rottenberg)											3
	Laccobius sp larva						2	9				
	Paracymus scutellaris											1

		(Rosenhauer)											
		Paracymus sp larva										1	
	Hygrobiidae	*Hygrobia hermanni (Fab.)*	1					1	1	3			
	Scirtidae (Helodidae)	*Larval sp A*										1	
	Elmidae (Elminthidae)	*Elmis aenea (Muller)*			1	1							
		Limnius volckmari (Panzer)			1								
		Limnius sp larva				1							
MEGALOPTERA	Sialidae	*Sialis lutaria*			1								
TRICHOPTERA	Rhyacophilidae	*Rhyacophila dorsalis*			2								
	Psychomyidae	*Lype reducta*										1	
	Hydropschidae	*Hydropsyche angustipennis*	1	8		3							
	Limnephilidae	*Limnophilus lunatus*	4		1		1				1		
		Limnephilus marmoratus	1					1					
	Leptoceridae	*Mystacides azura*				13							
	Goeridae	*Goera pilosa*				2							
LEPIDOPTERA	Pyralidae	*Nymphula sp.*					1				1	1	2
DIPTERA	Tipulidae	*Tipula sp.*					2			1		2	3
	Chironimidae		125	49	7	34	60	162		40	24	37	1
	Simulidae	*Simulium sp.*	15	58	1	9		1					
	Dixidae				1								
	Sciomyzidae	*Sepedon sp.*					1						
	Muscidae	*Limnophora sp.*						1					
	Chaoboridae	*Chaoborus sp.*					2						
	Ceratopogonidae	*Atrichopogon group*								4			
		Culicoides group									1		
	Ptychopteridae											1	

Appendix 2 Macrophyte species percentage cover values across sample sites

Common name	***Species/Genus***	**S2**	**S1P1**	**S1P2**	**S1P5**	**AH**	**BR**	**Bra**
Water plantain	*Alisma plantago-aquatica*	1	5					
Wild Angelica	*Angelica sylvestris*					1		
Fool's Watercress	*Apium nodiflorum*	1				1		
Lesser Water Parsnip	*Berula erecta*				1			
False Fox Sedge	*Carex otrubae*		1					
Pendulous Sedge	*Carex pendula*	1					1	
Remote Sedge	*Carex remota*						1	
Marsh Thistle	*Cirsium palustre*					1		1
Galingale	*Cyperus longus*					1		
Common Spikerush	*Eleocharis palustris*	1						
Rosebay Willowherb	*Epilobium angustifolium*					1		
Great Willowherb	*Epilobium hirsutum*	1	1	1		5	1	
Water Horsetail	*Equisetum fluviatile*						5	
Wood Horsetail	*Equisetum sylvaticum*						1	
Hemp Agrimony	*Eupatorium cannabinum*	5		1				
Marsh Bedstraw	*Gallium palustre*	2			1			1
Reed Sweet grass	*Glyceria maxima*					5		
Square-stalked St John's Wort	*Hypericum tetrapterum*		1					
Sharp-flowered Rush	*Juncus acutiflorus*			1				1
Jointed Rush	*Juncus articulatus*		1					
Toad Rush	*Juncus bufonius*					1		
Compact Rush	*Juncus conglomeratus*	1	1	1			1	1
Soft Rush	*Juncus effusus*	15	1			1		1
Greater Birdsfoot Trefoil	*Lotus uliginosus*	1	1	1		1		
Gipsywort	*Lycopus europaeus*	1	1			1	1	
Purple Loosestrife	*Lythrum salicaria*	5	1	1	2	1	1	1
Water mint	*Mentha aquatica*	1	1	5		1	5	
Water Forget-me-not	*Myosotis scorpioides*	1						
Sweet Gale	*Myrica gale*							1
Bog Asphodel	*Narthecium ossifragum*							1
Red Bartsia	*Odontites verna*		1					
Hemlock Water Dropwort	*Oenanthe crocata*			1		1	1	
Lousewort	*Pedicularis sylvatica*							

