Assessment of the Impact of Nutrient Removal on Eutrophic Rivers

R&D Technical Report P2-127/TR

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ASSESSMENT OF THE IMPACT OF NUTRIENT REMOVAL ON EUTROPHIC RIVERS

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EXECUTIVE SUMMARY

1. The river Wensum, a calcareous lowland river in north Norfolk, is a site of Special Scientific Interest and a candidate Special Area of Conservation under the EU Habitats Directive as a representative of the calcareous *Ranunculus* lowland stream habitat.

2. The two main urban sewage works discharging into the Wensum, from Fakenham and East Dereham, had phosphorus stripped from their effluents by sedimentation with iron salts, from autumn 1999, funded under the AMP 3 investment programme for the water industry.

3. This project was initiated in order to understand the consequences it would have for the river's physical and ecological condition. The specific objectives of the project are shown on pages 2-3 of this report. The project commenced one year before the phosphorus-stripping, to enable one year's 'before' data to be collected.

4. The spatial pattern of total phosphorus in the Wensum showed that the effluents had clearly increased the water column total phosphorus. The mills weirs caused total phosphorus to accumulate in the sediment behind them. Bioavailable phosphorus (BAP) was related to sediment type. Sediment total phosphorus was probably bound to iron.

5. Background levels of soluble phosphorus from sub-catchments not known to be impacted by any effluents were 9-15 µg/L (of PO_4-P) and total phosphorus 39-77 µg/L. These represent the runoff from diffuse agricultural sources; no examples could be found of 'natural' sites.

6. There were significant in-stream decreases of SRP and TP downstream from the two major STWs after phosphorus removal. The impact on sediment BAP was less convincing but this may be due to the high variability of BAP due to sediment heterogeneity.

7. The phosphorus control at Dereham and Fakenham led to a decrease of 64% and 76% respectively of the loads of total phosphorus (TP) in the River Wensum. Predicted TP loads were in accordance with observed loads and decreased relatively predictably after the phosphorus removal. The soluble reactive phosphorus (SRP) concentrations were relatively low over the post-stripping period, however this was also due to the high flow regime of the year 2000-2001. Concentrations of SRP at low flow (Q95) and mean flow regime (Q50) were predicted to be respectively extremely high (415-1039 µg/L of PO_4-P) and very high (140-273 µg/L) below the STWs.

8. Aquatic plants were not found to be an indicator of the level of nutrients in the lowland rivers investigated, this over a wide gradient of soluble reactive phosphorus (11-3600 µg/L of PO_4-P) and ammonium (16-434 µg/L of NH_4-N) in a nitrate-rich environment (6.4-12.9 mg/L NO_3-N). This is contrary to general opinion. The species composition of aquatic vascular plants was predicted by the

niche theory and species demographic processes, which were also found to partially interact.

9. There were only marginal differences in species richness between rivers in the wider survey (Wensum, Nar, Bure) that could be attributed to phosphorus enrichment. The spatial species turnover along rivers was high, resulting from both natural and anthropogenic factors.
10. Fifteen recommendations for conservation, river management and future water management arising from this study are made.

CONTENTS

ACRONYMS

AWS	Anglian Water Servives plc
ASIN	Arcsine transformation
BAP	Bio-available Phosphorus
BACI	Before and After Control Impact
BOD_5	5-day Biochemical Oxygen Demand
BUR	River Bure
CATCH	Catchment area
CI	Confidence Interval
CL	Confidence Limit
COSHH	Control Of Substances Hazardous to Health
DIST	Distance of the site from the source of the river
DoE	Department of the Environment
DW	Dry weight
EA	Environment Agency
EAWEN	EA monitoring sites of the Wensum catchment area
ECAP	Eutrophication Control Action Plan
EPC_0	zero Equilibrium Phosphate Concentration
F	Filtered
FBA	Fructose-biphosphate Aldolase
GOF	Goodness of Fit
JNCC	The Joint Nature Conservation Committee
LEAP	Local Environment Agency Plan
LOG	Logarithmic transformation
LOIS	Land Ocean Interaction Study
LTR	University of Leicester Herbarium
MAFF	Ministry of Agriculture, Fisheries and Food
MF	long term mean flow (Q50)
MTR	Mean Trophic Rank
NAR	River Nar
NCA	Natural Combination of Attributes
NERC	National Environmental Research Council
NF	Not Filtered
NGR	National Grid Reference
NH_4	Ionised Ammonia
NRA	National Rivers Authority
NSA	Nitrate Sensitive Area
OM	Organic Matter
OSPAR	Oslo and Paris Conventions

PCA	Principal Components Analysis
p.e.	Population equivalent
PP	Particulate Phosphorus
Q	Discharge
Q10	high flow (10 % chance exceedance discharge)
Q95	low flow (95 % chance exceedance discharge)
R&D	Research and Development
RDA	Redundancy Analysis
SAC	Special Area of Conservation
SA(E)	Sensitive Area (Eutrophic)
SD	Standard Deviation
SIMCAT	stochastic computer modelling developed by WRc
SPA	condensed spatial data derived from a PCA
SRP	Soluble Reactive Phosphorus
SS	Suspended Solids
SSD	Silty Sediment Deposit
SSSI	Sites of Specific Scientific Interest
STW	Sewage Treatment Work
SUP	Soluble Unreactive Phosphorus
TCa	Total Calcium (sediment)
TDCa	Total Dissolved Calcium of the column water
TDP	Total Dissolved Phosphorus (filtered water samples)
TFe	Total Iron (sediment)
TON	Total Oxidised Nitrogen
TP	Total Phosphorus
TPSED	Total Phosphorus content of the Sediment
TPWAT	Total Phosphorus concentration of the column Water
TSP	Total Dissolved Phosphorus (unfiltered water samples)
UGS	Unvegetated Gravel/Sand
UNESCO	United Nations Educational Scientific and Cultural Organisation
USGS	United States Geological Survey
USEPA	United States Environmental Protection Agency
UWWTD	Urban Waste Water Treatment Directives
VRP	Vegetated Riparian Plants
VSP	Vegetated Submerged Plants
WEB	Wendling Beck
WEN	River Wensum
WIS	River Wissey
WRc	Water Research Council
WW	Wet weight

1. INTRODUCTION

1.1 Background

Elevated concentrations of phosphorus in lowland rivers of the United Kingdom has been a matter of concern for several decades. That concern has largely focussed on the effects upon downstream lakes and reservoirs (Harper 1992; Royal Commission 1992; Environment Agency 2000). Concern for the ecological consequences in flowing waters themselves has developed only in the past decade and has been focussed more recently upon candidate Special Areas of Conservation (cSAC) under the EU Habitats Directive (92/43/EEC), where rivers themselves, independent of standing water bodies, may be designated as representative of a particular ecological type. The river Wensum, the subject of this report, is such a candidate SAC.

A good example of the history of the development of the UK strategy for managing lowland eutrophication comes from the adjacent catchment to the Wensum on the east – the Bure. In this catchment, one of the major systems of the Norfolk Broadland, detailed research into the causes and effects of eutrophication has been carried out by the Environment Agency (and its predecessor organisations) over the last 25 years, in collaboration with a range of partners. This work, extensively reported in the scientific literature (e.g. Moss 1983; Phillips *et al* 1999a), indicated that phosphorus was the key limiting nutrient in the slow-flowing rivers and shallow lakes of the area (see Section 2.1.3). Nutrient budgets have shown that sources of nitrogen species were mostly diffuse and in-stream nitrogen concentrations were dependent on the catchment land use and the extent of the riparian buffer zone. Although MAFF introduced a Pilot Nitrate Scheme in designated Nitrate Sensitive Areas (NSAs) before the implementation of the European Nitrate Directive (91/676/EEC), the control of nitrogen export from the land to the streams proved to be difficult, essentially because of the intricacies of the processes involved (Foundation of Water Research 1990; Environment Agency 2000). Removal of phosphorus from sewage effluents, and in one case diversion of the effluent away from the river system, has been used to manage the effects of phosphorus enrichment. Phosphorus control by dosing with ferric salts was initially undertaken on an experimental scale by Anglian Water, and its use was gradually extended to all the major Sewage Treatment Works (STW) on the rivers Bure and its tributary, the Ant. The treatment process was substantially upgraded by the addition of "Dynasand" filters to the treatment process in 1996. Phosphorus sources, however, were mostly concentrated at industrial and STWs effluents (e.g. Moss *et al* 1984; Johnes 1996) making its control a 'mere' technological challenge (e.g. Thomas & Slaughter 1992). Phosphorus removal at STWs has been therefore one of the measures implemented to protect the Norfolk Broads from eutrophication (see e.g. Moss 1983; Moss *et al* 1988; Harper 1992; Phillips *et al* 1999a). It started on the River Ant in 1978 and was extended to the River Bure from 1986 onwards (Thomas & Slaughter 1992). After phosphorus stripping, the agricultural impact became relatively more important in the River Ant and Bure catchments (Johnes 1996; Johnes *et al* 1996; Bramwell, unpublished). As a result of the substantial investment made at these works, significant improvements in the concentrations of phosphorus in the receiving waters have been recorded, together with reductions in the length and severity of algal blooms.

While the work undertaken in the Broads had shown that phosphorus removal can be an effective management tool for slow flowing and still waters, there was little evidence available to demonstrate the likely impact of phosphorus removal in faster flowing systems. The designation of the River Wensum as a Site of Specific Scientific Interest (SSSI), as well as a candidate SAC, and a voluntary commitment by Anglian Water plc to install phosphorus removal technology at the two largest treatment works on this river funded under the industry Asset Management Plan AMP 3, provided an opportunity to evaluate the impact of phosphorus reduction from point sources in this lowland river.

Eutrophication is a pan-European problem of major concern, leading to additional Regional commitments. The Urban Waste Water Treatment Directive – UWWTD (91/271/EEC), OSPAR convention (MMC 1992, 1998) and the Water Framework Directive (2000/60/EC) are the main ones directed at water. Other Directives offer methods of indirect control: The Habitats Directive (92/43/EEC), Freshwater Fish Directive (78/659/EEC) and the Surface Water Abstraction Directive (75/440/EEC).
In lowland England and Wales, phosphorus reduction at STWs has been mainly achieved by designation of Sensitive Areas (Eutrophic) – SA(E)s – under the Urban Waste Water Treatment Directive. This was the first national driver for eutrophication control, and, in the River Thames SA(E), pilot studies were conducted to assess the contributions of different sources. Initially the Department of the Environment (DoE) did not request phosphorus removal from STWs (Kinniburgh *et al* 1997; Neal *et al* 2000a). The Environment Agency strategy was to undertake, from 31 December 1998, pilot studies combining phosphorus removal at major STWs (point discharges with population equivalents greater than 10,000) situated upstream from SA(E)s, with in-depth studies through the catchment-based Local Environment Agency Plans (LEAPs) leading to introduction of pilot Eutrophication Control Action Plans (ECAPs, extensions of the more integrative LEAPs) to determine the most appropriate management (Environment Agency 2000).

The phosphorus removal systems in the river Wensum, installed at both Dereham and Fakenham STWs, were put in under a discretionary environmental programme, in anticipation of the implementation of the Urban Waste Water Treatment and Habitats Directive. This work was carried out in addition to Anglian Water’s financial support of this R&D project. The current phosphorus removal, at both works, has no statutory drivers and was agreed with the Environment Agency as a ‘best endeavours’ approach to achieve phosphorus concentrations as low as practicable, with no detriment to the work’s existing sanitary consents. Under the AMP 3 Environmental Quality Enhancement Programme both works will require UWWTD P removal (2 mg/l total phosphorus) by 31/12/04 and 1 mg/l TP (as an annual average) under the Habitats Directive by the same date. However, with recent agreement with the Environment Agency, the obligation date for Fakenham has been brought forward to 31/12/02.

It is in this context that the Environment Agency (Anglian Region), together with Anglian Water Services plc and English Nature, supported this research. The overall objective was to investigate the phosphorus dynamics of a calcareous eutrophic river system before and after removing phosphorus from major treated sewage effluents, using the River Wensum, North Norfolk, as an example.

The specific objectives of the R&D project P2-127 defined by the Environment Agency in 1998 were:

- to undertake a literature review and collate existing data on the River Wensum
- to collect baseline data for ecological conditions prior to, and following, the implementation of phosphorus removal at two major sewage treatment works discharging into the River Wensum;
- to determine ecological trends based on these data and provide information, which will assist in the justification of future phosphorus removal operations at other locations;
- to collect information on the interaction of aquatic macrophytes with phosphorus in flowing river systems to assist with future management of macrophytes in rivers;
- to produce a written scientific report in a form suitable for use by the sponsoring bodies in evaluating future nutrient management actions on the River Wensum and similar lowland river systems;
- to produce information for the general public about the project.

1.2 Study area

The natural landscape of the temperate climate of the county of Norfolk following the post-glacial recolonisation since *circa* 8000 BP in East Anglia would essentially be a mixed oak forest of *Quercus robur* (West 1970; Godwin 1975; Ferris *et al* 1995) although Mesolithic people might have already interacted with the natural succession of vegetation using fire (Smith 1970).

Forest clearance started with the introduction of farming activities in the Neolithic (*circa* 4000 BP in northwest Europe). A pollen diagram from Old Buckenham Mere (Norfolk) illustrates the development of agricultural practices from Neolithic to Norman times in the lowlands of East Anglia: cereals have been cultivated since the bronze age (*circa* 1000 BP) and a shift from pastoral to arable farming occurred in the Anglo-Saxon period (*circa* 400 AD; Turner 1970).

At present the rural landscape is dominated by arable land, although pasture in the floodplain and scattered woodlands still remain (Boar *et al* 1994). The rich soils of the area allow highly productive arable farming and an intensive program of land drainage started in the 1940s to expand cultivation in response to the increased demand for home-produced food during and after the Second World War. Financial incentives at both the national (Agriculture Act (1947) and european (European Union Common Agricultural Policy Beckett & Bull 1999) scale have further encouraged arable farming.

The area of the present study covers four lowland river basins (the land does not rise above 95 metres OD): the Rivers Wensum, Bure, Nar, and Wissey. It is characterised by Upper chalk solid geology overlain by quaternary deposits (chalk boulder clay; glacial sands and gravel). Pictures in Appendix 1.1 illustrate the type of rivers and the modification of the landscape.

What would have been the pristine conditions? Small streams would have run through woodland, trees would have fallen across the channel and backwaters. Aquatic plants would have been, for the most part, restricted to open waters not entirely shaded by the canopy and therefore their occurrence and abundance would have been very patchy. The rivers would have meandered in the floodplain. Geomorphic processes would have driven the patterns of vegetation succession in the rivers (e.g. Robertson & Augspurger 1999). The catchments would have had a high retention capacity of water, energy and matter. Nitrogen loss would mostly have been in an organic form (Perakis & Hedin 2002). Phosphorus sources would have come from the rock weathering of the upper chalk but the dissolved phosphate (the fraction most readily available for primary producers) would rapidly have co-precipitated with calcite (House 1990) or bound with iron hydroxide (Fox 1989) in contact with the oxygenated flowing surface water.

The pristine state of these rivers and the natural cycle of the phosphorus and nitrogen elements, however, has been progressively lost with human alteration of the landscape. The rivers have been heavily engineered: their beds have been lowered and straightened to drain the surrounding land more efficiently, channels have been re-directed, straightened, widened and weirs inserted to create impoundments from which hydro-power could be generated for water mills grinding corn from the middle ages. In the 20th Century large-scale gravel extraction started and in places still occurs, in the floodplain, at the edges of which seepage of pristine groundwater once occurred. The gravel pits have generally then been flooded leaving deep water lakes. Numerous settlements and their effluents have had dramatic effects on the ecology of the rivers (Hynes 1970; Sculthorpe 1967). Nowadays, most sewage is collected and treated but this is fairly recent (mid 20th Century). In-channel plant biomass and productivity is now high because light is no longer a limiting factor since the clearance of the riparian forest and the rivers are extremely enriched in nutrients (phosphate, nitrate, ammonium) in addition to being naturally rich in minerals (major cations and anions). Even after human-alteration of their hydrology rivers are still predominantly groundwater fed (spate flows are generally not very powerful). The management strategy related to macrophytes has, so far, consisted of weed cutting to prevent summer flood-flow blockages. Silt accumulates in-channel because of agricultural practices (bare soils during the winter), landscape modification (lack of riparian vegetation) and past engineering work (weirs) and some sections of the rivers need dredging to remove it.

In the 1990s several years of drought in Britain led to water shortages and exacerbation or the appearance of eutrophication problems in lowland rivers, mostly large growths of filamentous green algae (*Cladophora*, *Enteromorpha*) smothering aquatic vascular plants (English Nature & Environment Agency 2000). There is now the increasing risk of catastrophic events for the ecology of these rivers, dominated as they are by the growth of aquatic plants and filamentous algae. Events are oxygen depletion when plant respiration exceeds the natural re-oxygenation of the water by atmospheric exchange (Dodds & Welch 2000) or toxic blooms of cyanobacteria (Bowling & Baker 1996; Codd 2000). The risks are real: cyanobacteria have been recorded in calcareous English streams (Whitton & Lucas 1997) and the conditions for development of a bloom may be present during a drought in the sluggish part of many lowland rivers (Krokowski & Jamieson 2002). High concentrations of nitrate may also be harmful for human health through intake in drinking water (Hartman 1983; Bouchard *et al* 1992; Bowling & Baker 1996).

1.3 Conservation and restoration

The River Wensum is a prime example of a *Ranunculus*-dominated calcareous lowland river specified in the European Habitats Directive and is a proposed Special Area of Conservation (SAC) under the Council Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora (English Nature 2000) – part of the European Union's Natura 2000 network (92/43/EEC). Furthermore, it is one of 27 rivers in England that have been designated as SSSI by English Nature. The notification of the River Wensum as an SSSI is primarily a legal mechanism to protect its particular conservation interest because of the wildlife it supports: *Salmo trutta* (Brown Trout), *Cottus gobio* (Bullhead), *Lampetra planeri* (Brook Lamprey), *Austropotamobius pallipes* (White-Clawed Crayfish), *Vertigo moulinsiana* (Desmoulin's snail), *Boturus stellaris* (Bittern), *Anas strepera* (Gadwall), *Aythya ferina* (Pochard), *Pipistrellus pipistrellus* (Pipistrelle), *Myotis natteri* (Natterer's Bat), *Arvicola terrestris* (Water vole), *Lutra lutra* (Otter). English Nature and the Environment Agency agreed to establish a conservation strategy for each river SSSI to identify the factors that impact the biota, to raise key conservation issues and list actions which are required to maintain and enhance the special scientific interest of the river. An action plan has been set for the River Wensum (English Nature & Environment Agency 2000). Two actions given priority were to assess the impact of nutrients on the aquatic flora and to reduce the loads of phosphorus discharged into the river. These actions reflected the growing interest of the conservation agencies, the uncertainties about phosphorus dynamics in rivers and its effects on the aquatic flora that dominates the ecology of the lowland rivers of UK (Mainstone *et al* 1994, 2000).

There are several reasons why the action plan has focused on phosphorus rather than nitrogen. Dissolved orthophosphates is rapidly adsorbed or precipitated in an aerobic environment, therefore not readily available for the primary producers. Nitrate and ammonium however are very labile and are easily taken-up by primary producers. Phosphorus loads in the study region came predominantly from point source effluents rather than from the leaching of fertilisers applied throughout the catchment by farmers (Johnes 1996). Nitrate concentrations were very high and homogeneous throughout the four river basins (Environment Agency, unpublished data), therefore the impact of nitrate on aquatic plants could not be studied. The reduction of point source phosphorus inputs by precipitation at the main sewage treatment works was much easier to implement than the control of nitrogen in a first step of a restoration attempt for the River Wensum SSSI.

The phosphorus control action plan (English Nature & Environment Agency 2000) was an opportunity to study in depth the impact of phosphorus removal on the in-stream levels and fluxes of phosphorus.

1.4 Linking aquatic macrophytes and nutrients: biomonitoring

The bio-assessment of nutrient enrichment in calcareous lowland streams using aquatic vascular plants offers two main advantages: aquatic plants are the dominant autotrophs (the development of phytoplankton is small) and nutrients have a direct bottom-up effect upon plants (Hutchinson 1975; Grime 1979), although the interaction with epiphytes and the growth of epilithic algae may also be important. Submerged plants as bioindicators should allow the characterisation of the state of the river ecosystem through its biochemical, cytological, physiological or ecological characteristics (adapted from Blandin 1986, p. 224). Most of the actual research on the use of hydrophytes has focused (1) on the distribution of species or groups of species (phytosociological communities, species life-history based group of species); and (2) on the tissue nutrient content.

A number of indices based on aquatic plants have been developed in the past, focusing on different aspects. These are:-

1) Ellenberg's indicator values of light, moisture, pH, nitrogen and salt (Ellenberg 1988; Hill *et al* 1999);
2) overall assessment of the stream quality (damage rating - Haslam & Wolseley 1981; Haslam 1982);
3) stream water quality (Empain 1978; Harding 1981; Vrhovek *et al* 1984);
4) organic pollution (Husák *et al* 1989);
5) trophic ranking for standing water (Newbold & Palmer 1979; Palmer *et al* 1992; Palmer & Roy 2001) and streams (Haslam & Wolseley 1981; Holmes & Newbold 1984; Newbold & Holmes 1987) based on alkalinity.

At present the Environment Agency increasingly uses an indicator species-based method, known as 'Mean Trophic Rank' (MTR) – an assessment system using a 10-100 scale based upon scores and cover value of indicator species, developed by Holmes (Holmes 1995, 1996; Holmes *et al* 1999). It is used for quantifying the advance of eutrophication caused by rising nutrient concentrations and also its regression caused by nutrient control, principally at sewage treatment works. Previous versions of this index (Harding 1981; Holmes & Newbold 1984; Newbold & Holmes 1987) have already been integrated in a French index (Haury *et al* 1996). The standardised sampling strategy of the MTR is influencing greatly the European Commission of Normalisation (CEN/TC 230/WG 2/TG 3). It is also proposed to incorporate this tool in the forthcoming PLANTPACS – a predictive system to assess river water quality and ecological status using macrophytes (Maberly *et al* 2000). But so far the MTR has only been superficially appraised (Newman & Dawson 1996; Dawson *et al* 1999a) and independent tests (Demars & Harper 1998; Ali *et al* 1999) have been rare and inadequate.

Ideally, biomonitoring tools should be (1) as general as possible with respect to their biogeographic application; (2) reliably indicating human impact of a particular type; (3) based on biological mechanisms; and (4) derived objectively from a sound theoretical concept in ecology (modified from Dolédec *et al* 1999). The use of aquatic macrophytes is considered further in this way.

1.5 Organisation of this report

The achievements of the project aims are described as follows: –

- The chemical impact of the STWs as well as the physical impact of the weirs and impoundments associated with the mills along the River Wensum is described in Chapter 2.1;

- The background concentration of phosphorus in sites not impacted by known point sources and the impact of the point sources in the catchment of the River Wensum are quantified in Chapter 2.2;

- The water and sediment phosphorus loads prior and after phosphorus control at two major STWs (Fakenham and East Dereham) are calculated in Chapter 2.2 and 2.3;

- The aquatic vascular plants are linked with their riverine ecosystem along a gradient of phosphorus enrichment in the Rivers Wensum, Nar, Bure and Wissey in Chapter 3;

- The findings are summarised in Chapter 4;

- Conservation, management and policy issues related to eutrophication in lowland rivers arising from the work are discussed in Chapter 5 and recommendations for the Environment Agency, Anglian Water and English Nature made.

2. PHOSPHORUS

2.1 Human impacts on the spatial heterogeneity of phosphorus and sediment characteristics in a lowland calcareous river basin

2.1.1 Introduction

Soluble Reactive Phosphorus has been monitored for many years in the main lowland rivers by the Environment Agency (Robson & Neal 1997) but not the sediment's phosphorus content or the dynamics of phosphorus at the sediment-water interface. In the UK, it is only recently with the Land Ocean Interaction Study (LOIS) programme that in-stream phosphorus patterns and underlying processes were thoroughly investigated combining sediment sample collection, fluvarium experiments and the mass-balance approach (House & Warwick 1998a, 1998b, 1999; House & Denison 2000; Bowes & House 2001), mostly based on the pioneering work of House *et al* (1995a, 1997). House & Warwick (1999) studied the longitudinal profile of the river bed sediment of the River Swale. They found little heterogeneity of river bed sediment phosphorus content or sorption properties between sites. This contrasts with results showing a high heterogeneity of total phosphorus content in the sediment at 17 sites from different lowland rivers (Clarke 2000) and in the Great Ouse estuarine system (Prastka, 1996; Nedwell *et al* 1999).

Aquatic vascular plants (the dominant primary producers in lowland calcareous rivers of England) can uptake phosphorus from the sediment or the water through their roots and their leaves (Clarke 2000). The proportion of phosphorus uptake from the sediment may depend on the species and the surface water enrichment. The sediment bio-available phosphorus (BAP) is made up by the fluxes of dissolved phosphorus of the interstitial water in dynamic equilibrium with a fraction of particulate phosphorus (bound to the sediment – Sharpley 1993).

The present section investigates the impact of two large STWs and major industrial effluents, together with physical impoundment by weirs, on the spatial heterogeneity of river bed sediment characteristics with emphasis on the phosphorus, in the Wensum.

This chapter addresses the following questions: –

- Can different river bed types account for the within site variability of the sediment characteristics?
- Do point source effluents and weirs impact on site sediment characteristics?
- What is the impact of the mill structures on the river bed?
- Can total and bioavailable phosphorus in the sediment be predicted?

2.1.2 Data collection

The data were gathered during two sampling periods. Water and sediment samples were collected during the plant growth period from late April to early October 1999 along the river and above and below anthropogenic impacts (mill weirs, STWs, industrial effluents) at 34 sites (Fig 2.1, App 2.1). In-channel river bed patterns were grouped into four main types clearly distinguished in the field: (1) unvegetated gravel/sand (UGS); (2) silty sediment deposit (SSD); (3) vegetated submerged plants (VSP); (4) vegetated riparian plants (VRP). Two replicates were generally sampled at random within each of these types. In April 2000, 14 sites spanning the same range of habitat types were resampled to investigate a wider range of sediment characteristics and their relationships. This period corresponded to the start of the water plant growing season and the river was near bank-full at the time of the survey.

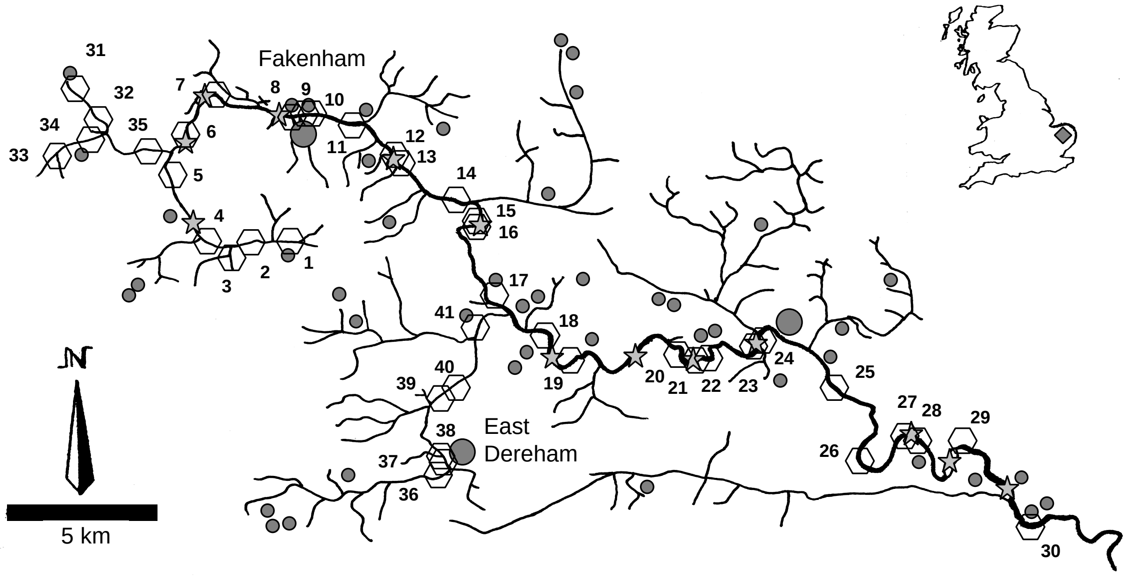


Figure 2.1 Location of the point source effluents (circles), weirs associated with the mills (stars) and sampling sites of Section 2.1 (hexagons) in the River Wensum catchment area.

2.1.3 Methods

Streamwater samples were collected in the mid channel by lowering down through the water column a 150 mL plastic bottle. Water samples were filtered through a 0.45-µm cellulose nitrate filter in the evening or the following morning. Sediment samples (5-10 cm deep) were collected by a plastic hand-scoop, then passed through a 2 mm sieve *in situ* and stored in sealed plastic bags or plastic pots. All the samples (1999 n=107; 2000 n=50) were kept in the dark as cool as possible (mainly 4-10°C) and were brought to the laboratory within three days.

Filtered water samples were analysed as soon as possible on return from the field by manual colourimetry for soluble reactive phosphorus (SRP, as in Murphy & Riley 1962) and total dissolved phosphorus (TDP, based on the acid persulphate digestion method of Rowland & Haygarth 1997). Absorbencies were measured using a HACH DR/2000 spectrophotometer. The total phosphorus (TP) was determined on unfiltered stream water samples in the same way as TDP. The sediment bio-available phosphorus (BAP) was measured with the iron oxide-impregnated paper strip technique of Sharpley (1993) modified by House & Denison (1997). This is because determination of the SRP of the interstitial water obtained by centrifugation of the sediment (e.g. Wiegleb 1984) underestimates the available fraction. The quantification of the inorganic phosphorus fraction by boiling for 20 minutes dry sediments in 1M HCl (e.g. Clarke 2000) may be too aggressive and may overestimate the available phosphorus.

Sediment total phosphorus (TP_{SED}) analysis was modified from the ignition method of Andersen (1976), Solorzano & Sharp (1980) and Rose (1995) – Demars (2002) has more detail.

Phosphorus can co-precipitate with calcium (House & Donaldson 1986), can be associated with oxidized iron (Fox 1989) and be present in organic matter. Therefore total calcium and total iron (respectively TCa and TFe) were measured by Atomic Absorption Spectroscopy (based on Standing Committee of Analysts 1983; 1987; 1992) and the proportion of organic matter (%OM) was determined with the ignition step of the TP_{SED} analysis (550ºC for 2 hours). Correlations between phosphorus and TCa, TFe, %OM may reveal which factor controls the phosphorus retention (although the redox potential, microbial activity, current velocity and bioturbation may control the phosphorus dynamics).

All glassware was washed before use, soaked in acid (10% nitric acid for at least 12 hours) and rinsed three times with deionised water and air dried. The linearity of the working standard solutions (n=7) using two independently prepared stock solutions was always $r^2>0.999$. The reliability of the measurements was generally good (Demars 2002).

The granulometry (sediment passed through 2000, 1000, 500, 250 µm sieves) of the sediment was obtained by dry sieving. Although it is not reliable for extracting precisely the proportion of silt and clay (<0.063 mm) it was a pragmatic way of assessing the proportion of coarse material (0.250-2 mm) thought to be inversely related to the proportion of silt (organic matter) and clay (minerals).

Transformations of data were applied whenever necessary (see Demars 2002).

2.1.4 Results

Can different river bed types account for the within site variability of the sediment characteristics?

All sediment characteristics across river bed types sampled in April 2000 show highly significant mean differences with BAP being the less pronounced (Fig 2.2). A common pattern emerges with unvegetated sand/gravel (UGS) being the less rich, silty sediment deposit (SSD) and vegetated riparian plant (VRP) being often the richest and vegetated submerged plant (VSP) being quite heterogeneous and intermediate. The highest levels of organic matter are found in the vegetated riparian plant (VRP).

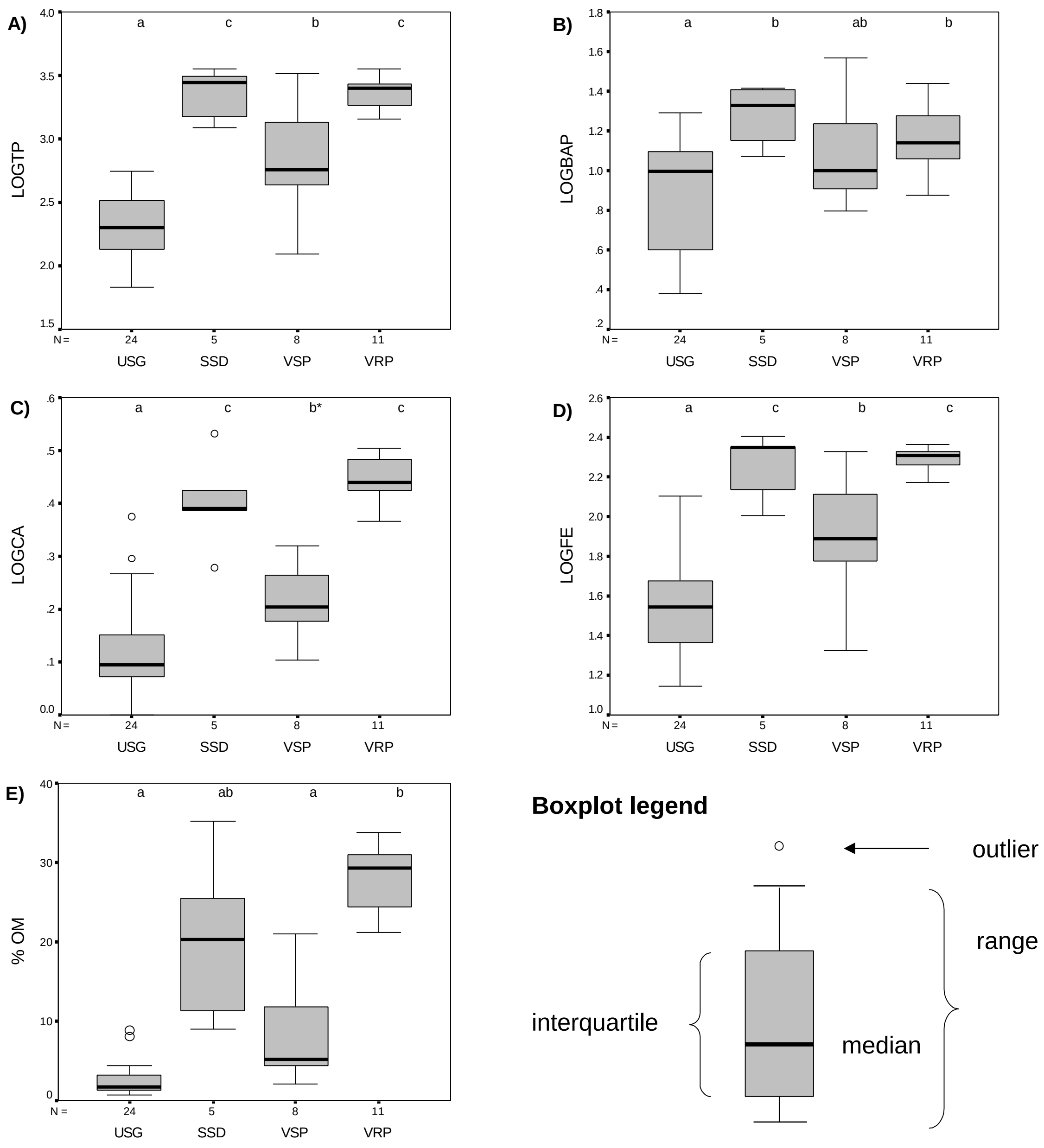


Figure 2.2 Differences in sediment characteristics among four river bed types sampled in April 2000 in the River Wensum. Significant differences (a=0.05) are denoted with a conventional lettering.
<u>Sediment characteristics</u>: A) total phosphorus (TP); B) bioavailable phosphorus (BAP); C) total calcium (Ca); D) total iron (Fe); E) organic matter (OM). Note the log transformation for all variables except OM.
<u>Habitats</u>: UGS = unvegetated gravel/sand; SSD = silty sediment deposit; VSP = vegetated submerged plants; VRP = vegetated riparian plants.

