



A review of lake restoration practices and their performance in the Broads National Park 1980–2013

2015

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1 Introduction

1.1 Context

Although it is increasingly clear that freshwater ecosystems are exposed to multiple stressors, including climate change, eutrophication arguably still represents the single biggest threat to their integrity (Smith *et al.*, 2006; Jeppesen *et al.*, 2012). Questions concerning the functioning, management and restoration of enriched lakes have therefore dominated aquatic ecology research for several decades. This report presents an analysis of the effectiveness of various restoration approaches used in the Norfolk Broads since the early 1980s.

The Broads are a series of sixty or so, mostly small, shallow lakes, formed by the flooding of mediaeval peat diggings, that lie in the valleys of the major rivers draining the eastern part of Norfolk and Suffolk (Figure 1.1). Their natural history is described by George (1992) and Moss (2001). The Broads have immense national significance, both as a resource for tourism, recreational boating and other water-based pursuits, and as a hotspot of aquatic and wetland biodiversity. However, intensification of agriculture and increases in human population density over the last 150 years have greatly increased pressures on the aquatic environment through cultural eutrophication. Restoration of the Broads is considered a priority in order to maintain their recreational and ecological value. A programme of research, monitoring and experimental management has therefore been ongoing for over 30 years led by the Broads Authority and supported by the Environment Agency and Natural England or their predecessors.

There are a number of recent reviews on the effectiveness of restoration approaches applied to shallow lakes in the Netherlands (e.g. Gulati *et al.*, 2008) and Denmark (Jeppesen *et al.*, 2012; Sondergaard *et al.*, 2007), building on interim results published in the 1990s. The last major review of restoration approaches used in the Norfolk Broads was commissioned in the mid-1990s by the Broads Authority and the Environment Agency, supported by EC LIFE funding (see Madgwick & Phillips, 1996). An update that takes account of the full range of measures applied and evaluates their success over decadal scales is therefore timely.

1.2 Eutrophication of the Broads

Research into the eutrophication process advanced rapidly in the 1960s and 70s to the extent that eutrophication as a phenomenon is now well understood and its effects are widely documented. In the Broads, as with many other shallow lakes in north west Europe, these effects include high levels of phytoplankton production (often including blooms of blue green algae), a reduction and eventually almost complete loss of aquatic plant cover and diversity, high rates of sedimentation, dominance by small open water zooplankton species and loss of plant-associated invertebrates, simplification of the structure of the fish community and dominance by a small number of age classes of planktivores.

Based on early palaeolimnological studies and historical records Moss (1980)

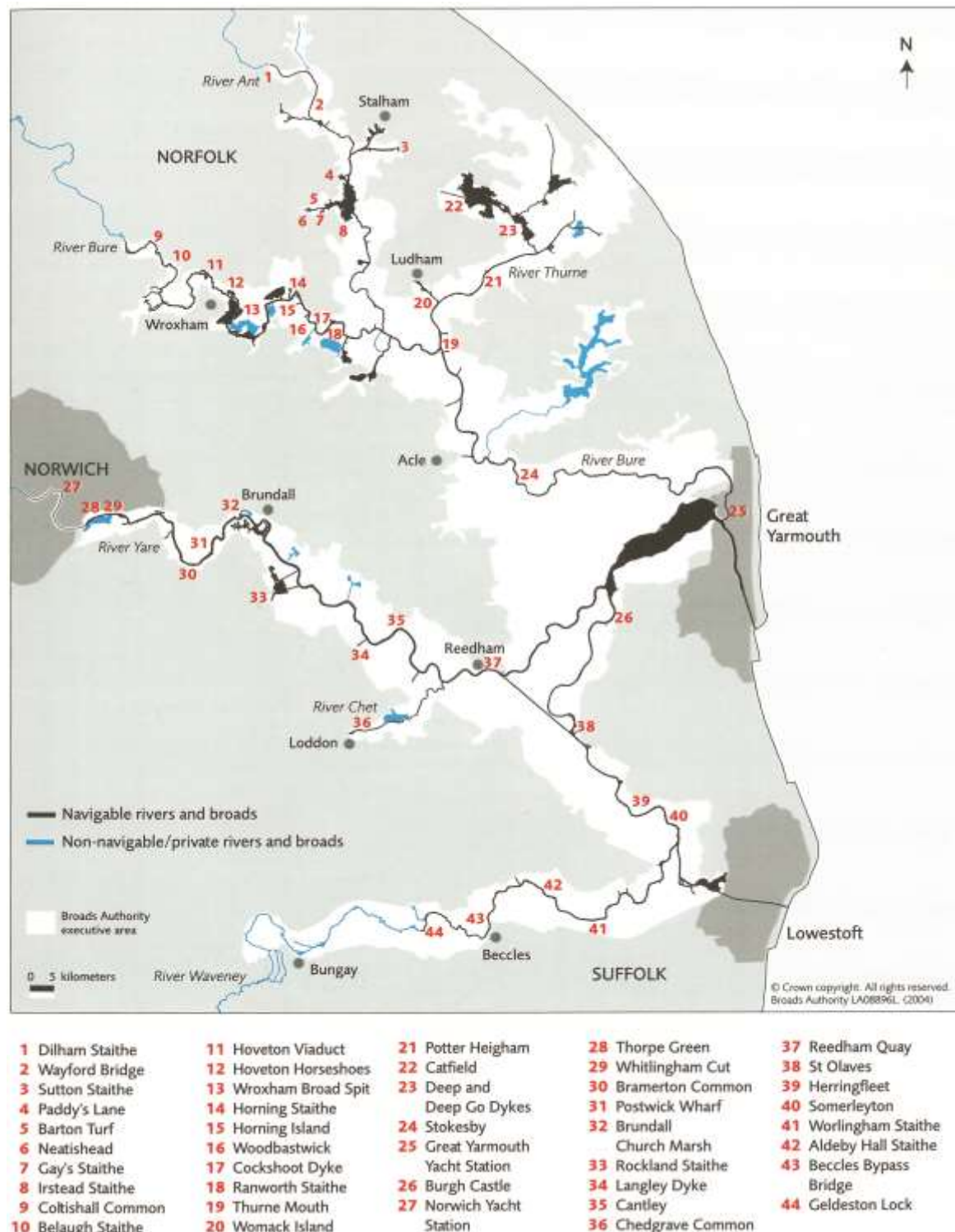


Figure 1.1. Geographical distribution of named broads

developed a three-phase model of ecological change for the Broads (Figure 1.2): Phase 1 (c. pre-1900) comprised clear water lake dominated by 'low- growing' plants, especially charophytes and *Najas marina*; Phase 2 (c. 1900–60s) was characterised by the loss of charophytes and development of tall, canopy-forming taxa (e.g.

Potamogeton pectinatus and *Ceratophyllum demersum*) as a result of eutrophication; Phase 3 (c. post-1950) consisted of sparse macrophytes and turbid water due to phytoplankton dominance. The three-phase model has been widely used to inform lake management and restoration in the Broads and elsewhere in Europe. This model has however, since been refined by Sayer *et al.*, (2010) and Madgwick *et al.*, (2011) who use more extensive palaeoecological evidence and historical records to demonstrate that Phase 1 vegetation was both more diverse than originally believed and characterised by a multilayered structure rather than being composed exclusively of low growing species (Figure 1.1).

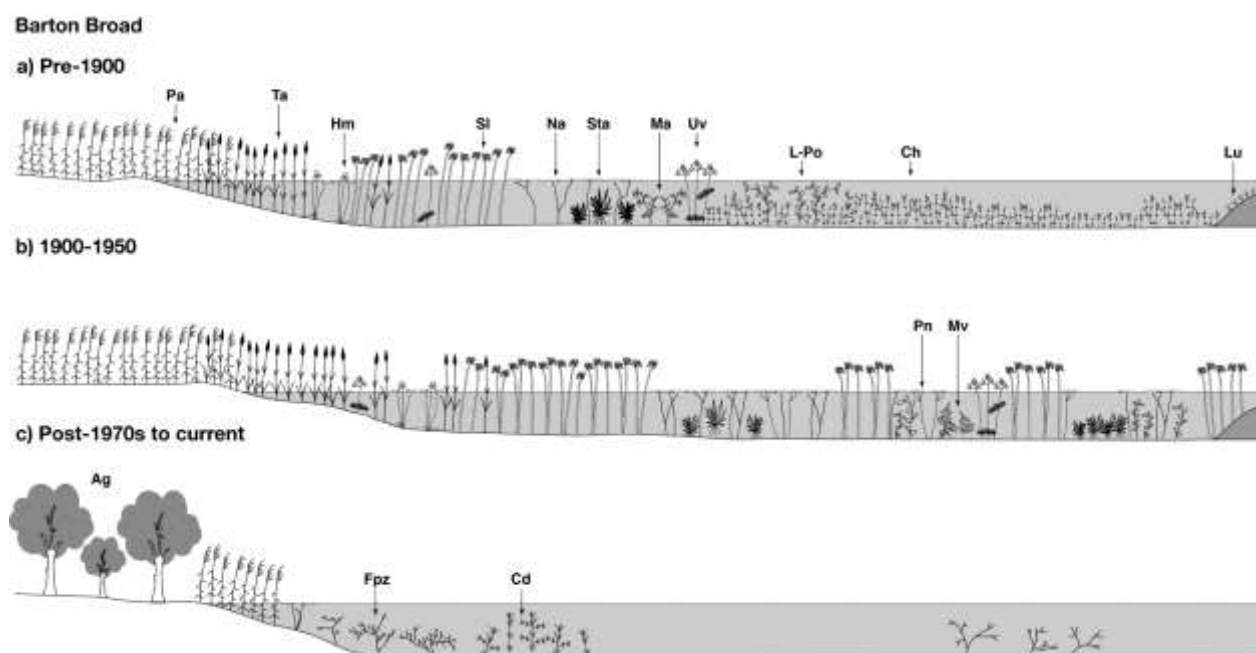


Figure 1.2. Reconstruction of macrophyte spatial relationships in Barton Broad over three time periods; pre-1900 (a), 1900-1950 (b), 1970s-current (c). Codes to plant names: Pa – *Phragmites australis*, Ta – *Typha angustifolia*, Hm – *Hydrocharis morsus-ranae*, Sl – *Schoenoplectus lacustris*, Na – *Nymphaea alba*, Sta – *Stratiotes aloides*, Ma – *Myriophyllum alterniflorum*, Uv – *Utricularia vulgaris*, L-Po – Large, broad-leaved *Potamogeton* taxa (e.g. *P. lucens*, *P. praelongus*, *P. alpinus*), Ch – *Chara* spp., Lu – *Littorella uniflora*, Pn – *Potamogeton natans*, Mv – *Myriophyllum verticillatum*, Ag – *Alnus glutinosa*, Fpz – Fine-leaved *Potamogeton* taxa (e.g. *P. pectinatus*, *P. pusillus*) and *Zannichellia palustris*, Cd – *Ceratophyllum demersum*. From Madgwick *et al.*, (2011).