Reed Canary-grass	*Phalaris arundinacea*			1		5	15	
Reed	*Phragmites australis*	30		5	70	15	30	
Redshank	*Polygonum persicaria*		1	1				
Tormentil	*Potentilla erecta*							1
Common Fleabane	*Pulicaria dysenterica*			1				
Buttercup	*Ranunculus acris*	1	1					
Lesser Spearwort	*Ranunculus flammula*	1		1				
Watercress	*Rorippa nasturtium-aquaticum*	1						
Sallow	*Salix caprea*	5			1	20	10	
Sea Clubrush	*Scirpus maritimus*	1						
Water Figwort	*Scrophularia auriculata*	1	1	1		1		
Bittersweet	*Solanum dulcamara*	1	1	1				
Branched Bur-reed	*Sparganium erectum*	1				1	5	
Marsh Woundwort	*Stachys palustris*					1		
Tansy	*Tanacetum vulgare*					1		
Greater Reedmace	*Typha latifolia*	40	1	50	1	25	30	
Stinging Nettle	*Urtica dioica*	1						
Water Speedwell	*Veronica anagallis-aquatica*		40	5				
Brooklime	*Veronica beccabunga*	1	1					
Common Duckweed	*Lemna minor*	1				1	1	
Bog Pondweed	*Potamogeton polygonifolius*	5						60
Water Starwort	*Calitriche obtusangula*			1				
	Calitriche stagnalis			1				
Floating Clubrush	*Eleogiton (Scirpus) fluitans*	1						
Parrots Feather	*Myriophyllum aquaticum*						1	

Appendix 3 Results of Bivariate correlation analysis using Spearmans rank correlation coefficient.

Spearman's rho		total	n	BMWP	ASPT	invert J	invert. D	macrophyte score (rarity)	macrophyte score (TRS)	EQI invert	EQI plants
total	Correlation Coefficient	1									
	Sig. (2-tailed)	.									
n	Correlation Coefficient	0.739	1								
	Sig. (2-tailed)	0.058	.								
BMWP	Correlation Coefficient	0.429	0.667	1							
	Sig. (2-tailed)	0.337	0.102	.							
ASPT	Correlation Coefficient	-0.342	-0.109	0.018	1						
	Sig. (2-tailed)	0.452	0.816	0.969	.						
invert. J	Correlation Coefficient	-0.464	-0.487	-0.179	-0.595	1					
	Sig. (2-tailed)	0.294	0.268	0.702	0.159	.					
invert.D	Correlation Coefficient	-0.327	-0.028	0.055	-0.596	0.837*	1				
	Sig. (2-tailed)	0.474	0.953	0.908	0.158	0.019	.				
macrophyte score (rarity)	Correlation Coefficient	0.414	0.682	0.667	-0.255	-0.108	0.257	1			
	Sig. (2-tailed)	0.355	0.092	0.102	0.582	0.818	0.578	.			
macrophyte score (TRS)	Correlation Coefficient	0.714	0.523	0.5	-0.559	0.036	0.091	0.144	1		
	Sig. (2-tailed)	0.071	0.229	0.253	0.192	0.939	0.846	0.758	.		
EQI invert	Correlation Coefficient	-0.342	-0.109	0.018	1.000**	-0.595	-0.596	-0.255	-0.559	1	
	Sig. (2-tailed)	0.452	0.816	0.969	.	0.159	0.158	0.582	0.192	.	
EQI plants	Correlation Coefficient	0.072	0.364	0.45	-0.055	-0.198	0.064	0.491	0.162	-0.055	1
	Sig. (2-tailed)	0.878	0.423	0.31	0.908	0.67	0.891	0.263	0.728	0.908	.