Do the sediment characteristics correlate to each other?

All the sediment variables are significantly correlated except the pairwise comparison BAP-TCa after Bonferroni correction at P < 0.05 (Fig 2.3). The tightest correlation is between TP_{SED} and TFe with r = 0.93. TP-BAP (r = 0.63) and TP-TCa (r = 0.83) were not as well correlated. BAP was generally poorly correlated to the other variables.

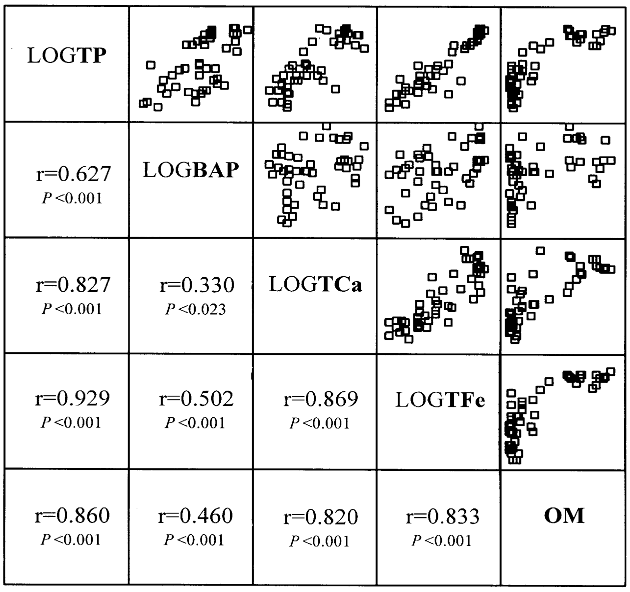


Figure 2.3 Pearson's correlation among sediment characteristics across all sites and river bed types. Relationships between pairs of sediment characteristics are significant at a=0.05 when *P* < 0.01 (after Bonferroni correction).

Do point source effluents and weirs impact on site sediment characteristics?

The bioavailable phosphorus (BAP) measurements (µg/g P of wet sediment < 2 mm) from year 1999 survey ranged from 1 to 54 µg/g and differed greatly between sites (Fig 2.4).

It is clear that point sources of phosphorus impacted greatly on the total phosphorus concentration in the water (Fig 2.5). It is not clear how much the point sources of phosphorus impact on the sediment phosphorus content, however, it seems that the weirs had larger impact than either the STWs or the industrial effluents on the TP_{SED} (Fig 2.6). This is clear from each pair of sites situated just above and below a weir (WEN140/150, WEN190/200, WEN260/270). BAP seemed to have been less impacted by the weir than by the STWs (Fig 2.6).

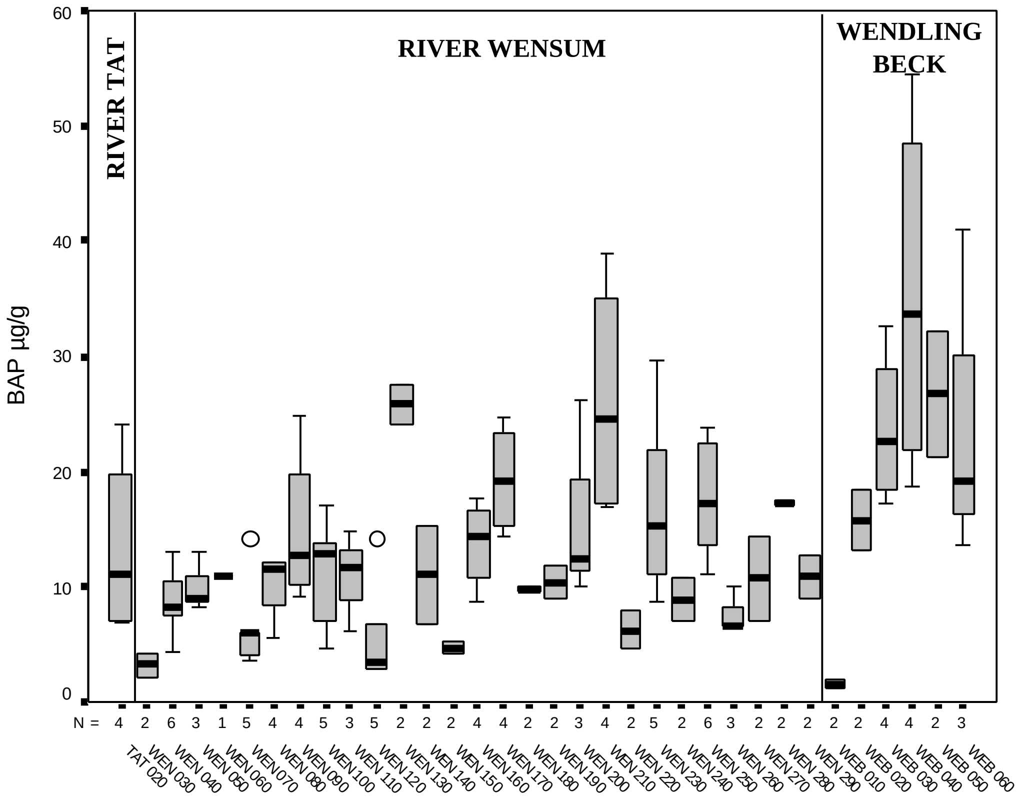


Figure 2.4 Longitudinal profile of bioavailable phosphorus (µg/g P wet weight) at 34 sites sampled in summer 1999 in the Rivers Tat, Wensum and the Wendling Beck. N = number of samples per site. Two samples were generally taken at random within the main river bed types.

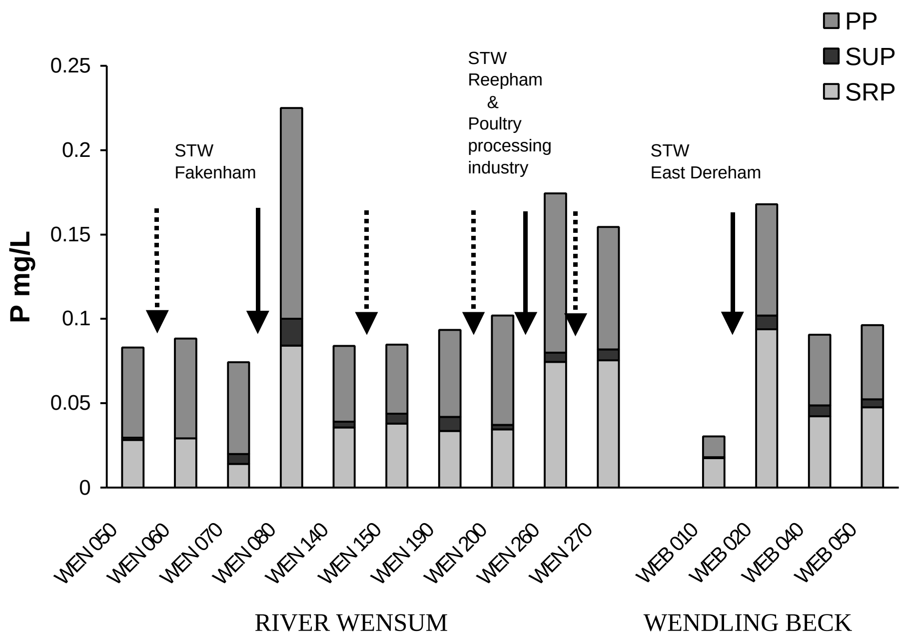


Figure 2.5 Solute forms of phosphorus in April 2000 along the River Wensum and the Wendling Beck. SRP = soluble reactive phosphorus; SUP = soluble unreactive phosphorus; PP = particulate phosphorus. SRP + SUP + PP = total phosphorus. Plain arrows represent major point source impacts and dashed arrows indicate impoundments and weirs associated with the mills.

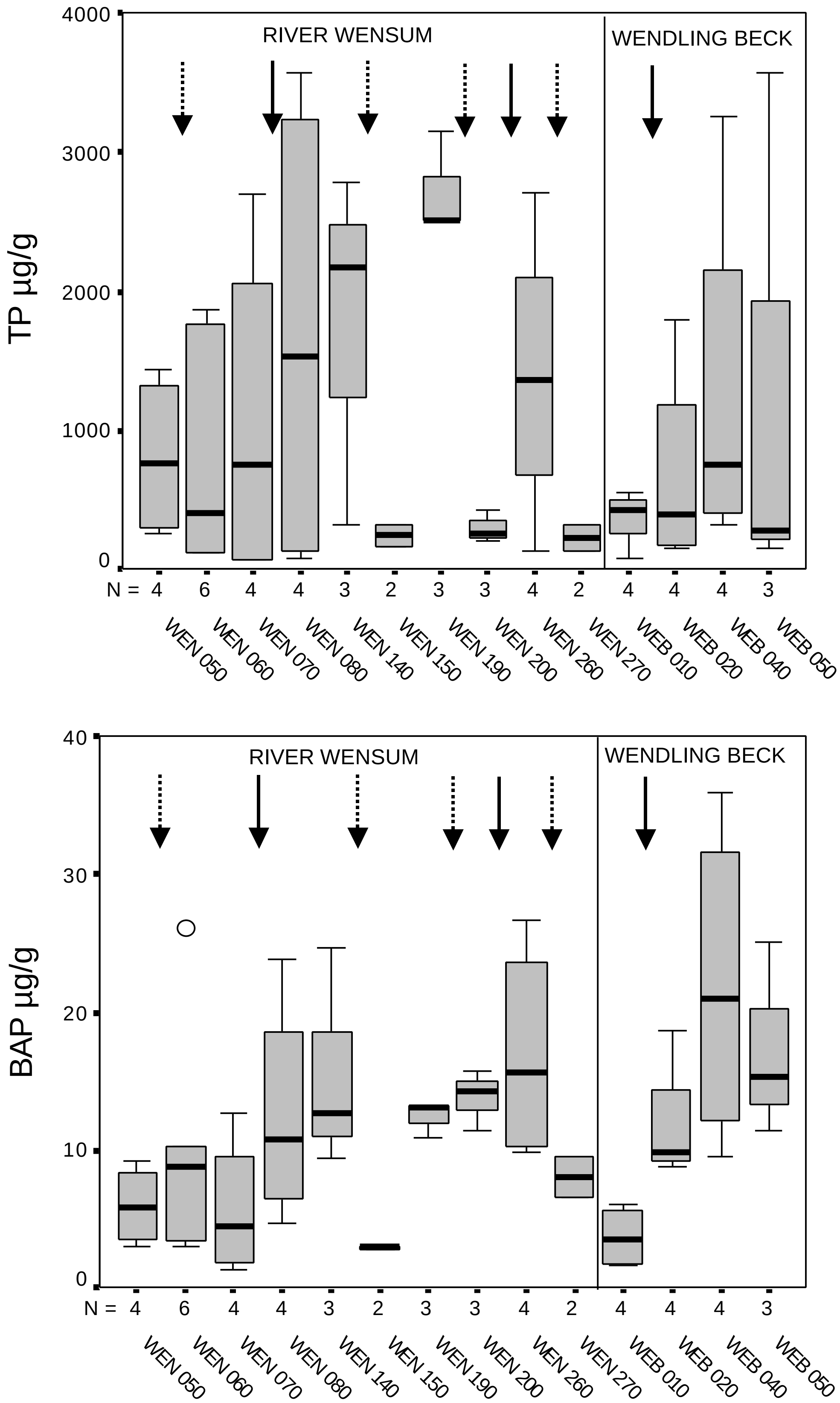


Figure 2.6 Total (TP as P µg/g DW) and bioavailable (BAP as P mg/g WW) phosphorus along the River Wensum and the Wendling Beck plotted by site, with anthropogenic impacts indicated by arrows (dashed = weirs; plain = major STWs and effluents). N = number of samples per site.

What is the impact of the mill structures on the river bed?

Figure 11 illustrates the great local variation in river width, which is also coupled with breaks in the river bed slope. This is the result of engineering works carried out for the ancient mill industry and it alters greatly the current velocity (Kemp *et al* 1999). The River Wensum is consequently divided (Fig 2.7) by the mill structures (impoundments and weirs) into long silty stretches, situated above the mill structures, which alternate with short gravelly sections. The observed mismatch in figure 2.7 between silt cover and width local variation comes from differences in sampling accuracy and problems in scaling the three different datasets (see Boar *et al* 1994).

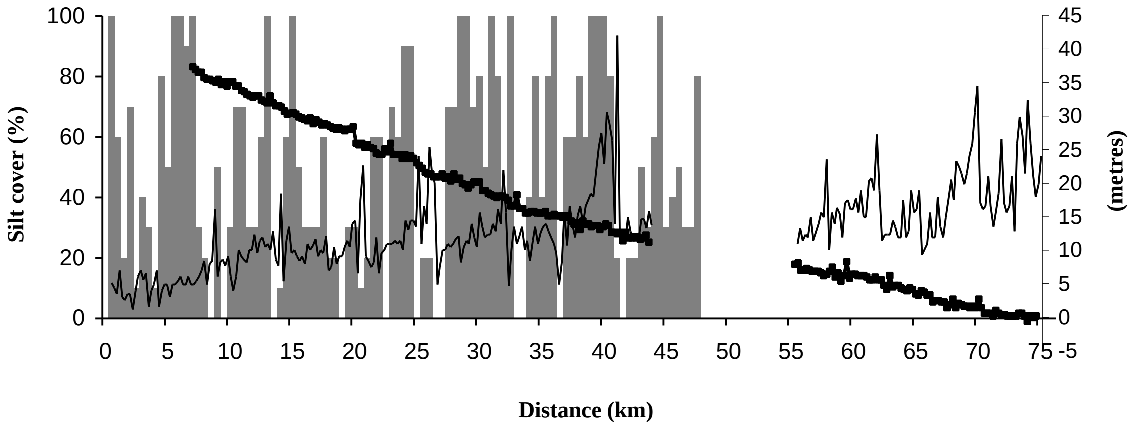


Figure 2.7 Longitudinal profile of the River Wensum: percentage of silt cover (grey area), bankfull width (line) and river bed altitude (linked squares). Modified from Boar *et al* (1994), with permission from the author. Note that there is a mismatch between the three variables due to the different accuracy and scaling of the three datasets. No silt cover data available for lower 27 km.

Can total and bio-available phosphorus in the sediment be predicted?

Organic matter was an extremely poor predictor of BAP (r^2 = 0.04, $P < 0.05$). but predicted sediment granulometry sufficiently well (adjusted r^2 = 0.81, $P < 0.001$) to use it as a surrogate for granulometry. The tight correlation TFe-OM and TP-OM would therefore indicate that the granulometry is also well correlated to TFe and TP_{SED}.

SRP was a weak predictor of BAP (r^2 = 0.44, $P < 0.001$) but could partly explain the high BAP measurements in the Wendling Beck below East Dereham STW (from site WEB020 to WEN060). This is confirmed by TP_{WAT} being a weak although significant predictor of BAP (r^2=0.34; P<0.001). BAP can significantly be better predicted by multiple regression with TP_{WAT} and TP_{SED} as independent variables (r^2=0.51; P<0.001). TP_{WAT} was not a significant predictor of TP_{SED}.

2.1.5 Discussion

In April 2000, the river was near bank-full. The proportion of particulate phosphorus (PP) was therefore higher and the concentration of total phosphorus (TP) lower than during low flows; but the general observed pattern of TP along the River Wensum is similar whatever the flow conditions are (Environment Agency, unpubl.; Demars, unpubl.).

The range of measured BAP concentration (1-54 µg/g) was smaller than findings from other English lowland rivers (House & Denison 1997; 1998), while TP_{SED} (66-3570 µg/g) was comparable to other British studies from lowland rivers (Rose 1995; House & Denison 1997, 1998; Clarke 2000; Clarke & Wharton 2001). The comparability of the results is limited by the different methodology used.

It is hard to attribute a causal relationship from the close relationships occurring between $LOGTP_{SED}$, LOGTCa, LOGTFe and OM. The tight relationship between iron and total phosphorus would suggest however, that iron may be responsible for the phosphorus retention in the sediment. The mechanisms involved could however be unravelled by further sediment phosphorus fractionations (e.g. de Groot & Golterman 1990; Paludan & Jensen 1995; Qualls & Richardson 1995; Golterman 1996) and manipulative experiments (see House & Denison 2000).

The impoundments along the river dramatically impact upon the dynamics of in-stream flow which results in extensive problems of siltation in the River Wensum (Fig. 11; Boar *et al* 1994). This explains the very high TP_{SED} concentrations recorded above the weirs and might have an effect on a mass-balance phosphorus budget of the River Wensum catchment area (see e.g. House & Warwick 1998a, 1999; House & Denison 2000; Bowes & House 2001). Thus the ecology of the river may not follow downstream changespredicted from river ecosystem theory (Vannote *et al* 1980; Ward & Stanford 1983).

Different plant species (or morphological types) are known to affect their associated sediments by altering the local flow dynamics within their patches (Haslam 1978; Sand-Jensen & Mebus 1996). This would explain the BAP variability in vegetated submerged plants (VSP). This explanation is however somewhat undermined by a recent survey at 17 sites in lowland rivers showing a high variability of inorganic and total phosphorus across sediments supporting the same species (Clarke 2000; Clarke & Wharton 2001).

2.1.6 Conclusion

The STW effluents had clearly increased the water column total phosphorus concentration. The weirs had a clear impact on the total phosphorus content of the sediment. Since these two forms of total phosphorus predicted the bioavailable phosphorus in the sediment (r^2=0.51), this means that the interaction of the weirs and STWs together had the greatest impact on sediment phosphorus availability. The within site variability of the sediment characteristics was tightly linked to the river bed types. The sediment total phosphorus was extremely well correlated to total iron (r=0.93) suggesting a strong association between the two parameters.

2.2 The impact of treated sewage effluents on the in-stream phosphorus dynamics of a calcareous lowland river basin

2.2.1 Levels of phosphorus enrichment in the lowland rivers of England

The pristine level of phosphorus in English lowland rivers is not known, although the mean phosphorus concentration found in a catchment entirely of woodland may provide a hint: 4 µg/L of soluble reactive phosphorus and 40 µg/L of total phosphorus in Warren House Stream (3.9 km^2), a sub-catchment of the Bure (Moss *et al* 1988). The concentration of phosphorus from a calcareous aquifer or springs would be a better indicator. In France, total phosphate from calcareous groundwater was 20 µg/L (12/07/1989, well n°1, Verdun; Sigg *et al* 1994). Numerous spring-fed streams from the calcareous aquifer of the Upper Rhine Plain (France) have been well monitored and offer a fair estimate of 8 (6.4-12.9) µg/L of orthophosphate (Carbiener *et al* 1990; see Stainton 1980 for comparison with SRP).

Reported SRP concentration from streams not impacted by point source effluents in lowland English rivers are rather high: 59 µg/L in the River Wey (House & Denison 1998); 106 µg/L in the Great Ouse (House & Denison 1997); 52 (39-76) µg/L in four tributaries of the Welland predominantly dominated by pasture (Evans 1999). This could be explained by unknown point sources (e.g. seepage from septic tanks) or differences in land use and landscape features (e.g. Johnes *et al* 1996). However in the flatter catchments of Norfolk (spring area 50-80m altitude), lower concentrations were measured: 20 (18-21) µg/L in tributaries of the River Nar running across woodland, pasture and arable land; 23 µg/L in Beachamwell Stream, a tributary of the Wissey (Demars, unpublished); 31 µg/L (total dissolved phosphorus) in Spixworth Beck (40.3 km^2) and 16 µg/L in Cockshoot Stream (0.6 km^2), both situated in the Bure catchment (Moss *et al* 1988); around 20 µg/L in the River Waveney (although here small point sources discharging into the drains were probably balanced by the buffering mechanism of the fen; Pitt 2001).

The vast majority of streams in the lowlands of England have high levels of nutrients because they are widely impacted by sewage treatment works (STWs) point sources and intensive agriculture diffuse sources (e.g. Casey & Newton 1973; Edwards 1973a; Osborne 1981; Moss 1983; Moss *et al* 1988; Heathwaite *et al* 1996; Muscutt & Withers 1996; House & Denison 1997, 1998; House *et al* 1997; Kinniburgh *et al* 1997; Pinder *et al* 1997; Russell *et al* 1998; Evans 1999; Haygarth & Jarvis 1999; Neal & Robson 2000; Neal *et al* 2000b). The contribution of atmospheric deposition (Owens 1976; Williams 1976; Harper 1992; Gibson *et al* 1995; Newman 1995) and rock weathering (Newman 1995) sources are comparatively insignificant, even for the standing waters of the Norfolk Broads (Osborne 1981).

2.2.2 Phosphorus dynamics

Mass balance approaches have shown that large quantities of phosphorus, coming from the STW effluents, must be stored in the sediment of the rivers (Osborne 1981; Moss *et al* 1988; House & Warwick 1998a), most likely within the riparian habitat and silty stretches (e.g. situated above the weirs, Chapter 2.1). Determination of the Zero Equilibrium Phosphate Concentration (EPC_0, the concentration of soluble reactive phosphorus at which the flux of phosphorus sediment adsorption equal the flux of phosphorus sediment release – House & Denison 2000) of river bed sediment generally showed a net uptake of soluble phosphorus, especially downstream from STW effluents where high concentrations of soluble phosphorus prevailed (House & Denison 1997, 1998; House & Warwick 1999). EPC_0 and adsorption affinities of river bed sediment showed a higher retention capacity of phosphorus by fine particles (silt and clay) rather than coarse material (House & Warwick 1999). A rapid uptake of soluble phosphorus by suspended sediment also occured especially during storm events (House *et al* 1995a; House & Denison 1998, House & Warwick 1998b).

In calcareous lowland rivers, dissolved phosphorus is controlled by the interaction of a buffer mechanism (dynamic equilibrium between dissolved phosphorus of the water and the phosphorus adsorbed by the sediment – Froelich 1988), coprecipitation with calcite and iron hydroxide (House & Donaldson 1986; Golterman 1988; Fox 1989; Hartley *et al* 1997; House & Denison 2000), redox potential (Mortimer 1971; Patrick *et al* 1973; Patrick & Khalid 1974; House & Denison 2000), current velocity (House *et al* 1995b) and biological interactions (Gachter *et al* 1988; Gachter & Meyer 1993; House *et al* 1995a). The mechanisms involved in the interaction of phosphate with sediment are not yet clear (Froelich 1988; House & Denison 2000; Bowes & House 2001; Golterman 2001).

It has been hypothesized that a substantial fraction of the sediment phosphorus moved along the river bed as bed load (Osborne & Moss 1977). Indeed, the constant remobilisation of in-stream sediment may not be negligible in groundwater-fed lowland rivers (Walling & Amos 1999). Net retention of phosphorus measured by House & Warwick (1998a) was also attributed to bed-sediment and primary producers. However, the impact of the fluxes of suspended sediment has probably been underestimated in many nutrient budgets (e.g. Osborne & Moss 1977; Moss *et al* 1988; Johnes 1996) because of the relatively low sampling effort compared to continuous monitoring (Wass *et al* 1997; Walling & Amos 1999). Net phosphorus losses have probably been overlooked due to the episodic nature of phosphorus export and the intricacies of the factors involved (House *et al* 1998; Bowes & House 2001). Moreover the net retention of phosphorus by aquatic plants was found rather negligible (House *et al* 2001).

The EPC_0 reported in the lowland rivers of England, impacted by major STWs, was much lower than the dissolved phosphate (House & Denison 1997, 1998; House & Warwick 1999), i.e. far from the equilibrium expected in natural rivers (Froelich 1988). The anthropogenic phosphorus input therefore exceeds the buffering capacity of the ecosystem (Froelich 1988).

2.2.3 Control

The River Wensum has four tributaries not impacted by point discharges. It is therefore ideal to evaluate the impact of diffuse sources alone, which is an important step to help setting pragmatic nutrient targets for the British lowland rivers (Moss *et al* 1988; Mainstone *et al* 1994, 2000; Dodds & Welch 2000; Environment Agency 2000).

As the mean dissolved phosphate concentration decreases to under 50 µg/L however, there is then a possibility of net release of phosphate from the sediment as the phosphate buffer mechanism is re-established with the EPC_0 value ranging from 0.6 to 40 µg/L (Froelich 1988). In the upper River Wey, a chalk stream not impacted by known point source effluents, with a mean soluble reactive phosphorus of 59 µg/L, a mean EPC_0 of 53 (14-79) µg/L was reported and alternative sorption/desorption of dissolved phosphate by the river bed sediment were observed between the seasons (House & Denison 1998). The role of the biota was also found to increase with SRP concentration in the column water approaching EPC_0 values (House *et al* 1995a).

2.2.4 Aims

The general aims of this section are to measure the relative contributions of the phosphorus sources, the interaction of phosphorus with sediment, and to investigate the impact of phosphorus removal at STWs in a lowland calcareous river basin. The specific purposes of this study are: -

- to find the range of phosphorus concentrations within the River Wensum basin;
- to find the level of phosphorus concentrations due to diffuse sources alone;
- to study the impact of STWs on some water and sediment characteristics, especially phosphorus;
- to study the relationships between sediment characteristics;
- to predict the phosphorus content in the suspended solids and river bed sediments;
- to assess the impact of phosphorus removal below the STWs on the in-stream water and sediment forms and levels of phosphorus.

2.2.5 Data collection

The collection of water and sediment was conducted as in section 2.1.2. Stream water samples were filtered *in situ* as soon as collected using a filtering unit and filter first rinsed with site water. Two random replicates of sediment (5-10 cm deep) were collected using a plastic hand-scoop, then passed *in situ* through a 2mm sieve, and the <2mm fraction stored in plastic pots. All the water and sediment samples (each season n=18) were generally collected within 30 hours and kept in the dark as cool as possible (mainly 4-10°C) while being transported to the laboratory.

2.2.6 Methods

In the laboratory, samples were analysed, unless otherwise stated, as in section 2.1.3 for ammonium (NH_4 – following the method of Havilah *et al* 1977), SRP, TDP and TP_{WAT}. Suspended solid concentrations were measured by filtration of 500 mL of water through 0.45-µm cellulose nitrate membranes. A subsample (50 mL) of filtered water was acidified for subsequent analysis of total dissolved calcium (TDCa). Five sediment characteristics were determined: bioavailable phosphorus (BAP); total phosphorus (TP_{SED}); total calcium (TCa); total iron (TFe); and proportion of organic matter (%OM). NH_4/SRP/BAP and TDP/TP analyses were generally started within 48 and 62 hours respectively after collection.

A preliminary survey was carried out in April and 29th June 2000 at 42 sites (Fig 2.8 and App 2.2) to identify the spatial heterogeneity of total phosphorus (TP) and Soluble Reactive Phosphorus (SRP) concentrations in the river Wensum catchment. Temporal variability (3-4 October 2000, 30-31 January 2001, 10-11 April 2001, 28-29 June 2001) of the water and sediment characteristics were further investigated at nine sites along a cumulative gradient of anthropogenic phosphorus input: four sites impacted only by diffuse sources (diffuse – WTR005, BLW010, PSB010 and RPS030); two sites impacted by distant (> 5 km) small Sewage Treatment Works (distant STW – WEB010 and WEN070) and three sites impacted by an effluent from a nearby (< 1 km) large STW (nearby STW – WEN080, WEB020 and RPS040 – Fig 2.8). The geomorphology of the river stretches was comparable between sites. Only the major river bed type was sampled - unvegetated sand/gravel - to avoid direct interactions with the primary producers (i.e. filamentous algae and vascular plants) and to obtain comparable measurements. The period covered a range of hydrological events (see Fig. 2).

To assess the efficiency of phosphorus removal, water samples were collected in 1999 (19/05, 20/08, 12/10), 2000 (18/04, 29/06, 3/10) and 2001 (31/01, 11/04, 29/06), 200 metres above and 100 metres below the major STWs at WEN070/WEN080 (Fakenham) and similarly at WEB010/WEB020 (East Dereham – no data in October 1999). Sediment samples were collected each year during spring at the two pairs of sites.

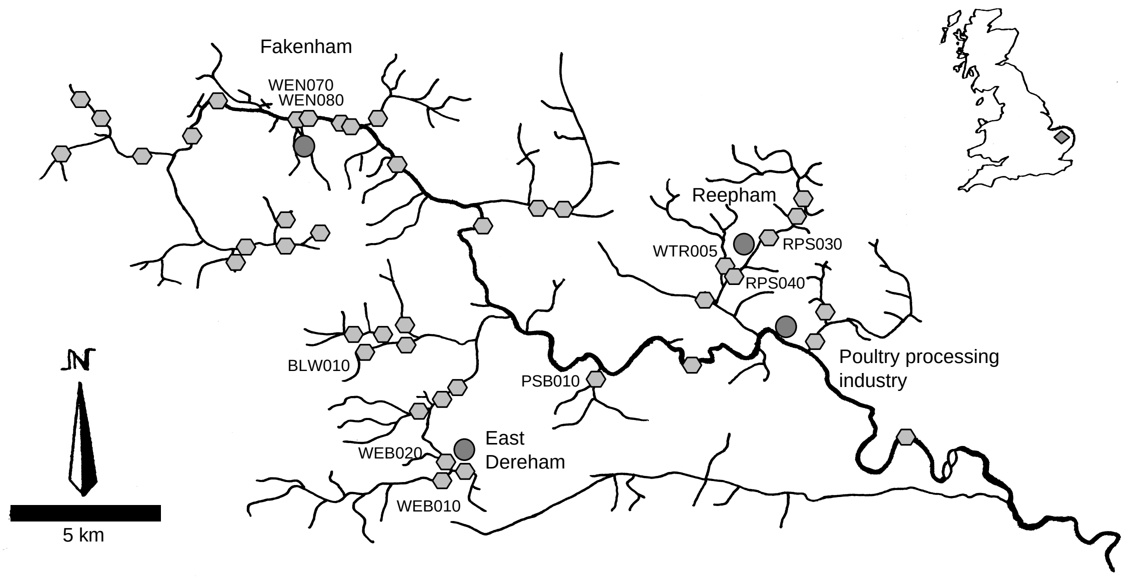


Figure 2.8 Location of the main point source effluents (circles) and sampling sites of Section 2.2 (hexagons) in the River Wensum catchment area.

2.2.7 Results

Range of solutes and sediment characteristics in the River Wensum basin

Figure 2.9 shows the lognormal distribution of the mean SRP and TP resulting from the preliminary surveys across the catchment of the River Wensum. Six sites (14%) had a SRP concentration lower than 25 µg/L and 14 lower than 50 µg/L (33%). Only three sites (7%) had a TP lower than 50 µg/L and 14 sites (33%) lower than 100 µg/L. Table 2.1 summarizes the mean concentration of the solutes and sediment characteristics for the selected sites.

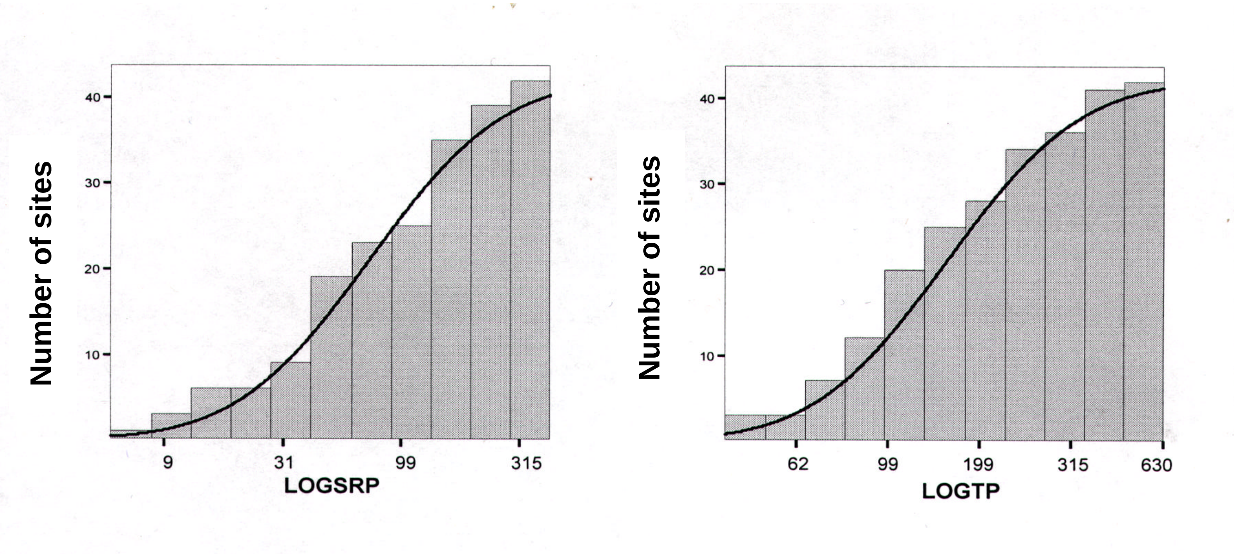


Figure 2.9 Lognormal distribution of SRP and TP (µg/L PO_4-P) in 42 sites of the River Wensum catchment.