Subsequent to the development of the three phase model Scheffer *et al.*, (1993) adapted the concept of alternative stable states to shallow lakes, pointing out that at intermediate fertility various internal buffering mechanisms (e.g. spatial refugia, nutrient sequestration in macrophyte tissue) would be likely to sustain a state of clear

water with macrophyte dominance, rather than turbid conditions with phytoplankton dominance (Figure 1.3). Only at comparatively low fertility would macrophyte dominance be ensured (resembling Phase 1 above), whereas at very high fertility algal dominance would always occur. It was postulated that various triggers (e.g. exotic grazers, saline incursions, water level fluctuations, toxicity caused by agrochemicals) would induce a rapid switch from the desirable plant-dominated state (Phase 2 above) to the undesirable algal-dominated state (Phase 3) by causing a break down in internal buffering mechanisms. Since it is rarely possible to recreate the fertility associated with a spontaneous switch to plant dominance ($TP < 30 \mu\text{g l}^{-1}$), the goal of restoration is effectively to identify and remove the triggers responsible for the switch to algal dominance and to re-establish the buffering mechanisms that favour the persistence of clear water, macrophyte-dominated lakes.

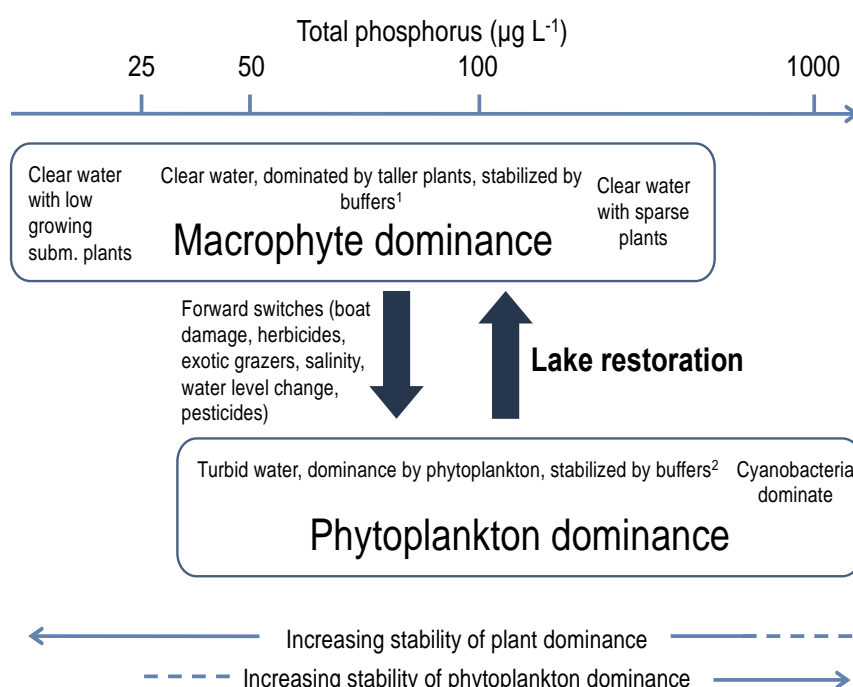


Figure 1.3. Summary of alternative stable states model for shallow lakes based on Scheffer *et al.*, (1993).

1.3 Indicators of restoration success

A combination of multiple palaeolimnological studies, collation of historical macrophyte data, recent observations of macrophyte communities in shallow lakes of the Upper Lough Erne region, Northern Ireland (which show remarkable similarities to the Broads c. pre-1950), and several ecological studies in the Broads and

neighbouring shallow lakes, allow us to outline a number of chemical/ecological features that might be used to judge the success of recent restoration work.

The features listed below likely correspond to the Broads prior to at least the 1940s-50s. As with shallow lakes elsewhere (Bakker *et al.*, 2013) it is recognised that restoration of the Broads to their pristine state (perhaps before 1850-1900) is probably not feasible in the current day.

1. Total phosphorus (TP) concentrations below 40 $\mu\text{g l}^{-1}$.
2. Clear water with chlorophyll concentrations of <30 $\mu\text{g L}^{-1}$ that will typically allow light penetration to depths of about 1.5m thereby ensuring that macrophytes are not light limited over the majority of the area of a lake. The presence of clear water signifies effective top down control of phytoplankton by zooplankton grazing or effective sequestration of nutrients in macrophyte beds.
3. Diverse macrophyte communities (c. 10+ species) covering much of the lake area, including species with different seasonalities. In particular it is important that a lake contains a number of late season/overwintering species that can prevent large seasonal declines in macrophyte biomass and consequent incursions of phytoplankton (Sayer *et al.*, 2010). In this way a lake remains clear throughout the year. The occurrence of *Chara* spp. and other species such as *Ceratophyllum demersum* and *Najas marina* are very important in this respect, with the best lakes supporting multiple species of Characeae. As well as ensuring macrophyte dominance over long periods of the year, a high diversity of species also buffers inter-annual fluctuations in macrophyte biomass. Under such conditions phytoplankton populations lack windows of opportunity, spatially, seasonally and inter-annually.
4. A structured marginal zone of plants extending some distance from the lake edge into open water. Prior to the 1950s, many of the Broads supported extensive swamps with a succession from *Phragmites australis* → *Typha angustifolia* → *Schoenoplectus lacustris* (this was especially evident at Barton Broad – Madgwick *et al.*, 2011). Scattered through this were beds of water-lilies and islands of *S. lacustris* and floating “hover” of *P. australis* were common hundreds of metres off-shore. This community created extensive patches of low-energy water and in turn shelter for free-floating (e.g. *Stratiotes aloides*, *Hydrocharis morsus-ranae*, *Ricciocarpus natans*, weakly anchored (e.g. *Myriophyllum verticillatum*, *Utricularia vulgaris*), or poorly streamlined plant species (e.g. *Potamogeton lucens*, *Potamogeton coloratus*, *Potamogeton compressus*, *Persicaria amphibia*). In pen water these taxa are vulnerable to physical damage, or uprooting from organic-rich sediments by wave action. The restoration of marginal swamps will likely be essential for the maintenance of high macrophyte diversity; including occurrence of all of the above submerged/floating plants which are variously rare or extinct at present in the lakes (though populations survive in some Broadland ditches). In addition this kind of habitat structure provides excellent habitat for ancestral Broads fishes such as tench and pike. A diagrammatic depiction of structured former Broads edge habitat (based on reconstructions of Barton Broad but probably typical of many broads on the basis of sediment core and historic al

record data) is given in Figure 1.2. In addition a photograph showing *Schoenoplectus*-water-lily islands in Barton Broad in the 1940s is given in Figure 1.3. A similar vegetation structure still occurs in many lakes in the Upper Lough Erne system in Fermanagh, Northern Ireland.

5. Low populations of pelagic Cladocera (especially *Daphnia* spp.), due to low amounts of algal food and high macrophyte biomass. *Daphnia* spp. tend to be low to non-existent in broads cores prior to major eutrophication (see Davidson *et al.*, 2008; 2011). A good sign of recovery in the Broad's in the longer-term will be dominance of the cladoceran community by plant-associated species (e.g. *Ceriodaphnia*, *Eurycercus lamellatus*, *Acroperus harpae*).
6. A typical macrophyte-associated fish community, potentially including rudd, perch, pike, eel and especially tench, which spans a range of feeding guilds.



Figure 1.3. “Open waters” of Barton Broad in the 1940s showing islands of *Schoenoplectus lacustris* and *Nymphaea alba*.

1.4 Achieving restoration objectives

In simple terms shallow lake restoration attempts to reactivate a series of internal buffering mechanisms that support macrophyte dominance which in turn promotes desirable changes in higher trophic levels. By mediating higher level interactions a positive feedback is initiated that reinforces macrophyte dominance. Strictly speaking lake restoration should be termed lake rehabilitation since the target is most likely to be the Phase 2 assemblages described in 1.2 rather than the pristine state of Phase 1. It is widely recognised that in balancing the competing demands of food, water,

energy and ecosystem security, returning ecosystems to a truly pristine state is no longer a realistic objective.

Figure 1.4 shows the interlinkages between restoration objectives and the 'entry points' of different restoration techniques. Essentially these involve bottom up control of nutrients, initially by external load reduction (e.g. through diversion, isolation or tighter effluent standards), often followed up by sediment removal to address internal loading problems associated with P release from sediments. The immediate effectiveness of these approaches can be assessed in terms of a reduction in water column TP concentrations.