Spearman's rho		total	n	BMWP	ASPT	invert J	invert. D	macrophyte score (rarity)	macrophyte score (TRS)	EQI invert	EQI plants
residency time	Correlation Coefficient	0.406	0.309	0.464	0.456	-0.493	-0.403	0.456	-0.174	0.456	-0.464
	Sig. (2-tailed)	0.425	0.551	0.354	0.364	0.321	0.428	0.364	0.742	0.364	0.354
Area (m2)	Correlation Coefficient	0.214	0.342	0.429	-0.234	0.321	0.455	0.414	0.214	-0.234	-0.45
	Sig. (2-tailed)	0.645	0.452	0.337	0.613	0.482	0.305	0.355	0.645	0.613	0.31
Elevation	Correlation Coefficient	0	-0.054	0.607	0.09	0	-0.145	0.27	0.179	0.09	0.559
	Sig. (2-tailed)	1	0.908	0.148	0.848	1	0.756	0.558	0.702	0.848	0.192
Distance from source (m)	Correlation Coefficient	0	0.054	-0.607	-0.09	0	0.145	-0.27	-0.179	-0.09	-0.559
	Sig. (2-tailed)	1	0.908	0.148	0.848	1	0.756	0.558	0.702	0.848	0.192
Age (years)	Correlation Coefficient	0.019	0.245	-0.412	-0.057	-0.037	0.238	-0.16	-0.075	-0.057	-0.387
	Sig. (2-tailed)	0.968	0.596	0.359	0.904	0.937	0.607	0.731	0.873	0.904	0.391
Time since clearance	Correlation Coefficient	0.288	0.245	0.126	-0.455	0.396	0.468	0.264	0.252	-0.455	-0.609
	Sig. (2-tailed)	0.531	0.596	0.788	0.306	0.379	0.29	0.568	0.585	0.306	0.147
% shaded	Correlation Coefficient	0.371	0.729	0.667	0.037	-0.111	0.245	0.505	0.334	0.037	-0.112
	Sig. (2-tailed)	0.413	0.063	0.102	0.937	0.812	0.596	0.248	0.465	0.937	0.811
% grazed	Correlation Coefficient	-0.289	-0.51	-0.144	-0.437	0.577	0.294	-0.364	0.289	-0.437	0.291
	Sig. (2-tailed)	0.53	0.243	0.758	0.327	0.175	0.522	0.422	0.53	0.327	0.526
% emergent vegetation	Correlation Coefficient	0.214	0.198	0.571	0.018	0.179	0.127	0.216	0.286	0.018	-0.45
	Sig. (2-tailed)	0.645	0.67	0.18	0.969	0.702	0.786	0.641	0.535	0.969	0.31
pH	Correlation Coefficient	-0.25	-0.378	0.036	-0.306	0.714	0.455	-0.45	0.393	-0.306	-0.36
	Sig. (2-tailed)	0.589	0.403	0.939	0.504	0.071	0.305	0.31	0.383	0.504	0.427

Spearman's rho		total	n	BMWP	ASPT	invert J	invert. D	macrophyte score (rarity)	macrophyte score (TRS)	EQI invert	EQI plants
conductivity	Correlation Coefficient	0.107	-0.288	0.143	-0.577	0.5	0.182	-0.144	0.571	-0.577	0.18
	Sig. (2-tailed)	0.819	0.531	0.76	0.175	0.253	0.696	0.758	0.18	0.175	0.699
Reference pH (8/8/03)	Correlation Coefficient	-0.182	-0.248	-0.164	-0.853*	0.909**	0.852*	0.073	0.236	-0.853*	-0.009
	Sig. (2-tailed)	0.696	0.592	0.726	0.015	0.005	0.015	0.876	0.61	0.015	0.984
Reference Conductivity (8/8/03)	Correlation Coefficient	0.071	-0.234	0.429	-0.234	0.321	0.018	0.234	0.214	-0.234	0.126
	Sig. (2-tailed)	0.879	0.613	0.337	0.613	0.482	0.969	0.613	0.645	0.613	0.788
# Odonata sp	Correlation Coefficient	-0.436	-0.248	0.218	0.505	-0.091	-0.139	0.33	-0.655	0.505	0.266
	Sig. (2-tailed)	0.328	0.592	0.638	0.248	0.846	0.766	0.469	0.111	0.248	0.564
# Coleoptera sp	Correlation Coefficient	-0.306	0.091	-0.234	0.382	-0.378	-0.11	-0.136	-0.306	0.382	0.536
	Sig. (2-tailed)	0.504	0.846	0.613	0.398	0.403	0.814	0.771	0.504	0.398	0.215
alkalinity	Correlation Coefficient	-0.086	-0.406	-0.543	-0.551	0.314	-0.118	-0.551	0.429	-0.551	0.086
	Sig. (2-tailed)	0.872	0.425	0.266	0.257	0.544	0.824	0.257	0.397	0.257	0.872
plant J	Correlation Coefficient	0.286	0.667	0.714	-0.126	0.107	0.455	0.595	0.357	-0.126	-0.018
	Sig. (2-tailed)	0.535	0.102	0.071	0.788	0.819	0.305	0.159	0.432	0.788	0.969
plant D	Correlation Coefficient	0.393	0.667	0.893**	-0.018	-0.071	0.2	0.757*	0.321	-0.018	0.144
	Sig. (2-tailed)	0.383	0.102	0.007	0.969	0.879	0.667	0.049	0.482	0.969	0.758
	N	7	7	7	7	7	7	7	7	7	7

** Correlation is significant at the .01 level (2-tailed).

* Correlation is significant at the .05 level (2-tailed).