Impact of STWs on water and sediment characteristics

Figure 2.10 illustrates the impact of treated sewage effluents on the variability of the soluble nutrients at different seasons. The significance of the observed patterns is summarized in Table 2.2. Soluble reactive phosphorus (LOGSRP), total phosphorus (LOGTP) and ammonium (NH4) concentrations were strongly impacted by the STW. All the phosphorus forms except the particulate phosphorus (LOGPP) were temporally variable. The interaction factor treatment x season was also significant for total phosphorus (LOGTP) and soluble unreactive phosphorus (LOGSUP). Total dissolved calcium concentrations (LOGTDCa) remained stable. The random factor (sites nested within treatment) was generally highly significant.

Table 2.1 Summary of the mean solute concentrations and sediment characteristics at the selected sites surveyed in 2000-2001. All units are expressed as the atomic mass of the element, i.e. P, N, Ca, Fe. All sediment vaiables in DW except BAP (WW). N represents the number of seasons sampled. Location of sites is displayed on Fig 2.8.

Sites	Solutes								Sediment characteristics				
	SRP µg/L	SUP µg/L	PP µg/L	TP µg/L	TDCa mg/L	NH4 µg/L	SS mg/L	PSS mg/g	TP µg/g	BAP µg/g	TCa mg/g	TFe mg/g	OM %
N	6	6	6	6	3	4	4	4	4	4	3	3	4
WTR 005	9	10	30	48	139	69	10.2	4.9	94	2.3	8.4	2.0	1.0
BLW 010	14	10	15	39	133	37	2.8	19.3	186	4.1	10.8	3.6	1.4
PSB 010	15	13	20	47	151	70	7.3	3.2	285	4.0	21.0	4.2	1.4
RPS 030	14	12	50	77	136	65	9.6	5.8	161	4.9	0.9	3.5	1.1
WEB 010	49	12	18	79	130	42	4.4	9.6	183	4.2	9.6	4.1	1.2
WEN 070	55	11	31	97	134	48	5.4	6.1	152	3.6	12.1	2.9	1.5
WEN 080	141	12	56	210	134	90	4.3	20.3	171	5.6	5.7	2.1	0.8
WEB 020	282	19	150	451	144	339	5.1	44.9	233	10.8	4.1	3.1	1.1
RPS 040	384	15	95	493	131	133	8.3	11.0	310	14.0	3.7	3.2	0.8

Table 2.2 Two way anova probabilities performed on the solutes (see Fig 2.10). Factors: T = treatment; R(T) = sites as nested random factor within treatment; S = season; and T x S = interaction factor season x treatment.

	T	R(T)	S	T x S
LOGSRP	< 0.001	0.004	0.005	0.069
LOGTP	0.001	< 0.001	0.018	0.004
LOGSUP	0.125	0.001	< 0.001	0.003
LOGPP	0.071	< 0.001	0.449	0.532
LOGTDCA	0.389	0.176	0.802	0.094
LOGNH4	0.035	0.001	0.112	0.291
LOGSS	0.668	0.001	< 0.001	0.647
LOGPPS	0.067	0.011	< 0.001	0.667

Figure 2.11 shows the impact of treated sewage effluents on the variability of the sediment characteristics at different seasons. All the variables were very constant over time except total iron (TFe), which exhibited lower concentrations in the autumn (October 2000). Total calcium (TCa) changed significantly between the seasons, although only at sites impacted by STWs. BAP concentrations were higher downstream from the large STWs, but did not differ significantly between the 'diffuse' and 'distant STW' treatments.

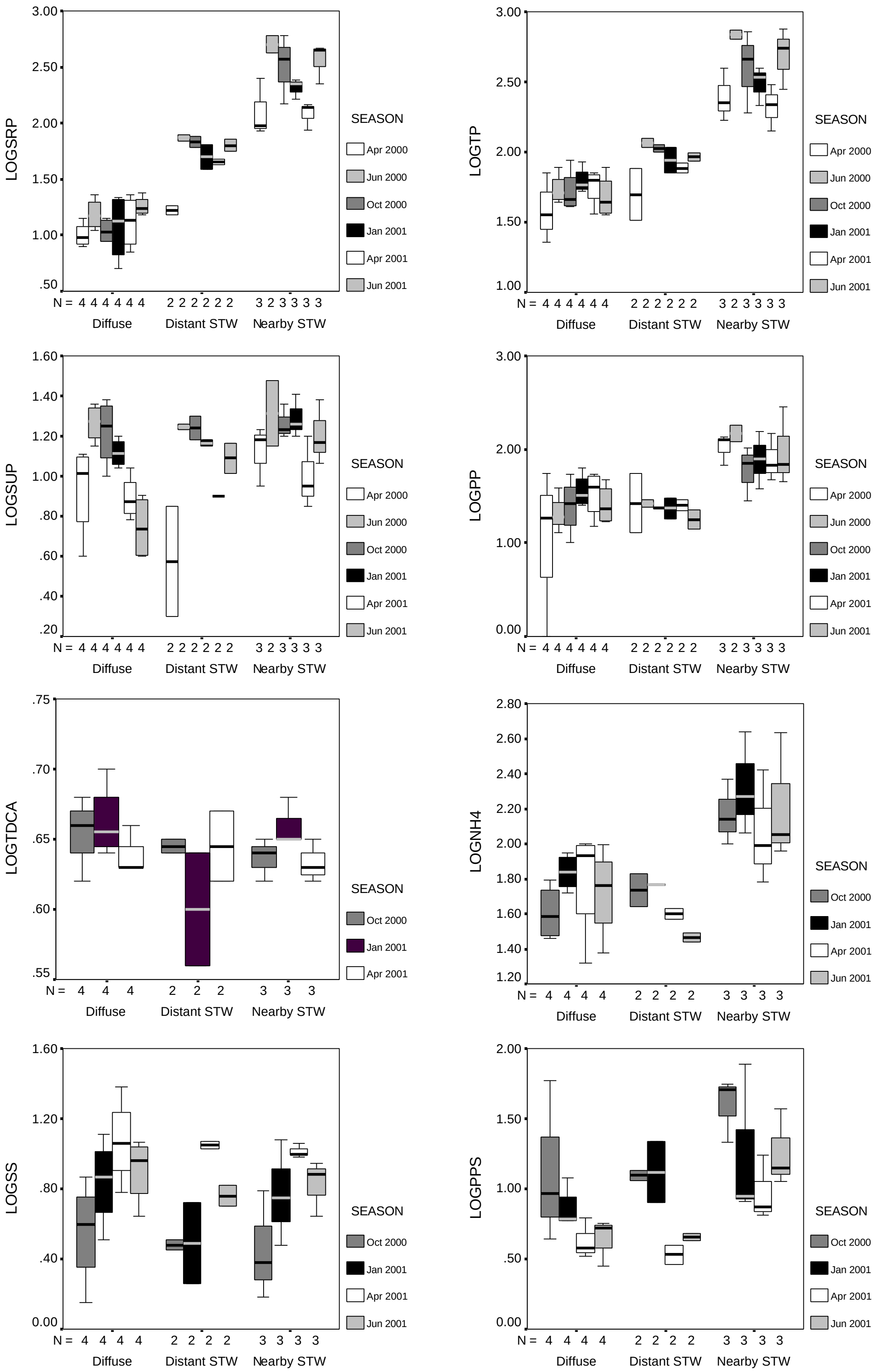
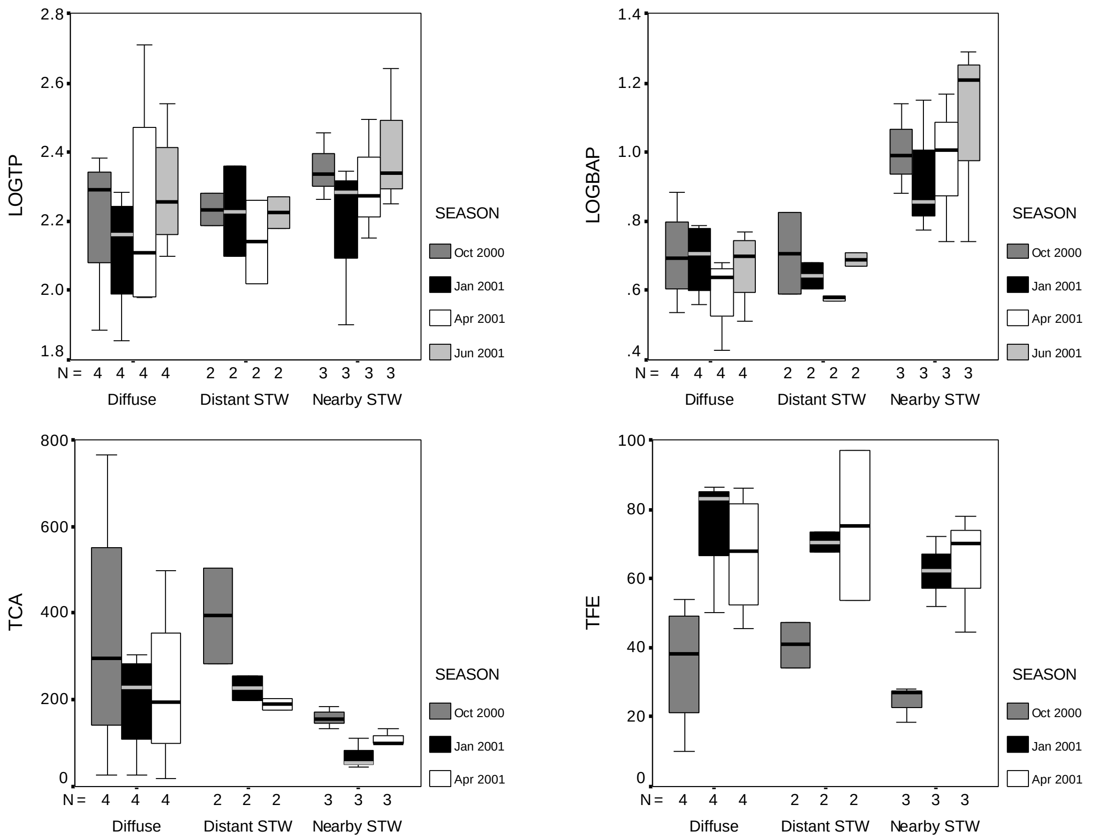
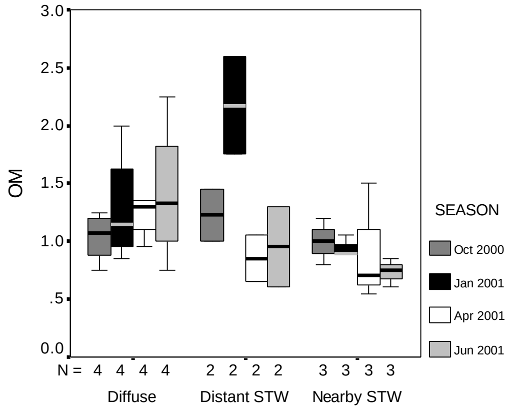


Figure 2.10 **Boxplots of the solutes showing the treatment heterogeneity and time variability. Diffuse = sites impacted only by diffuse sources; Distant STW = sites impacted by distant small point sources of pollution; Nearby STW = sites impacted by nearby effluents of large sewage treatment works. The box represents the interquartile range and the position of the median. All variables presented and analysed after LOG (x+1) transformation. Units: SRP + SUP + PP = TP in μg/L PO_4-P; TDCa in mmol/L; NH_4 in μg/L NH_4-N; SS in mg/L; PPS in mg/g PO_4-P DW. Probabilities in Table 2.2.**





	T	R(T)	S	T x S
LOGTP	0.713	0.002	0.288	0.849
LOGBAP	0.039	< 0.001	0.095	0.615
TCa	0.420	0.001	0.014	0.726
TFe	0.642	0.001	< 0.001	0.648
OM	0.108	0.406	0.076	0.090

Figure 2.11 Impacts of STWs and seasonality onto the sediment characteristics of a unvegetated gravel river bed type. Units: LOGTP, LOGBAP in µg/g PO_4-P DW, WW respectively and analysed after LOG (x+1) transformation; TCA, TFe as µmol/g; OM as % DW sediment. Two way Anova probabilities are displayed in the inserted table. Factors: T = treatment; R(T) = sites as nested random factor within treatment; S = season; and T x S = interaction factor season x treatment.

Relationships between sediment characteristics

No significant relationships were found between sediment characteristics at P<0.05 after Bonferroni adjustment but two trends were detected with TP-BAP and BAP-TCa respectively positive and negative (Fig 2.12).

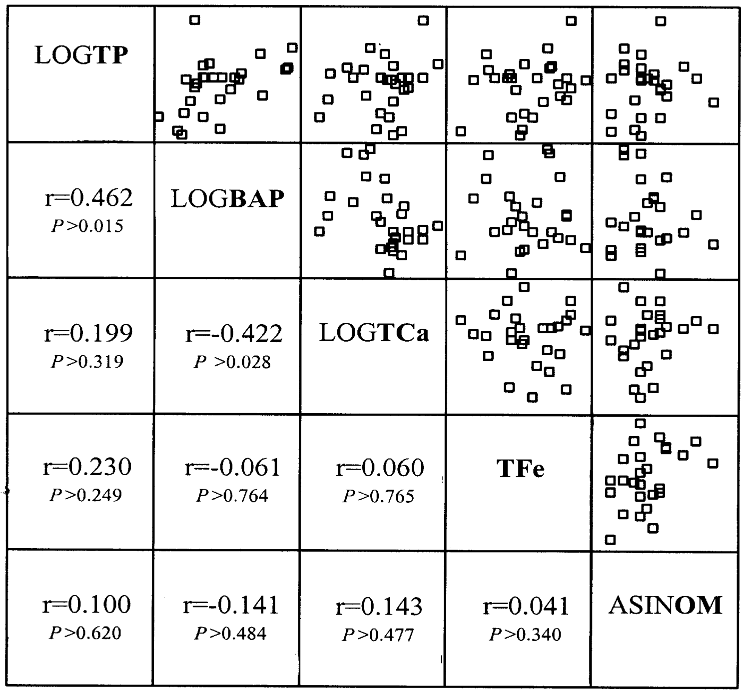


Figure 2.12 Correlation between pairs of sediment characteristics in unvegetated gravel/sand sediment. BAP and TP are marginally correlated. There is a negative trend between TCa and BAP. There is, however, no correlation between TP and TFe as observed before (Section 2.1) across all types of river bed sediments.

Predicting the sediment forms of phosphorus

Suspended solid concentration (LOGSS) was inversely related to phosphorus content (LOGPPS; r^2=0.455 - Fig 2.13). Total phosphorus in the water ($LOGTP_{WAT}$) was a fair predictor of the sediment bioavailable phosphorus (LOGBAP; r^2=0.64; P<0.001, Fig 2.14a), but did not predict a significant amount of the sediment total phosphorus ($LOGTP_{SED}$) variability (Fig 2.14b). The bioavailable phosphorus (LOGBAP) was significantly better predicted when using both total phosphorus forms ($LOGTP_{WAT}$ and $LOGTP_{SED}$) as independent variables (r^2=0.75; P<0.001).

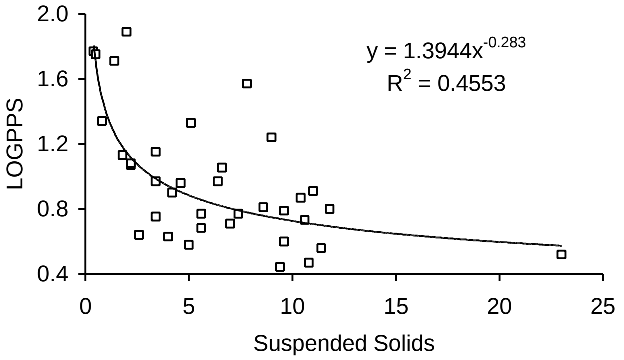


Figure 2.13 Suspended solid concentration (mg/L) as predictor of the concentration of phosphorus in the suspended solids (PPS). PPS (mg/g DW) were LOG(x+1) transformed.

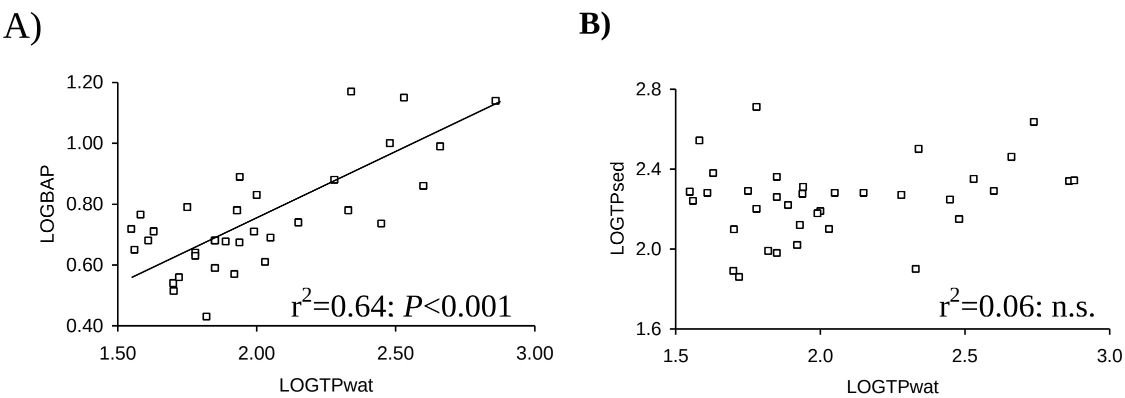


Figure 2.14 Total Phosphorus in the column water (TP_{WAT}) as predictor of (A) sediment bioavailable phosphorus (BAP) and (B) Total Phosphorus in the sediments (TP_{SED}). All variables were LOG(x+1) transformed. Units: TP_{WAT} as µg/L PO_4-P; BAP as µg/g PO_4-P WW; TP_{WAT} as µg/g PO_4-P DW.

Impact of phosphorus removal

The phosphorus stripping programme at the major STWs had significant impact on the concentration of soluble reactive phosphorus (LOGSRP) and total phosphorus (LOGTP; Fig 2.15) and their loads (Fig 2.16). Phosphorus removal decreased in-stream sediment BAP below East Dereham (P=0.22) but it was less clear what happened at Fakenham (Fig 2.17). The load of SRP (expressed in kg of phosphorus per day) was significantly higher above Fakenham STW in 2001 (Fig 2.16).

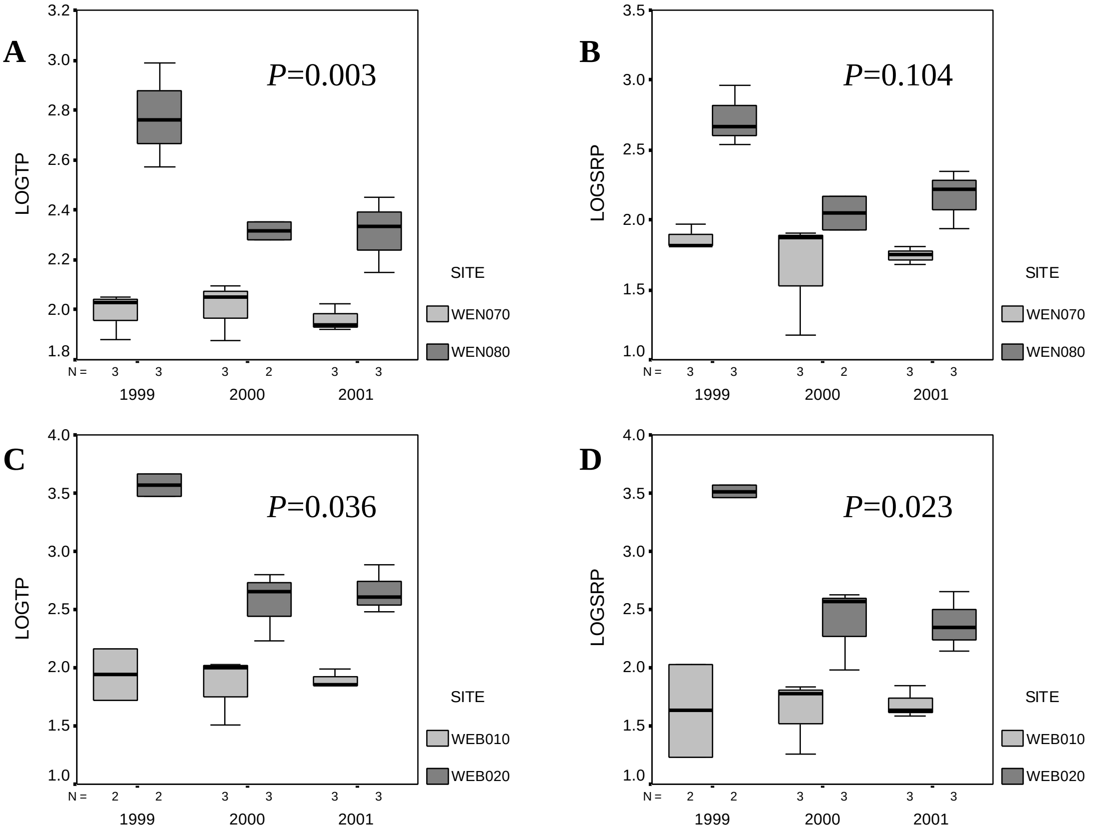


Figure 2.15 **Impact of phosphorus removal showing the decrease of in-stream SRP and TP (mg/L PO_4-P) above (WEN070) and below (WEN080) Fakenham STW (A, B) and above (WEB010) and below (WEB020) East Dereham STW (C, D). SRP, TP were LOG(x+1) transformed.**

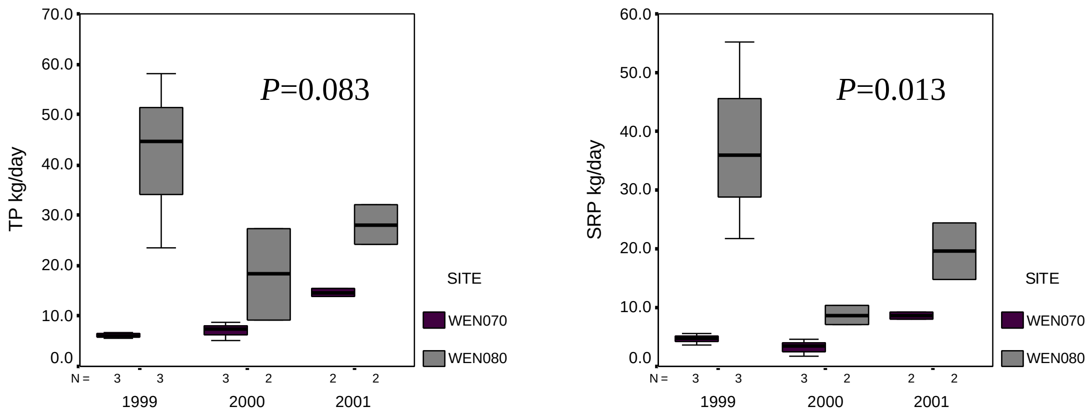


Figure 2.16 **Impact of phosphorus removal showing the decrease of in-stream phosphorus loads (kg/d PO_4-P) above (WEN070) and below (WEN080) Fakenham STW.**

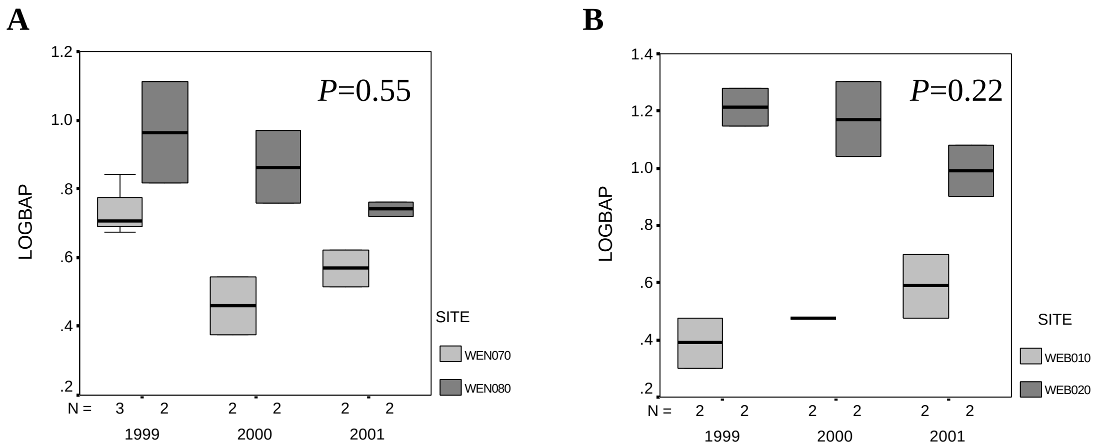


Figure 2.17 Impact of phosphorus removal on sediment bioavailable phosphorus (BAP as µg/g PO_4-P) above (WEN070) and below (WEN080) Fakenkam STW (A) and above (WEB010) and below (WEB020) East Dereham STW (B). BAP was LOG(x+1) transformed.

2.2.8 Discussion

Range of solutes and sediment characteristics in the river Wensum

The phosphate buffer mechanism is likely to operate in 33% of the sites, which had SRP concentrations lower than 50 µg/L. The TP concentration of the River Wensum largely exceeded US recommended levels (USEPA 2000; USGS 1999a,b). Therefore, it could impact downstream standing waters in the valley (Schindler 1990), vegetation of the floodplain (Prach *et al* 1996; Trémolières *et al* 1998; Lepe 1999), as well as estuarine (Nedwell & Raffaelli 1999), coastal ecosystems (Froelich 1988; Dederen 1992; Livingston 2000; House *et al* 1998) and oceans on the long term (Tyrrel 1999).

Diffuse sources of phosphorus

The diffuse sources of SRP concentration (13 µg/L) found in this study were among the lowest recorded in East Anglia and have to date not been detected in the 'Harmonised Monitoring Scheme' of the Environment Agency of the Anglian Region (Environment Agency, unpublished; Evans 1999). Environment Agency (EA) analyses are probably higher for two reasons. First the water samples routinely analysed by the EA were not filtered before SRP analysis (Muscutt & Withers 1996; Mainstone *et al* 2000). Second the SRP concentration measurements were overestimated (Chapter 2.3), potentially due to analytical problems with silica interference (Neal *et al* 2000c). The mean concentrations (n=6) of SRP (9-15 µg/L) and especially TP (39-77 µg/L) measured in sub-catchments non-impacted by known point sources of phosphorus were in the same range as those measured by Moss *et al* (1988) and probably reflect the diffuse sources from agricultural land use (Johnes 1996).

Solutes and sediment characteristic dynamics

The seasonality of SRP and TP was not significant for the 'diffuse' treatment (sites not impacted by point sources of pollution) but storm events were not monitored. During storm events a peak of total phosphorus occurs, mostly particulate phosphorus bound on suspended solids, but only intensive sampling would detect it (Tunney *et al* 1997; House & Denison 1998; House & Warwick 1998b, 1998c; Walling & Amos 1999). Seasonality for the other two treatments (sites impacted by STWs) can be explained by the discharge variability diluting the phosphorus effluents (Edwards 1973b; Jarvie *et al* 1998).

The statistical significance of the random nested factor (sites within treatment) indicates that the sites differ significantly from each other. This may be explained by a variety of factors not investigated such as the differences in effluent phosphorus load, the differences in valley slope gradient, the land use or ecotone structure between the sub-catchment (Boar *et al* 1994; Johnes 1996).

The seasonal non-variability of TDCa agrees with previous results from the River Wensum where TDCa was only very weakly predicted by the discharge (r^2=0.13; P<0.010) and was in fact relatively constant (110-140 mg/L; Edwards 1973b).
The values of the sediment characteristics recorded fall in the range of the same river bed type (unvegetated gravel/sand – UGS) investigated in 1999 and April 2000 (Section 2.1). The temporal stability of the phosphorus forms in this river bed type contrasts with previous temporal studies although the methodology is not strictly identical (House & Denison 1997, 1998).

The observed negatively related trend of BAP and TCa might reflect the antagonistic effect of high concentration of SRP in the interstitial water with calcite crystal-growth (House *et al* 1986a,b; House 1987, 1990, 1999). The sediment total phosphorus content was not related to the minerals analysed (calcium, iron) as in the previous study looking at a wider range of river bed types (Section 2.1, Fig 2.3).
The high levels of SRP and PPS concentration just below the STWs may be respectively adsorbed or trapped, as would be indicated by the fair prediction of BAP with TP_{WAT}.

Impact of phosphorus removal

The BACI design would have been strengthened with the addition of a co-variable to partial out the fluctuating phosphorus concentration of the STW effluent (see Fig 4 in Moss *et al* 1988), although this was stabilised with the implementation of the phosphorus removal (Chapter 2.3, Fig 2.21). Such a BACI design was recently used by measuring the boron as well as the phosphorus concentration (Neal *et al* 2000b).

This study did not investigate the impact of phosphorus removal on the longitudinal profile of the River Wensum, especially below the main STWs, but EA data (Environment Agency, unpublished) showed that the phosphorus stripping at major STWs has not been sufficient to bring the dissolved phosphate concentration to the level of the EPC_0 (10-60 µg/L), which could be seen as an appropriate target level for English lowland rivers. The EPC_0 is however not a fixed value and will come down as SRP concentrations decrease (W.A. House, personal communication). The relatively constant SRP concentration in streams not impacted by STWs[1] may indicate the existence of a phosphate buffer mechanism (*sensu* Froelich 1988) which would operate at around 13 ±6 SD µg/L (mean SRP concentration). However the coarse materials of the bed of these streams did not exactly represent the whole catchment of the River Wensum where aquatic macrophytes, in-stream riparian habitat, and the weirs lead to large siltation of the main channel. In these habitats, the conditions resemble more those of shallow lakes (characterised by a higher retention capacity) and there the physico-chemical equilibrium would be altered by the bioturbation activity of the macrofauna (Phillips & Jackson 1990; Harper 1992, p. 267; Demars, personal observation). Another major problem is the large impact of small STWs scattered in the catchment (Muscutt & Withers 1996; Section 2.1, Fig 2.1).

The impact of phosphorus stripping on the BAP may only become significant after a few years due to the concentrations of SRP below the treated sewage effluents remaining high and the unknown sediment mobility.

2.2.9 Conclusion and perspectives

The design of the present study allowed the mean diffuse sources of phosphorus concentrations (SRP=13, TDP=24 and TP=53 µg/L) to be determined and showed the sediment phosphorus forms (TP and BAP) to be less impacted than the water column concentrations by upstream STWs. These findings are important because the current sampling strategy of the Environment Agency does not allow the diffuse source level of phosphorus (atmospheric deposit, natural weathering and diffuse anthropogenic sources) to be determined. The monitoring of small streams not impacted by point sources of phosphorus and deep aquifers would be necessary to find the background of phosphorus levels (as in the Danish monitoring strategy; Kronvang *et al* 1993). The strategy was used in the UK in the LOIS (Land Ocean Interaction Study) programme at a scale which included a wide range of Eastern river basins (Leeks & Jarvie 1998).

The present study focused on distinguishing patterns at different levels of anthropogenic impacts. It did not investigate the underlying processes that are critical to understand ecosystem functioning (e.g. Phillips *et al* 1978, Schindler 1990, Phillips *et al* 1994, Jeppesen *et al* 1998, Jones & Mulholland 2000). Such processes have only recently been addressed in the UK by House *et al* (1995a), House & Denison (1997, 1998) and the Land Ocean Interaction Study (LOIS) through a combination of fieldwork, fluvarium experiments and a mass-balance approach (House & Warwick 1998a, 1998b, 1999; House & Denison 2000; Bowes & House 2001). More work should now follow on the dynamics of phosphorus at multiple spatial scales (García-Ruiz *et al* 1998; Chapters 2.1 and 2.3) and temporal scales (e.g. House & Denison 1997, 1998; House & Warwick 1998b, 1998c, 1999). The interaction of the biological activity upon the phosphorus dynamics in British lowland rivers, especially at the interfaces biota-sediment and water-sediment would require further investigations as initiated by Hartley *et al* (1997) and House *et al* (2001).

This study detected significant in-stream decreases of SRP and TP downstream from major STWs (serving over 10,000 equivalent people) after phosphorus removal. The

[1] The constancy of the SRP should be checked site by site because of the R(T) significance (Table 2.2)

impact on sediment BAP was less convincing but this may be due to the high variability of BAP (due to sediment heterogeneity) and the low number of replicate making the statistical test very weak. This is probably because the dissolved phosphate concentration was not brought down to the level of the EPC_0 (10-60 µg/L). The latter figure has been suggested as an appropriate target level for English lowland rivers, but in the Wensum the anthropogenic siltation of the main channels and the numerous small STWs scattered in the catchment may prevent rapid success. Success after phosphorus removal might otherwise be expected in rivers because of the naturally low retention capacity of such ecosystems compared to lakes, although little is known about the sediment mobility.

2.3 Phosphorus control: problems at the catchment scale of impact assessment

2.3.1 Introduction

Phosphorus removal at sewage treatment works has been very successful and has led to considerably reduced in-stream loads (Harper 1992; Thomas & Slaughter 1992; Phillips *et al* 1999a). However the effectiveness of phosphorus control in the Norfolk Broads was limited because of internal phosphorus release from the sediment (Harper 1992; Phillips *et al* 1994). This may not happen in rivers where the sediment retention time might be much shorter. Although phosphorus stripping alone was far from solving all the problems of the Broads, the same reasoning has been applied to highly eutrophic lowland rivers by the Environment Agency.

An example of this occurs in the calcareous lowland river Wensum where several negative consequences of nutrient enrichment have been reported. These are development of dense growths of blanket weed and other filamentous algae during the summer months; risk of impoverishment of the flora; depletion of dissolved oxygen levels (English Nature & Environment Agency 1998, 2000). Particular concerns were raised by English Nature about the protected *Ranunculus* (Habitats Directive), although its value in the chalk stream ecosystem has recently been questioned (Langford, 2001).

2.3.2 Aims

The present study set out to investigate the impact of phosphorus control at two designated qualifying discharges under the UWWTD on the water quality of the River Wensum. It aimed to provide the water quality data necessary to consider the potential implications of phosphorus control on the ecology of the river, especially that of the submerged aquatic vascular plants. The following questions were addressed:

- What has been the phosphorus removal efficiency at Fakenham and East Dereham sewage treatment works?
- What has been the impact of phosphorus control on the river water quality and was it predictable?
- What are likely to be the bioavailable phosphorus concentrations over a range of flow regimes?

- What were the concentrations of nitrogen? How do they compare with the concentrations of phosphorus?
- What were the concentrations of chlorophyll *a*?