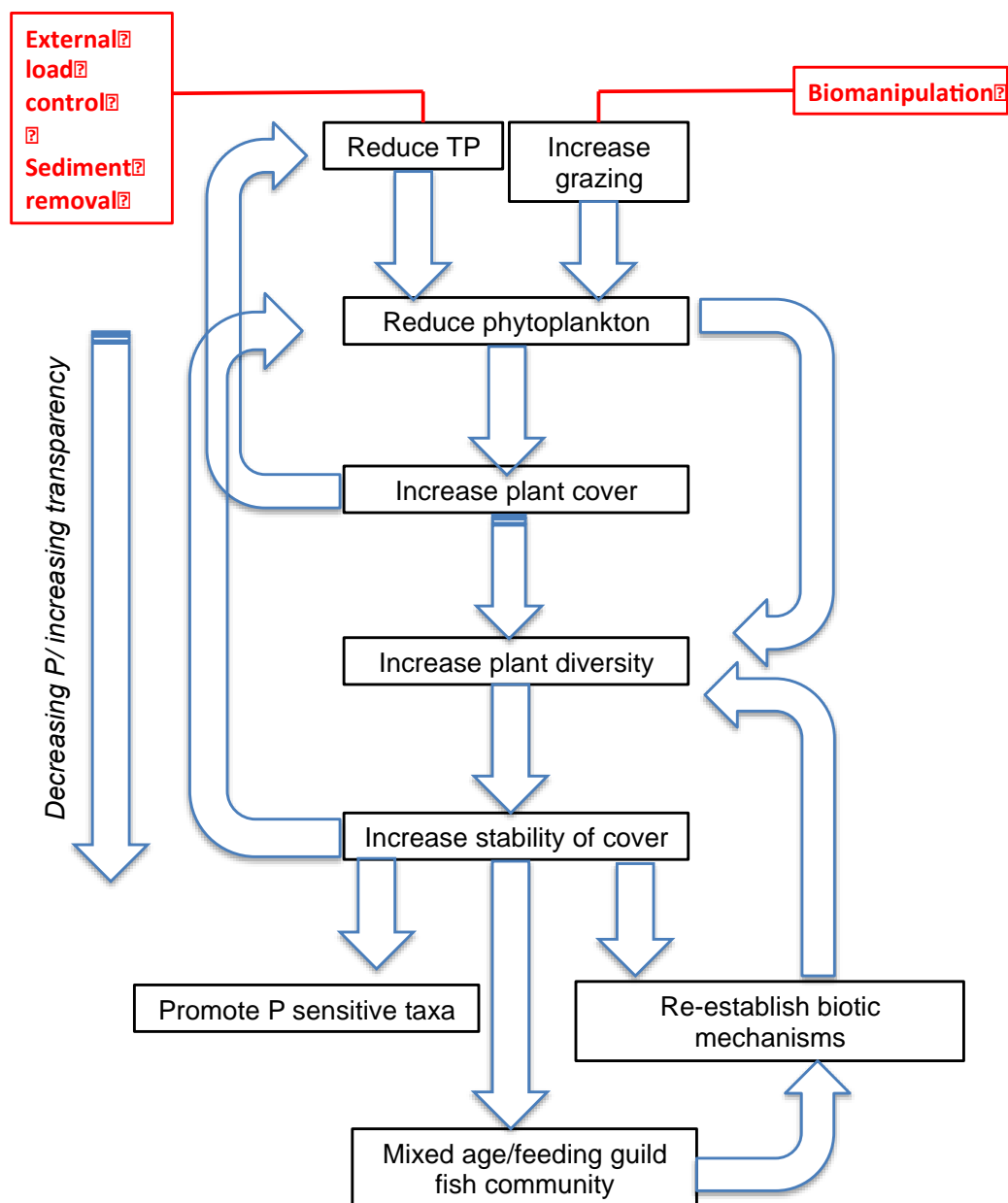


Figure 1.4 Relationships between restoration objectives and the 'entry points' of the major restoration techniques

A reduction in TP availability will lead to a reduction in phytoplankton biomass (expressed in terms of chlorophyll *a* concentrations). However, this will only be of ecological significance if light penetration increases sufficiently for the development of rooted macrophytes. To stimulate increases in water clarity top down control of phytoplankton through grazing by zooplankton has often been attempted through biomanipulation. This seeks to significantly reduce the standing crop of zooplanktivorous fish populations by direct removal or to reinstate controls on zooplanktivorous fish through the stocking of piscivorous species. The initial effectiveness of biomanipulation can be gauged from changes in chlorophyll *a* concentrations or the ratio of chlorophyll to TP.

Over the longer term the effectiveness of these approaches can be assessed firstly in terms of increasing plant cover indicating a shift towards macrophyte dominance. Further positive indicators of this stage being achieved will be provided by increased plant diversity (indicating scope for spatial and temporal niche partitioning, reduced dominance by a small number of nutrient-demanding growth forms and connectivity with a viable propagule bank or with external propagule sources). Following increased diversity, plant cover is more likely to stabilise which is reflected in a reduction in the scale of inter-annual fluctuations in plant cover. Ultimately, as nutrients are effectively sequestered in plant biomass, alongside continuing declines in external and internal load, more P sensitive taxa should begin to establish. In the broads potential indicators of this stage include *Chara* spp. and *Najas marina*. With the increase in stability of vegetation cover and the structural complexity afforded by multiple growth forms the scope for habitat partitioning by fish will increase, thereby promoting the development of a mixed feeding guild and age structured community. Suitable indicators of this are likely to include the presence of rudd and tench, alongside pike and larger perch. The presence of a mixed fish community, coupled with an increase in populations of waterbirds following the increase in plant abundance and availability of plant-associated invertebrate prey, may be important in establishing biotic mechanisms such as herbivory which will help to prevent dominance of submerged vegetation by small numbers of canopy forming species.

1.5 Data sources

The work presented here is based on an integrated analysis of data from a number of independent sources. This allows a longer term and cross-broads perspective to be gained that differs from the intensive but short term and lake-specific approach followed in most previous studies. The sources used include (i) water chemistry data based on mostly monthly samples collected and analysed by the Environment Agency; (ii) macrophyte surveys undertaken annually by the Broads Authority in late summer which are available for the majority of lakes and less frequently for smaller or isolated sites and those without conservation designations; (iii) sediment chemistry data collected either by the NRA, as part of the LIFE project, or commissioned by the Broads Authority, for a selection of broads based on sampling before and after sediment removal; (iv) records of fish removal or capture, mostly from isolated broads, based on annual fish surveys undertaken by ECON on behalf of the Broads Authority; (v) monthly counts of water birds for a selection of sites based on the

Wetland Bird Survey (WeBS) co-ordinated by the British Trust for Ornithology; (vi) analyses of sediment cores or core base material, mostly for plant and animal macrofossils, from a selection of sites undertaken by the Environmental Change Research Centre at UCL; (vii) historical records of aquatic plants from the broads collated by Madgwick (2009); (viii) long term climate data for Lowestoft collected and verified by the UK Meteorological Office.

1.6 Report organisation

This report firstly considers (section 2.1) the interventions that have occurred in the Broadlands in relation to specific management targeted at lake restoration. This is supported by an analysis of climatic changes (section 2.2) that have occurred over the same period as these potentially have a significant bearing on the interpretation of results, including perceived responses of water chemistry or macrophytes to management.

Section 3 deals with spatial patterns in water chemistry and macrophytes across the broadlands network before considering how key parameters have changed temporally within different natural groupings of broadlands (isolated, riverine, and near brackish water bodies on the Upper Thurne). The relationship between physicochemistry and macrophytes is then considered in more detail to identify key thresholds that restoration actions should aim to achieve in order to promote a shift to macrophyte-dominance.

Section 4 explores the effects of the three major restoration methods that have been applied – external load reduction, sediment removal and biomanipulation – in terms of the restoration objectives set out in this section and considers the factors that constrain the effectiveness of these methods. Section 4 concludes with a consideration of emerging and experimental approaches.

Section 5 draws together the major themes from section 4, providing a catalogue of current achievements and challenges and how these relate to the restoration undertaken to date. A final section details some outstanding questions that we consider to be a priority for future research and the implications of these for future monitoring.

A separately available Annex contains dossiers on individual broadlands covering the larger or more significant water bodies and those with a well documented management history and associated monitoring of water chemistry and aquatic vegetation. The broadlands covered in these dossiers include Alderfen, Barton, Cockshoot, Cromes, Hoveton Great, Hoveton Little and Pound End, Heigham Sound, Hickling, Horsey, Martham North and South, Ranworth, South Walsham, Trinity Broadlands, Upton and Wroxham.

2 Overview of changes in the Broads over the study period

2.1 Management activities

The management activities carried out in individual broads have already been catalogued by Kelly (2013) and it is not intended to repeat this information in the same detail. The various forms of management have been undertaken for a combination of reasons, including recreation and navigation, but have been primarily intended to reduce the effects of eutrophication, with the aim of restoring the shallow lake ecosystem to a clear water macrophyte-dominated state.

The various actions undertaken to date and their purpose are summarised in Figure 2.1. The locations of the relevant sites are shown in Figure 1.1. These actions can be considered as falling into four major groups that are arranged chronologically from top to bottom. This sequence roughly follows that which has been adopted in shallow lake restoration projects in Denmark and the Netherlands.

In the first instance various measures have been undertaken to reduce external load. These include the diversion of inflows, the physical isolation of a lake from the main river (where there is no statutory right of navigation), and the introduction of phosphorus stripping at the main sewage works in the region to meet tighter effluent standards.

External load reduction has been followed up in many cases by sediment removal (also referred to as suction dredging, de-silting and mud-pumping). This has typically followed modest responses of TP or chlorophyll concentrations to external load reduction measures, leading to the conclusion that internal loading, due to phosphorus release from sediments, is maintaining the nutrient supply. Sediment removal is the single commonest intervention applied across the broads. Although internal load reduction was the primary motivation for the earlier phases of sediment removal at sites such as Alderfen, Barton, Cockshoot and Hoveton Little Broad, more recently (2005 onwards) sediment removal has been widely applied to smaller hydrologically isolated sites subject to chronic loss of depth and where there is a specific need to restore habitat for rooted macrophytes. In such cases palaeolimnological investigations have often been used to identify the desirable depth to expose sediment containing high densities of charophyte oospores.

The third category of intervention falls under the heading of biomanipulation. In the broads this has mainly involved the removal of zooplanktivorous fish, although elsewhere the selective removal of benthivores and/or stocking of piscivores has also been undertaken. The primary short term purpose of biomanipulation is to achieve clear water conditions suitable for the growth of macrophytes by restoring top down control of phytoplankton populations through grazing by zooplankton. Biomanipulation has often been undertaken following a poor response to sediment removal, for example at Cockshoot Broad, the earliest of the broads to be biomanipulated (1989). In the longer term, having secured a stable high cover of macrophytes, biomanipulation seeks to achieve a mixed age, mixed feeding guild fish community. This would include plant-associated species such as rudd and tench,

alongside piscivores such as pike, perch and eels, that can be maintained without continuing intervention.