Spearman's rho		residency time	Area (m2)	Elevation	Distance from source (m)	Age (years)	Time since clearance	% shaded	% grazed	% emergent vegetation	pH
residency time	Correlation Coefficient	1									
	Sig. (2-tailed)	.									
Area (m2)	Correlation Coefficient	0.812*	1								
	Sig. (2-tailed)	0.05	.								
Elevation	Correlation Coefficient	-0.058	-0.179	1							
	Sig. (2-tailed)	0.913	0.702	.							
Distance from source (m)	Correlation Coefficient	0.058	0.179	-1.000**	1						
	Sig. (2-tailed)	0.913	0.702	0	.						
Age (years)	Correlation Coefficient	-0.031	0.225	-0.954**	0.954**	1					
	Sig. (2-tailed)	0.954	0.628	0.001	0.001	.					
Time since clearance	Correlation Coefficient	0.58	0.919**	-0.432	0.432	0.406	1				
	Sig. (2-tailed)	0.228	0.003	0.333	0.333	0.366	.				
% shaded	Correlation Coefficient	0.691	0.815*	-0.148	0.148	0.33	0.636	1			
	Sig. (2-tailed)	0.128	0.025	0.751	0.751	0.47	0.125	.			
% grazed	Correlation Coefficient	-0.891*	-0.433	0.433	-0.433	-0.454	-0.364	-0.599	1		
	Sig. (2-tailed)	0.017	0.332	0.332	0.332	0.306	0.422	0.155	.		
% emergent vegetation	Correlation Coefficient	0.87*	0.857*	0.179	-0.179	-0.15	0.685	0.704	-0.289	1	
	Sig. (2-tailed)	0.024	0.014	0.702	0.702	0.749	0.09	0.077	0.53	.	
pH	Correlation Coefficient	-0.348	0.321	0.107	-0.107	-0.094	0.324	0.074	0.577	0.464	1
	Sig. (2-tailed)	0.499	0.482	0.819	0.819	0.842	0.478	0.875	0.175	0.294	.

Spearman's rho		residency time	Area (m2)	Elevation	Distance from source (m)	Age (years)	Time since clearance	% shaded	% grazed	% emergent vegetation	pH
conductivity	Correlation Coefficient	-0.638	-0.143	0.571	-0.571	-0.617	-0.09	-0.371	0.866*	0.071	0.607
	Sig. (2-tailed)	0.173	0.76	0.18	0.18	0.14	0.848	0.413	0.012	0.879	0.148
Reference pH (8/8/03)	Correlation Coefficient	-0.642	0.236	-0.073	0.073	0.048	0.395	-0.132	0.588	-0.018	0.527
	Sig. (2-tailed)	0.169	0.61	0.877	0.877	0.919	0.381	0.778	0.165	0.969	0.224
Reference Conductivity (8/8/03)	Correlation Coefficient	0.116	0.143	0.821*	-0.821*	-0.898**	0.018	-0.148	0.433	0.429	0.321
	Sig. (2-tailed)	0.827	0.76	0.023	0.023	0.006	0.969	0.751	0.332	0.337	0.482
# Odonata sp	Correlation Coefficient	0.309	-0.073	0.564	-0.564	-0.591	-0.33	-0.132	-0.147	0.073	-0.346
	Sig. (2-tailed)	0.551	0.877	0.187	0.187	0.163	0.469	0.778	0.753	0.877	0.448
# Coleoptera sp	Correlation Coefficient	-0.618	-0.667	-0.18	0.18	0.312	-0.691	-0.243	0	-0.775*	-0.45
	Sig. (2-tailed)	0.191	0.102	0.699	0.699	0.496	0.086	0.599	1	0.041	0.31
alkalinity	Correlation Coefficient	-0.754	-0.771	0.143	-0.143	-0.213	-0.429	-0.841*	0.878*	-0.6	0.429
	Sig. (2-tailed)	0.084	0.072	0.787	0.787	0.686	0.397	0.036	0.021	0.208	0.397
plant J	Correlation Coefficient	0.638	0.857*	-0.036	0.036	0.225	0.667	0.964**	-0.433	0.714	0.179
	Sig. (2-tailed)	0.173	0.014	0.939	0.939	0.628	0.102	0	0.332	0.071	0.702
plant D	Correlation Coefficient	0.812*	0.75	0.321	-0.321	-0.168	0.487	0.852*	-0.433	0.75	0
	Sig. (2-tailed)	0.05	0.052	0.482	0.482	0.718	0.268	0.015	0.332	0.052	1
	N	6	7	7	7	7	7	7	7	7	7
**	Correlation is significant at the .01 level (2-tailed).										
*	Correlation is significant at the .05 level (2-tailed).										