2.3.3 Material and methods

Study area

The river Wensum is a calcareous stream in Norfolk rising in the north and flowing eastwards to join the Yare at Norwich. More information about the ecology, geology and climate of the river Wensum can be found in Chapter 1, Baker *et al* (1978); Baker & Lambley (1980); Boar *et al* (1994). The gauging stations, monitoring sites of the Environment Agency, as well as the three main phosphorus effluents in the catchment area of the River Wensum are displayed on Figure 2.18 and further details listed in Table 2.3-2.5.

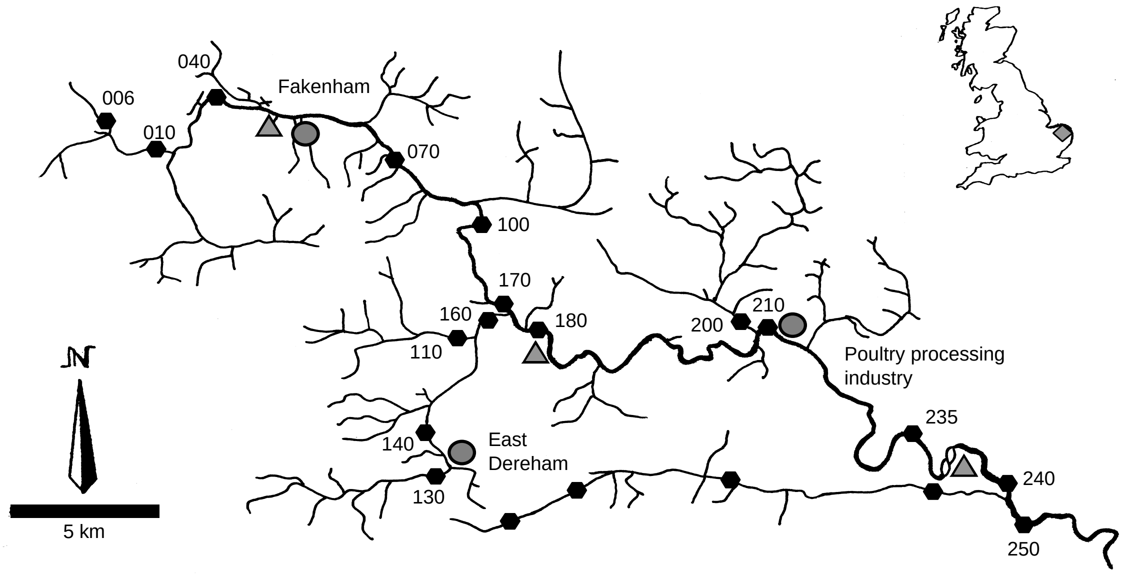


Figure 2.18 Catchment area of the River Wensum. Hexagons = monitoring sites of the Environment Agency; triangles = gauging stations; ovals = major urban (Fakenham and Dereham STWs) and industrial (poultry processing) phosphorus effluents.

Table 2.3 Gauging station characteristics. MF = mean flow; Q95 and Q10 = 95% and 10% chance flow exceedance.

	National Grid Reference	Catchment area (km^2)	Start of records	MF (m^3/s)	Q95 (m^3/s)	Q10 (m^3/s)	Rainfall mm
Fakenham	TF 919 294	161.9	May 1966	0.85	0.23	1.59	698
Swanton Morley	TG 020 184	397.8	Oct 1969	2.64	0.89	4.86	no data
Costessey Mill	TG 177 128	570.9	Feb 1960	3.98	1.23	7.41	672

Table 2.4 Characteristics of the selected monitoring sites of the Environment Agency prior to and after phosphorus removal. The mean discharge at ungauged sites was derived from the continuous flow record at Fakenham using the following catchment area ratio: catchment area of studied site / catchment area of Fakenham gauging station (results are displayed in the column titled ratio).

		area km^2	gauging site	ratio	discharge m^3/s prior	after
Black Water	EAWEN110	29	Fakenham	0.18	0.135	0.225
Wendling Beck	EAWEN130	37	Fakenham	0.23	0.171	0.288
	EAWEN140	45	Fakenham	0.28	0.210	0.354
River Wensum	EAWEN040	145	Fakenham	0.90	0.660	1.136
	EAWEN070	197	Fakenham	1.22	0.906	1.549
	EAWEN180	398	Swanton Morley	1.00	2.345	4.301
	EAWEN240	571	Costessey Mill	1.00	3.452	no data

The qualifying discharges

Anglian Water Services (AWS) provided the phosphorus concentrations (total and soluble) of the crude sewage and final effluent of the two qualifying discharges (Fakenham & East Dereham STWs) before and after phosphorus removal.

Observed loads at EA monitoring sites

Seven sites were selected for their specific location (upstream / downstream of the qualifying discharges and the main industrial effluent) and the availability of physico-chemical data. Phosphorus has been monitored monthly at all sites by the EA since January 1990 and twice a month since March 2000. No data were available prior to phosphorus removal at EAWEN130 (above East Dereham STW); therefore it was extrapolated (TP EAWEN040 prior x TP EAWEN130 after ÷ TP EAWEN040 after). Observed annual loads were estimated from the mean phosphorus concentration and mean discharge at the site on day of sampling. Discharge at ungauged sites on the river

Wensum was estimated from Fakenham gauging site thought to have the most similar flow characteristics (Tables 2.3 and 2.4). Because of the relatively low frequency of sampling only two periods were distinguished: prior (1990-1999) and after (2000-2001) phosphorus removal. Different phosphorus concentrations were supplied by the Environment Agency over time: SRP from non-filtered (NF) samples for the whole period 1990-2001; soluble reactive phosphorus (SRP) from filtered (F) samples for 2000-2001; and finally TP only at a few sites for 1995-1996 and 2000-2001 at most of the sites. The quality of the data could then be investigated, especially during the overlapping period 2000-2001. All the concentrations were transformed into TP based on a site-specific and period-specific SRP/TP ratio (Table 2.6). These ratios were derived from EA data as well as personal data collected during the project throughout the catchment area of the River Wensum (1999-2001; Section 2.2; Demars, unpublished).

Estimation of the loads from all known effluents

The Environment Agency (EA) monitored the total soluble phosphorus (TSP) of most STWs' final effluents between July 1993 and May 1996 to calculate the mean concentration of total soluble phosphorus (TSP). The TSP was converted to total phosphorus (TP) based on a TSP/TP ratio of 0.9 (AWS data; Table 3). The loads were estimated with the derived mean flow (from the population equivalent, p.e. or dry weather flow in accordance with SIMCAT instructions; NRA 1995) and if, neither was available, from the consented maximum daily flow. For the STW effluents not monitored an estimate of the TSP discharged per population equivalent - p.e. (0.028 mg/L/p.e.) was derived from the mean of all discharges with a p.e. less than 500 to calculate the loads. Industrial effluent loads were estimated from the monitored (same period as above) mean TP concentrations (derived from mean TSP) and consented maximum daily flow.

Table 2.5 Characteristics of the known STWs and industrial effluents. The three STWs without National Grid Reference (NGR) are situated on the River Tud joining the Wensum between EAWEN240 and EAWEN250. p.e. = population equivalent. The data displayed were collected, analysed and calculated by the Environment Agency. TP loads expressed as kg/d PO_4-P.

STWs effluents	NGR	TP kg/d	p.e.
Billingford	TG 018 203	0.02	62
Bylaugh	TG 036 183	4.99	2913
Church Farm Barns	TG 066 203	0.21	no data
East Rudham	TF 835 285	1.09	560
East Dereham	TF 977 135	31.59	17475
Fakenham	TF 921 289	28.60	13439
Foulsham	TG 026 243	1.49	998
Foxley Laurence Road	TG 038 218	0.01	33
Foxley Norwich Road	TG 036 215	0.00	19
Hockering		1.07	562
Little Fransham Crown Lane	TF 901 126	0.01	48
Little Fransham Glebe Close	TF 905 120	0.16	111
Mattishall		3.93	3344
North Elmham	TF 998 213	1.96	1122
North Tuddenham		0.00	24
Park Homes Haveringland Hall	TG 160 210	0.07	111
Reepham	TG 104 227	7.19	4017
RAF Sculthorpe	TF 833 312	1.69	1089
Sparham Norwich Road	TG 071 197	0.00	27
Sparham Wells Close	TG 081 181	0.00	27
Stibbard Council	TF 981 285	0.01	42
Swanton Morley	TG 013 183	0.86	502
Swanton Novers	TG 019 321	0.02	54
Swanton Novers Nursing Home	TG 024 317	0.01	50
Weasenham All Saints	TF 849 221	0.01	39
Weasenham St Peters	TF 853 223	0.06	96
Wendling	TF 936 136	0.00	25
West Raynham	TF 871 251	0.03	71
Wherry Housing Assoc	TG 135 189	0.18	no data
Total at EAWEN250		**85.26**	
Industrial effluents			
Bernard Matthews, Gt Witchingham	TF 939 216	24.07	
Bernard Matthews, Stanfield	TF 115 183	0.61	
Church Farms Mushrooms, Fransham 'A'	TF 902 122	0.00	
Church Farms Mushrooms, Fransham	TF 902 122	0.01	
Crisp Maltings LTD, Gt Ryburgh	TF 964 272	4.26	
Crisp Maltings LTD, Gt Ryburgh (batch)	TF 956 276	4.83	
Pioneer Aggregates Langar Br Quarry	TF 954 285	1.39	
Redland Aggregates, Sparham	TG 083 181	2.78	
Tannery House fish Farm, Worthing	TF 997 201	4.70	
Total at EAWEN250		**42.65**	
Total STWs + Industrial effluents		**127.91**	

Table 2.6 Phosphorus ratio extracted from spot-sampling and Environment Agency monitoring data.

	SRP/TP ratio	
	prior	after
EAWEN110	0.50	0.50
EAWEN130	0.80	0.71
EAWEN140	0.90	0.67
EAWEN040	0.80	0.77
EAWEN070	0.85	0.76
EAWEN180	0.85	0.66
EAWEN240	0.80	0.70

Prediction of soluble reactive phosphorus concentration at different flow regimes

The mean flow (MF = Q50), 95% (Q95) and 10% (Q10) chance flow exceedance discharges were calculated for each site from the three gauging stations using the catchment area ratio as above. For each of these sites and flow regime combinations, was derived a tentative SRP/TP ratio based on the findings of Section 2.2 and the EA data.

The predicted loads from all known point sources after phosphorus removal were used to work out the TP concentration (mg/m^3) using the discharge and making the appropriate transformations. Then the TP concentrations were transformed into SRP concentration with the specific SRP/TP ratio. Finally the contribution of the diffuse sources was added: 13 mg/m^3 (Section 2.2). The equation was the following:

$SRP_i = [(L_i*1000000/(Q_i*3600*24))*(SRP/TP)_i\ ]+13$ **Equation 2.1**

with L_i = loads TP (kg/d PO_4-P) at site i,
Q_i = discharge m^3/s at site i,
SRP = concentration of soluble reactive phosphorus (mg/m^3 PO_4-P) at site i,
$(SRP/TP)_i$ the specific site x flow regime ratio,
i an EA monitoring site.

Patterns in the long term records of the Environment Agency

The Environment Agency long term records of flow, phosphorus and nitrogen forms, as well as levels of chlorophyll *a*, were compiled for each site over the period 1990-2001 to illustrate the impact of the point sources and different temporal variations (annual, before and after phosphorus control). Very few outliers were taken out of the dataset when it was more likely to be a mistake in the database rather than a real value (i.e. concentration well under the limits of detection or one order above the peak values). On relatively few occasions the levels of the solutes were under the limits of detection (except for ionised ammonia where it was more frequent). The value retained was half of the value of the limit of detection. The long term records of phosphorus illustrate the exact values retained for the calculation of the loads.

Two issues that arose from examining these data were the quality of the phosphorus measurements and the impact of discharge on the concentration of phosphorus prior to and after phosphorus control. These are dealt with at the end of the results section.

2.3.4 Results

Phosphorus control at the qualifying discharges

Prior to phosphorus removal, East Dereham and Fakenham STWs were respectively discharging 31.6 and 28.6 kg of phosphorus per day representing 42% of the point sources situated above EAWEN250 (including therefore the River Tud). The phosphorus concentration of the crude sewage averaged 14.8 (±5.6 SD) mg/L at Fakenham (n=77, Oct/1999- Feb/2000) and 16.8 (±5.3 SD) mg/L at East Dereham (n=64, Oct/1999- Feb/2000). The pattern of variation of phosphorus input into the two STWs was similar for the two STWs (Fig 2.19).

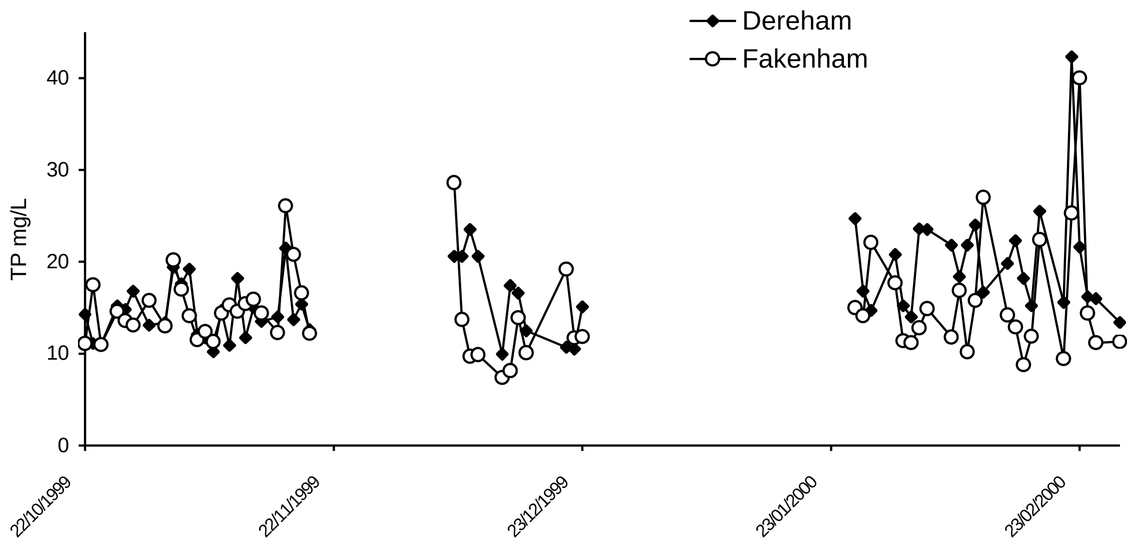


Figure 2.19 Total phosphorus (TP) concentrations of the crude sewage entering the STWs.

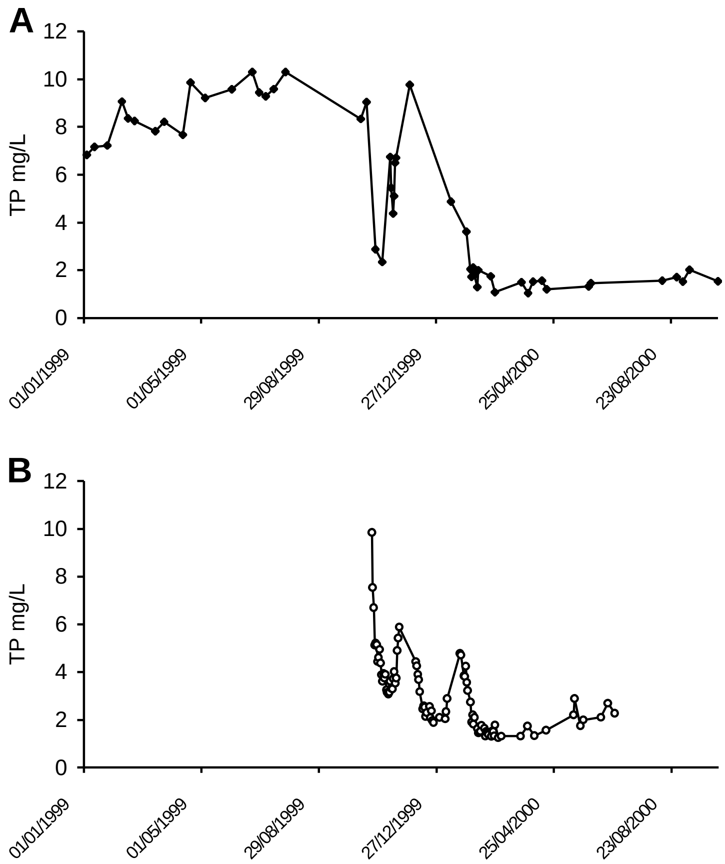


Figure 2.20 Total phosphorus (TP) concentration of the final effluent coming out of East Dereham (A) and Fakenham (B) STWs. Trials were run in 1999. The stripping target of 2 mg/L of TP could not be maintained at Fakenham STW without compromising the existing sanitary consents.

The improvement of the phosphorus removal at the STWs is shown in Figure 3. After several trials TP concentrations of the final effluent stabilised at about 3.5 mg/L at Fakenham STWs and 2.0 mg/L at Dereham (Fig 2.20). The effective removals of phosphorus at the STWs were therefore 77% at Fakenham and 88% at Dereham. The new daily loads thus decreased to 9.35 P kg/d (64% reduction) at Fakenham and 6.95 P kg/d at Dereham (76% reduction). The qualifying discharges' contribution dropped from 42% (1990-1999) to 18% (2000-2001) of the total point source effluents situated above EAWEN250.

A continuous monitoring of the final effluent at Fakenham STW showed that the phosphorus concentrations were stable over a 36 hour period (Fig 2.21).

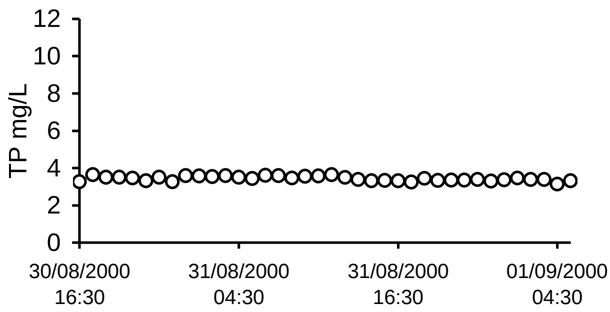


Figure 2.21 Total Phosphorus (TP) concentration of the actual final effluent coming out of Fakenham STW.

Observed and predicted impacts of phosphorus control on in-stream loads

The observed loads at the sites situated below the qualifying discharges dropped sharply after the phosphorus removal (Table 2.7), although this was confounded by high rainfall over the period 2000-2001 (the discharge was roughly 1.7 times mean flow; see Table 2.3). The three sites situated above the STWs, however, displayed higher phosphorus loads.

The predicted phosphorus loads based on point sources and diffuse sources matched relatively closely the observed loads prior to and after phosphorus stripping (Table 2.7; Fig 2.22). The residual discrepancy did not show particular patterns of loss or gain (Table 2.7).

Table 2.7 Phosphorus loads prior to and after phosphorus control at the STWs. Predicted loads were calculated by adding the point sources and estimated diffuse sources.

	observed TP kg/day		point sources TP kg/day		diffuse sources TP kg/day		predicted TP kg/day		discrepancy TP kg/day	
	prior	after	prior	after	prior	after	prior	after	prior	after
EAWEN110	0.9	1.6	0.6	0.6	0.6	1.0	1.2	1.6	-0.4	-0.1
EAWEN130	1.5	2.6	0.2	0.2	0.8	1.3	0.9	1.5	0.6	1.1
EAWEN140	41.1	8.6	31.8	7.1	1.0	1.6	32.7	8.8	8.4	-0.2
EAWEN040	6.5	10.1	2.9	2.9	3.0	5.2	5.9	8.1	0.6	2.0
EAWEN070	46.5	17.4	42.0	22.7	4.2	7.1	46.1	29.8	0.4	-12.4
EAWEN180	114.5	56.7	82.6	38.7	10.7	19.7	93.3	58.4	21.2	-1.7
EAWEN240	127.7	no data	122.9	79.0	15.8	no data	138.7	no data	-11.0	no data

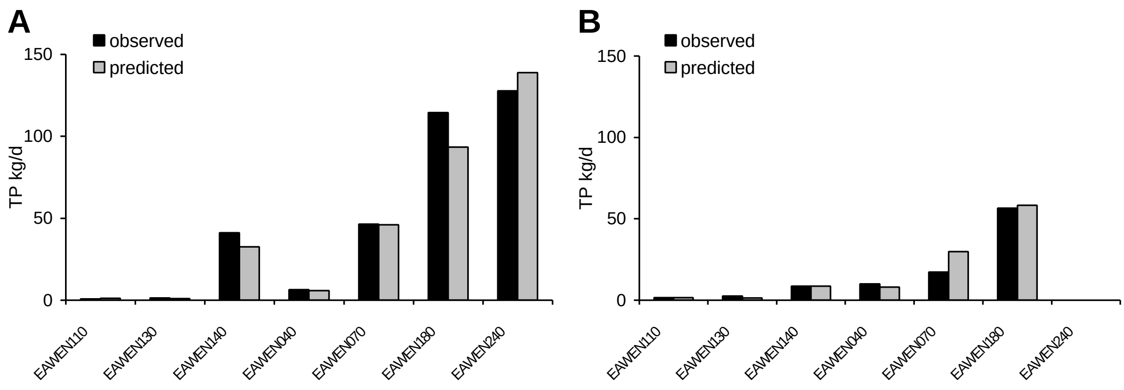


Figure 2.22 Total Phosphorus (TP) loads observed and predicted at seven monitoring sites prior to (A) and after (B) phosphorus control.

The contribution of the diffuse sources at the different sites reflected the higher discharge over 2000-2001 as well as the phosphorus removal at the STWs (Table 2.8). Although the two processes had an opposite effect on the loads, the phosphorus control had such a large effect on the sites situated below that both effects could be observed although at different sites (compare Fig 2.22a and 2.22b). The relative contribution of diffuse sources was boosted from 12% to 34% at EAWEN180 by the two processes. It means that the point sources were still really dominating phosphorus loads, and this even above the main industrial effluent. The loads could not be calculated at EAWEN240 because the flow data were not yet (4 Jan 2002) available for the period 2000-2001 at Costessey Mill (EA, personal communication).

Predicted phosphorus concentrations at different flow regimes

Below the discharges, the predicted concentrations of SRP at low flow (Q95) and mean flow (MF) regime were respectively extremely high (415-1039 µg/L) and very high (140-273 µg/L) – see Table 2.8. The observed SRP concentration over the period 2000-2001 provided a means of testing the predicted value at a flow regime intermediate between MF and Q10. There was a satisfactory level of agreement between predicted and observed values although there was a clear underestimation at the two sites (EAWEN130 and EAWEN040) situated above the qualifying discharges (Fig 2.23).

Table 2.8 Predicted SRP concentration at different flow regime (from Equation 2.1): mean flow (MF = Q50), 95% (Q95) and 10% (Q10) chance flow exceedance. Tentative SRP/TP ratio was derived for each flow x site combination. The observed SRP concentration at the 2000-2001 mean discharge (Q2000-01) are for comparison.

	MF (m^3/s)	Q95 (m^3/s)	Q10 (m^3/s)	SRP/TP ratio MF	SRP/TP ratio Q95	SRP/TP ratio Q10	SRP µg/L MF	SRP µg/L Q95	SRP µg/L Q10	Q 2000-01	observed SRP
EAWEN110	0.15	0.04	0.29	0.50	0.55	0.40	36	107	23	0.23	40
EAWEN130	0.20	0.05	0.37	0.65	0.70	0.60	20	41	16	0.29	74
EAWEN140	0.24	0.06	0.45	0.75	0.80	0.65	273	1039	134	0.35	188
EAWEN040	0.77	0.21	1.43	0.65	0.70	0.60	41	126	27	1.14	79
EAWEN070	1.04	0.28	1.94	0.75	0.80	0.65	203	763	101	1.55	99
EAWEN180	2.64	0.89	4.86	0.75	0.80	0.65	140	415	73	4.30	101
EAWEN240	3.98	1.23	7.41	0.75	0.80	0.65	185	608	93	no data	162

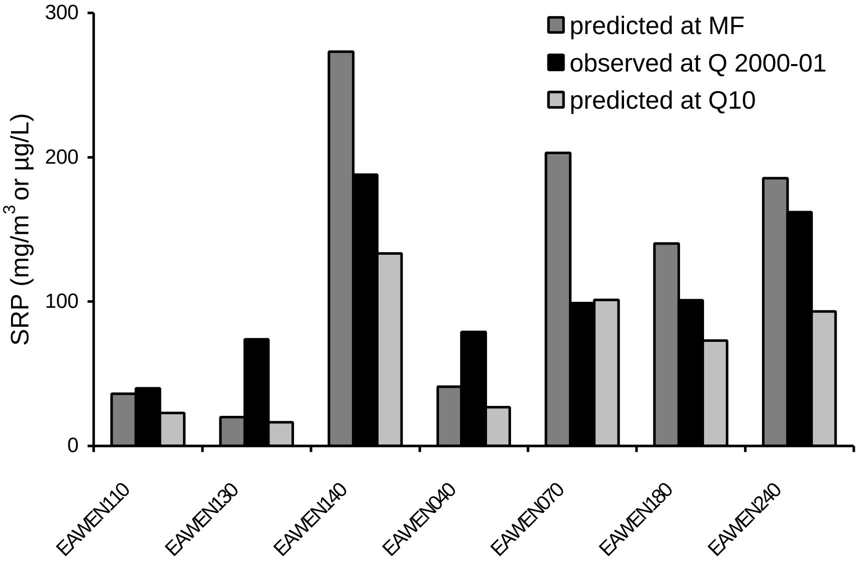


Figure 2.23 Observed and predicted soluble reactive phosphorus (SRP) concentrations at seven monitoring sites after phosphorus control. Observed concentrations indicated that the predicted concentrations were relatively reliable although the latter were underestimated above the STWs at Dereham (EAWEN130) and Fakenham (EAWEN040).

Long term flow records of the Environment Agency

The mean daily flow record from Fakenham gauging station is presented to illustrate of the flow variation over 45 years (Fig 2.24). The years of the drought in 1976 and during the nineties stand out.

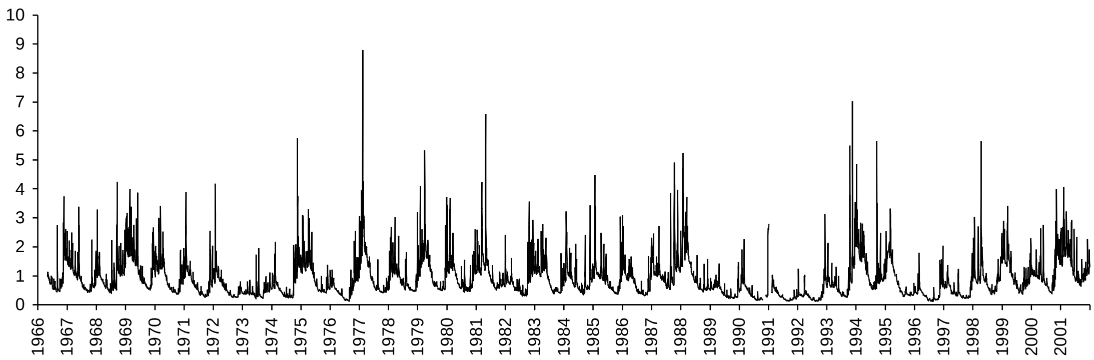
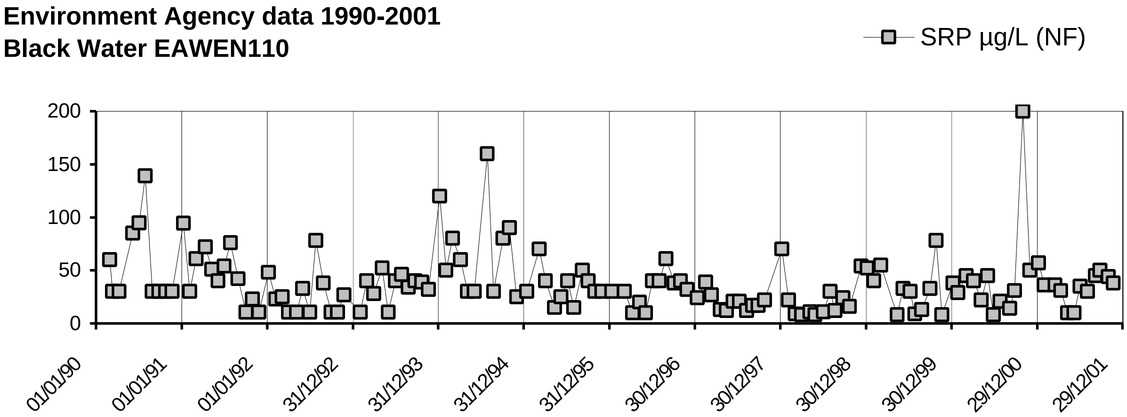


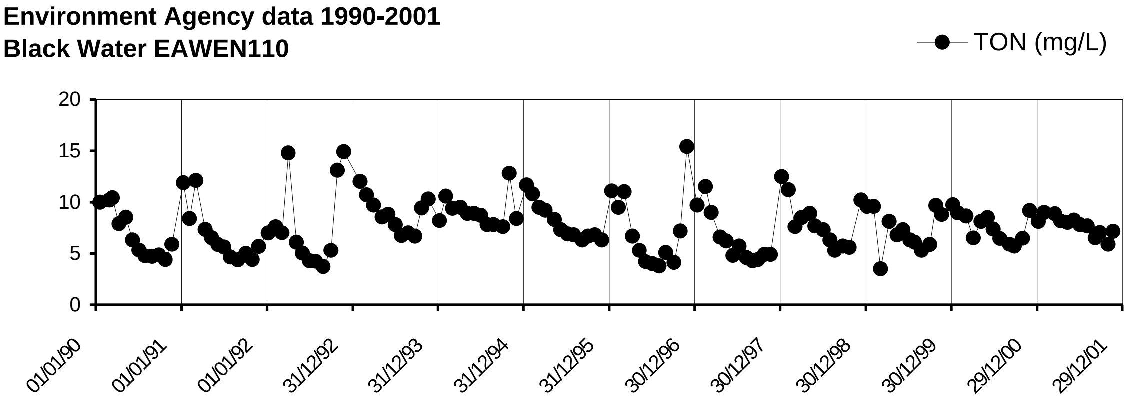
Figure 2.24 Daily flow (m^3/s) at Fakenham (NGR TF919294 – 162 km^2) between 1966 and 2001, River Wensum. Note the droughts in 1976 and in the nineties.

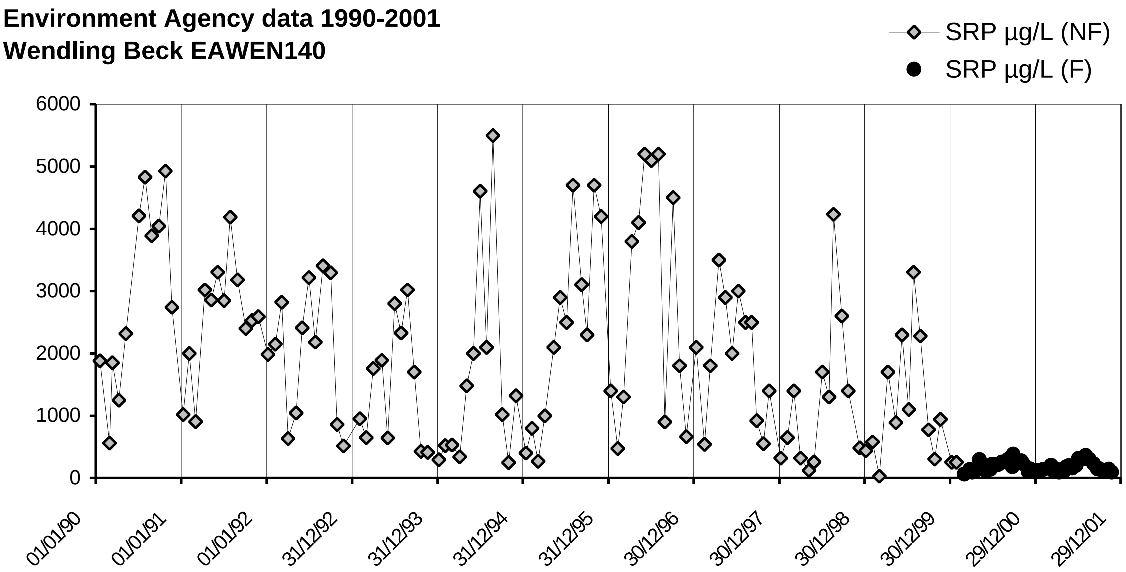
Long term solute records of the Environment Agency

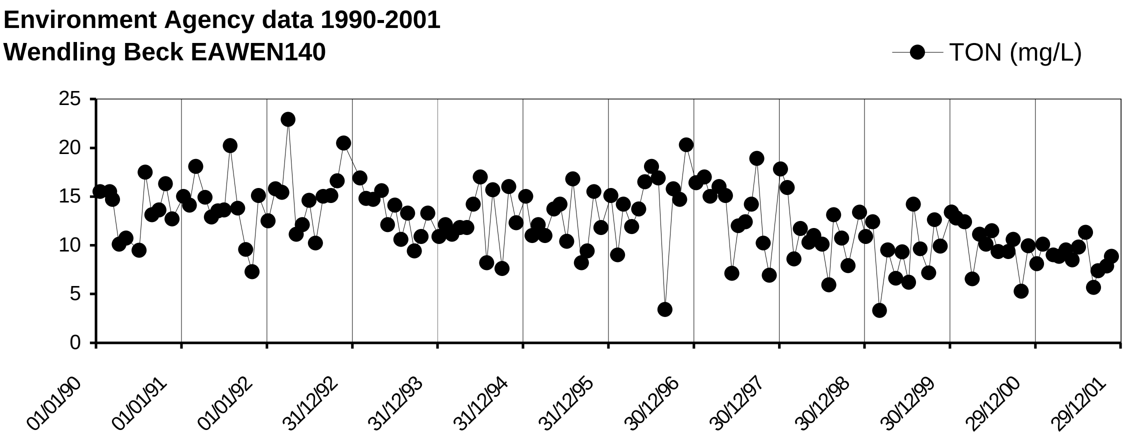
The levels of SRP (PO_4-P) varied irregularly between the sites because of the point sources. The maximum SRP concentrations reached more than 500 µg/L in the Tat, 1000 µg/L in the Wensum and exceeded 5000 µg/L in the Wendling Beck, during the drought summers of the nineties. The maximum values recorded in 2000-2001 were however much more modest due to the decreased loads and high flow regime (see above). The seasonal trend observed in 2000-2001 of SRP was not evident in the previous years when SRP was measured on unfiltered samples. (see Fig 2.25)

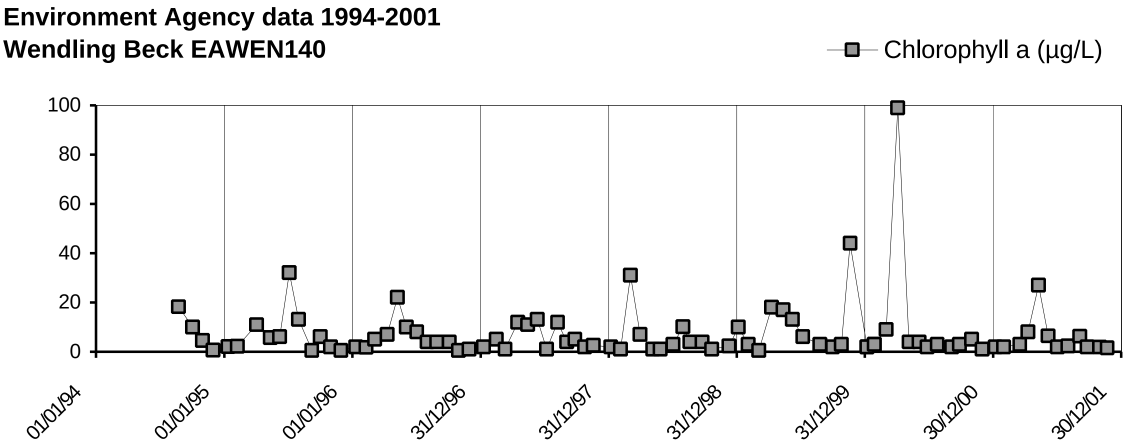
Figure 2.25 Nutrient concentrations at selected monitoring sites of the Environment Agency: soluble reactive phosphorus (SRP) measured from filtered (F) and non-filtered (NF) samples; total oxidised nitrogen (TON, mostly nitrate); ionised ammonia (NH_4) and chlorophyll *a*.