The final category deals with a group of approaches that are often more experimental in nature and driven mainly by the need to overcome factors constraining the recovery of macrophyte populations, or in some cases factors which are considered to threaten the stability of existing macrophyte populations. These techniques include deliberate transplanting of aquatic or emergent plants, often in conjunction with bird exclosures to prevent grazing, mainly by coot, swans or feral geese. On the Thurne Broads in particular fences or coir mattresses have been widely used to combat goose grazing. Trials have been undertaken at a small number of sites using cobweb brushes to act as artificial plant beds and increase refugia from predation for zooplankton or invertebrates. Finally, clearance of alder and willow scrub from lake edges has been undertaken at many sites, often with limited success, in an attempt to stimulate expansion of reedswamp, reduce shading of shallow water habitat and to reduce terrestrialsation of the existing reed margin.

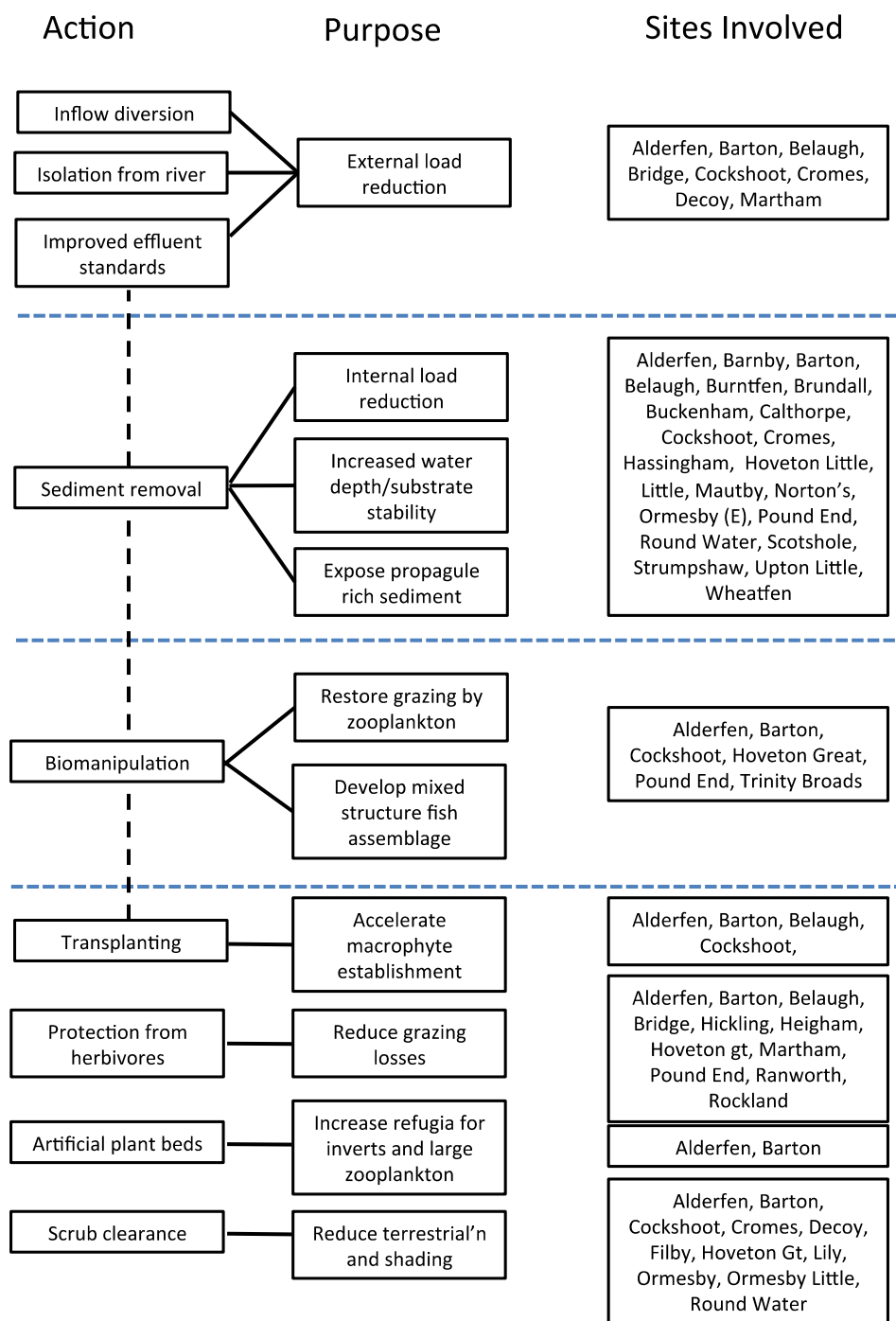


Figure 2.1. Summary of major restoration measures employed in the broads, their purpose and the sites affected. Information extracted mainly from Kelly (2013).

2.2 Climatic changes

2.2.1 Data Sources

It is important to consider the potential impact local climatic changes have had when interpreting the long-term monitoring data and identifying possible responses to management.

Monthly maximum and minimum air temperature, rainfall and sunshine duration data were obtained from a site at Lowestoft from the meteorological office public archive (Meteorological Office). Environment Agency daily mean river flow data were taken from gauging stations on the River Ant (Honing Lock) and the River Bure (Horstead Mill). Data for the North Atlantic Oscillation (NAO) were taken from (Hurrell and National Center for Atmospheric Research Staff, 2013).

2.2.2 Summary

Over the period of monitoring there have been significant changes in local climate. The most marked is a sudden increase in temperature which occurred at the end of the 1980s. There was also an increase in rainfall from the late 1990s which following groundwater re-charge resulted in a marked increase in river discharge from 1999. Thus over the period of monitoring the Broads have experienced a shift from a dry and cold period to wetter and warmer conditions.

These changes have clearly influenced the long-term trends of several water quality parameters in the Broads. Examples are periods of elevated phosphorus concentrations during the summers when river discharge was low (see 3.1.1.1) and elevated nitrate during wetter years.

The increase in temperature which occurred at the end of the 1980s may also have aided the re-establishment of macrophytes as it can be shown that the duration of sunshine is correlated with macrophyte cover in broads with well-established macrophytes (see 3.2.3.) and that temperature is a significant predictor of macrophyte cover in broads where they are re-establishing (see 3.2.3).

2.2.3 Temperature & hours of sunshine

There have been several significant fluctuations in air temperature but the most marked was a rise of about 1°C between 1987 and 1988 (Figure 2.3.3-1). This change has been seen widely across northern Europe (Gerten and Adrian 2000) and has been attributed to a prolonged positive phase of the NAO (Hari *et al.*, 2006). However, the temperature increase has been more sustained than the NAO, extending to 2010 (Figure 2.3.3-2). This sudden increase followed a particularly cold winter in 1986 when many broads froze for a sufficiently long period for ice-skating to take place (Phillips pers obs). A similarly cold winter occurred in 1979 but the more recent cold winters of 1992 and 2010 were not as severe (Figure 2.3.3-3). The abrupt temperature change of 1987/1988 was apparent in both trends for both winter and spring (Figure 2.3.3-3 & Figure 2.3.3-4) but less so during the summer (Figure 2.3.3-5) where the modelling suggests a continuing increasing trend. However, a

bivariate GAM model which considers change in seasonality with the long term trend clearly shows the transition from a cold to warm period around 1990 (Figure 2.3.3-6). The significance of this change is also clearly seen when a longer time period is considered (Figure 2.3.3-7) and it is clear that over the period of monitoring the Broads area has been subject to a significant increase in temperature with fewer and less severe cold winters and warmer summers since 1990.

Associated with this abrupt increase in temperature is a similar increase in the hours of sunshine (Figure 2.3.3-8). However, unlike the long term temperature record, which suggests that conditions since 1990 are much warmer, hours of sun returned to conditions more typical of the period. Thus the period from 1970-1990 was not only colder than average but also had significantly more overcast days.

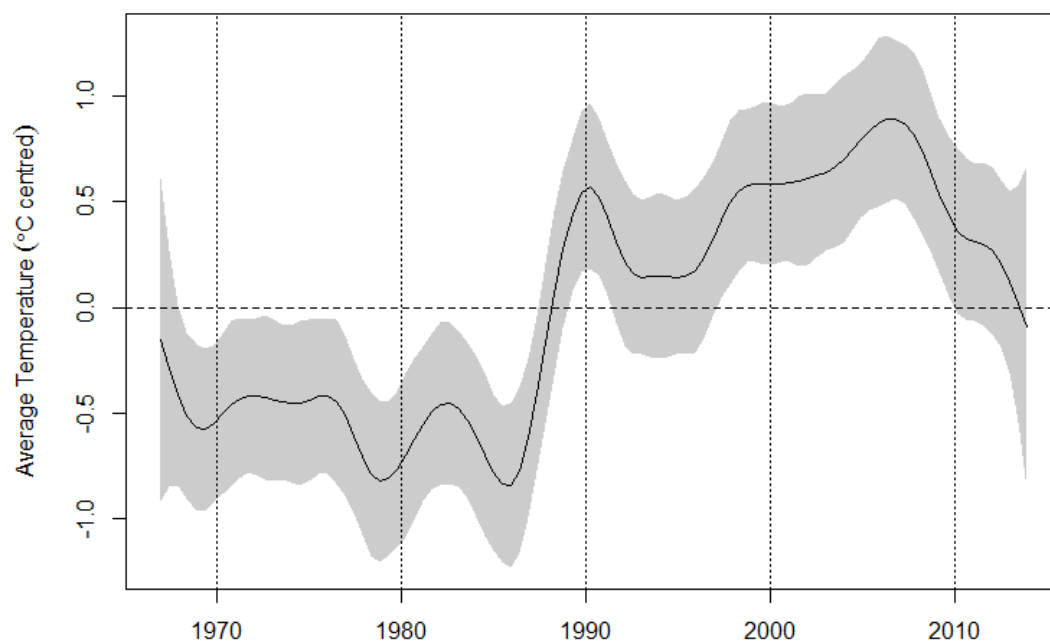


Figure 2.2.3-1 Trend in monthly mean air temperature derived from GAM model, values expressed relative to the average for the period (1967-2013)