Spearman's rho		conductivity	Reference pH (8/8/03)	Reference Conductivity (8/8/03)	# Odonata sp	# Coleoptera sp	alkalinity	plant J	plant D
conductivity	Correlation Coefficient	1							
	Sig. (2-tailed)	.							
Reference pH (8/8/03)	Correlation Coefficient	0.546	1						
	Sig. (2-tailed)	0.205	.						
Reference Conductivity (8/8/03)	Correlation Coefficient	0.714	0.236	1					
	Sig. (2-tailed)	0.071	0.61	.					
# Odonata sp	Correlation Coefficient	-0.182	-0.287	0.436	1				
	Sig. (2-tailed)	0.696	0.533	0.328	.				
# Coleoptera sp	Correlation Coefficient	-0.414	-0.294	-0.667	0.009	1			
	Sig. (2-tailed)	0.355	0.523	0.102	0.984	.			
alkalinity	Correlation Coefficient	0.943**	0.471	0.257	-0.406	0.029	1		
	Sig. (2-tailed)	0.005	0.346	0.623	0.425	0.957	.		
plant J	Correlation Coefficient	-0.214	0.091	0	-0.073	-0.306	-0.886*	1	
	Sig. (2-tailed)	0.645	0.846	1	0.877	0.504	0.019	.	
plant D	Correlation Coefficient	-0.107	-0.091	0.321	0.236	-0.414	-0.886*	0.893**	1
	Sig. (2-tailed)	0.819	0.846	0.482	0.61	0.355	0.019	0.007	.
	N	7	7	7	7	7	6	7	7

** Correlation is significant at the .01 level (2-tailed).

* Correlation is significant at the .05 level (2-tailed).

Appendix 4 Principal Components Analysis Results

Principal Component Analysis: Plant cover

Eigenanalysis of the Correlation Matrix

Eigenvalue	14.040	13.349	9.852	8.760	6.925	4.074
Proportion	0.246	0.234	0.173	0.154	0.121	0.071
Cumulative	0.246	0.481	0.653	0.807	0.929	1.000

Principal Component Analysis: Coleoptera sp only

Eigenanalysis of the Correlation Matrix

Eigenvalue	2.2039	1.7082	1.1303	1.0138	0.9801	0.7607
Proportion	0.245	0.190	0.126	0.113	0.109	0.085
Cumulative	0.245	0.435	0.560	0.673	0.782	0.866

Principal Component Analysis: Coleoptera, Odonata and Trichoptera only

Eigenanalysis of the Correlation Matrix

Eigenvalue	2.1438	1.5625	1.3456	1.0302	0.8248	0.7251
Proportion	0.238	0.174	0.150	0.114	0.092	0.081
Cumulative	0.238	0.412	0.561	0.676	0.767	0.848

Principal Component Analysis: All Invertebrate sp

Eigenanalysis of the Correlation Matrix

Eigenvalue	23.758	17.358	15.092	14.104	10.296	9.531
Proportion	0.233	0.170	0.148	0.138	0.101	0.093
Cumulative	0.233	0.403	0.551	0.689	0.790	0.884

Appendix 5 Cluster Analysis Results

Pond sites with plant data.

Correlation Coefficient Distance, Single Linkage

Amalgamation Steps

Step	Number of clusters	Similarity level	Distance level	Clusters joined		New cluster	Number of obs. in new cluster
1	6	87.08	0.258	5	6	5	2
2	5	79.61	0.408	1	4	1	2
3	4	78.27	0.435	3	5	3	3
4	3	77.76	0.445	1	3	1	5
5	2	67.77	0.645	1	2	1	6
6	1	60.66	0.787	1	7	1	7

Pond sites with invertebrate data.

Correlation Coefficient Distance, Single Linkage

Amalgamation Steps

Step	Number of clusters	Similarity level	Distance level	Clusters joined		New cluster	Number of obs. in new cluster
1	6	84.68	0.306	1	4	1	2
2	5	84.52	0.310	1	5	1	3
3	4	78.78	0.424	2	3	2	2
4	3	77.99	0.440	1	2	1	5
5	2	76.77	0.465	1	6	1	6
6	1	73.14	0.537	1	7	1	7

All sites with invertebrate data.