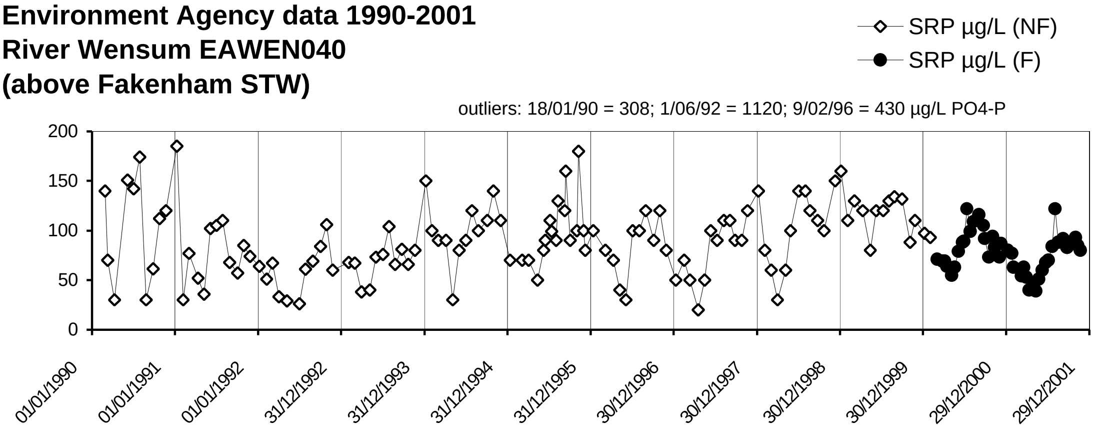


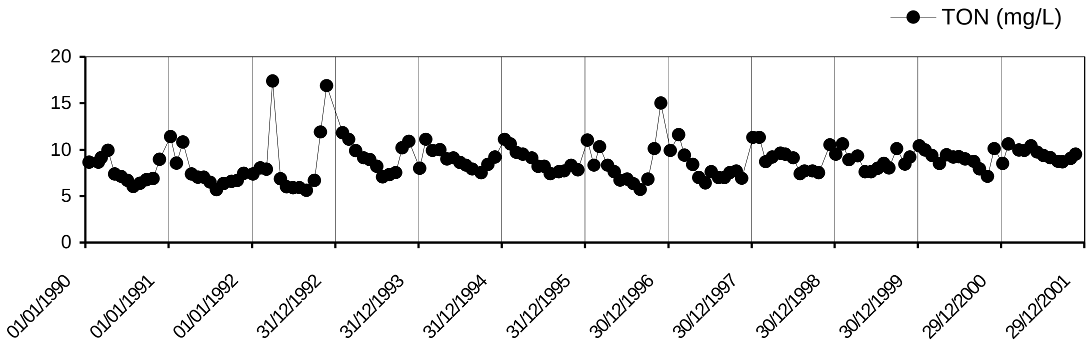

Environment Agency data 1990-2001
Wendling Beck EAWEN140
SRP µg/L (NF)
SRP µg/L (F)
6000
5000
4000
3000
2000
1000
0
01/01/90
01/01/91
01/01/92
31/12/92
31/12/93
31/12/94
31/12/95
30/12/96
30/12/97
30/12/98
30/12/99
29/12/00
29/12/01

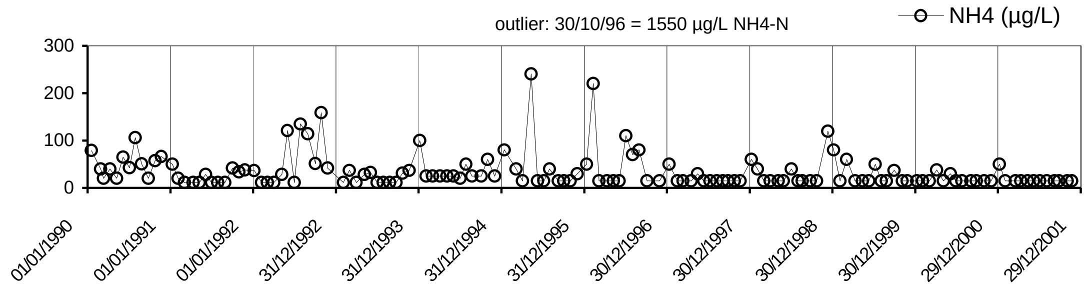

Environment Agency data 1990-2001
Wendling Beck EAWEN140
TON (mg/L)
25
20
15
10
5
0
01/01/90
01/01/91
01/01/92
31/12/92
31/12/93
31/12/94
31/12/95
30/12/96
30/12/97
30/12/98
30/12/99
29/12/00
29/12/01

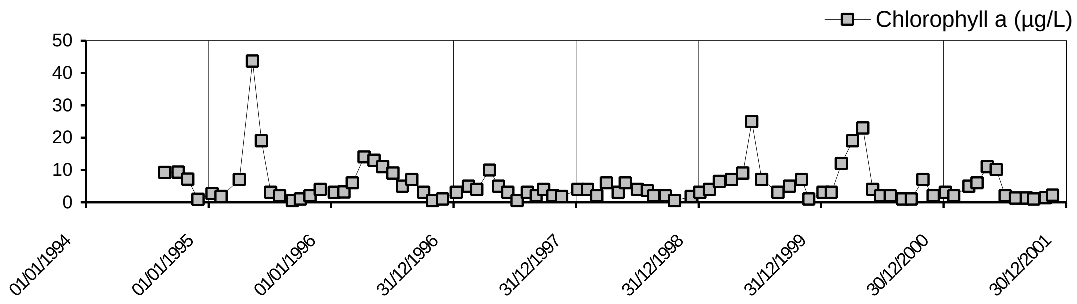

Environment Agency data 1994-2001
Wendling Beck EAWEN140
Chlorophyll a (µg/L)
100
80
60
40
20
0
01/01/94
01/01/95
01/01/96
31/12/96
31/12/97
31/12/98
31/12/99
30/12/00
30/12/01

Environment Agency data 1990-2001
River Wensum EAWEN040
(above Fakenham STW)

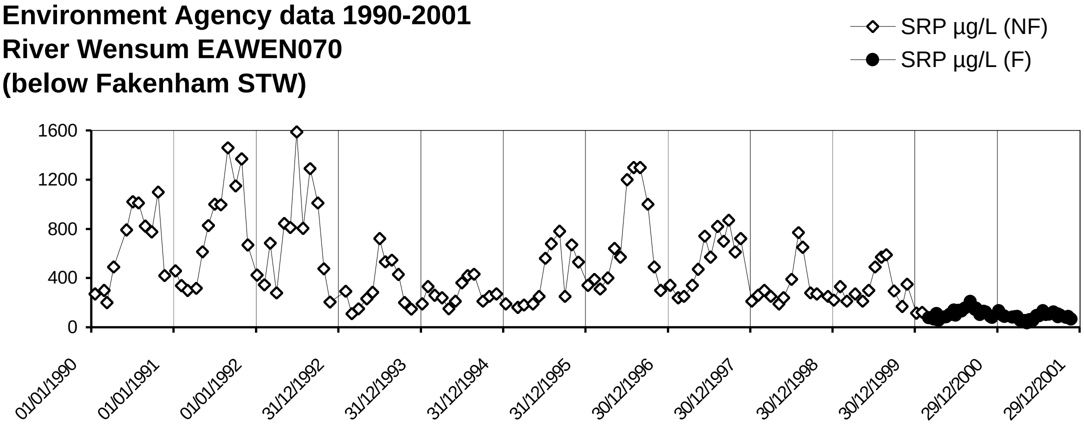


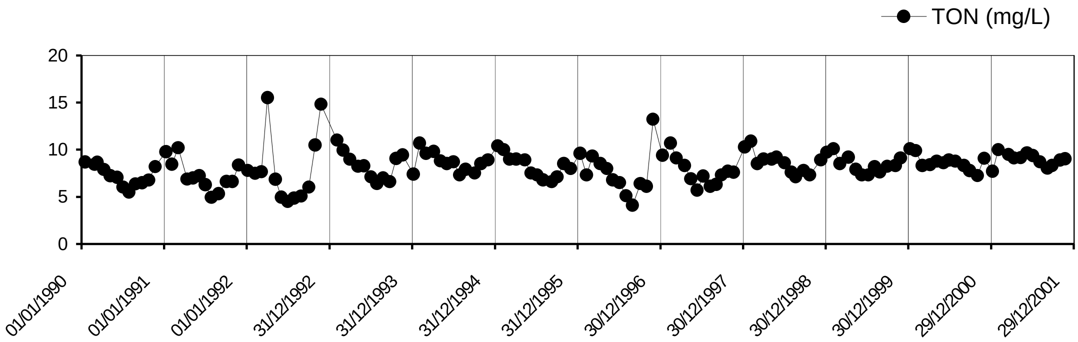


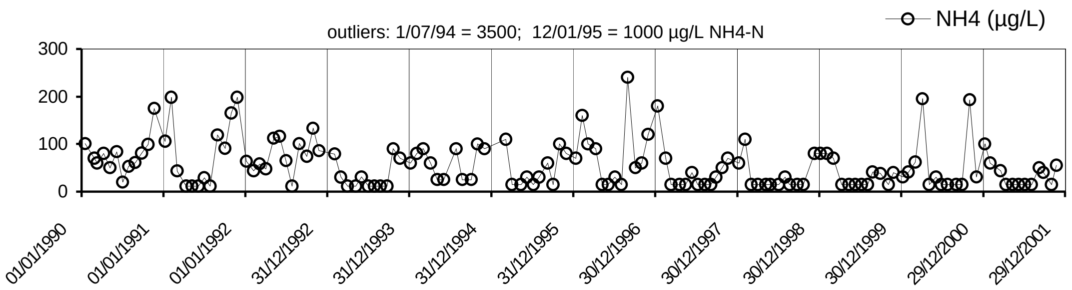


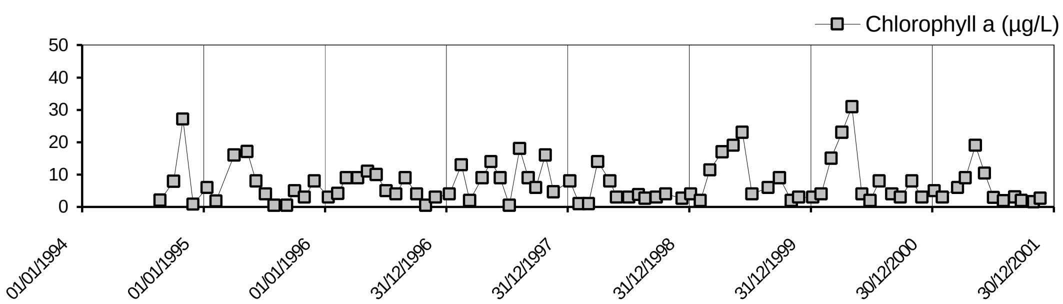


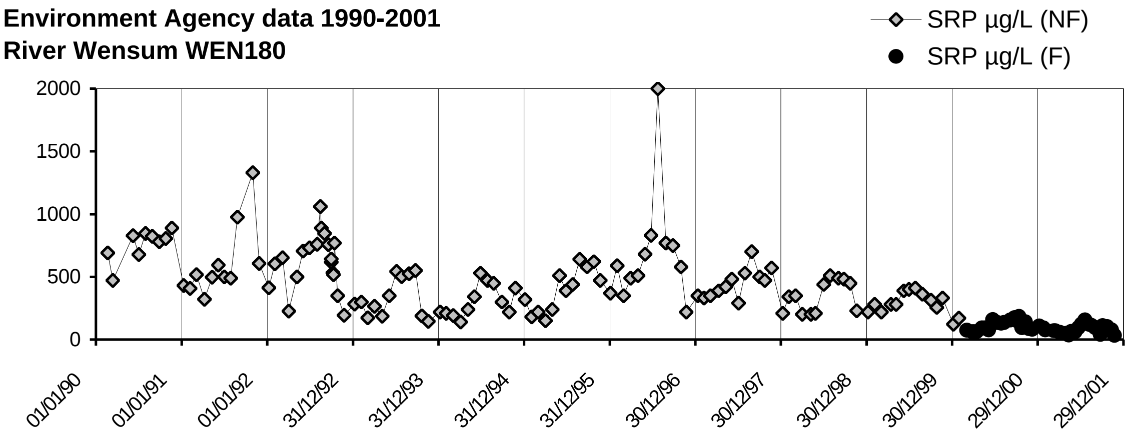
Environment Agency data 1990-2001
River Wensum EAWEN070
(below Fakenham STW)

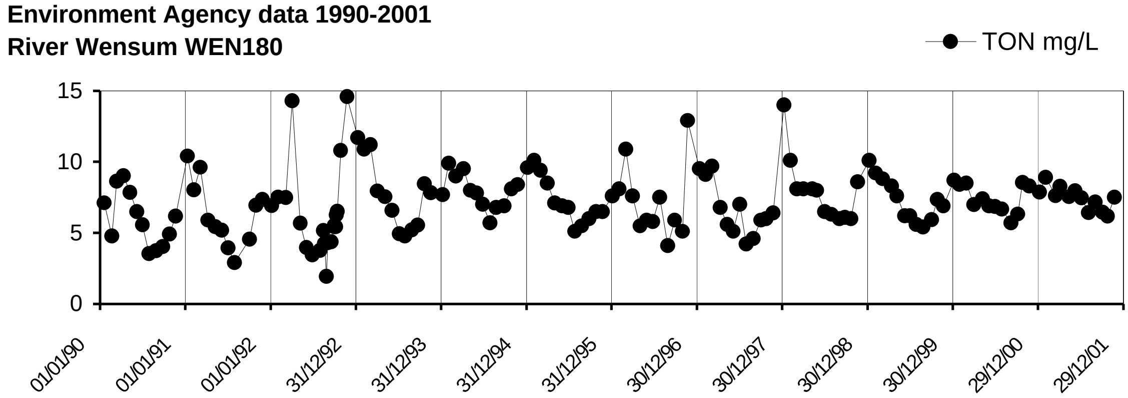


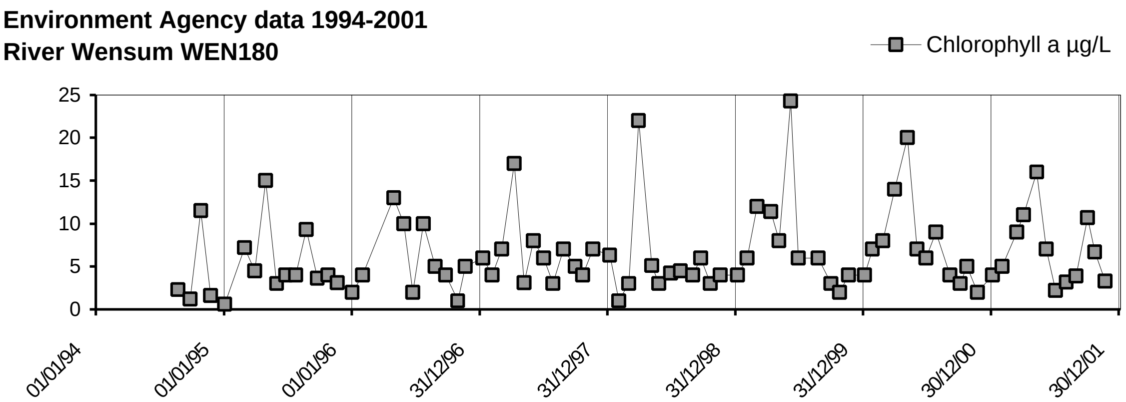












Total oxidised nitrogen (TON – mostly nitrate) concentrations displayed a sinusoidal pattern with constant period of one year and an apparent increasing amplitude downstream (towards the lowest values) and general decreasing amplitude with higher flow regimes. TON values oscillated around 5-10 mg/L and ranged within two (summer) to fifteen (winter) mg/L. These trends were not visible at EAWEN140 situated downstream from Dereham STW. No long term trends were detected.

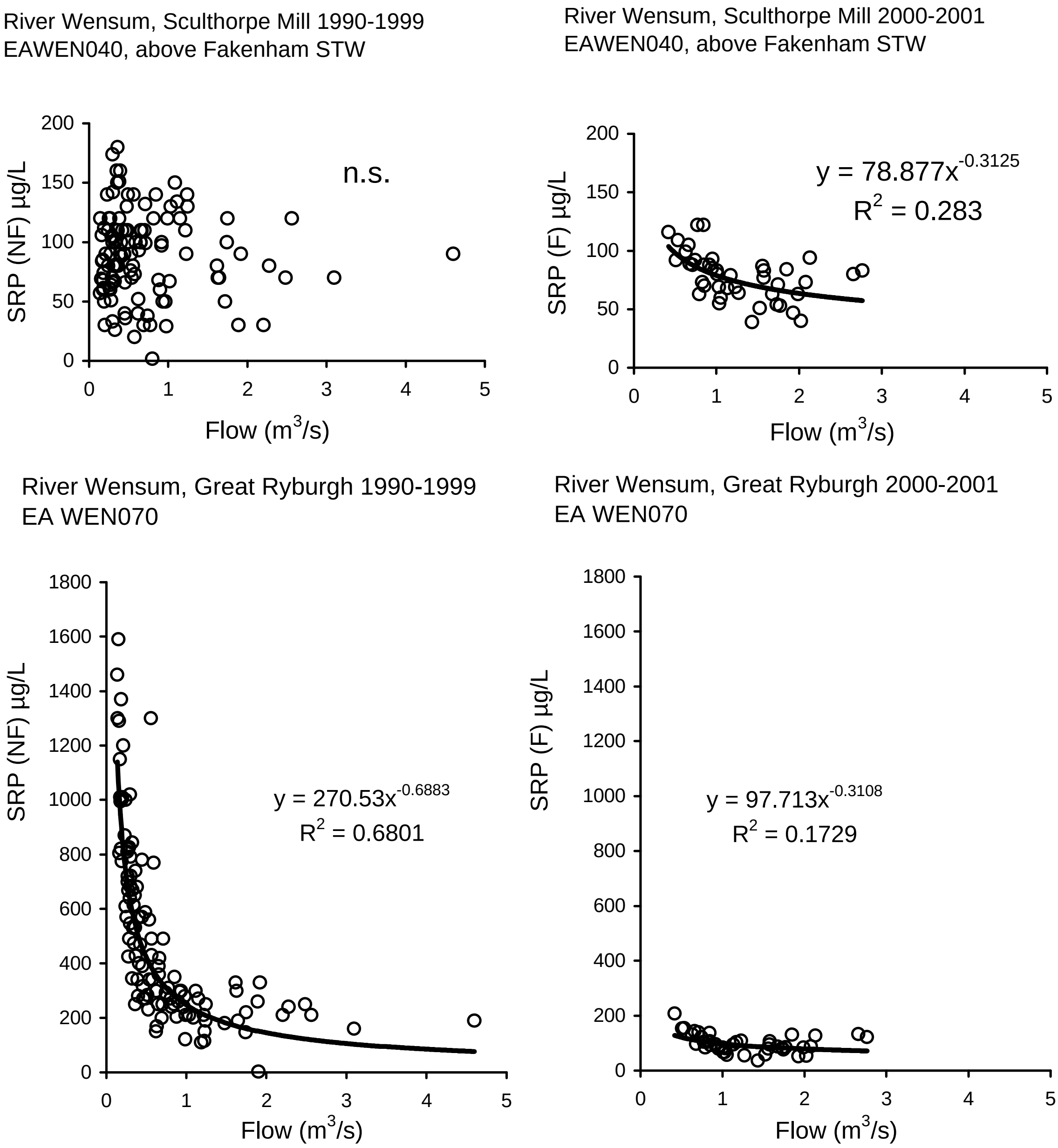
The levels of ionised ammonia (NH_4) were most of the time under the limit of detection above Fakenham STW (EAWEN040). NH_4 concentrations were however higher below the STW (EAWEN070). Maximal values peaked at 100-250 µg/L.

The levels of chlorophyll *a* (generally under 10 µg/L) reached an annual spring peak between 15-50 µg/L whatever the concentration of phosphorus although the peaks seemed more conspicuous in the lower catchment (compare sites EAWEN040 and EAWEN180). No substantial difference seemed to emerge between dry and wet years.

Impact of discharge on SRP concentrations prior to and after phosphorus control

The SRP concentrations were often relatively well predicted by the discharge using a power function (Fig 2.26). The relationship however did not hold at sites with relatively low SRP concentrations (EAWEN110 and EAWEN130) – Fig 2.26. The relationship was also weaker over the period 2000-2001 (Fig 2.26). At EAWEN040 the relationship was only significant when using SRP measurements based on filtered samples.

Figure 2.26 Flow as predictor of Soluble Reactive Phosphorus (SRP) analysed from filtered (F) and non-filtered (NF) samples at selected Environment Agency sites. n.s. = not significant. SRP as PO_4-P.



Artefacts in Environment Agency phosphorus data

SRP concentrations measured from non-filtered samples were of doubtful quality because they often exceeded the TP concentrations (Fig 2.27).

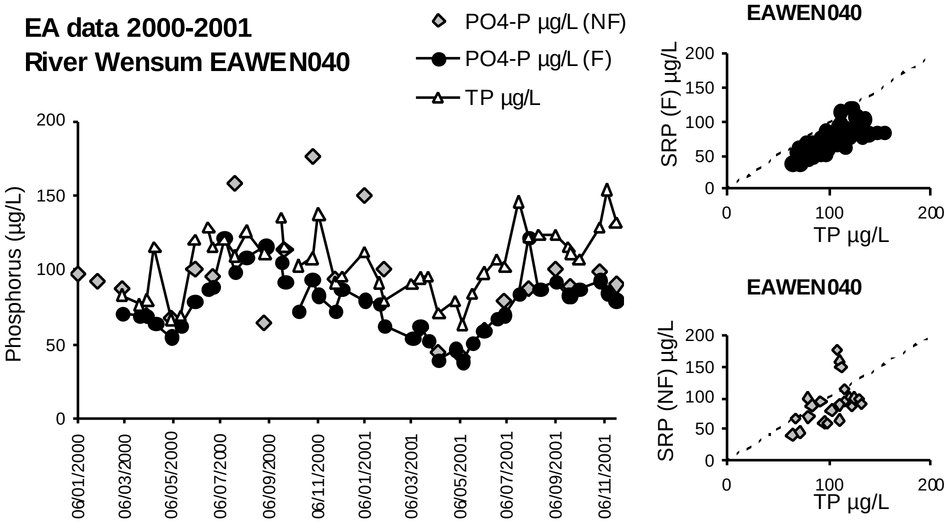


Figure 2.27 Environment Agency data. Soluble Reactive Phosphorus (SRP) concentrations analysed from filtered (F) and non-filtered (NF) water samples and Total Phosphorus (TP).

2.3.5 Discussion

Phosphorus stripping efficiency at the STWs

The similar pattern of variation of phosphorus input (crude sewage) in the two qualifying discharges did not reflect the suspected (AWS, personal communication) flushes of phosphorus from the laundry at Fakenham. These suspected peaks from the laundry were one of the reasons put forward by AWS to explain why Fakenham STW phosphorus removal performed less well than Dereham STW.

Reliability of the loads

The reliability of the calculated loads (observed and predicted) deserves more attention. Prior to phosphorus control, the mean coefficient of variation of TSP concentrations in the final effluent was 23.6 % (± 6.45 SD) for the main STWs and 67 % (± 38 SD) for the main industrial effluents. This can be due to daily or hourly variation (Moss *et al* 1988). Another source of error in the reported mean is that the monitoring was conducted during the standard working hours of AWS and EA (i.e. not at night or over the week-end). The reliability of the chemical analysis was not known and this needs to be investigated further in the light of the above reported anomalies (SRP>TP) and the analytical protocols used by the EA (Neal *et al* 2000c). The terminology was also not very clear and confusing in the EA database: phosphorus soluble reactive, phosphorus soluble reactive (filtered), orthophosphates, total soluble phosphorus.

The daily mean flow data could be influenced by (1) industrial and agricultural groundwater abstraction (although this seemed limited for the Wensum; Boar *et al* 1994, Hiscock *et al* 2001); (2) the complexity of the measurement and regulation of the flow regime (the data for Costessey Mill still need to be processed in order to account for gate movements upstream of the gauging station; EA, personal communication); (3) crump weirs at Swanton Morley display sometimes non-modular flow as a result of summer weed growth downstream (Center for Ecology and Hydrology, http://www.nwl.ac.uk/ih/nrfa/gauging_station_network /index.htm).

Measured loads were found to be much lower than predicted loads in different nutrient budget studies carried out in the Broads (Osborne & Moss 1977; Osborne 1981; Moss *et al* 1988; Johnes 1996). This virtual loss has been attributed to river bed sediment uptake and the underestimation of the bed load movement during flood events by the spot-sampling regime (Phillips *et al* 1999a). The observed loads of the present study were based on a much longer time series of spot-sampling covering a wide range of flow events (especially before phosphorus removal). Therefore the calculations were much less likely to be affected by the spot-sampling regime to the same extent as the previous studies.

The reliability of the predicted and observed loads based on monthly phosphorus data should be further tested against a continuous record because wide variations can occur depending on the sampling frequency and methods of calculation (Phillips *et al* 1999b; Webb *et al* 1997, 2000).

The determination of the diffuse source background is difficult to assess in practice because non-impacted streams (by point-sources), representative of the whole catchment, are virtually non-existent in Norfolk (Moss *et al* 1988; Fig 2.1). The most relevant data used in this study were from Section 2.2, similar to the adjacent catchment of the River Bure (Moss *et al* 1988). The diffuse sources of phosphorus expressed in kg/ha/year of cachment area were similar to previous studies carried out in Norfolk (see Table 2.9). The values published by Osborne and Moss (1977) were overestimated (Osborne 1981). However, it is not because all the studies found similar results that these figures are true. Indeed they share common limitations including (1) a low frequency of spot-sampling regime over a relatively short period of time and (2) fairly simple and pragmatic calculations that do not take into account explicitly the processes responsible for the observed concentrations. None of these studies (including the present one) can therefore provide well-estimated observed loads against which the predicted loads could be tested.

Table 2.9 Export coefficients of the diffuse sources derived for Norfolk river catchment areas. TP expressed in kg/ha/year P.

Locality	year	TP kg/ha/year	Reference
River Ant (Neatishead stream)	1975	0.729	Osborne & Moss (1977)
Alderfen Broad		0.107	Phillips (1977)
River Ant (Barton Broad)	1974-1975	0.231	Osborne (1981)
	1975-1976	0.215	
River Ant (drainage dykes)	1974-1975	0.139	
	1975-1976	0.129	
River Bure (Spixworth Beck)	1979-1980	0.121	Moss et al (1988)
River Bure (Wroxham)	1971	0.113	Johnes (1996)
	1981	0.098	
	1988	0.121	
	1996	0.125	
River Wensum (Costessey)	1988	0.136	
River Wensum (tributaries)	1990-1999	0.075-0.101	this study
	2000-2001	0.130-0.181	

Predicted levels of SRP

The predicted concentrations of SRP may not therefore be very accurate. However under low flow regime (Q95) the levels are predicted to be still extremely high for the sites situated below the main effluents. These figures are in accordance with the long term records showing extremely high peaks during the droughts of the nineties, even though phosphorus had been partially controlled by then (Fig 2.25). The large underestimation of SRP concentration at sites EAWEN130 and EAWEN040 may either come from the inaccuracy of the tentative SRP/TP ratio or form the underestimation of phosphorus loads coming from small STWs.

The effect of discharge

Discharge has been an effective predictor of SRP concentration because of the dilution effect of the point sources effluent. This was first observed in the River Wensum and Yare by Edwards (1973a,b). The relationship was weaker (or not significant) in 2000-2001 because of the relative high flow regime during that period and to a lower extent because of the decrease of phosphorus loads.

Limitations of the data

Where the diffuse sources of TP were predominant, then the relationship discharge-SRP did not seem to hold, however it was not clear whether this was real (as suggested in Section 2.2 for sites not impacted by known effluents) or an artefact produced by the unreliability of the SRP (NF) measurements (Fig 2.27). This unreliability of the SRP (NF) also blurred the seasonal trends of the period 1990-1999 (Fig 2.25, site EAWEN040).

Chlorophyll *a*

The cause of the spring peak of chlorophyll *a* has not been investigated although two origins are possible: (1) periphyton washed off because the short water residence time would not allow the phytoplankton to develop very much and benthic algal growth generally occurred in spring (Demars, personal observation); and (2) phytoplankton because of the connected standing water bodies (Pensthorpe Waterfowl Park, gravel pits, fisheries) especially in the lower part of the catchment (Moss 1977; Moss *et al* 1984; Basu & Pick 1996). The absence of peaks may be due to the low frequency of the spot-sampling regime.

Ecological implications

The levels of nitrogen (TON and NH_4) and phosphorus in the column water and the sediment (Demars, unpublished) of the River Wensum were extremely high. Neither phosphorus nor nitrogen has been depleted in the past 12 years. The present level of nutrients in the river Wensum therefore, seems to be in excess of the need by the primary producers. The levels of chlorophyll *a* were low and controlled by factors other than nutrients. Most of the standing biomass (and productivity) is contributed by aquatic vascular plants and filamentous algae however, and it is on these primary producers that the role of nutrients should be tested.

This raises fundamental questions:-

What is the fraction of phosphorus readily available to aquatic macrophytes?

What are the nutrient levels (phosphorus, nitrogen) under which the production of macrophytes is likely to be limited in riverine ecosystems?

What are the risks of nutrient enrichment?

What would be the most cost efficient management strategy?

These are addressed in Chapters 3 and 4.

2.3.6 Conclusion

The phosphorus control at Dereham and Fakenham led to a decrease of 64% and 76% respectively of the loads of total phosphorus (TP) discharged to the River Wensum. Predicted TP loads were in accordance with observed loads and decreased relatively predictably after the phosphorus removal. However further work is needed to test the validity, accuracy and precision of the observed TP loads against which the predicted loads could be more accurately tested. The soluble reactive phosphorus (SRP) concentrations were relatively low over the post-stripping period, however this was also due to the high flow regime of the year 2000-2001. Prediction at 95% chance exceedance of the flow regime (Q95) showed that the SRP concentration will still peak at very high levels should a drought occur in the future. The long term variations, as well as the predicted ones, of nutrient levels are still too high to expect any biological changes in the river.

Further problems remain to be tackled. The important ones are whether the point source phosphorus control can bring the desired improvements in water quality against the background of catchment sources (Moss *et al* 1988) and whether we can define a target level of phosphorus in calcareous streams.

3. LINKING AQUATIC VASCULAR PLANTS WITH THEIR RIVERINE ENVIRONMENT

3.1 Aquatic plant composition

3.1.1 Introduction

Many descriptive studies have examined patterns, at different scales, of riverine aquatic vascular plant distribution in the United Kingdom (e.g. Butcher 1927, 1933; Haslam 1978, Holmes 1983). Some studies have tried to test the match between plant composition and the riverine environment but datasets were too complex (e.g. Wiegleb 1984), the exploratory variables were not contrasted with each other (e.g. Dawson & Szoszkiewicz 1999), or the results were not interpretable because of flawed study design and analysis (e.g. Carbiener *et al*. 1990; Robach *et al*. 1996).

Ecological theories have been used as a basis for the present study. The species environmental requirements (niche *sensu* Hutchinson 1957) and species demographic processes (*sensu* MacArthur & Wilson 1967; Harper 1977) are the two ecological principles at work in the building of species assemblages. Both perspectives have received considerable support in ecological studies, but have neither been formally unified (Hubbell 2001) nor integrated successfully into a single testing procedure for aquatic organisms in a river network (e.g. Power *et al*. 1988; Hinch *et al.* 1991; Riis 2000) until very recently (Olden *et al*. 2001).

The habitat of the species can be characterised by two gradients: the spatial heterogeneity (adversity gradient) and the temporal variability (disturbance gradient) of the environment (Townsend & Hildrew 1994). Both gradients can be characterised by three components: biotic (competitors, consumers), chemical (alkalinity, nutrients, pollutants) and physical (geomorphological). Here these three components are used as as a means of testing the niche theory.

Aquatic plant species are more likely to disperse along rivers in a downstream direction (Wiegleb 1980; Wiegleb 1984; Demars & Harper 1998) than between catchment areas (Wiegleb 1983), which can be seen as biogeographical islands (MacArthur & Wilson 1967). Here, the structure of the river network will be used as a surrogate for species probability of colonisation with the aim of testing the importance of dispersal processes in the building of plant assemblages.