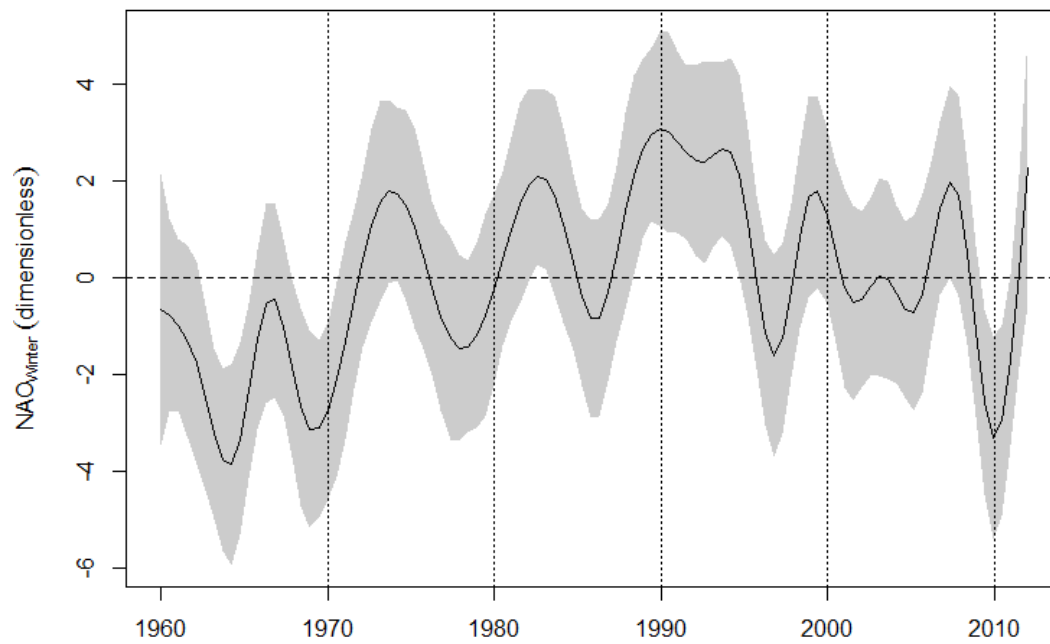


Figure 2.2.3-2 Trend in winter NAO index derived from GAM model, values expressed relative to the average for the period (1960-2012)

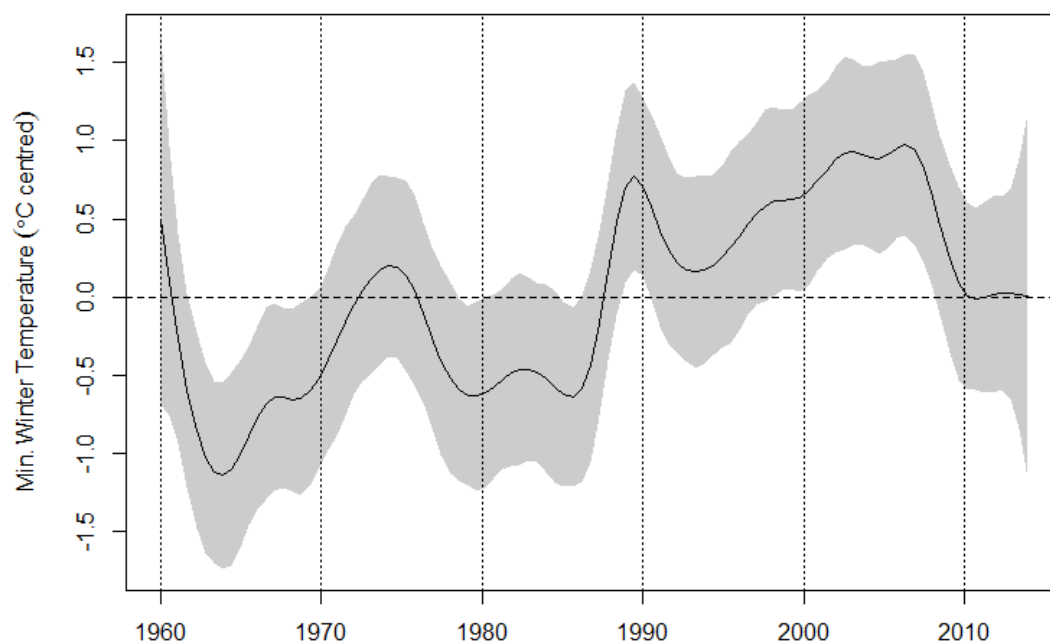


Figure 2.2.3-3 Trend in mean monthly minimum air temperature for winter (November – February) derived from GAM model, values expressed relative to the average for the period (1960-2013)

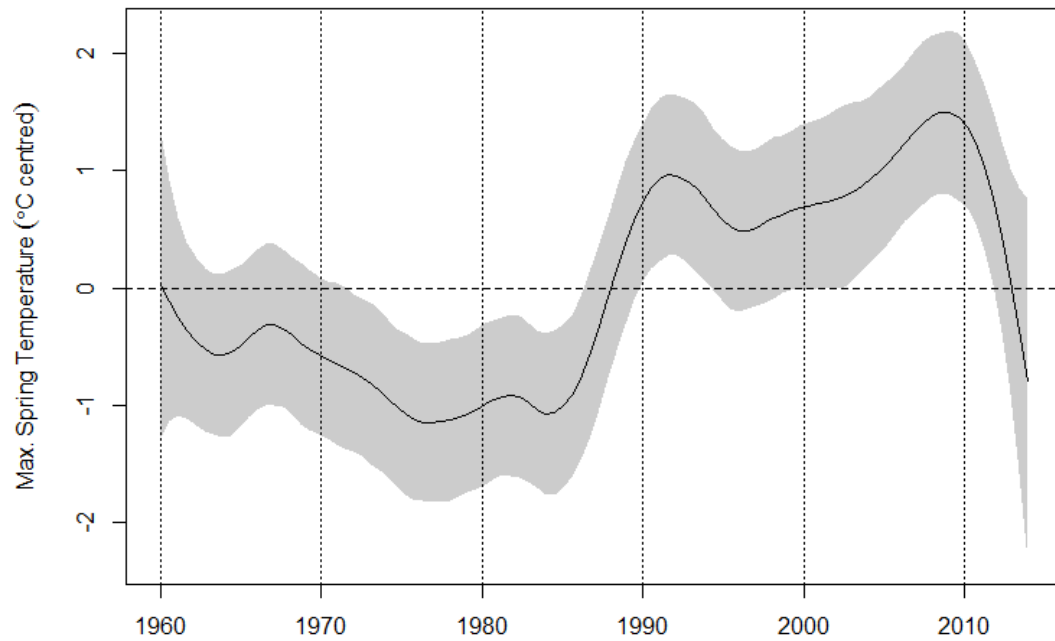


Figure 2.2.3-4 Trend in mean monthly maximum air temperature for spring (March – May) derived from GAM model, values expressed relative to the average for the period (1960-2013)

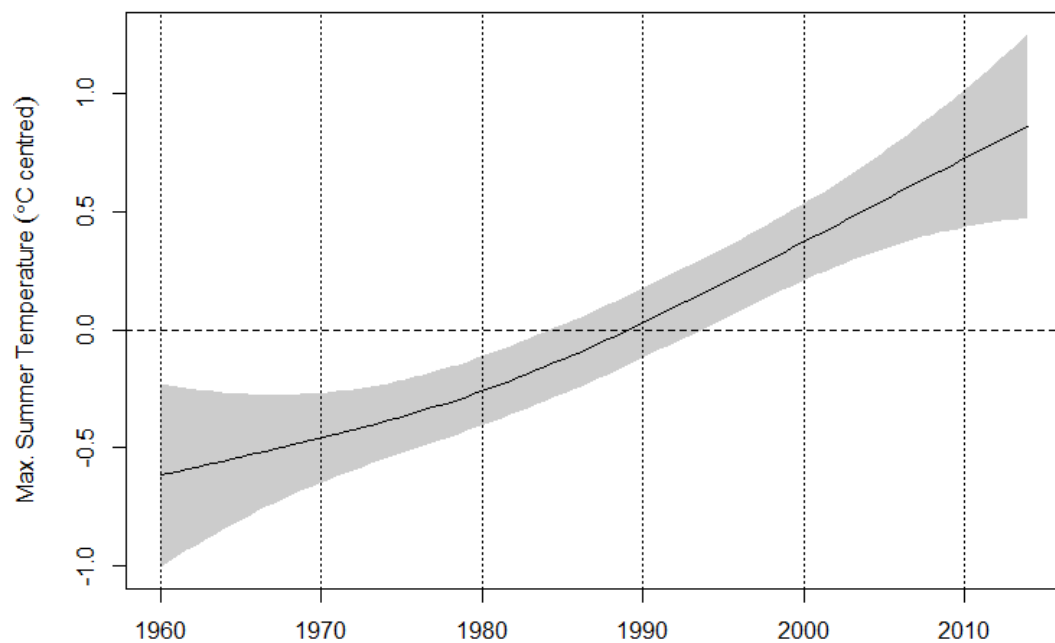


Figure 2.2.3-5 Trend in mean monthly maximum air temperature for summer (June–October) derived from GAM model, values expressed relative to the average for the period (1960-2013)

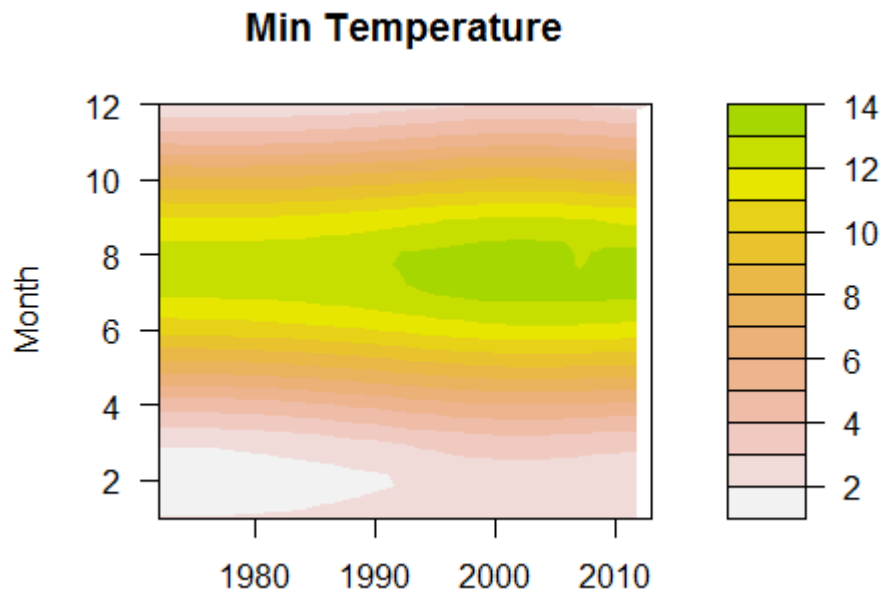


Figure 2.2.3-6 Contour plot showing changes in monthly minimum air temperature (1970-2013) with season and year using bivariate GAM model.