Correlation Coefficient Distance, Single Linkage

Amalgamation Steps

Step	Number of clusters	Similarity level	Distance level	Clusters joined		New cluster	Number of obs. in new cluster
1	8	84.68	0.306	3	6	3	2
2	7	84.52	0.310	3	7	3	3
3	6	84.08	0.318	1	4	1	2
4	5	82.70	0.346	1	3	1	5
5	4	82.54	0.349	1	2	1	6
6	3	78.78	0.424	1	5	1	7
7	2	76.77	0.465	1	8	1	8
8	1	73.14	0.537	1	9	1	9

Appendix 6 RIVPACS EQI and fauna predictions

RIVPACS ecological quality index for Bourne Stream site 1

Bourne stream1 "Spring" and "Summer"

Sample biases : Spring : 2.00 Summer : 2.00 Autumn : 0.00
Estimated overall BIAS (Mean underestimation of number of taxa) : 2.04

	-- Uncorrected for Bias --						--- Corrected for Bias ---				
	Exp(E)	Obs(O)	O/E	SD	lcl	ucl	Obs(O)	O/E	SD	lcl	ucl
No. Taxa	27.2	25	0.92	0.06	0.80	1.03	27.04	0.98	0.07	0.85	1.13
ASPT	5.03	4.40	0.87	0.04	0.81	0.95	4.54	0.90	0.04	0.83	0.99

Ecological quality band of the sample :

	----------- Uncorrected for Bias ----------					--- Corrected for Bias ----		
	TAXA --		--- ASPT --		GQA	TAXA	ASPT	GQA
Band	Lower Limit	%Prob	Lower Limit	%Prob	%Prob	%Prob	%Prob	%Prob
a	0.85	84.8	1.00	0.2	0.0	97.4	.6	1.6
b	0.70	15.2	0.90	24.6	24.8	2.6	50.8	50.8
c	0.55	0.0	0.77	75.2	75.2	0.0	47.6	47.6
d	0.45	0.0	0.65	0.0	0.0	0.0	0.0	0.0
e	0.30	0.0	0.50	0.0	0.0	0.0	0.0	0.0
f	0.00	0.0	0.00	0.0	0.0	0.0	0.0	0.0
Face band		a		c	c	a	b	b

==

RIVPACS ecological quality index for Bourne Stream site 2

Bourne stream 2 "Spring" and "Summer"

Sample biases : Spring : 2.00 Summer : 2.00 Autumn : 0.00
Estimated overall BIAS (Mean underestimation of number of taxa) : 2.04

		-- Uncorrected for Bias --					--- Corrected for Bias ---				
	Exp(E)	Obs(O)	O/E	SD	lcl	ucl	Obs(O)	O/E	SD	lcl	ucl
No. Taxa	27.9	26	0.93	0.06	0.81	1.04	28.04	0.99	0.07	0.86	1.14
ASPT	6.02	4.81	0.80	0.03	0.75	0.86	4.92	0.82	0.03	0.75	0.88

Ecological quality band of the sample :

	----------- Uncorrected for Bias ----------					--- Corrected for Bias ----		
	TAXA --		--- ASPT --		GQA	TAXA	ASPT	GQA
Band	Lower Limit	%Prob	Lower Limit	%Prob	%Prob	%Prob	%Prob	%Prob
a	0.85	88.2	1.00	0.0	0.0	98.2	0.0	0.0
b	0.70	11.8	0.90	0.2	0.2	1.8	1.0	1.0
c	0.55	0.0	0.77	85.0	85.0	0.0	91.2	91.2
d	0.45	0.0	0.65	14.8	14.8	0.0	7.8	7.8
e	0.30	0.0	0.50	0.0	0.0	0.0	0.0	0.0
f	0.00	0.0	0.00	0.0	0.0	0.0	0.0	0.0
Face band		a		c	c	a	c	c

===

RIVPACS III **PREDICTION OF FAUNA**

Bourne stream1 "Spring" and "Summer"

Environmental data used:

Latitude	50 degrees 44 minutes N
Longitude	1 degrees 55 minutes W
Altitude	46 m
Distance from source	.5 km
Discharge category	1
Mean width	.9 m
Mean depth	17.0 cm
Substratum composition:	
Boulders + cobbles	0 %
Pebbles + gravel	16 %
Sand	16 %
Silt	68 %
mean substratum (phi)	5.24
Slope _	10.0 m/km
Alkalinity_	168.0 mg/l CaCO3
Mean air temperature	10.63 C
Annual air temperature range	12.15 C