The habitat templet concept (Southwood 1988) hypothesises that the spatial and temporal gradients provide the frame upon which evolution forges characteristic assemblages of species traits. Historical events (e.g. glaciation), phylogenetic interdependence between traits, biogeographical area and spatial autocorrelation have all been implicitly assumed not to constrain the match between present-day species trait combinations and present-day environmental gradients (Townsend & Hildrew 1994; Bornette *et al*. 1994). Formulated predictions have so far been relatively poorly supported (e.g. Bornette *et al.* 1994; Willby *et al.* 2000; Wootton & Gee 2001). However, a significant match between species' attributes and environmental gradients would provide insights into the potential mechanisms involved in the species distribution within the considered templet.

A number of existing concepts have helped to shape the sampling strategy of this study, which then constrained the way of structuring and analysing the data. Figure 3.1 illustrates the simplified flow and organisation of the work. Many studies have used multivariate analysis models to try to link species (and species' traits) with environmental gradients, or spatial and temporal datasets (e.g. Jongman *et al.* 1995; ter Braak 1996; Chessel *et al.* 1997; Legendre & Legendre 1998; Blanc 2000). This study set out to build an holistic and pragmatic statistical strategy.

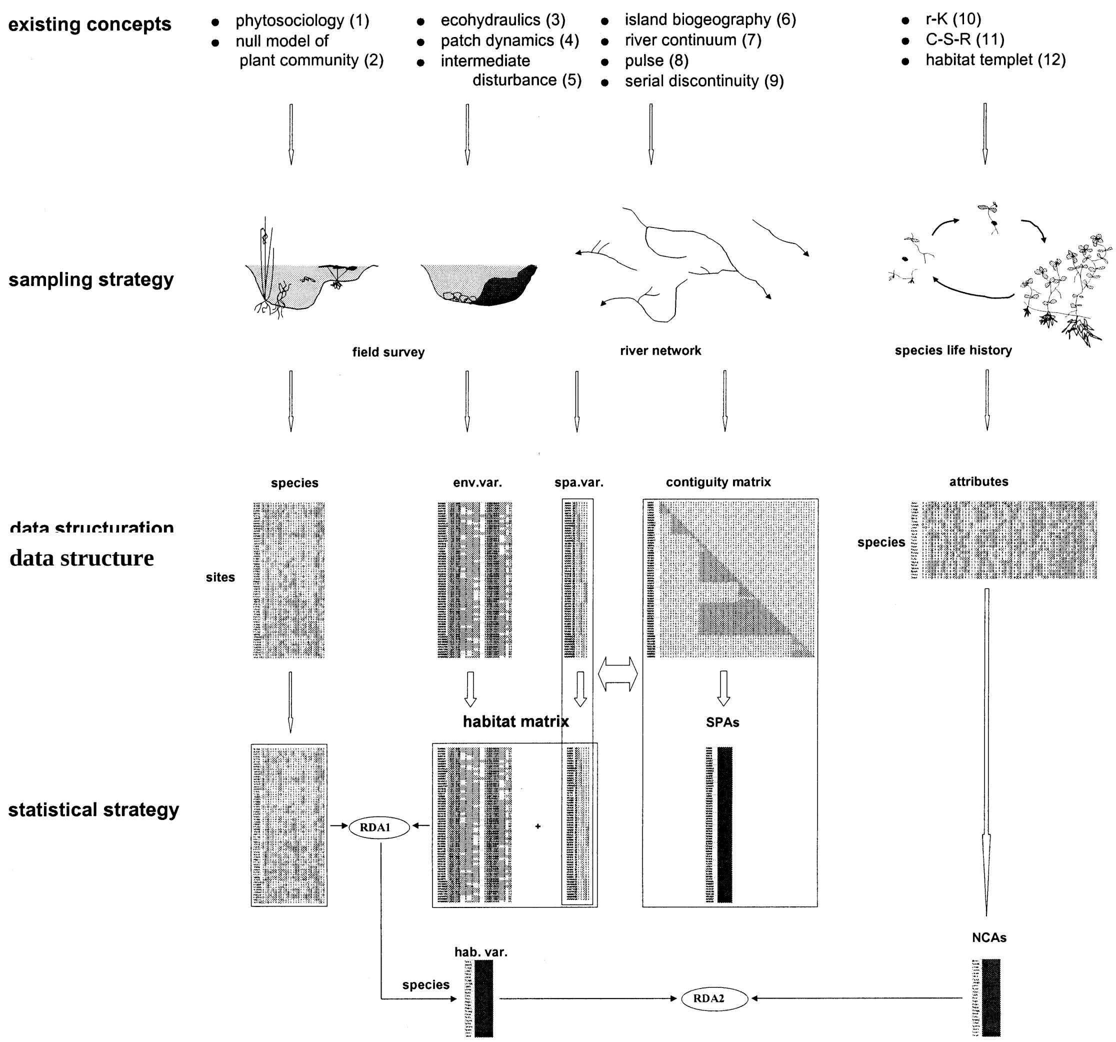


Figure 3.1 Flow diagram illustrating the organisation and sequence of thoughts. (1) Whittaker 1962; (2) Wilson & Gitay 1999; (3) Kemp *et al.* 1999; (4) Townsend 1989; (5) Grime 1973; (6) MacArthur & Wilson 1967; (7) Vannote *et al.* 1980; (8) Tockner *et al.* 2000; (9) Ward & Stanford 1983; (10) Pianka 1970; (11) Grime 1977; (12) Southwood 1988. env. = environmental; spa. = spatial; hab. = habitat; var. = variables; NCAs = Natural Combination of Attributes; RDA = Redundancy Analysis; SPAs = condensed spatial variables.

The intention of the study was to cover the potential impact of interspecific competition and environmental stochasticity on plant distribution, by testing for niche differentiation and colonisation probability, respectively (see Begon *et al.* 1996; p.300-304, 775, 801). The general null hypotheses were: –

Ho (niche) – H_No Species composition and species attribute assemblages do not match environmental gradients

Ho (dispersal) – H_Do Species composition and species attribute assemblages are not constrained by the structure of the river network (i.e. species probability of colonisation)

If H_No and H_Do are both rejected, then there are no interactions between the two perspectives.

More precise null hypotheses and their implicit assumptions could be set, although only once the gradients of the habitat templet of the system under study are known (see section 3.3).

Lowland rivers in Britain, as well as in Europe, have been re-profiled, straightened and embanked for several centuries, therefore there is little connectivity between them and the floodplain standing waters (mostly artificial) and valley fens (once feeding the river with pristine water, but now drained for farming or removed by gravel extraction) – Krause (1971); Bravard *et al.* (1986); Petts et al. (1989); Bornette & Amoros (1991); Bornette *et al*. (1994); Bornette *et al*. (1998); Piégay *et al*. (2000). The flow dynamics have often been further impacted by impoundments, channel structures and weirs constructed for the operation of water mills (Boar *et al*. 1994), contributing further to the loss of their natural pulse (Tockner *et al*. 2000). Thus the ecology of such rivers will not follow patterns present in pristine systems (Vannote *et al*. 1980) with distance downstream (Ward & Stanford 1983).

3.1.2 Methods

Study region

The study of a limited geographical area gives the advantage of having rivers with a common history (e.g. impact of glaciation, species introduction). The region was also restricted to the chalk solid geology overlain by boulder clay and glacial deposits. Five calcareous rivers draining lowland catchments (maximum elevation of the land 95m OD), mostly agricultural, were selected for their different levels of nutrient enrichment: the rivers Wissey (275 km^2 above Northwold); Bure (313 km^2 above Horstead); Nar (153 km^2 above Marham); Wensum (571 km^2 above Costessey) and the Wendling Beck (a tributary of the Wensum) - Norfolk, UK (Fig 3.2). Phosphorus removal at Sewage Treatment Works (STWs) has been implemented on three rivers: Bure (1986), Nar (1993), the Wensum and the Wendling Beck (Autumn 1999). The hydrographs of these catchments were very similar and the concentrations of Chlorophyll *a* small (rarely exceeding 10 µg/L; Environment Agency, unpublished). Weed cutting effort and boat traffic impact (on the lower Wissey) could not be quantified however (Environment Agency, unpublished). But the impact of boat traffic on the lower Wissey was likely to be negligible in terms of the resuspension of fine particles (Moss 1977; Demars, personal observation). More information about the geology, ecology, hydrology and physico-chemistry of these river basins is given in Edwards (1971), Baker *et al*. (1978), Baker & Lambley (1980), Moss (1983), Moss *et al*. (1988), Petts (1994), Boar *et al*. (1994), and Hiscock *et al*. (2001).

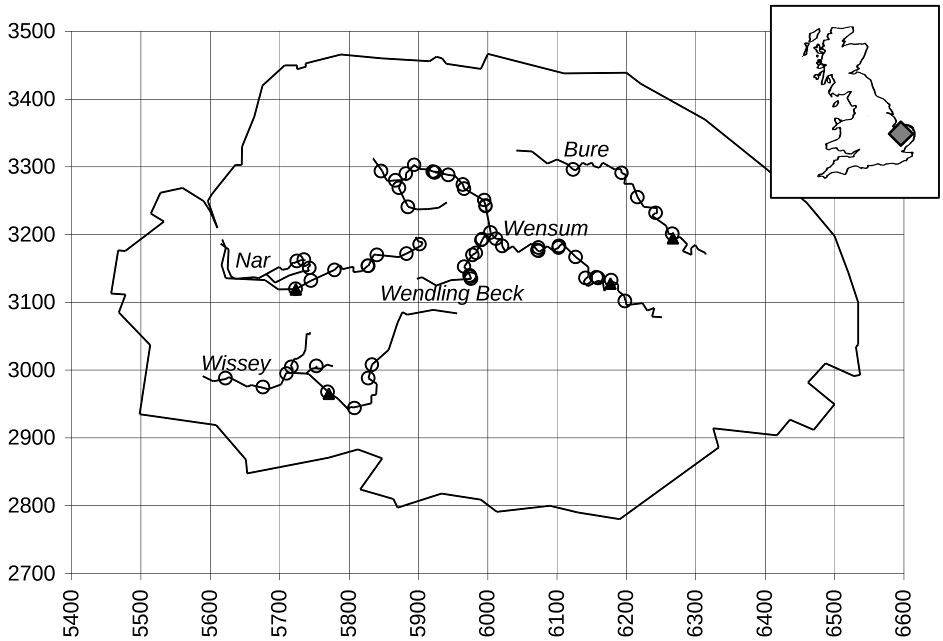


Figure 3.2 Location of sampling sites in the study area (Norfolk, UK).

Data collection

Sixty-two sites (App 3.1) were selected at Environment Agency 'Harmonised Monitoring Scheme' locations where long term chemical datasets (1990-1999) were available. These sites were approximately evenly distributed along the water course. Pairs of sites were also sampled above and below the main STWs and weirs in the River Wensum catchment area. Two snapshot water samples (winter and summer) were collected from sites mostly situated in tributaries not impacted by STWs and not covered by the Environment Agency monitoring. The mean ranges, from the monitoring data of the Environment Agency, were: temperature (9.8-11.8°C), alkalinity (4.4-5.3 meq/L), pH (7.7-8.3), conductivity (647-874 µS/cm), turbidity (2.6-7.7 FTU – Wass *et al*. 1997). The physico-chemical characteristics were homogeneous across the sites investigated and typically reflected the influence of the chalk aquifer and land use (Johnes 1996; Sear *et al*. 1999), at least during the period 1990-1999 (covering about 120 monthly measurements). The low Biochemical Oxygen Demand (1.0-2.3 mg/L O_2) reflected the efficiency of the STWs. No notifiable saprogenic problems were recorded during the survey. All the streams were polluted by nitrate in the studied region: total oxidised nitrogen ranged from 6.4 to 12.9 mg/L NO_3-N and was never close to depletion even during the summer. Therefore the impact of nitrate could not be studied. Two chemical gradients however were retained for the analyses: phosphorus (soluble reactive phosphorus - SRP) because it could limit the growth of plants in some streams and ammonium (NH_4) because high levels may be toxic for aquatic vascular plants (e.g. Dendène *et al.* 1993).

The physical characteristics of the sites were recorded in the same way as in the Environment Agency's Mean Trophic Rank (MTR) standardised sampling methodology (Environment Agency, 1996). These were shade intensity, width, depth and the substrata (gravel, sand, silt, clay) acting here as a surrogate for current velocity (Kemp *et al*. 1999). Instead of choosing a 100 metre stretch length as unit site, about 500 m^2 of channel, geomorphologically homogeneous and largely unshaded, were surveyed for aquatic plants.

To achieve an exhaustive survey within a site, plants were surveyed by snorkelling in zigzags going against the current for about 40 minutes (although no strict time limit was imposed). An index of cover-abundance (*sensu* Braun-Blanquet 1932; see Poore 1955) was allocated for each taxon. Herbarium sheets of all taxa recorded for this study have been deposited in the University of Leicester Herbarium (LTR). Nomenclature follows Stace (1997). *Ranunculus trichophyllus* and *R. penicillatus* ssp. *pseudofluitans* var. *pseudofluitans* were aggregated with the non-flowering specimens. An isozyme technique was developed to identify the non-flowering population of *Callitriche* (Demars & Gornall 2002). *Callitriche obtusangula* and *C. platycarpa* were found very frequently and were aggregated because a few sites were not screened and it was not possible to give the index of cover-abundance for each species when often two grew in the same site. *Lemna minor* and *L. minuta* were pooled together for the analysis as *L. minuta* was probably overlooked in the first summer. Filamentous green algae taxa (e.g. *Cladophora*, *Enteromorpha*, *Vaucheria*) were aggregated because the abundance/dominance of some of these taxa can be very variable and the identification down to species level problematic. Characeae, mosses and liverworts were identified to species but excluded from this study as the frequency of each taxon was too low.

The River Wensum catchment area was surveyed mostly during summer 1999 and the other rivers during summer 2000. A subset of 12 sites in 2000 and 16 sites in 2001 were resampled in the River Wensum and the Wendling Beck to investigate the temporal variability of the species composition.

Data structure

Matrices for the spatial study

The Braun-Blanquet index of cover-abundance was coded numerically from 1 to 6. Taxa recorded in less than 3 sites (rare and transient species) were eliminated as were all the strictly emergent riparian species (e.g. *Glyceria maxima*; *Phragmites australis*; *Typha latifolia*) to exclude statistical and sampling bias. It was felt that the riparian emergent plants might reflect the environmental conditions of the bank more than the in-stream water quality. This resulted in the species matrix 'sites x species' (62 rows x 25 columns; App 3.2). The resulting list of species is displayed in the legend of Fig 3.5 and App 3.1.

The catchment identity was binary coded as a dummy variable (CATCH). The distance (DIST) from the source of each site (in km) was measured from 1/50000 Landranger maps with a map measurer. Shade intensity and substrate modalities were coded (**1**<20; **2** 20-40; **3** 40-60; **4** 60-80; **5**=80%). Mean width and depth were expressed in metres. Soluble reactive phosphorus (mg/L PO_4-P) and ammonium (mg/L NH_4-N) were log transformed. Filamentous algae were regarded as a biotic constraint on aquatic plant development and so were included in the habitat matrix (coded as the plants). This resulted in the habitat matrix 'sites x habitat characteristics' (62 rows x 16 columns; App 3.3 and 3.4).

Matrices for the temporal study

The annual survey, carried out in 12 sites in the Wensum and the Wendling Beck, has accumulated observations over a three year period (1999, 2000, 2001). Replicated sites were given a code for each year. Aquatic vascular species were retained if they occured at least twice (either across sites or years). This resulted in the annual species turnover matrix 'replicated sites x species' (36 rows x 23 columns). A binary-coded matrix was produced to identify the 12 sites and the 3 years. This resulted in the annual identifier matrix 'pseudo-sites x identifiers' (36 rows x 12+3 columns).

The same approach was applied to the biannual survey carried out in another four sites that were added to the previous 12 leading to a total of 16 sites sampled twice (1999 and 2001). This resulted in the biannual species turnover matrix (32 rows x 23 columns) and the biannual identifier matrix (32 rows x 16+2 columns).

Attribute matrix

The attribute matrix 'species x attributes' (25 rows x 58 columns; App 3.5) was derived from Willby *et al*. (2000). The list of attributes and attribute codes is reproduced in Table 3.5.

Data analysis

Two standardized principal components analysis (PCA) were run using the habitat matrix: one including all the variables (i.e. columns); and one excluding DIST and the dummy variable CATCH to investigate the strength of the habitat variables and their relationships. Another standardized PCA was run using the species matrix to investigate how the species relate to each other (Jongman *et al*. 1995; ter Braak & Smilauer 1998).

Then redundancy analyses (RDAs), based on a correlation matrix, were performed to (1) investigate which variables from the habitat matrix would significantly explain the species composition alone; (2) include in the final RDA the significant variables; and (3) finally quantify the variance partitioning of the selected habitat variables, as well as their interaction factors. The significance of the axes of the final RDA model (named RDA1) were also tested one after another with the Monte Carlo permutation test.

The potential effect of time alone on the species composition was also investigated with RDA. The Monte Carlo random permutation test was based on a random cyclic shift restricted to the replicated time series, which consisted of each resurveyed site. Partial RDAs enabled changes in composition between different combinations of years: 1999-2000, 2000-2001 and 1999-2001 to be examined. The analyses were also repeated on presence-absence species matrices.

It is more complicated to test whether attributes match habitat variables because it requires to produce a three-table ordination (species, habitat and attribute). The ‘double CCA’ approach (Lavorel *et al*. 1998, 1999) was used although here a linear model was more appropriate, i.e. ‘double RDA’. The habitat x species table, produced by the RDA1 model, was tranposed into the species x habitat table. Species scores from the initial six attribute matrix PCA axes were used to rationalize the number of potential predictive variable (58), this choice being based on the pattern of decay in eigenvalues and the amount of variance explained by the combined axes. These axes could be seen as Natural Combination of Attributes (NCAs; Lavorel *et al*. 1999). This resulted in the NCA matrix ‘species x NCAs’ (25 rows x 6 columns). Then redundancy analyses (RDAs), based on a correlation matrix, were performed to (1) investigate which NCAs (all linearly independent from each other) would significantly explain the projection of habitat variables in the species space; (2) run the final RDA model (named RDA2); and (3) test the number of significant axes. The NCAs were then translated back to attributes themselves by taking the matrix product between the attribute scores of the attribute matrix PCA and the NCAs scores of the RDA2 model (Lavorel *et al*. 1999).

The potential effect of time alone on the NCAs was also investigated with a RDA model based on the NCAs matrix and the species x time table derived from the temporal RDA models. The test was an unrestricted Monte Carlo random permutation.

All the multivariate analyses were performed with Canoco 4.0 (ter Braak & Šmilauer 1998) and ordination graphs with Canodraw 3.1 (Smilauer 1992), unless otherwise stated. Further details and analyses are contained in Demars (2002).

3.1.3 Results

Definition of the templet

The PCA ordination diagram of the environmental variables (Fig. 3.3a) shows two relatively independent gradients: geomorphological characteristics on the first axis (GRAVEL, SILT, DEPTH, WIDTH) and nutrient enrichment (SRP, NH4) mostly along the second axis. The biotic component (ALGAE) occupies an intermediate position between the level of nutrients and the physical characteristics of the river channel. Because of the relative independence of the three components (physical, chemical and to a lesser extent biotic), the dataset is appropriate for defining the templets on which species’ and NCAs’ matches will be tested. The inclusion of the spatial variables shows the relationship of the catchments with the environmental variables (Fig. 3.3b).

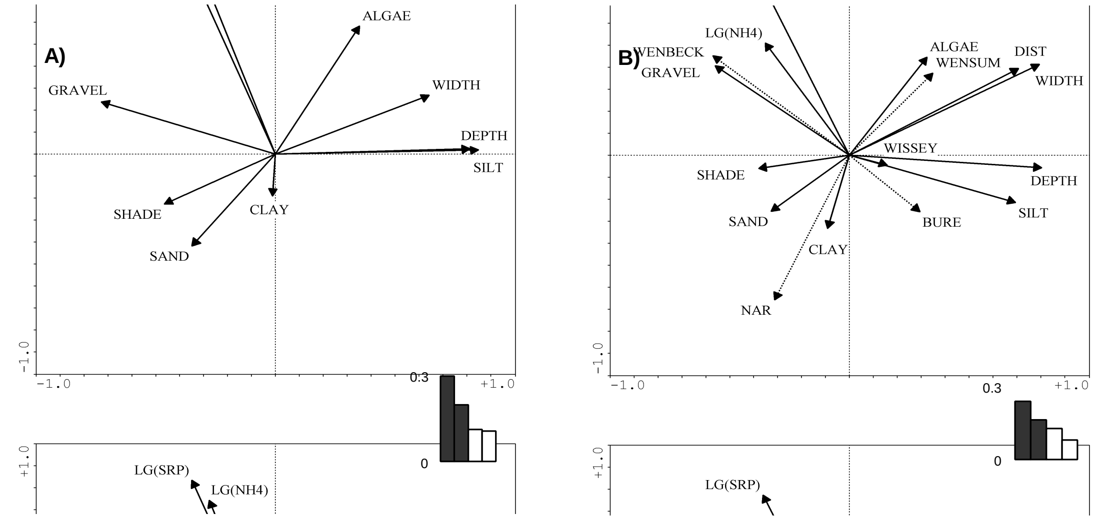


Figure 3.3 Principal components analysis (PCA) ordination diagram of (A) the environmental variables alone and (B) all habitat variables (environmental + spatial). Diagram A shows a relative independence of the geomorphological (gravel, depth, silt width), nutrient (soluble reactive phosphorus, ammonium) and biotic (algae) factors. Axis 1 of the PCA is horizontal and axis 2 is vertical. Insets depict the pattern of decay of the eigenvalues (strength of the PCA axes).

System specific implicit assumptions

It is important to consider the implicit assumptions about the system under study before formulating the specific null hypotheses. (1) Species and species attributes (intrinsic properties) are independent of the spatial and temporal gradients (extrinsic properties, *sensu* Steneck & Dethier 1994). (2) There is no lateral connectivity with the standing water bodies. (3) When the catchment dummy variable was used to identify the biogeographical area, the test implied that the sites were independent. (4) If the contiguity matrix is used to code for the spatial structure of the river network, then sites are assumed evenly distributed along the river corridors. (5) Weirs situated along the rivers have no effect on species dispersal. (6) Weed cutting (notably sections of the river Wensum) and boat traffic (essentially lower Wissey) have no effect on the species distribution. (7) Species have an equal probability of dispersion. (8) Historical constraints are similar between catchments. (9) Biological species attributes are relevant to tested environmental gradients. (10) The temporal species turnover between 1999 and 2000 is negligible. (11) The species distribution is neither affected by the regional species pool, nor (12) by the nutrient availability in the sediment.

System specific hypotheses

There is no match of the species composition or NCAs with:-

(1) the nutrient gradient (H_No nutrients);

(2) the biotic gradient (H_No biotic);

(3) the geomorphological gradient (H_No geomorphology);

(4) the longitudinal connectance (H_Do connectance);

(5) the individual river basins (H_Do biogeographical island);

(6) time (Ho time).

There is no interaction between the above specific H_No and H_Do (H_{NxD}o interaction).

Species match with environmental gradients and river network structure

The first step of the RDA1 model lists the environmental and spatial variables in order of the variance they explain singly (Table 3.1, marginal effect). Species composition was significantly related to the distance, width, depth, silt, gravel and catchment variables, but not to algae, nutrient levels, shade, sand and clay (after Bonferroni correction). There was also no significant additional effect of the nutrients, algae and shade taken all together (partial RDA with distance, catchment, depth and silt as covariables; axis 1 eigenvalue: 0.028, P>0.11; all canonical eigenvalues: 0.058, P>0.23). The stepwise selection showed that four variables were explaining 30.2% of the variance: distance, catchment, depth and silt (Table 3.1). It was 75% of the sum of all canonical eigenvalues (0.405) of all 12 variables (16 columns) considered in the RDA1 model. The interaction factor (DEPTH+SILT) x DIST was highly significant (P<0.003). The interactions of DIST x CATCH and (DEPTH+SILT) x CATCH were not significant (P>0.1). The variance partition is presented in Figure 3.4.

The first four axes of the final RDA1 model were significant (AX1 P<0.01, AX2 P<0.01, AX3 P<0.02, AX4 P<0.02) and explained 24.7% of the variance (82% of the sum of all canonical eigenvalues of the selected variables). The species – habitat variables biplot ordination diagram is presented in Figure 3.5. The fourth axis mostly distinguished the species composition of the Wendling Beck from the Bure catchment (not represented for brevity). The species x habitat table produced by the RDA1 model enables each species relationship with all habitat variables included in the final model to be examined in detail (Table 3.2).

The constrained ordination diagrams (RDA1, RDA1*bis*) of the species were similar to the unconstrained species PCA ordination diagram. It meant that the selection of exploratory variables was good.

Temporal effect on species composition

The RDA model based on the 'annual turnover matrix' showed that a significant change of species composition had occurred from 1999 to 2001 (P<0.013). But this species turnover was slow because the changes in species composition between 1999-2000 (P>0.378) and between 2000-2001 (P>0.285) were not significant. The RDA model based on the 'biannual turnover matrix' explained 3% of the variance and confirmed the significant shift between year 1999 and year 2001 (P<0.001). Analyses ran on the presence-absence species matrices gave similar results.

Table 3.1 Percentage of explained variance by each habitat variable using a redundancy analysis model (RDA1): singly (marginal effect) and after stepwise selection.

	Marginal effect		Stepwise selection		
	% expl. var.	*P*	order	% expl. var.	*P*
DIST	11.1	< 0.001	**1**	**11.1**	< 0.001
WIDTH	10.7	< 0.001	8	1.2	0.522
DEPTH	9.3	< 0.001	**2**	**5.3**	< 0.001
WENSUM	5.3	< 0.001	**3**	**4.0**	< 0.001
SILT	5.0	< 0.001	**6**	**2.8**	**0.002**
GRAVEL	4.1	0.001	9	1.0	0.654
NAR	3.9	0.004			
WISSEY	3.9	0.004	**4**	**2.7**	**0.007**
WENBECK	3.3	0.011	**5**	**2.7**	**0.006**
BURE	3.0	0.033	**7**	**1.6**	**0.198**
ALGAE	2.8	0.039	**n.s.**		
SHADE	2.8	0.045	**n.s.**		
LG(NH4)	2.4	0.112	**n.s.**		
SAND	2.2	0.132	**n.s.**		
LG(SRP)	2.2	0.148	**n.s.**		
CLAY	0.7	0.944	**n.s.**		

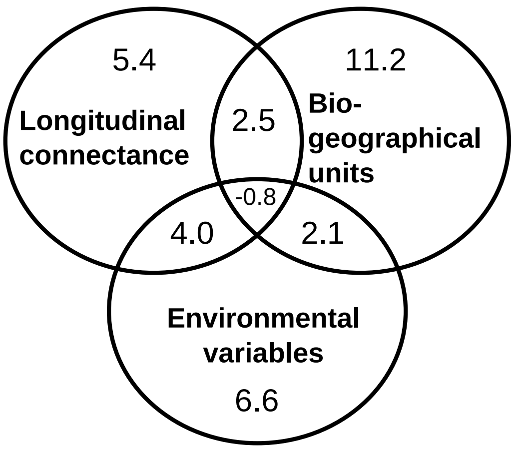


Figure 3.4 Variance partitioning of the environmental and spatial variables. All values expressed in % of explained variance.

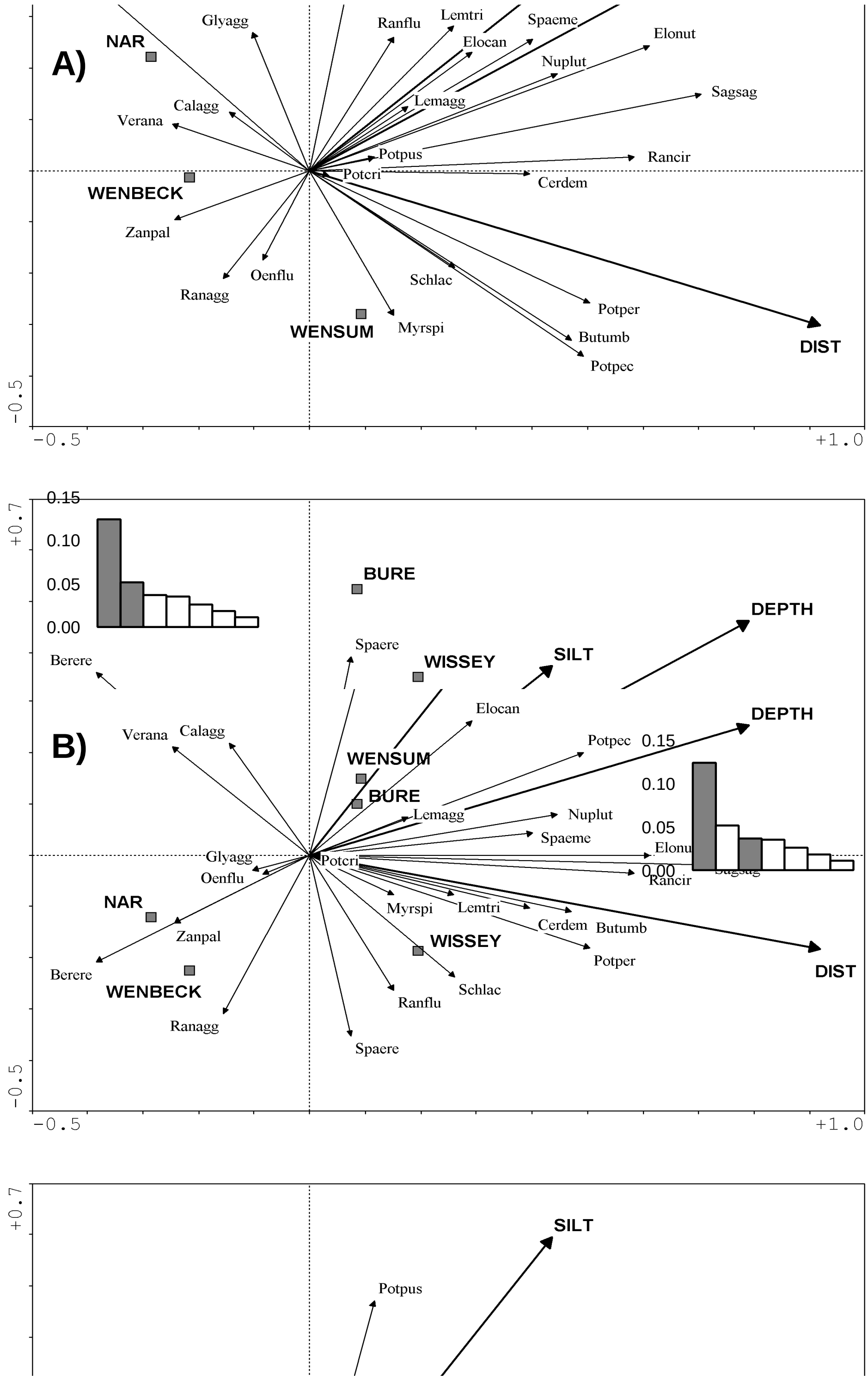


Figure 3.5 Species and habitat variables projection into the Redundancy Analysis RDA1 ordination diagram: (A) axes 1x2 and (B) axes 1x3. Axis 1 is horizontal in both diagrams. Insets depict the pattern of change in eigenvalues and highlight which axes are displayed (shaded chart bar). Complete list of species: Berere = *Berula erecta*; Butumb = *Butomus umbellatus*; Calagg = *Callitriche platycarpa*/*obtusangula*; Cerdem = *Ceratophyllum demersum*; Elocan = *Elodea canadensis*; Elonut = *E. nuttallii*; Glyagg = *Glyceria fluitans*/*notata*/*declinata*; Lemagg = *Lemna minuta*/*minor*; Lemtri = *L. trisulca*; Myrspi = *Myriophyllum spicatum*; Nuplut = *Nuphar lutea*; Oenflu = *Oenanthe fluviatilis*; Potcri = *P. crispus*; Potpec = *P. pectinatus*; Potper = *P. perfoliatus*; Potpus = *P. pusillus*; Ranagg = *Ranunculus penicillatus* ssp. *pseudofluitans* var. *pseudofluitans* / *trichophyllus*; Rancir = *R. circinatus*; Ranflu = *R. fluitans*; Sagsag = *Sagittaria sagittifolia*; Schlac = *Schoenoplectus lacustris*; Spaeme = *Sparganium emersum*; Spaere = *S. erectum*; Verana = *Veronica anagallis-aquatica*; Zanpal = *Zanichellia palustris*.

Table 3.2 Species x habitat variables table derived from the RDA1 final model. Shaded cells denote respectively positive (score = 0.3) and negative (score = -0.2) associations between species and habitat variables. DIS = distance; DEP = depth; SIL = silt; BUR = Bure; WEN = Wensum; WIS = Wissey; WEB = Wendling Beck; Cfit = Cumulative fit per species as fraction of variance of species.

	DEP	SIL	DIS	CATCHMENTS					Cfit
				BUR	NAR	WEN	WIS	WEB	
Berere	-0.18	-0.08	-0.36	0.28	0.52	-0.46	-0.09	-0.03	46
Butumb	0.24	-0.04	0.59	0.02	-0.21	0.39	-0.18	-0.17	42
Calagg	0.03	0.05	-0.16	0.31	0.01	0.06	-0.01	-0.34	29
Cerdem	0.17	0.31	0.39	-0.06	-0.10	0.01	0.21	-0.08	31
Elocan	0.45	0.30	0.16	0.24	-0.06	0.03	0.00	-0.18	25
Elonut	0.62	0.30	0.47	0.10	-0.21	-0.04	0.31	-0.11	45
Glyagg	0.07	0.05	-0.13	0.23	0.32	-0.23	-0.10	-0.10	18
Lemagg	0.28	0.15	0.15	0.11	0.16	0.06	0.04	-0.40	20
Lemtri	0.42	-0.05	0.18	0.39	-0.16	-0.11	0.15	-0.13	38
Myrspi	-0.02	-0.13	0.27	0.07	-0.24	0.29	-0.03	-0.18	21
Nuplut	0.41	0.38	0.33	0.16	-0.19	-0.07	0.08	0.11	33
Oenflu	-0.12	-0.21	-0.04	-0.09	-0.09	0.10	-0.13	0.17	8
Potcri	0.04	-0.07	0.03	0.09	-0.13	0.03	0.03	-0.01	3
Potpec	0.32	0.16	0.54	-0.15	-0.24	0.56	-0.12	-0.31	46
Potper	0.28	0.00	0.58	-0.18	-0.11	0.31	0.05	-0.23	41
Potpus	0.22	0.36	-0.01	-0.05	-0.08	0.18	-0.07	-0.07	27
Ranagg	-0.27	-0.35	-0.02	-0.08	0.12	-0.03	-0.30	0.28	27
Rancir	0.44	0.30	0.58	0.29	-0.22	0.12	0.01	-0.18	44
Ranflu	0.16	-0.04	0.06	-0.08	-0.13	-0.25	0.66	-0.10	45
Sagsag	0.56	0.46	0.61	0.08	-0.23	0.03	0.32	-0.19	57
Schlac	0.05	-0.08	0.35	-0.07	-0.11	0.14	0.07	-0.09	17
Spaeme	0.48	0.22	0.27	0.21	-0.16	-0.08	0.15	-0.03	26
Spaere	0.17	-0.03	0.00	0.05	0.26	-0.48	0.22	0.15	33
Verana	-0.11	0.10	-0.27	0.08	0.17	-0.02	-0.05	-0.18	16
Zanpal	-0.25	-0.27	-0.15	0.09	0.02	0.01	-0.05	-0.06	17

Natural Combination of Attributes

Six NCAs summarised 66% of the attribute variance. The main pattern of variation are summarised in Table 3.3.