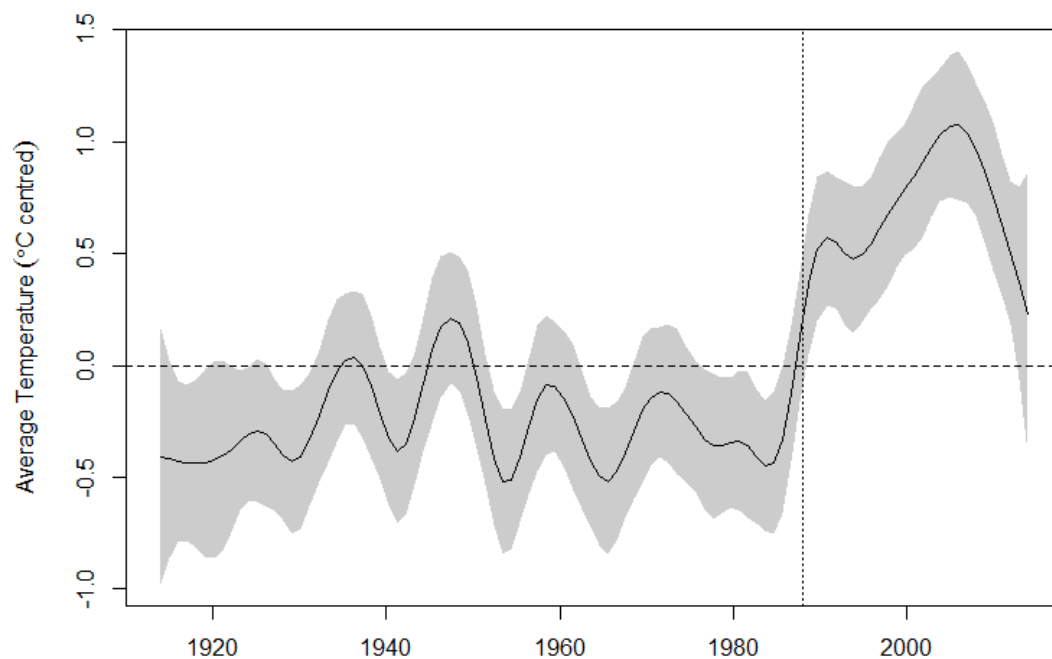


Figure 2.2.3-7 Long-term trend in monthly mean air temperature derived from GAM model, values expressed relative to the average for the period (1914-2013). Vertical line marks the abrupt temperature shift in 1987/1988

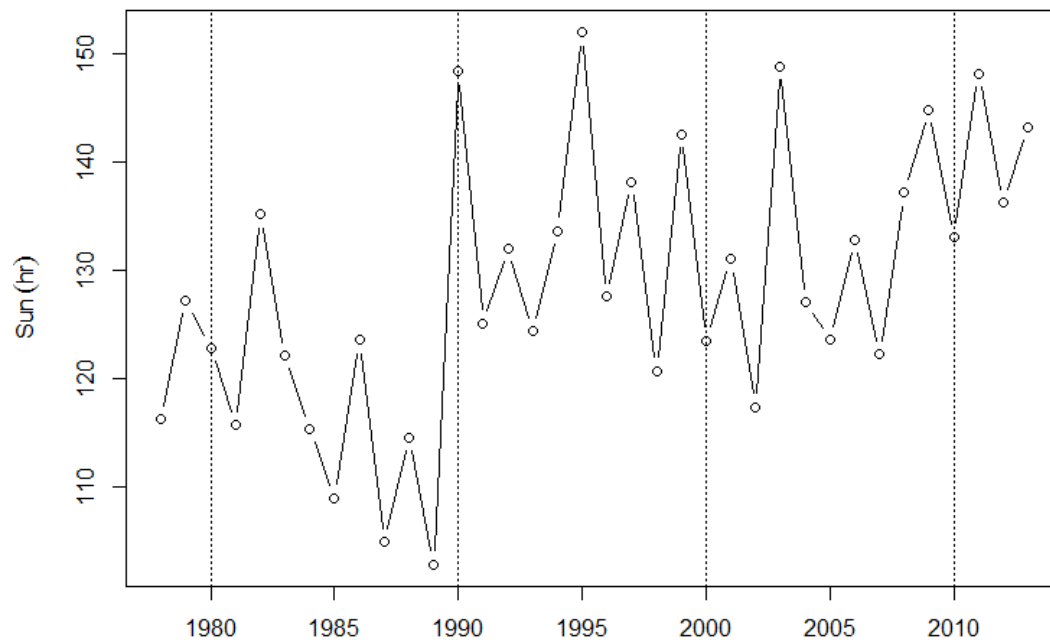


Figure 2.2.3-8 Change in monthly average hours of sun (1978-2013)

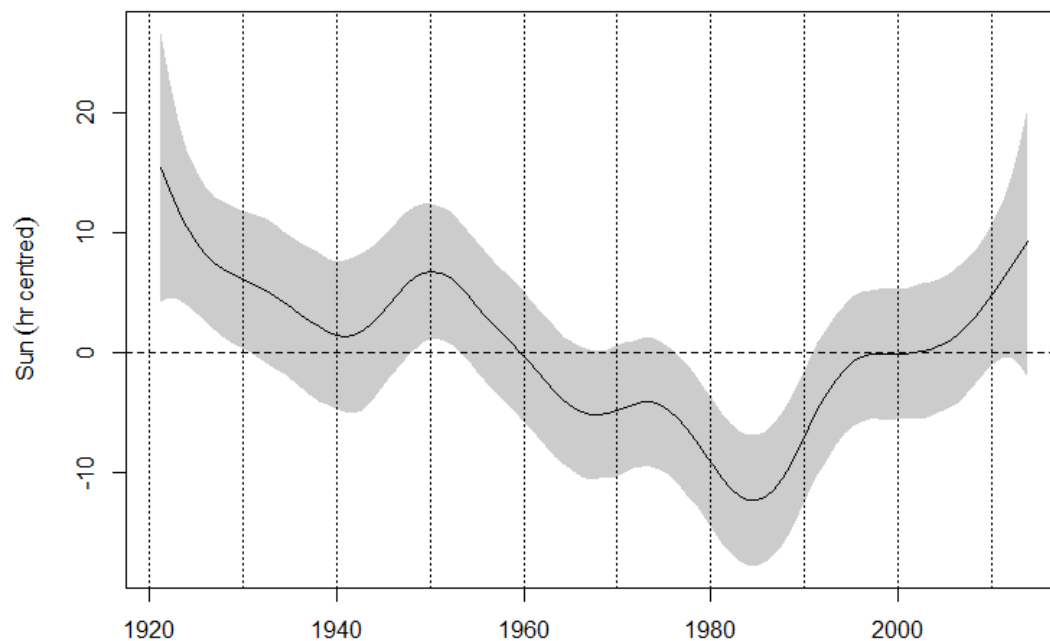


Figure 2.2.3-9 Long-term trend in monthly hours of sun derived from GAM model, values expressed relative to the average for the period (1920-2013).

2.2.4 Rainfall and River Discharge

Monthly mean rainfall increased from the mid-1990s (Figure 2.3.4-1) although the change was not as marked as the change in temperature, with above average rainfall not occurring until 1999 (Figure 2.3.4-2). Inter-annual changes are most marked during the summer (June-October) with a sequence of periods of lower summer rainfall at 6-8 year intervals (Figure 2.3.4-3). The cumulative effect of these repeated

periods of lower rainfall significantly reduced groundwater re-charge resulting in lower than average river flow prior to 1999 in both the rivers Ant and Bure, with particularly low flows in the summers of 1991 and 1996 (Figure 2.3.4-4 & Figure 2.3.4-5)

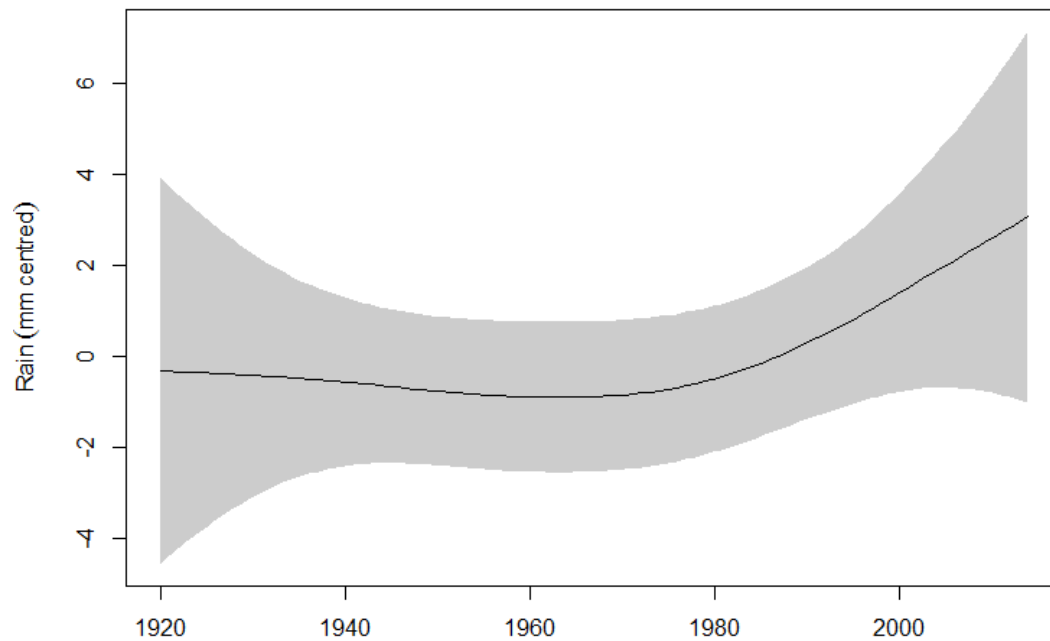


Figure 2.2.4-1 Long-term trend in average monthly rainfall derived from GAM model, values expressed relative to the average for the period (1920-2013).