Probability of group membership	Bourne stream1	31	91.4%
		5	6.9%

Predicted BMWP families in decreasing order of probability of capture (Bourne stream 1)

100.00%	Chironomidae	37.00%	Caenidae
100.00%	Oligochaeta	36.60%	Valvatidae
100.00%	Gammaridae (incl. Crangonyctidae & Niphargidae)	29.80%	Leptophlebiidae
99.90%	Limnephilidae	27.50%	Dendrocoelidae
99.90%	Baetidae	23.90%	Scirtidae (=Helodidae)
99.40%	Tipulidae	23.90%	Perlodidae
98.70%	Sphaeriidae	15.20%	Polycentropodidae
96.90%	Planariidae (incl. Dugesiidae)	14.40%	Leuctridae
94.40%	Glossiphoniidae	10.80%	Psychomyiidae (incl. Ecnomidae)
93.30%	Lymnaeidae	10.50%	Beraeidae
89.60%	Elmidae	10.40%	Gerridae
87.90%	Simuliidae	9.60%	Leptoceridae
85.90%	Dytiscidae (incl. Noteridae)	9.50%	Calopterygidae
85.90%	Hydrobiidae (incl. Bithyniidae)	9.20%	Brachycentridae
84.20%	Ephemerellidae	9.20%	Libellulidae
83.10%	Erpobdellidae	9.10%	Nepidae
82.80%	Asellidae	2.80%	Lepidostomatidae
82.50%	Physidae	2.70%	Heptageniidae
78.80%	Hydrophilidae (incl. Hydraenidae)	1.70%	Philopotamidae
73.50%	Piscicolidae	1.50%	Taeniopterygidae
71.10%	Rhyacophilidae (incl. Glossosomatidae)	1.20%	Perlidae
62.90%	Nemouridae	1.00%	Odontoceridae
58.40%	Ancylidae (incl. Acroloxidae)	0.60%	Gyrinidae
55.40%	Haliplidae	0.40%	Cordulegasteridae
55.20%	Planorbidae	0.40%	Chloroperlidae
50.50%	Goeridae	0.10%	Coenagriidae
48.20%	Sialidae	0.10%	Dryopidae
46.00%	Corixidae		
40.10%	Hydropsychidae		Exp(E)
38.90%	Ephemeridae	BMWP score	137.1
38.90%	Sericostomatidae	No. taxa	27.2
37.00%	Hydroptilidae	ASPT	5.03

RIVPACS III **PREDICTION OF FAUNA**

bourne stream 2 “Spring” and “Summer”

Latitude	50 degrees 44 minutes N
Longitude	1 degrees 54 minutes W
Altitude	28 m
Distance from source	2.8 km
Discharge category	1
Mean width	.9 m
Mean depth	17.0 cm
Substratum composition:	
Boulders + cobbles	0 %
Pebbles + gravel	45 %
Sand	45 %
Silt	10 %
mean substratum (phi)	.24
Slope	10.0 m/km
Alkalinity	120.0 mg/l CaCO3
Mean air temperature	10.63 C
Annual air temperature range	12.10 C

Probability of group membership bourne stream 2	8	40.7%
	9	38.2%
	7	12.3%
	19	4.5%
	30	1.8%
	32	1.7%

Predicted BMWP families in decreasing order of probability of capture (Bourne stream 2)