Table 3.3 Natural combination of attributes (NCAs).

Natural Combination of Attributes	Negatively related attributes	Positively related attributes
NCA1	Multiple apical growth point, small leaves, reproduction by fragmentation, water pollinated, medium body size (e.g. *Ranunculus circinatus*, *Elodea canadensis*, *Ceratophyllum demersum*)	High root:shoot biomass, plastic growth form (anchored, with floating, emergent or heterophyllous large leaves), waxy leaf texture, single basal growth point, vegetative reproduction by stolons (e.g. *Schoenoplectus*, *Sagittaria*, *Potamogeton pectinatus*)
NCA2	Anchored, submerged growth form with soft non-waxy leaves, large body size, high number of reproductive organs (e.g. *Ranunculus fluitans*, *Potamogeton perfoliatus*, *Myriophyllum spicatum*)	Free floating surface growth form, small size, very high number of reproductive organs, vegetative reproduction by budding, single basal growth point, low flexibility, small fruit size (e.g. *Lemna minor*/*minuta*)
NCA3	Medium size fruit, self pollination, early production of reproductive organs	Low number of reproductive organ, large fruit size, rigid leaves
NCA4	Capillary leaves, perennial	Entire leaf , annual, pollination by air bubbles,
NCA5	mid period of production of reproductive organs, medium number of reproductive organs, high body flexibility	none
NCA6	none	Medium body size, biennial/short lived perennial, and anchored heterophylly growth form

Relationships between NCAs and habitat variables

The results produced by the general model of RDA2 testing the hypothesis that NCAs were not related to the habitat variables are presented in Table 3.4. One major finding is that the habitat variables do not predict the most robust NCAs (NCA1, NCA2). Then trends (marginally significant after Bonferroni correction) were found in the general model as well as in each specific model testing for the geomorphology (DEPTH, SILT), longitudinal connectance (DIST), and biogeographical island (CATCH) null hypotheses. The canonical axes of the final RDA2 model including NCA3 and NCA5 as explanatory variables were significant (*P*=0.007) and explained 20.2% of the variance of the habitat variables.

The triplot ordination diagram illustrates the relationships between the NCAs (as well as the attributes), species and habitat variables (Fig 3.6). The longitudinal gradient (DIST) was marginally negatively related to NCA5 and positively to NCA3, while the geomorphological gradient (DEPTH) was only marginally positively related to NCA3 (see above for the shift in attributes along these NCAs and Fig 3.6).

Table 3.4 Percentage of explained variance by each NCAs using RDA2 model. Since all the variables were linearly independent, no stepwise selection was carried out. Tables: spe x hab = species x habitat; spe x env = species x environmental variables; spe x catch = species x catchments; spe x dist = species x distance. All the tables were derived from a series of RDA1 models (see Fig 3.1).

RDA1 model settings		tables	explor. var.	% expl. Var	*P*
variables	co-variables				
DIST		spe x hab	**NCA5**	**10.8**	**0.029**
DEPTH			**NCA3**	**9.4**	**0.052**
SILT			NCA2	4.8	0.311
CATCH			NCA4	3.3	0.547
			NCA6	2.3	0.741
			NCA1	1.5	0.889
DEPTH	DIST	spe x env	**NCA4**	**14.9**	**0.032**
SILT	CATCH		**NCA3**	**12.8**	**0.065**
			NCA2	3.8	0.395
			NCA5	0.8	0.801
			NCA6	0.5	0.856
			NCA1	0.4	0.889
CATCH	DIST	spe x catch	**NCA3**	**10.3**	**0.030**
	DEPTH		NCA1	5.2	0.293
	SILT		NCA4	4.7	0.347
			NCA5	3.8	0.467
			NCA6	3.1	0.557
			NCA2	1.7	0.797
DIST	DEPTH	spe x dist	**NCA5**	**19.2**	**0.027**
	SILT		**NCA4**	**15.6**	**0.051**
	CATCH		**NCA3**	**15.4**	**0.051**
			NCA1	10.0	0.121
			NCA6	0.4	0.781
			NCA2	0.1	0.868

Table 3.5 Species traits and their subdivision into attributes (from Willby *et al* 2000)

traits	attributes	codes	
growth form	free floating surface	01	frflsr
	free floating submerged	02	frflsb
	anchored, floating leaves	03	anflle
	anchored, submerged leaves	04	ansule
	anchored, emergent leaves	05	anemle
	anchored, heterophylly	06	anhete
vertical shoot architecture	single apical growth point	07	siapgr
	single basal growth point	08	sibagr
	multiple apical growth point	09	muapgr
leaf type	tubular	10	tubula
	capillary	11	capill
	entire	12	entire
leaf area	small (<1cm2)	13	LA 1
	medium (1-20 cm2)	14	LA 2
	large (20-100 cm2)	15	LA 3
	extra large (>100 cm2)	16	LA 4
morphology index (score)	1 (2)	17	MI 1
	2 (3-5)	18	MI 2
	3 (6-7)	19	MI 3
	4 (8-9)	20	MI 4
	5 (10)	21	MI 5
rooting at nodes		22	nodal
high below-:above-ground biomass		23	root
mode of reproduction	rhizome	24	rhizom
	fragmentation	25	fragmn
	budding	26	buddg
	turions	27	turion
	stolons	28	stolon
	tubers	29	tuber
	seeds	30	seed
number of reproductive organs /year/individual	low (<10)	31	RO 1
	medium (10-100)	32	RO 2
	high (100-1000)	33	RO 3
	very high (>1000)	34	RO 4
perennation	annual	35	annual
	biennial/short lived perennial	36	shlipe
	perennial	37	perenn
evergreen leaf		38	winter
amphibious		39	amphib
gamete vector	wind	40	wind
	water	41	water
	air bubble	42	airbub
	insect	43	insect
	self	44	self
body flexibility	low (<45°)	45	BF 1
	intermediate (>45-300°)	46	BF 2
	high (>300°)	47	BF 3
leaf texture	soft	48	soft
	rigid	49	rigid
	waxy	50	waxy
	non-waxy	51	nowaxy
period of production of reproductive organ	early (March-May)	52	early
	mid (June-July)	53	mid
	late (August-September)	54	late
	very late (post September)	55	verlat
fruit size	< 1 mm	56	F 1
	1-3 mm	57	F 2
	>3 mm	58	F 3

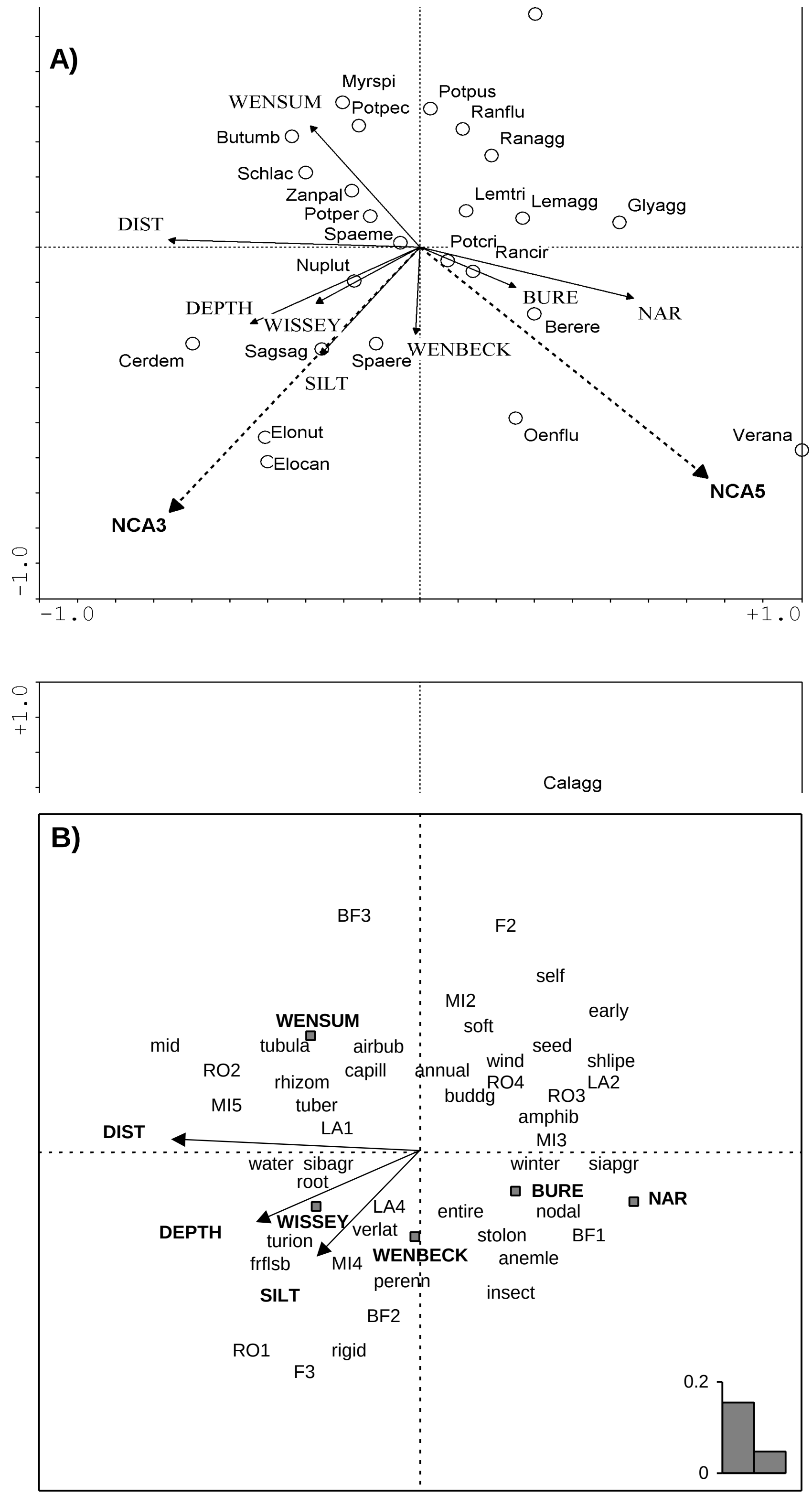


Figure 3.6 Redundancy Analysis (RDA2) triplot (species, attribute and habitat variables) ordination diagram depicting the distribution of (A) species, habitat variables and NCAs (see Table 3.3), and (B) selected trait attributes and explanatory habitat variables (drawn in Excel). Inset depict patterns of decay of eigenvalues. See attribute codes in Table 3.5 and species set of attributes in App 3.5)

Temporal effect on Natural Combination of Attributes (NCAs)

The NCAs did not shed any light on the observed species turnover during the three year period 1999-2000-2001 (RDA model: NCAs singly P>0.115; first axis P>0.588).

System specific null hypotheses

The results of the specific hypothesis-testing are summarised in Table 3.6.

Table 3.6 Results of the system specific null hypotheses-testing. n.a. = not applicable

System specific null hypotheses	variables	species	NCAs
$H_N o$ (nutrients)	SRP, NH_4	accepted	accepted
$H_N o$ (biotic)	algae	accepted	accepted
$H_N o$ (geomorphology)	depth, silt	rejected	marginally rejected
$H_D o$ (longitudinal connectance)	distance	rejected	marginally rejected
$H_D o$ (biogeographical island)	catchment	rejected	marginally rejected
Ho (time)	year	rejected	accepted
$H_{NxD} o$ (interaction factors)	distance x depth, silt	rejected	n.a.
	catchment x distance	accepted	n.a.
	catchment x depth, silt	accepted	n.a.

3.1.4 Discussion

Nutrient enrichment

The gradient of soluble reactive phosphorus (PO_4-P) and ammonium (NH_4-N) were respectively 11-3600 µg/L and 16-434 µg/L. Only 13% of the sites however had a mean SRP below 50 µg/L and 55% were exceeding 200 µg/L (Fig 3.7).

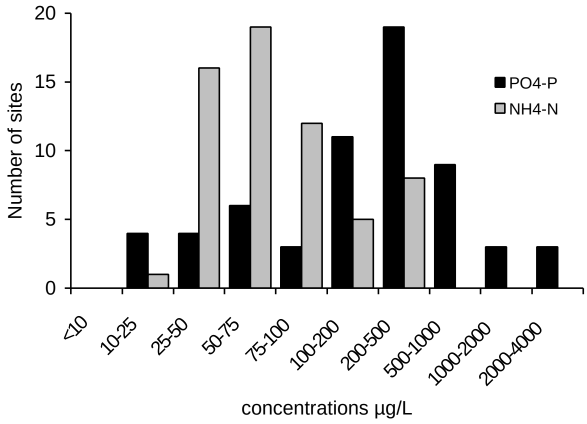


Figure 3.7 Soluble reactive phosphorus and ammonium gradient. Note that the class ranges are unequal.

This is the widest range that can be expected in this region (Section 2.2; Evans 1999; Environment Agency, unpublished) although it was inevitably biased towards very high concentrations. The mean sediment bioavailable phosphorus (as defined in Section 2.1) was not a significant predictor of aquatic plant composition in 34 sites sampled in 1999 within the river Wensum catchment (Demars, unpublished). Moreover, intensive sediment analyses in the river Wensum catchment showed little variation in the sediment bioavailable phosphorus from unvegetated gravel river bed type (Section 2.2). Recently, Clarke (2000) and Clarke & Wharton (2001) also failed to link species and sediment nutrient characteristics in British lowland rivers (although the total inorganic fraction of sediment nutrients may not be a good indicator of the nutrient availability). Letters within the European Commission of Normalisation (CEN/TC 230/WG 2/TG 3), publications sponsored by English Nature (e.g. Mainstone *et al.* 2000) and the Environment Agency (e.g. Dawson *et al.* 1999a), as well as literature reviews (e.g. White & Hendricks 2000) all claim that aquatic plant assemblages respond to nutrient enrichment in lotic ecosystems. However the English, French and German primary literature show many scientific gaps and logic flaws which suggests that their results, either accepting or rejecting the null hypothesis that aquatic plants respond to a gradient of nutrient in riverine ecosystem, must be reassessed.

Competition with filamentous green algae

The peak of development of filamentous green algae can be missed (blooms can occur in early spring) or may not occur every year (Marker 1976a, 1976b). This study spanned three wet years with summer spate flows (Fig 2.24). Therefore the instream current velocity (hydrologic stress), as well as storm events (hydrologic disturbance) were probably controlling the algal primary productivity (Biggs 2000). This may explain why algal growth was not found to impact upon plant assemblages during this study.

Geomorphological perspective

The mean depth of the river stretches was an important surrogate for light availability at the surface of the river bed and hydrostatic pressure. The substrate (especially gravel and silt) also reflected the in-stream current velocity (Kemp *et al.* 1999) and the sediment redox potential. The significance of these factors confirmed the findings of the more descriptive studies (e.g. Butcher 1927, 1933; Sirjola 1969; Edwards 1969; Haslam 1971; Lejmbach 1977). Shade was not an important factor in this study because the sites were all (except two) very open. The groundwater influence is likely to be negligible in most of the sites because of the geological nature of the river bed (boulder clay), but it could explain the presence of extensive submerged beds of amphibious species (e.g. *Berula erecta*, *Veronica anagallis-aquatica*) in some riffles mainly situated in the upper catchments where carbon dioxide supersaturation of the water facilitates their development (Sand-Jensen & Frost-Christensen 1999) and where the redox potential of the sediment would be rather high limiting the potential toxicity of ammonium, other metals (iron, manganese) or sulphide.

Demographic perspective

The distance of a site from the source of the streams (DIST) and the catchment identity (dummy variable CATCH) have been used as surrogate to investigate the species probability of colonisation. It was pragmatic although by no means an ideal quantification. The contiguity matrix better reflected the relationship of the sites within the river network, but did not allow the effect of catchment identity (i.e. island biogeography) from the longitudinal connectance to be separated. Moreover it assumed the sites to be strictly evenly distributed which was not the case (Fig 3.9) although the issue is more complex because of the potential barrier effect of weirs on species' probability of dispersal.

Weed cutting might also affect species population dynamics (Ham *et al.* 1982; Sabbatini & Murphy 1996). Although artificial standing waters present in the catchments are relatively unconnected by water, lateral dispersion might still occur by bird and human transport. The strong significance of DIST and CATCH as predictor of plant assemblages in a geomorphologically and chemically homogeneous river network should motivate further study on population structure and gene flow (Barrett *et al.* 1993; Gornall *et al.* 1998). Population structure and gene flow studies using isozymes showed that the landscape features were determining the distribution of *Callitriche* genotypes in the upper Wensum catchment (Demars, unpublished). However, the lack of genetic variability (especially in the first order streams) suggested that either the plants were spreading vegetatively or the technique was not sensitive enough. The challenge now is to find appropriate molecular markers (Ouborg *et al.* 1999).

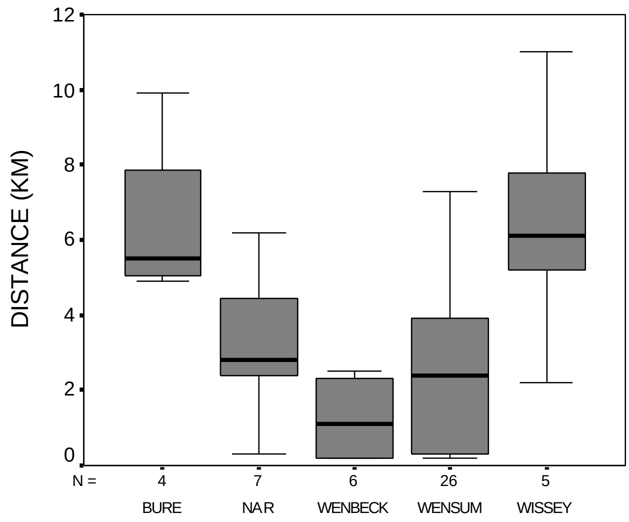


Figure 3.9 Between site distance across the five river catchments. The box represents the interquartile range and the position of the median.

Interaction of the niche theory and species demographic processes

The detection of the interaction of longitudinal connectance (surrogate for species probability of dispersal along the river corridor) and the physical habitat (depth and substrata surrogate for current velocity) is of prime interest. It means that the interacting underlying mechanisms of colonisation processes and interspecies competition have a specific impact on plant composition in riverine ecosystems. Further investigations on plant community structure revealed some aspect of this interaction and stressed the need to elaborate a unified concept in river ecology of plant community encompassing the niche theory and species demographic processes at multiple temporal and spatial scales (Demars 2002).

Regional species pool

It is unlikely that a relationship between species assemblages and anthropogenic impact will be found if the most sensitive species (stenoecious species) are scarce or absent. Scarce aquatic vascular plant species, which might be found in the calcareous riverine system of Norfolk (especially outside the Broads), include *Potamogeton coloratus*, *P. friesii*, *P. berchtoldii*, *P. pusillus*, *P. compressus*, *Groenlendia densa*, *Hydrocharis morsus-ranae*, *Hippuris vulgaris*, *Lemna gibba*, *Myriophyllum verticillatum* (Wheeler 1975, 1980; Baker *et al*. 1978; Baker & Lambley 1980; Preston & Croft 1997; Beckett & Bull 1999).

Temporal effect

Environmental variables, which would have enabled the significant change in species composition from year 1999 to 2001 to be interpreted, should have been collected. The physical habitat of the sites however barely changed over this period and the phosphorus removal at the two major sewage treatment works in autumn-winter 1999 had only a limited effect on the in-stream phosphorus concentrations (Section 2.1). Therefore the significant change in species composition is more likely to have been due to the combined effect of the hydrological events and stochastic species dispersal (Wiegleb *et al*. 1989). Analyses based on presence-absence data confirmed that the changes were really a species turnover, not merely a change of species cover-abundance.

History

It was not possible to extract sufficient information from the literature to assess whether the species distribution at the regional scale had significantly changed within the riverine system of Norfolk (Tansley 1911; Nicholson 1914; Petch & Swann 1968; Baker *et al*. 1978; Baker & Lambley 1980; Boar *et al*. 1994; Beckett & Bull 1999; Environment Agency, unpublished). However the general decline of species which are now scarce has been attributed to several causes: the loss of habitat (essentially destruction of spring flush, fens and valley fens – Haslam 1965; Baker *et al*. 1978), eutrophication of lakes (Jupp & Spence 1977; Phillips *et al.* 1978; Blindow 1992; Vestergaard & Sand-Jensen 2000; Riis & Sand-Jensen 2000; Sand-Jensen *et al*. 2000), and pollution in riverine ecosystems (Jones 1949; Carbiener 1969; Kohler *et al*. 1971; Weber 1976; Köck 1981; Wiegleb 1981a). The impact of species introduction (e.g. *Azolla filiculoides* in 1903, *Elodea canadensis* in the 1850s, *E. nuttallii* in 1978, *Crassula helmsii* in 1987 - Beckett & Bull 1999) cannot be assessed because of the lack of systematic surveys. These historical events have probably constrained the resulting distribution and abundance of the present day species (Preston & Croft 1997) but cannot be integrated because of the lack of extensive and reliable information.

Natural Combination of Attributes (NCA)

The NCAs were easy to interpret because the information was more condensed than the attribute-based group of Willby *et al*. (2000). The position and orientation of the NCAs into the habitat templet could be predicted. This would be a more realistic test than trying to predict the position of each trait which does not take into account the phylogenetic interdependence of the traits (Lavorel *et al*. 1998, 1999) although strong evolutionary convergences in aquatic vascular plants exist (Cook 1990; Willby *et al*. 2000).

Relationships between NCAs and habitat variables

This study is the first field-based test of the attribute matrix compiled by Willby *et al*. (2000) in a riverine ecosystem. Although the test was based on the robust technique of Lavorel *et al*. (1998, 1999) the results were poor. Willby *et al*. (2000) discussed more fully the ecological and methodological reasons for such mismatches. Moreover, the

NCAs (or traits) were supposed to be independent of the biogeographical areas (Charvet *et al*. 1998, 2000; Dolédec *et al*. 1999) but here NCA3 was marginally related to the catchment units (biogeographical island) and three NCAs (NCA3, NCA4, NCA5) were also marginally related to the distance of the site from the source (a surrogate for longitudinal connectance or probability of dispersal). The results might change as more information on traits becomes available and trait definitions improve (see discussion in Willby *et al*. 2000).

Scientific problems and future research issues

The species (and species traits) are not strictly independent of the environmental variables considered (Kemp *et al*. 1999). More than that, they can have an impact on the sediment characteristics (Sand-Jensen 1998) and in-stream flow velocity (Sand-Jensen & Mebus 1996; Sand-Jensen & Pedersen 1999). This tautological problem arises again with the attributes assumed to be independent of the environmental conditions (see Steneck & Dethier 1994; Kautsky 1988): some traits in Bornette *et al*. (1994), Grime *et al*. (1997) and Willby *et al*. (2000) are rather tautological: e.g. tolerance to variation in humidity, tolerance to sedimentation (Bornette et al. 1994); amphibious (Willby *et al.* 2000). Although the past evolutionary constraints (e.g. glaciation, landscape changes) might have been reasonably similar between the catchments, the effects of local past management history (ancient organic pollution, installation of sewage treatment works, construction of weirs, engineering work on the channel, long term weed cutting or boat traffic) on the riverine population dynamics of plants might have differed significantly between sites. The habitat templet concept assumes implicitly that demographic processes are negligible. It was not the case in this study (see also Wooton & Gee 2001). The lack of potentially relevant traits (e.g. ecophysiological) or their coarse grain definition cannot be assessed (sensitivity of the model). The model presented here has only found significant relationships (patterns) and cannot pretend to elucidate the causal relationships (ecological processes and biological mechanisms). It is an holistic model that can be applied on any riverine systems but its inductive nature means that the results cannot be generalised to all river networks. Finally the variables chosen as surrogate for the species population dynamics may not be completely ideal. Indeed, if a species is only found in a particular catchment; is it because it has failed to disperse, grow and reproduce in the other catchments, or is it because of unknown competitive exclusions (e.g. allelochemical)?

Further efforts should focus on the identification of the *Ranunculus* subg. *batrachium* taxa at a European scale because their plasticity is still puzzling expert botanists (Richard Lansdown, personal communication). The identified populations of *Ranunculus trichophyllus* and *R. penicillatus* ssp. *pseudofluitans* var. *pseudofluitans* were never recorded within the same stream and therefore their aggregation has biaised the statistical analysis. This was also observed to a lesser extent with the aggregated species of *Lemna* and *Callitriche* in the upper catchments.

Aquatic vascular plant assemblages are likely to be constrained by the refugia of the watershed (Sedell *et al*. 1990), interaction of landscape features such as habitat heterogeneity (Li & Reynolds 1995), habitat connectivity (Bornette & Amoros 1991, Bornette *et al*. 1998) and their intrinsic demographic dynamics (Tilman & Kareiva 1997; Hanski 1999; Casagrandi & Gatto 1999; Bell 2001; Bell *et al* 2001; Hubbell 2001). This was reflected by the holistic approach of this study, where the general null hypotheses Ho (niche) and Ho (dispersal) on species composition were both rejected and their interaction found to be partially significant. The NCAs approach gave very poor results: the niche theory was marginally supported by only one NCA, while the species demographic processes supposedly independent of the NCAs was also found to be marginally supported.

Implication for biomonitoring

A scientific biomonitoring programme (*sensu* Yoccoz *et al* 2001) should span over all types of aquatic habitats, be systematic, be supplemented by *in-situ* manipulative experiments to find out key processes and laboratory studies to work out the underlying biological mechanisms. This can be a realistic goal at the scale of an ecoregion although the definition of the size of the ecoregion, the degree of resolution (grain of the study) and sampling intensity would depend on the resources available.

Aquatic vascular plant composition in the calcareous rivers of Norfolk did not indicate the level of nutrients (phosphate, ammonium). Wiegleb (1984) had already deliberately rejected unifactorial interpretation because the bioindication concept based on plant composition does not take into account the complexity of ecosystems and neglected basic theoretical questions. The use of traits appears at the moment rather limited, contrary to previous ideas (e.g. Demars 1997). Alternative approaches should focus on ecosystem theories (e.g. Odum 1969; Margalef 1991; Jørgensen 1992; Ågren & Bosatta 1996) as illustrated by the ecohydrological concept (Zalewski *et al*. 1997).

3.1.5 Conclusion

Aquatic plants were not found to be an indicator of the level of nutrients in the lowland rivers investigated, this over a wide gradient of soluble reactive phosphorus (11-3600 µg/L of PO_4-P) and ammonium (16-434 µg/L of NH_4-N) in a nitrate-rich environment (6.4-12.9 mg/L NO_3-N). This is contrary to general opinion. The species composition of aquatic vascular plants was predicted by the niche theory and species demographic processes, which were also found to partially interact. The habitat templet concept, as currently applied to aquatic vascular plants, was found to have little, if any, predictive power. Moreover it was not found to be strictly independent of the similar biogeographical areas investigated, i.e. the intrinsic population dynamics of the species.

3.2 Aquatic plant community structure

3.2.1 Introduction

The previous section showed that the aquatic vascular plant composition in the calcareous lowland rivers of Norfolk is driven by a set of habitat characteristics and by the species demographic dynamics. The identification of structural patterns in aquatic vascular plant assemblages delimited by habitat factors would enable speculation on the underlying mechanisms generating these patterns, from which experimental designs and concepts could then emerge. The term community therefore is not intended to reflect a species assemblage identified from a null hypothesis of non-randomness in species co-occurrence (e.g. Wilson 1987, 1995). Rather, community is defined here as an assemblage of aquatic vascular plants that represent the structure of a river network. The structure of the community can be defined by the species richness, species evenness and the variety of species ecomorphological types (attribute-based group of species, *sensu* Willby *et al.* 2000).

These calcareous lowland rivers differed widely in their soluble reactive phosphorus levels (mean ± 95%CL µg/L PO_4-P): Bure (47 ± 3), Nar (193 ± 72), Wissey (338 ± 54), Wensum (375 ± 17) and Wendling Beck (1438 ± 161). Therefore, although the gradient of phosphorus did not influence the composition of species in the calcareous lowland rivers of Norfolk, there might be some differences in the community structure.

Other descriptors could provide information about the underlying processes determining community structure: the strength of the relationship between the percentage of sites at which a species is present and mean species abundance at these sites, the longitudinal and temporal species turnover and species evenness compared between communities (as defined above).

3.2.2 Hypotheses

The hypotheses of this part of the study were: –

? the species richness is similar between rivers with contrasting phosphorus levels;

? the species richness is equally driven by the species diversity and species evenness

? the spatial species turnover along rivers is highly variable because of the various anthropogenic impacts;

? the temporal species turnover is low because the calcareous lowland rivers have a relatively stable flow (i.e. spates rarely remove completely the plants);

? there is a significant relationship between occurrence and abundance of the species;

3.2.3 Methods

Fieldwork

All the rivers share a common history, similar hydrological and physico-chemical conditions, and flow mostly within agricultural catchments. This study investigates the same dataset as in Section 3.1 in which the study area, survey technique and data collection are fully described. Here all the vascular aquatic plants recorded were taken into account (33 taxa in total). This study focused, unless otherwise stated, on the sites situated on the main rivers (54 in total). The 500 m^2 area of the site (i.e. sample) optimised the trade-off between sampling effort and the geomorphological homogeneity of the sampling unit (Wiegleb 1981b). For practical reasons, the temporal scale correspond to an inter-annual scale at the optimal development of the vegetation during summer time. It does not take into account the seasonal variability well documented elsewhere (Butcher 1933; Ladle & Casey 1971; Dawson *et al.* 1978; Barrat-Segretain & Amoros 1995, 1996; Henry *et al.* 1996; Haury & Gouesse Aidara 1999; Champion & Tanner 2000).

Species richness

A species-area relationship was constructed with a randomisation procedure: the order of the 62 sites sampled was randomised, then the cumulated species richness (from 1 to 33) was plotted against the cumulated number of sites (from 1 to 62), and the process was repeated 10 times. The number of species was also plotted against the distance of the sites from the source and a log regression curve best fitted the data. To make the species diversity of the rivers comparable, randomisation procedures were used to standardize for the number of sites surveyed per river and the mean distance of the sites from the source (Sand-Jensen *et al.* 2000; Demars 2002).

The calculation of Shannon's diversity and equitability (evenness) index allowed the two component of species richness to be separated. Further analyses of plant community structure are in Demars (2002).

Species turnover

The spatial (along rivers) and temporal (inter-annual) species turnover St was calculated as in Tokeshi (1990):

$$St = 0.5 \sum_{i=1}^{n} \left| P_i(t) - P_i\,(t+1) \right|$$

where $P_i(t)$ and $P_i\,(t+1)$ are the proportional abundance of species i in sample t and t+1 respectively, and n is the total number of species occurring on the two occasions.

Species occurrence-abundance

The percentage of species occurrence across the 62 sites of Norfolk streams was plotted against the mean local species abundance (i.e. mean site cover-abundance) to investigate distribution patterns of aquatic vascular plants and try to relate it to its underlying mechanisms by examining species traits. The mean local species abundance should not be log transformed for further analysis as in Demars (2002, p. 183, 188) because of the non-linear nature of the cover-abundance index (Demars 2002, p. 199).

3.2.4 Results

Species richness

The cumulated number of new taxa recorded in the randomly ranked 62 sites followed a logarithmic distribution (Fig 3.10). The species richness showed a marked increase with the distance from source (Fig 3.11). The mean species richness was significantly higher in the Nar (+2.4), lower in the Wendling Beck (-1.0) and Wensum (-1.4), and did not differ significantly in the Bure and Wissey, compared to the overall mean species richness, this after correction for the number of sites sampled and taking into account the mean distance of the surveyed sites (Table 3.7).

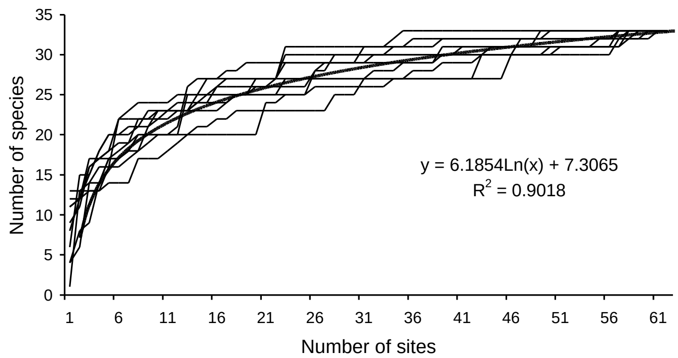


Figure 3.10 **Species area relationship derived from 10 randomized runs.**

The Shannon indices

The regression curve of the species richness predicted by the distance of the sites from the source of the river (Fig 3.11) and Shannon's diversity index had a similar shape (Fig 3.12). However the Shannon's equitability index was always high (except on one occasion) and was not related to the distance (Fig 3.13). The Shannon indices therefore pointed out two main characteristics of this study: the increase in species diversity is mostly due to the increase of species richness; and the very high species evenness illustrated species co-dominance rather than dominance.

The hypothesis that species richness is equally driven by the species diversity and species evenness is therefore rejected.

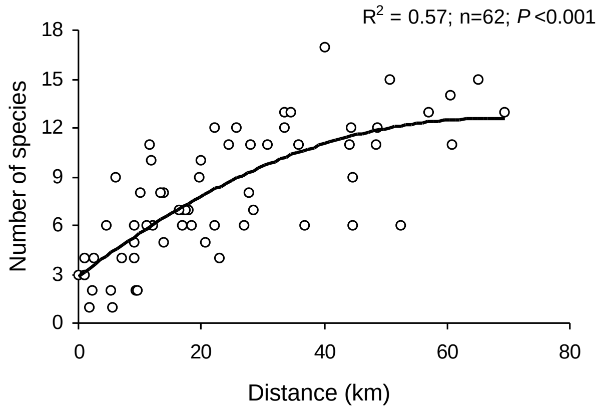


Figure 3.11 Distance of the sites from the source as predictor of species richness

Table 3.7 Comparison of the species richness between rivers impacted by different levels of phosphorus. n = number of sites sampled; d = mean distance of the sites from the source; S = number of species, corrected for the number of sites sampled (S_n) and mean distance of the site from the source (S_d); CL = confidence limits. S_n differ significantly from S_d when there is no overlap of the 95% CL.