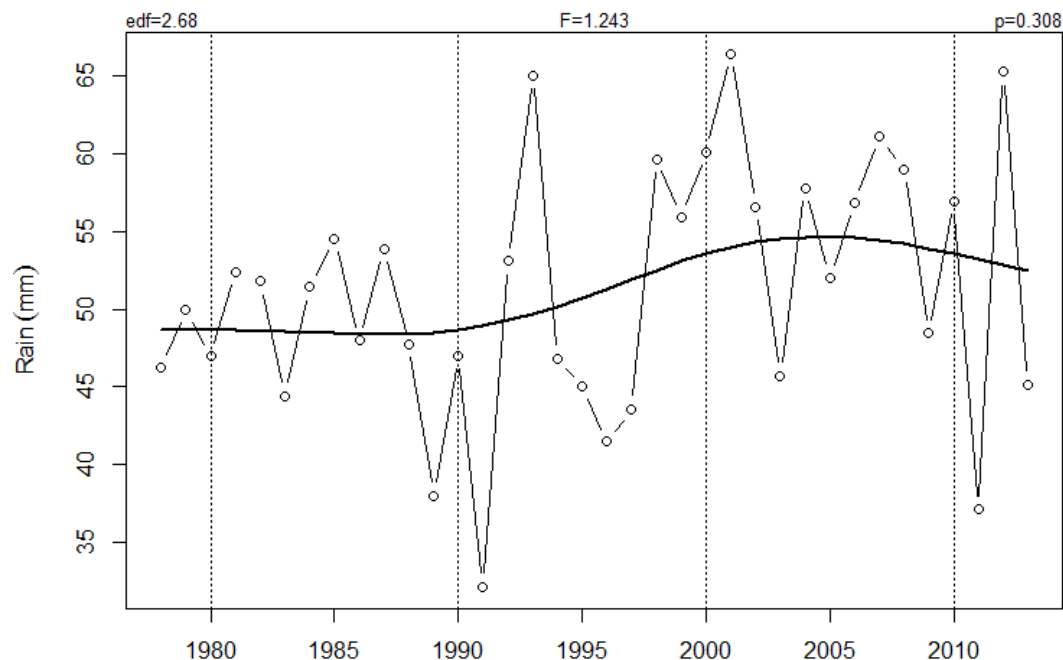


Figure 2.2.4-2 Changes in average monthly rainfall (1978-2013)

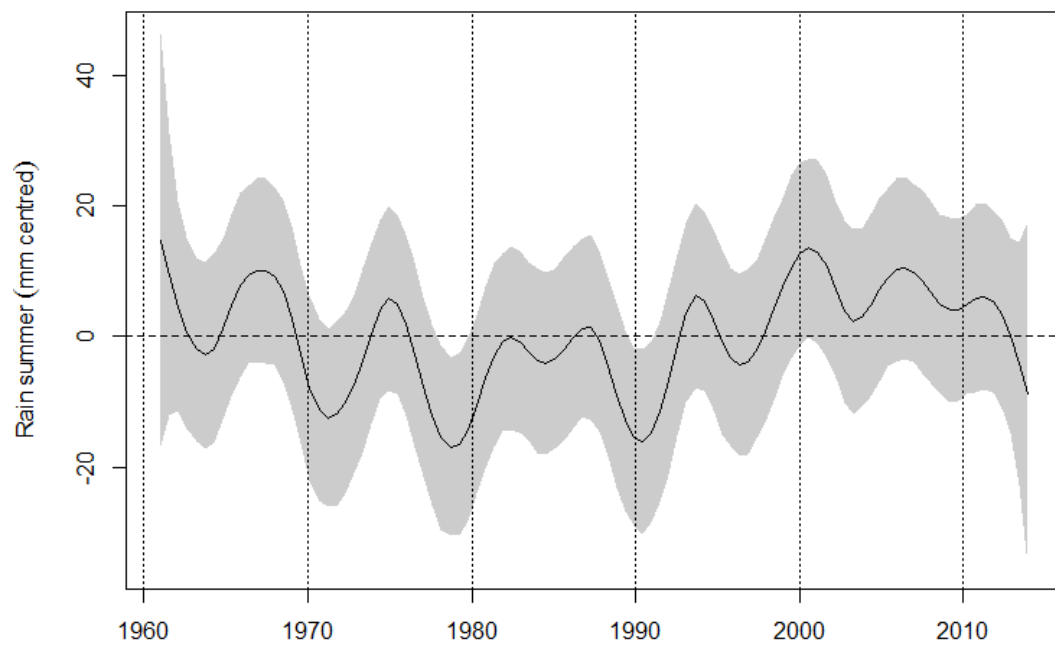


Figure 2.2.4-3 Trend in average monthly summer (June=October) rainfall derived from GAM model, values expressed relative to the average for the period (1960-2013).

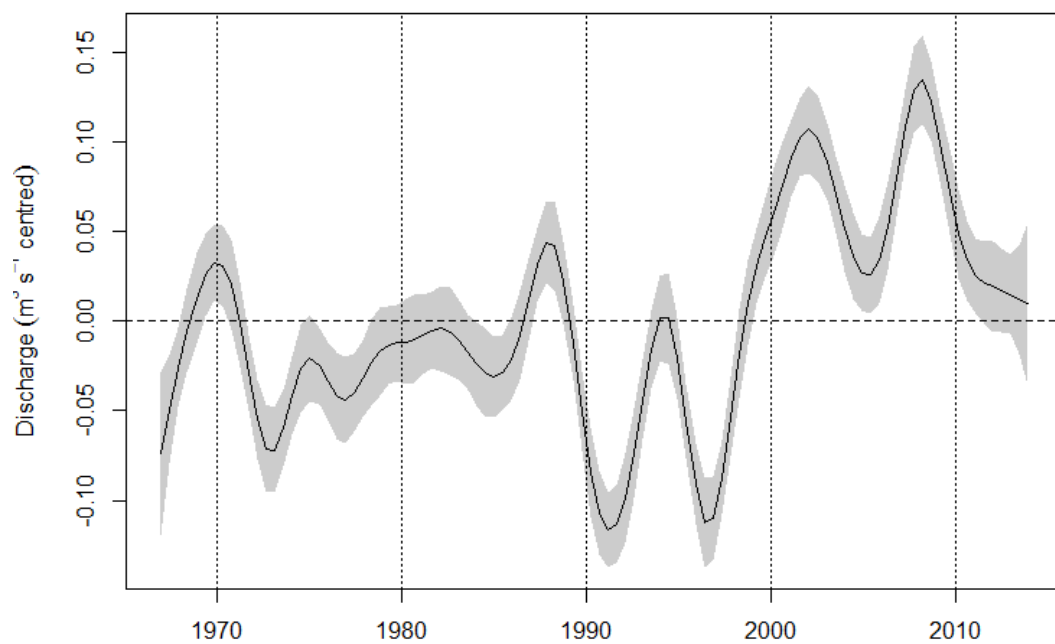


Figure 2.2.4-4 Trend in mean daily flow for River Ant at Honing Lock, derived from GAM model, values expressed relative to the average for the period (1960-2013).

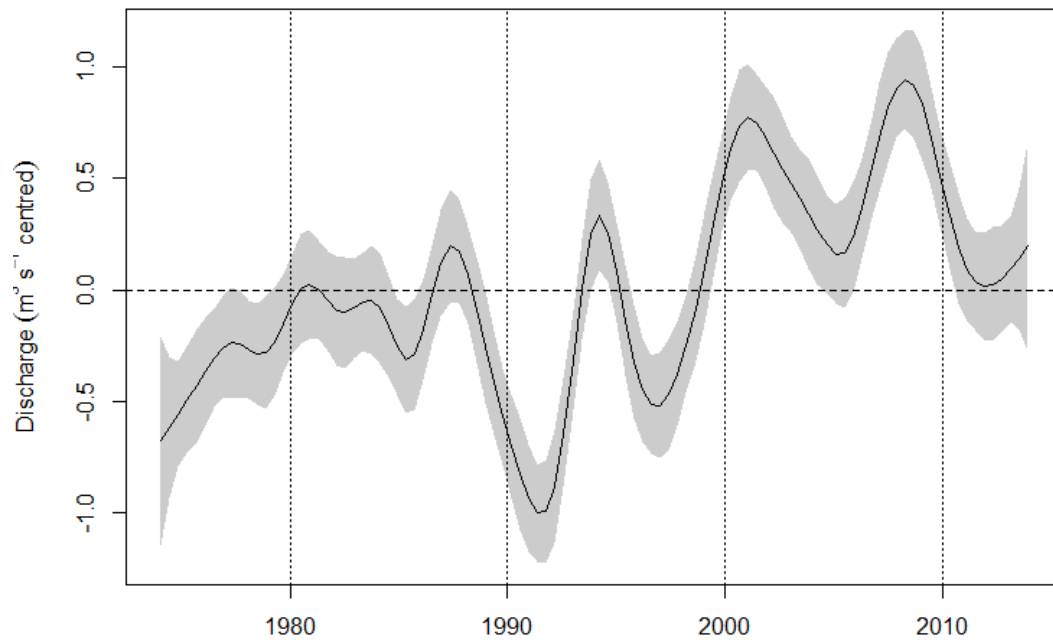


Figure 2.2.4-5 Trend in mean daily flow for River Bure at Horstead Mill, derived from GAM model, values expressed relative to the average for the period (1974-2013).