100.00%	Chironomidae	38.30%	Gyrinidae	1.90%	Neritidae
99.90%	Oligochaeta	36.80%	Asellidae	1.60%	Astacidae
99.80%	Gammaridae (incl. Crangonyctidae & Niphargidae)	36.40%	Haliplidae	0.90%	Philopotamidae
96.40%	Sphaeriidae	35.90%	Psychomyiidae (incl. Ecnomidae)	0.90%	Perlidae
96.10%	Baetidae	33.80%	Lepidostomatidae	0.30%	Aphelocheiridae
96.00%	Tipulidae	30.00%	Cordulegasteridae	0.30%	Unionidae
96.00%	Limnephilidae	27.20%	Scirtidae (=Helodidae)		
94.30%	Simuliidae	26.90%	Calopterygidae		
90.30%	Elmidae	26.80%	Caenidae		
82.80%	Dytiscidae (incl. Noteridae)	25.30%	Hydroptilidae		
78.60%	Leptophlebiidae	24.20%	Corixidae		
78.50%	Ephemerellidae	23.90%	Chloroperlidae		
77.40%	Rhyacophilidae (incl. Glossosomatidae)	20.40%	Piscicolidae		
77.40%	Hydrobiidae (incl. Bithyniidae)	17.50%	Taeniopterygidae		
74.20%	Glossiphoniidae	15.80%	Beraeidae		
73.40%	Nemouridae	15.10%	Planorbidae		
71.70%	Ancylidae (incl. Acroloxidae)	12.10%	Odontoceridae		
70.90%	Leuctridae	12.00%	Coenagriidae		
70.00%	Hydropsychidae	8.00%	Notonectidae		
67.90%	Ephemeridae	7.90%	Physidae		
66.40%	Sericostomatidae	7.80%	Gerridae		
66.40%	Hydrophilidae (incl. Hydraenidae)	7.80%	Dryopidae		
62.20%	Perlodidae	4.90%	Dendrocoelidae		
57.10%	Polycentropodidae	4.50%	Molannidae		Exp(E)
52.10%	Sialidae	3.90%	Capniidae	BMWP score	168.3
49.90%	Heptageniidae	3.80%	Libellulidae	No. taxa	27.9
49.70%	Erpobdellidae	3.80%	Aeshnidae	ASPT	6.02
45.60%	Lymnaeidae	2.90%	Valvatidae		
44.40%	Planariidae (incl. Dugesiidae)	2.50%	Brachycentridae		
42.70%	Goeridae	2.10%	Phyrganeidae		
39.30%	Leptoceridae	1.90%	Hirudinidae		

Appendix 7 PSYM Index of Biotic Integrity

Site	S2				S1P1				S1P2			
Metric	Observed	Predicted	EQI	IBI	Observed	Predicted	EQI	IBI	Observed	Predicted	EQI	IBI
Plants												
No. of submerged + marginal plant species	23	25.15	0.91	3	15	14.78	1.01	3	16	16.20	0.99	3
Number of uncommon plant species	1	4.29	0.23	0	0	2.40	0.00	0	1	2.61	0.38	1
Trophic Ranking Score	8.53	8.57	1.00	3	8.96	8.70	1.03	3	8.15	8.72	0.93	2
Invertebrates												
ASPT	4.45	5.14	0.87	3	4.38	5.09	0.86	3	4.25	5.10	0.83	2
Odonata+Megaloptera (OM) families	4	3.30	1.21	3	2	3.21	0.62	2	3	3.12	0.96	3
Coleoptera families	2	3.80	0.53	2	4	3.74	1.07	3	4	3.75	1.07	3
Sum of individual metrics				14				14				14
Index of Biotic Integrity (%)				0.78				0.78				0.78

Site	S1P5				AH				BR			
Metric	Observed	Predicted	EQI	IBI	Observed	Predicted	EQI	IBI	Observed	Predicted	EQI	IBI
Plants												
No. of submerged + marginal plant species	4	18.60	0.22	0	19	27.78	0.68	2	15	28.34	0.53	2
Number of uncommon plant species	1	2.97	0.34	1	0	4.86	0.00	0	0	4.98	0.00	0
Trophic Ranking Score	8.6	8.76	0.98	3	8.64	8.76	0.99	3	8.18	8.77	0.93	2
Invertebrates												
ASPT	4.25	5.10	0.83	2	4.05	5.13	0.79	2	4.48	5.11	0.88	3
Odonata+Megaloptera (OM) families	2	3.16	0.63	2	1	3.29	0.30	1	3	3.27	0.92	3
Coleoptera families	1	3.76	0.27	1	2	3.82	0.52	2	4	3.82	1.05	3
Sum of individual metrics				9				10				13
Index of Biotic Integrity (%)				0.50				0.56				0.72

Site	**Bra**			
Metric	Observed	Predicted	EQI	IBI
Plants				
No. of submerged + marginal plant species	10	14.61	0.68	2
Number of uncommon plant species	0	2.88	0.00	0
Trophic Ranking Score	3.7	6.30	0.59	0
Invertebrates				
ASPT	4.83	5.30	0.91	3
Odonata+Megaloptera (OM) families	3	3.80	0.79	3
Coleoptera families	2	3.78	0.53	2
Sum of individual metrics				10
Index of Biotic Integrity (%)				0.56

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