River	n	d	S_n (95%CL)	S_d (95%CL)	S_n-S_d	SRP (± 95%CL)
Bure	5	24.4	10.4 (9.1-11.7)	8.6 (7.7-9.5)	<=>	47 (± 3)
Nar	7	14.5	9.1 (9.0-9.1)	6.6 (5.8-7.4)	>	193 (± 72)
Wissey	7	33.8	10.1 (9.9-10.2)	10.2 (9.2-11.2)	<=>	338 (± 54)
Wensum	26	35.9	9.1 (9.0-9.3)	10.5 (9.5-11.5)	<	375 (± 17)
Wendling Beck	7	11.9	5.0 (4.9-5.1)	6.0 (5.2-6.8)	<	1438 (± 161)

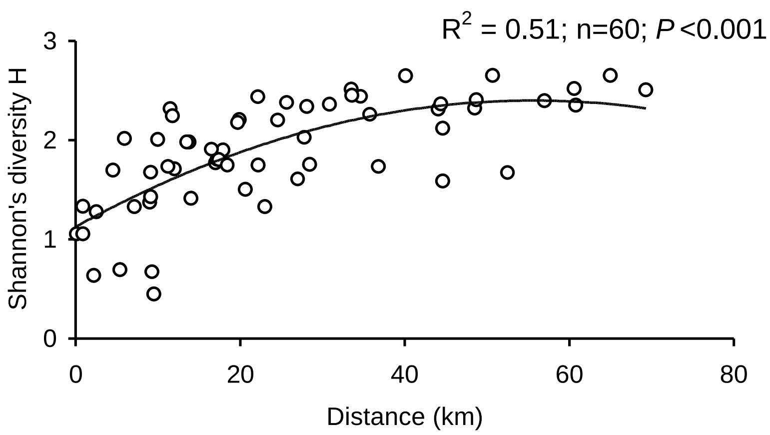


Figure 3.12 Distance of the site from the source as predictor of species diversity.

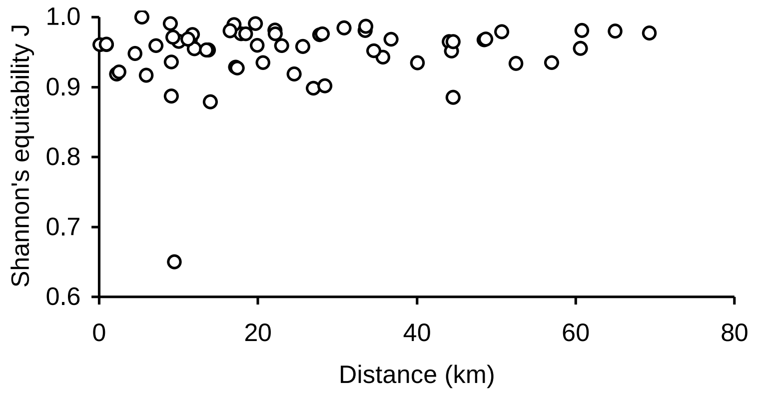


Figure 3.13 **Distance of the site from the source as predictor of species evenness.**

Species turnover

The spatial species turnover along the rivers was relatively high. The mean (±SD) was 0.51 (±0.03) and the range was 0.23 to 0.78. There seemed to be no longitudinal pattern or difference between rivers (Fig 3.14). The temporal species turnover in the Wendling Beck and the Wensum was low. The means (±SD) were respectively for 1999-2000, 2000-2001, 1999-2001: 0.010 (±0.007), 0.017 (±0.010), 0.025 (±0.015).

Species occurrence-abundance

No significant trend was detected between the percentage of species occurrence plotted against the mean local species abundance (Fig 3.15). This is mostly because rare species had a wide range of abundances.

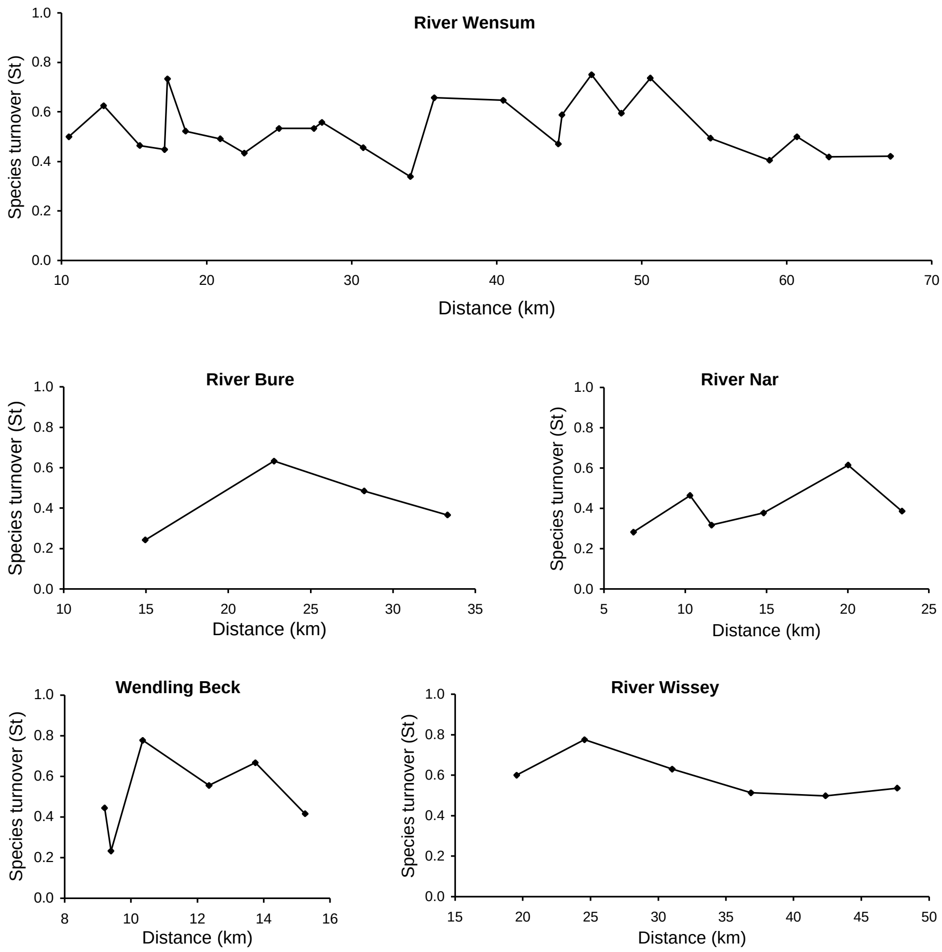


FIGURE 3.14 SPECIES TURNOVER ALONG THE RIVERS.

3.2.5 Discussion

Species richness

Since the plateau phase of the species area diagram was practically reached (Fig 3.10), it is expected that sampling more sites in the lowland rivers of Norfolk would not reveal a substantial number of other species.

The differences in species richness between rivers were rather marginal although significant and therefore the hypothesis that species richness is similar among all the rivers studied can be rejected. The River Bure had marginally more species (S_n) than the corrected overall average (S_d). It was not significant because the 95% CL was substantially larger than the other rivers due to a small statistical artefact (rather than a higher variability of species diversity). This would indicate that there might be a trend between phosphorus enrichment and species diversity. However the rivers differed also by other factors which were not contrasted (Fig 3.3b) in this analysis and great care should be taken in the use of this result.

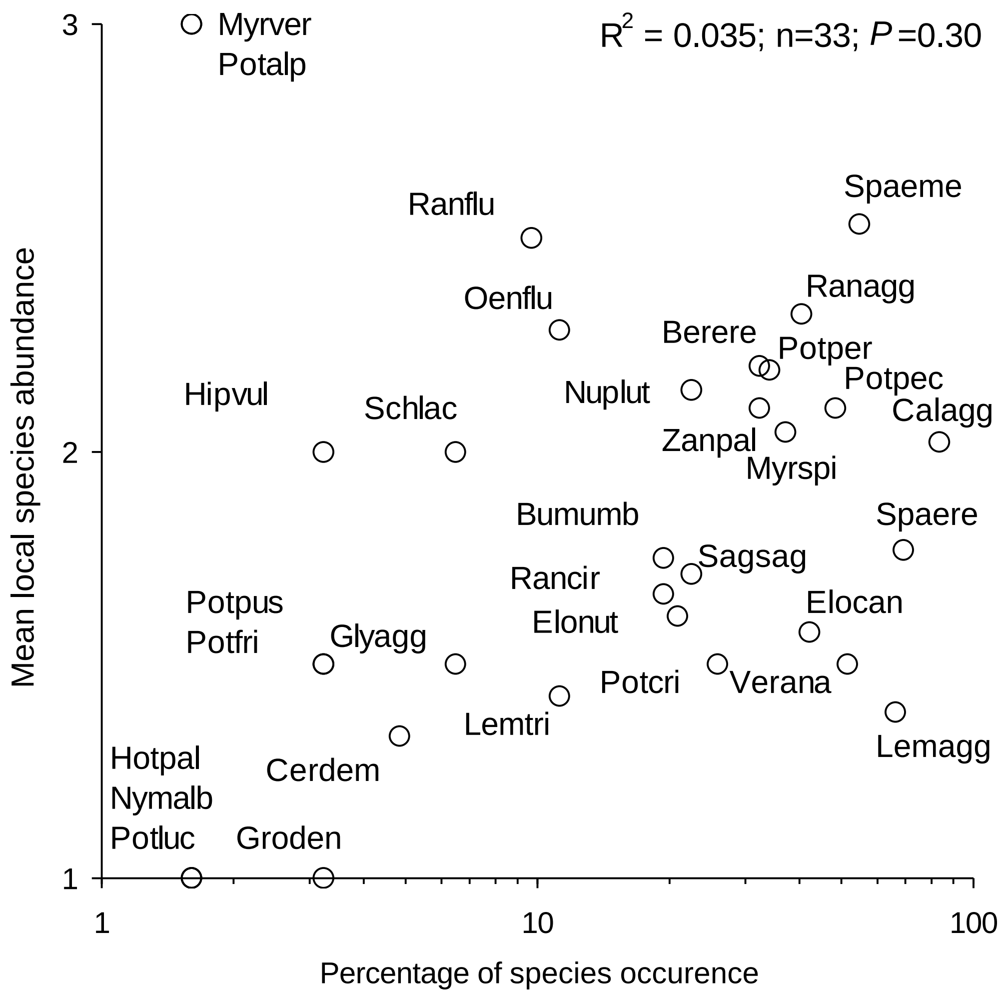


Figure 3.15 Percentage of species occurrence against mean local species abundance.

The gradient of phosphorus was extremely biaised towards very high values because all the tributary sites were excluded from this analysis. The most important point is that within the range of the observed levels of phosphorus in this study, the species diversity does not seem to be massively impacted as it would have been in shallow lakes.

The increase of species richness with the longitudinal connectance seems to be a paradigm of these lowland rivers: there is an accumulation of species at the bottom of the rivers (Demars & Harper 1998). This pattern was also mirrored by the number of attribute-based groups (Willby *et al*. 2000; Demars 2002). Competition should therefore be higher in the lower catchment where species packing may occur (Begon *et al*. 1996, p. 303).

Species turnover

The high spatial turnover of the species may explain why the spatial autocorrelation was not previously detected (Section 3.1). In fact the spatial autocorrelation is probably confounded by overriding factors such as the presence of the weirs on the river

changing the depth, in-stream flow dynamics and substrata (Chapter 2.1) or other factors not considered (e.g. local species dynamics).
The temporal species turnover was found here to be low over three years. However, a 10 year study showed a high species change at sites in lowland rivers (Wiegleb *et al*. 1989). Therefore, a longer term survey may be necessary to reveal the true picture. There is also a bias (not discussed in Demars 2002): the scale of the survey did not allow to demonstrate whether the patches of plants were stable over time or whether a high within site dynamic occurred. Mapping or splitting the surveyed reach would have allowed to quantify the within site variability (Butcher 1933; Jones 1955; Scott *et al*. 2002).
The hypotheses concerning the species turnover are not valid and therefore cannot be tested. The high spatial species turnover results from natural processes interacting with the anthropogenic impacts (e.g. weirs, alien species) and the temporal species turnover was more based on unknown reality (what is known about plants scouring in these systems apart from casual observations?).

Species occurrence-abundance

The relationship between percentage of species occurrence and mean local species abundance was not significant (rejection of the last hypothesis of Chapter 3.2.2). This is similar to the finding of Riis (2000) although here species with high occurrence also had a wide range of mean local abundance. It may reflect the ability of the rare species to reproduce vegetatively and form clonal patches once an individual is established (resistance) and the inability of some frequent species to colonise further the niche (resilience). The size of the patch or probability of colonisation would depend on the position of the sample (site) in the river network, the extent of development of competitive species and the number of species or number of attribute-based group of species (*sensu* Willby *et al*. 2000) which would drive the level of competition within a sample. Metapopulation dynamics may be an underlying process shaping aquatic vascular plant community as suggested by Riis (2000), however the probability of colonisation would also depend on the species biological traits, especially long dispersal and ability to occupy the available niche once established.

Implications

Because of the stochasticity of the total niche (*sensu* Hutchinson 1957) in riverine ecosystems, species are less likely to compete for resources and therefore the intrinsic demographic parameters of the species, i.e. species dispersal ability would be the dominant processes explaining the observed patterns. One consequence is that alien colonisers may readily establish and persist in these riverine ecosystems but would not dominate the whole ecosystem. The spatial demographic model (Tilman & Kareiva 1997; Hanski 1999) and neutral model (Bell 2001; Bell *et al*. 2001; Hubbell 2001) would then explain the observed patterns better. How stochastic or how variable, however, is the total niche in these rivers?

Models of species abundance patterns reflect the gradient from niche control (e.g. geometrical series model), dominance control and founder control (e.g. composite model, Tokeshi 1990) to mobility control (e.g. random assortment model, Tokeshi

1990) of the patch dynamic concept of stream community ecology (Townsend 1989), developed along two axes of spatial heterogeneity and temporal variability. The river habitat templet concept (Southwood 1988, Townsend & Hildrew 1994) was developed within the same frame, where the fittest species within each community may also be predicted by the set of their intrinsic biological traits. Where do the samples (sites of Norfolk rivers) fit into that picture? At local scale (one sample), the mobility control is likely to be more important in the upper catchment where the probability of colonisation is low. However at the bottom of these rivers, the probability of colonisation increases because of the longitudinal connectance and increase of habitability, diversity, connectivity for aquatic plants in the wider floodplain with a relative decrease of the pulse impacts. Therefore species competition is likely to occur and the niche theory would interact with the species demographic processes (successional mosaic, competitive lottery). The increase of competition in these rivers may therefore not come primarily from a reach scale low disturbance and low habitat diversity but rather be due to macroecological (biogeographical scale) and microecological (patch scale) processes of plant dispersal.

3.2.6 Conclusion

There were only marginal differences in species richness between rivers that could perhaps be attributed to phosphorus enrichment. The spatial species turnover along rivers was high, resulting from natural and anthropogenic factors. The temporal (annual) species turnover was low over a three year period but may not be negligible in the long term. There was no significant trend between species occurrence and mean local abundance of the species (contrary to the claims of Demars 2002). This lack of emergent pattern could be explained by the species intrinsic dispersal ability and vegetative reproduction in an unsaturated ecosystem. Shannon indices therefore pointed out two main characteristics of this study: the increase in species diversity is mostly due to the increase of species richness; and the very high species evenness illustrated species co-dominance rather than dominance.

The patch dynamic concept and habitat templet concept make predictions at local scales. However macroecological (biogeographical scale) and microecological (patch scale) processes may dictate the in-stream local processes: e.g. the probability of colonisation of a sample by a range of species would depend on the regional species pool (occurrence and abundance of the species) and habitat connectivity or extrinsic means of dispersal (e.g. animal, man). This study therefore reinforces the findings of Chapter 3.1, which show the interaction of niche theory and species demographic processes and stress the need for a unified theory linking niche and demography that could predict the observed patterns in riverine lowland ecosystems. The management implications are that the diversity of wetland habitats for aquatic plants should be preserved and restored at the regional scale (e.g. Norfolk).

4. CONCLUDING DISCUSSION

4.1 Phosphorus enrichment and control

The pristine state of the calcareous lowland rivers of Norfolk has been progressively transformed ever since the first development of farming and human settlements. Only with the post-war intensification of agriculture however, has there been a dramatic increase in both concentration and proportion (relative to total phosphorus) of the readily-available soluble reactive phosphorus (SRP) in lowland rivers (see Table 4.1; Chapter 3.1).

Table 4.1 Concentrations and proportion of soluble reactive phosphorus (relative to total phosphorus) in the lowland calcareous rivers of Norfolk. ‡ from Moss *et al.* (1988) in subcatchments of the River Bure; † this study, from four subcatchments of the River Wensum.

Catchment area	entirely covered by woodland ‡	impacted by agriculture	impacted by point source effluents †
SRP (µg/L PO_4-P)	4	16 ± 9 SD ‡ 13 ± 6 SD †	up to 500-5000
SRP/TP ratio (%)	10	26 ‡ 27 ± 12 SD †	generally 80-90

The levels of total phosphorus in the sediment were highest in the riparian fringe of vegetation above the weirs where silt and clay accumulated. The levels of sediment bioavailable phosphorus were highest at sites impacted by both the treated sewage effluents and weirs associated with mills. (Chapter 2.1)

Phosphorus removal effectively reduced the TP load contribution of the two STWs from 42% to 18% of the effluents in the whole catchment of the River Wensum. However the levels of SRP predicted at mean flow (140-273 µg/L PO_4-P) and low flow (415-1039 µg/L) downstream from the STWs, still remained excessive. Diffuse sources contributed 12% of the phosphorus budget prior to phosphorus control over the period 1990-1999 and 34% after phosphorus stripping in 2000-2001, although this increase was also due to the higher than average rainfall of the study period. The numerous and scattered small effluents contributed more than half of the phosphorus budget and therefore were the main cause of the high predicted SRP concentrations (Chapter 2.3).

4.2 Aquatic vascular plants: do they respond to phosphorus enrichment?

This study revealed that neither the species composition nor the species traits (life history trade-offs) or community structure of aquatic vascular plants showed a clear relationship with SRP despite the wide range of concentrations present (11-3600 µg/L PO_4-P). Sediment bioavailable phosphorus was not a significant predictor of the species composition of the River Wensum either, which reinforced the findings of Clarke (2000) (Chapter 3.1).

The nutrient content of plant tissues might provide another perspective on the nutrient availability in the environment. Critical tissue content under which plant growth or photosynthesis was altered has been determined for a few aquatic plant species by means of controlled experiments (e.g. Gerloff & Krombholz 1966; Colman *et al*. 1987; Madsen *et al*. 1998). Scattered data from the literature would suggest that in European lowland riverine ecosystems aquatic angiosperms are not limited by either nitrogen (N>2% dry weight) or phosphorus (P>0.2% dry weight) despite in-stream SRP concentrations as low as 10 µg/L – e.g. Casey & Downing 1976; Kern-Hansen & Dawson 1978; Peltre *et al*. 1993; Robach *et al.* 1995, 1996; Clarke 2000; House *et al*. 2001; Madsen & Cedergreen 2002. Aquatic plants have evolved to cope in an environment where bioavailable phosphorus is scarce (SRP <5(-10) µg/L PO_4-P; Table 4.1). Considering this, along with the critical tissue content data discussed above, it is proabable that hydrophytes are not directly controlled by the level of phosphorus encountered in today's lowland calcareous rivers impacted by agriculture and point source effluents (Table 4.1). This conclusion is reinforced by mass balance studies which have shown that plant phosphorus retention was small compared to the phosphorus fluxes: – in the River Frome (Westlake 1968) 2.5%; the Bere Stream (Ladle & Casey 1971) 1.1% and the River Thame (House *et al*. 2001) <1% (instream SRP 400-2000 µg/L PO_4-P).

The prospect of using aquatic vascular plants to indicate the trophic state of lowland rivers seems unlikely to succeed; not because aquatic plants cannot indicate the levels of phosphorus but because in reality all the rivers are enriched beyond the concentrations at which plants might show a response.

4.3 Aquatic vascular plants in a riverine environment: pattern recognition.

The species composition at 62 sites sampled in four major catchment of Norfolk showed that distribution responded to the biogeographical units and distance of the site from the source (both surrogate for species probability of colonisation) and channel depth/substrata (species environmental requirements).

However, even the most conspicuous geomorphological factors hardly predicted the Natural Combination of Attributes (NCAs). Moreover, the NCAs were marginally linked to the spatial structure of the river network surrogate for species probability of dispersal. Therefore the prospect of using the species' attribute approach as currently applied to aquatic vascular plants is not good. The lack of relationships was also probably due to the limited number of species (25) included in the analyses.

The lack of dominance within plant communities suggested that demographic processes as well as inter-species competition might have occurred at the local scale as predicted by the patch dynamics concept. However the different probabilities of colonisation of the sites as well as the regional species pool may dictate the reach scale processes of plant community structure and composition in a directional river network. This study indicates the need for a unification of the niche and demography theories at multiple spatial and temporal scales.

The Mean Trophic Rank methodology (Holmes *et al.* 1999) provided a sampling protocol that has been applied throughout England, Wales and Northern Ireland. This extensive dataset, made available by Dawson *et al.* (1999), together with other invaluable but less-available sources of data such as the JNCC database and the Atlas

2000 of the Flora of the British Isles could be used to give more insights through modelling using statistical (e.g. Heegaard *et al*. 2001; Willby *et al*. 2001; Demars 2002), Artificial Intelligence (pioneered by Bill Walley; http://www.cies.staffs.ac.uk/), or neutral macroecological dynamic models (e.g. Bell 2000; Hubbell 2001).

4.4 Inherent limits of the study

This project spanned three years during which no apparent substantial problems of eutrophication were observed in the studied rivers. However, during the droughts of the nineties, extensive filamentous algal growths were reported from riffles and impounded river sections of East Anglian rivers (e.g. pictures in Kemp 2000) as well as cyanobacterial bloom (e.g. Krokowski & Jamieson 2002). Unfortunately these observations were not systematically recorded for these rivers and the ecological consequences could not be assessed.

This study has identified some patterns but not the mechanisms. For example, the sediment granulometry was found to be linked with species composition, independently of the other measured factors. However, had it be measured, the redox potential of the sediment interstitial water might have been a better predictor. The redox potential itself mirrors important chemical equilibria and bacterial activity of the sediment. In the well oxygenated gravel river bed type, nitrate concentration of the interstitial water did not differ significantly from the water column. However, in the silty riparian habitat ammonium concentrations were extremely high and nitrate absent. This ammonium can be used by some riparian plants very effectively but may also be toxic at such high levels for other species. The same reasoning applies for a wide range of interconnected factors.

A species area curve (Demars 2002) suggested that it was unlikely that many taxa growing in the studied rivers had been missed. However a number of species have declined both regionally and nationally and were absent from this survey (Preston & Croft 1997; Becket & Bull 1999). Pristine lowland rivers disappeared hundreds of years ago in Norfolk. Areas of groundwater upwelling may constitute the last refuge for many wetland and aquatic species.

Macroalgae were not identified even down to genus levels. Their population is more dynamic than aquatic vascular plants with large growths, which often occur for only a few weeks during spring or during summer droughts. Epiphytes were only mentioned in the introduction (Chapter 1.4). Although it is suggested that they have played a role in the potential mechanism of submerged macrophyte decline in the Broads (Phillips *et al* 1978) and members of the scientific committee mentioned their importance, epiphytes remained low on the list of priorities, because of the need to understand the broader issues of aquatic vascular plant ecology in the Wensum first.

The subsidiary activity in the project objectives, to '*Review the effects of eutrophication on diversity, distribution and abundance of macrophyte flora*' only features as a recommendation in the discussion of this report (Chapter 3.4.1). Hundreds of key references from the primary literature were consulted and these still have to be presented on a sound basis and in a coordinated way. This is a large task because of (1) the complexity of the biogeochemical cycles, (2) the different ecophysiological capacity of species of aquatic plants, (3) the scale of studies published and (4) the fact that key references are published in several different languages (notably German, French and Czech). The objective was not possible to complete in the timescale available for this Technical Report and will be completed independently at Macaulay Institute and completed by the end of 2002.

4.5 Health and safety

The COSHH risk assessment sheets associated with the chemicals and the protocols, the risk assessment sheets associated with the field procedures, were completed before any laboratory work was started.

The field survey risk assessment, involving snorkelling with a dry suit, was reviewed half way through the project. Fieldworkers accordingly took the First Aid at Work course with the British Red Cross and followed this with Life Saving lessons at Wigston Swimming pool in Leicester. An audit was carried out by the Environment Agency at Marham (NGR 53 723 120), the lowest site surveyed on the River Nar, during summer 2000 and the techniques found acceptable.

The foot-and-mouth outbreak did not disturb the fieldwork because sampling had fortuitously been undertaken at the end of January 2001, just before the outbreak. No case of foot-and-mouth had been recorded in Norfolk two months after the start of the outbreak and so the landowners, after assessment of the risks, permitted the seasonal sampling to continue under a strict protocol. (App 4.1).

It is clear in retrospect, that the study would not have been possible under the agreed budget without the fortuitous assistance in the field of a European exchange student and a completed Ph.D. prior to post-doctoral employment. The tendering rules of the Environment Agency make it extremely unlikely that a tender fully costing a second field worker would have been acceptable.

5. RECOMMENDATIONS FOR CONSERVATION, RIVER MANAGEMENT AND FUTURE POLICY

The conservation strategy for the River Wensum SSSI aims at completing priority actions such as (A) '*identification of the ecological impacts of eutrophication on the River Wensum*' and (B) '*development of a holistic strategy to identify the sources of phosphate and nitrate and mechanism for the control of eutrophication*' (English Nature & Environment Agency 2000, p. 27). This R&D project report, following the completion of phosphorus removal at the main STWs by Anglian Water, represents substantial progress towards the above conservation aims.

This section raises conservation, management and policy issues related to eutrophication in lowland rivers and lists recommendations for the Environment Agency, Anglian Water and English Nature.

These recommendations should be read in conjunction with those suggested in the EA R&D Technical Report E39 (Dawson *et al*. 1999a), R&D Project Record W1/017/01 (Maberly *et al*. 2000), R&D Technical Report E1-SO1/TR (Clarke & Wharton 2001b), the Conservation Strategy for the River Wensum SSSI (English Nature & Environment Agency 2000, p. 27) and in particular the management strategy of the EA for tackling eutrophication (Environment Agency 2000). It is recommended that:-

R1 Long term monitoring of phosphorus should be set up at key sites to establish local reference conditions (particularly of the least impacted stream and groundwater sites).

R2 Long term monitoring of stream conditions such as flow, temperature, dissolved oxygen and turbidity should be set up, on a continuous basis using data loggers, at key sites in conjunction with regular field surveys of macrophyte development during spring and summer. This would complement the UK Environmental Change Network (Sykes *et al*. 1999) on the specific topic of eutrophication.

R3 The purposes, reliability and efficiency of the field survey techniques of aquatic plants in rivers should be reviewed (Standing Committee 1987; Sykes *et al*. 1999; Richard Lansdown, pers comm., Aquatic and Wetland Plant Forum http://www.ardeola.demon.co.uk/).

R4 Process-based dynamic models of sediment and nutrient transport from the land to the river and along the river should be backed. The ecological, economical and safety issues of the impact of siltation above the weirs associated with the mills on the Wensum (and similar rivers) should be reviewed.

R5 The development of process-based dynamic models, linking the different compartments of an ecosystem at the reach scale, should also be backed.

R6 Plant species' demographic dynamics, the catchment-based habitat heterogeneity and the regional species' pool should be taken into account for conservation and management purposes. It is important that the regional habitat diversity (e.g. spring fen, valley fen, macrophyte-dominated shallow lakes) be preserved and restored as much as possible to insure the persistence of the riverine flora (Riis 2000; Sand-Jensen *et al*. 2001).

R7 Studies of species' dispersal abilities should be promoted. The development of molecular markers to unravel the gene flow of aquatic vascular plants along rivers and across aquatic habitats seems promising.

R8 Screening of ecophysiological traits of macrophytes that would give a sound biological basis when using them as bioindicators of ecological integrity is necessary. This would complement what is already funded by the NERC, and it is suggested that the response of aquatic plants to changing redox potential in the sediment should be tested with different levels of phosphorus, nitrogen, metal and sulphide under controlled pH (e.g. Armstrong *et al*. 1996; Smolders & Roelofs 1996).

R9 Studies using methods other than those involving aquatic vascular plants should be supported, to understand the biological response of nutrient enrichment. Examples are the study of algae (Borchardt 1996; Kelly *et al*. 2001), of decomposers (Grattan & Suberkropp 2001), the whole-stream metabolism (Young & Huryn 1999; Mulholland *et al*. 2001) and food webs (Biggs *et al*. 2000; Rosemond *et al*. 2001).

R10 Studies of the interaction between eutrophication and other pollutions should be supported. such as toxic contaminants (Stansfield *et al*. 1989; Mingazzini 1993, 1997a, 1997b; Gunnarsson *et al*. 1995; Tang *et al*. 1997; Skei *et al*. 2000).

R11 The topic of nutrient control should be addressed from the perspective of sustainable development. Managing nutrient enrichment is a task that cannot be reduced to the 'mere technological' challenge of stripping the phosphorus from the major STWs (Chapters 1.1, 2.3). Several crucial issues must be considered, divided into the apparent and the hidden costs.

Apparent costs include the purchase of the stripping system, its maintainance and the use of material (chemical solution) and energy (electricity). There are several potential recovery technologies under development that have not yet made their way on a commercial basis due to their novelty and inherent relative cost. In this study the phosphorus was simply precipitated chemically and collected with the sludge. However there is growing concerns about the capacity of the land to recycle this sludge (Towers *et al*. 2002) especially with the necessary compliance of the European Nitrate Directive (91/676/EEC). This lead to another issue which is the impact and cost of diffuse pollution (e.g. Edwards & Withers 1998; D'Arcy *et al*. 2000). The hidden costs of land management, such as the degradation of ecological conditions or society structure and quality of life (Hindmarch & Pienkowski 1999; Everard *et al*. 2002) should be acknowledged and brought into the equation.

Eutrophication problems are exacerbated during droughts. The extent to which water quality issues interact with water resources such as this however, are not yet holistically addressed. Ecological properties such as flow regime or landscape structure are not considered part of the solution, but could be part of the sustainable management of riverine ecosystems. Ecohydrology is a new concept, supported by UNESCO, that suggests solutions by enhancing the retention of water and nutrients throughout the catchment area, with the use of ecosystem properties as tools, based on sound scientific knowledge (Zalewski *et al*. 1997; Zalewski 2000). The use of ecosystem properties for restoration purposes is an acceptance of the breakdown in ecosystem homeostasis (capacity of the ecosystem to resist against anthropogenic disturbances - Odum 1985, Schindler 1987, 1990). Anthropogenic impacts have so altered the functioning of ecosystems that scientists may soon no longer be able to study natural ones (Vitousek *et al.* 1997; Perakis & Hedin 2002). A good example is the difficulty of finding a pristine lowland river in the UK.

Eutrophication is a long term problem and should be treated with long term, low cost, catchment-scale solutions. These are: to limit the use of phosphorus in detergents (USGS 1999a), the widespread uptake of best agricultural practice, the restoration of ecotones and landscape structure (buffer strip, wetlands), the limitation of abstractions, the removal of the effects of the mill structure to restore river bed habitat heterogeneity (Kemp *et al*. 1999) and the encouragement of natural resource developments such as riparian forestry or extensive fish farming, both with native species. Local economies and the social structure of the countryside if, rather than being based on subsidies, relied upon ecologically sustainable land management, could lead to better quality of life and restoration of rivers. The Common Agricultural Policy (CAP)'s so-called 'second pillar' (rural development and environmental measures) recently advocates more sustainable development based on plans drawn in the UK such as the England Rural Development Programme (MAFF 2000). The EA recently specified in a consultation document its intended contributions and objectives to sustainable development and English Nature also has strong commitment to this. The water industry investment mechanism, Asset Management Planning (AMP), from which this project arose, is more reductive (Everard 2002) and holistic approaches have been recommended (e.g. Everard 2001).

R12 The Water Framework Directive

The implementation of the WFD (2000/60/EC) requires member states to determine the status of water bodies (e.g. groundwater, lake, running waters) using ecological criteria as well as the traditional physico-chemical parameters. This still poses questions, which have not been answered by this limited research. Questions arise from it about nutrients and about macrophytes:-

Nutrient criteria

1. How can we define nutrient thresholds?
2. Do we need to express them in concentration or load units?
3. Do we need international or regional thresholds?

Aquatic macrophytes

1. How can macrophytes contribute to indicate the quality of riverine ecosystems across catchments?

National environmental agencies have set up longterm physico-chemical monitoring programs. Few have tried to monitor the in-stream biological response of nutrient enrichment or control however (e.g. the United States

Geological Survey 1999). Biological responses of nutrient enrichment in streams have not been studied in depth, but a number of literature reviews have suggested nutrient targets for streams including nitrogen and phosphorus (Mainstone *et al*. 1994, 2000; Dodds *et al*. 1997, 1998; United States Environmental Protection Agency (USEPA) 1998, 2000; Dodds & Welsh 2000; Environment Agency 2000; Grubbs 2001). Since the background levels of nutrients can vary naturally (e.g. Holloway *et al*. 1998; Evans 1999), USEPA produced different nutrient criteria for every ecoregion of the USA (USEPA 2000).

R13 Riverine ecosystems are characteristed by a low retention capacity of energy and matter. Although eutrophication in rivers might only be detectable under specific conditions (e.g. summer droughts), nutrients travel downstream and may have dramatic impacts subsequently on standing waters, floodplains, estuaries, and marine ecosystems. Risk assessment and management of eutrophication for rivers should acknowledge this continuum.

R14 Cooperation between scientists, conservation agencies, water companies and policy makers should continue to be supported and coordinated, so that each participant can grasp the holistic context of water in order to contribute more effectively towards sustainable development in the context of "thinking globally, acting locally".

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