3 Spatial and temporal patterns in physicochemistry and aquatic vegetation and their inter-relationship

3.1 Spatial and temporal patterns

3.1.1 Water chemistry

3.1.1.1 Total Phosphorus

Since the start of water quality monitoring in the Broads in the mid-1970s the annual mean total phosphorus (TP) concentrations of many of the broads have decreased substantially (Figure 3.1.1-1). In the broads linked to the main rivers (Ant, Bure and Thurne) TP concentrations have decreased significantly (Table 3.1.1-1), following the introduction of phosphorus removal from effluents from the early 1980s and subsequent tightening of effluent standards (Figure 3.1.1-1ab). In the isolated broads, which are not influenced by these effluents there has generally been less significant change (Figure 3.1.1-1d) or changes are a reflection of other local management actions and environmental conditions (Figure 3.1.1-1c and e). Only the Trinity Broads show a significant overall increasing trend (Table 3.1.1-1 & Figure 3.1.1-1c)

There are clear influences of river discharge and hence local rainfall on the long-term trends. During the particularly low flow periods of the early 1990s there are clear interruptions in the declining trend of TP in the riverine broads and Thurne Broads (Figure 3.1.1-1a & b). This may be a reflection of longer periods of summer retention and a greater influence of phosphorus released from sediments (see 4.2.2), but it is also a result of upstream movement of more phosphorus enriched water as a result of tidal action from the lower reaches of the rivers, coupled with lower dilution. This is illustrated in Figure 3.1.1-2 where the minimum annual mean TP in the rivers and riverine broads of the Bure and Ant increases with chloride concentration. Significant tidal surges are not uncommon in the Broadland rivers (see 3.1.1.2) and, at least during the early 1990s, water from further downstream, which had higher TP concentration due to the continued discharge of phosphorus enriched effluents, influenced many of the riverine broads further upstream.

In the isolated broads, Alderfen, Cockshoot and Cromes, changes in TP show similar overall trends, with an increase in the 1990s and a subsequent decline from 2000. These changes are more complicated to explain (see details for each broad in Annex dossiers) but the highest concentrations occurred in the 1990s during the period of exceptionally low river flows. All of these broads have small effective catchments, either naturally (Cromes & Alderfen) or as a result of isolation (Cockshoot) and the combined effect of low discharge, and probably low groundwater level, may have magnified the influence of the release of phosphorus from the sediment as a result of lower dilution and very limited summer water exchange. The extent of phosphorus release from the sediment of these sites is discussed below (4.2.2), but it was only after 2000 when river flows, and by implication ground water levels, markedly increased that lower TP concentrations were observed (Figure 3.1.1-1e). Similar relationships between TP concentration and water level were noted for Filby Broad (Annex: Trinity Broads dossier section 4.1), although in the Trinity Broads TP

concentrations also appear to have increased following manipulation of the fish community in 2005. Benthic feeding fish can influence sediment derived phosphorus either directly via re-suspension of the sediment or indirectly via predation of benthic chironomids which can influence bioturbation and in different circumstances either increase or decrease phosphorus retention (Breukelaar *et al.*, 1994; Hansen *et al.*, 1997; Tátrai 1988).

Table 3.1.1-1 Mann Kendall test for monotonic trend of mean annual total phosphorus time series for monitored broads

	Mann Kendall Test			
	Tau	2-sided pvalue		
Barton	-0.713	0.000	***	1972-2012
Hoveton Little	-0.695	0.000	***	1983-2012
Hoveton Great	-0.667	0.002	***	1983-2012
Wroxham	-0.613	0.000	***	1983-2012
Ranworth	-0.573	0.000	***	1978-2012
Cockshoot	-0.527	0.000	***	1978-2012
S Walsham	-0.481	0.000	***	1971-2012
Hickling	-0.428	0.000	***	1983-2012
Upton	-0.390	0.001	***	1979-2012
Alderfen	-0.317	0.007	**	1972-2012
Cromes	-0.109	0.472	ns	1972-2012
Horseý	0.129	0.299	ns	1978-2012
Rollsby	0.150	0.211	ns	1982-2012
Ormesby	0.194	0.099	*	1981-2012
Martham S	0.209	0.108	ns	1978-2012
Ormesby Little	0.281	0.018	**	1978-2012
Filby	0.308	0.009	**	1978-2012
Martham N	0.347	0.007	**	1978-2012

The continued importance of phosphorus release from the sediment of the majority of these broads is illustrated by the seasonal pattern of TP concentration (Figure 3.1.1-3). This shows that the maximum TP concentration occurs in the summer (June – August) rather than the winter, which would have been expected if the catchment was the primary phosphorus source. A few of the broads (Martham N & S, Upton, Cromes and Cockshoot) have relatively small seasonal variation and in these no clear summer peak was observed. These are the broads with the lowest average TP (Figure 3.1.1-4) and are the ones that have either always been dominated by macrophytes or have had higher macrophyte cover over the last 10 years and have had their sediment removed.

In summary the Broad's now have a wide range of TP concentration (Figure 3.1.1-4), all lower or largely unchanged from those found at the start of the monitoring period. The lowest values, which as a 10 year average (2003-2012) meet the WFD targets for High status, are found in the macrophyte dominated Upton and Martham Broad's.

The highest concentrations, which are close to the Moderate/Poor boundary, are found in South Walsham and Filby broads. However the majority (the Bure broads, Hickling, the isolated Alderfen and Cromes broads and the Trinity broads) are now all in Moderate status. A few of these are relatively close to the Good/Moderate boundary (Ormesby, Decoy, Hoveton Little Pound End) while Barton, Horsey are Good and Cockshoot is at the High/Good boundary.

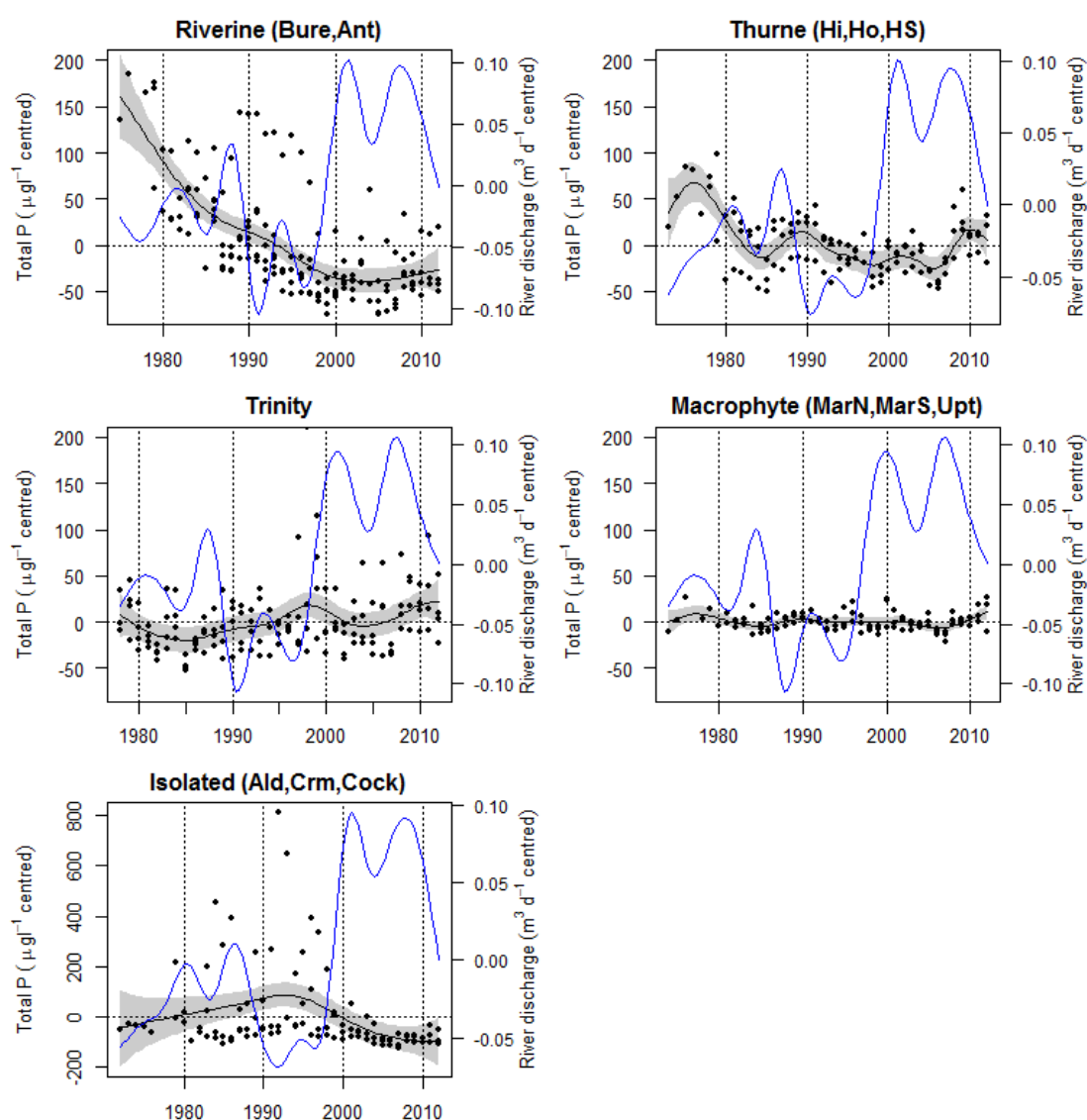


Figure 3.1.1-1 Trend of total phosphorus (black line) in different groups of broads compared to river discharge (blue line) a) riverine broads (Barton, Wroxham, Hoveton Great & Little, Ranworth South Walsham); b) Thurne broads (Hickling, Horsey, Heigham Sound), c) Trinity broads; d) Macrophyte dominated broads (Martham N & S, Upton), e) Small isolated broads (Alderfen, Cromes N, Cockshoot). Lines are GAM smooths of annual mean data, values expressed relative to the average for the period of data available for each broad.

There is no indication that the mean annual TP concentration in the majority of these lakes is continuing to decrease in response to previous management actions. Mann Kendall tests for trend show no significant change since 2003, the only exceptions being Martham South where there has been a slight but significant increasing trend, the reasons for which are discussed in the dossier for that site (see Annex), but are not fully understood. For Martham South the increase in TP is most likely a result of changes in tidal mixing with the more phosphorus enriched lower River Thurne. However, the 2013 WFD classification (2010-2012) for TP was only Good status and given that these broads had previously achieved High status for phosphorus, and that they contain the most diverse macrophyte communities it would be a cause for concern if the trend of increasing TP concentration continued.

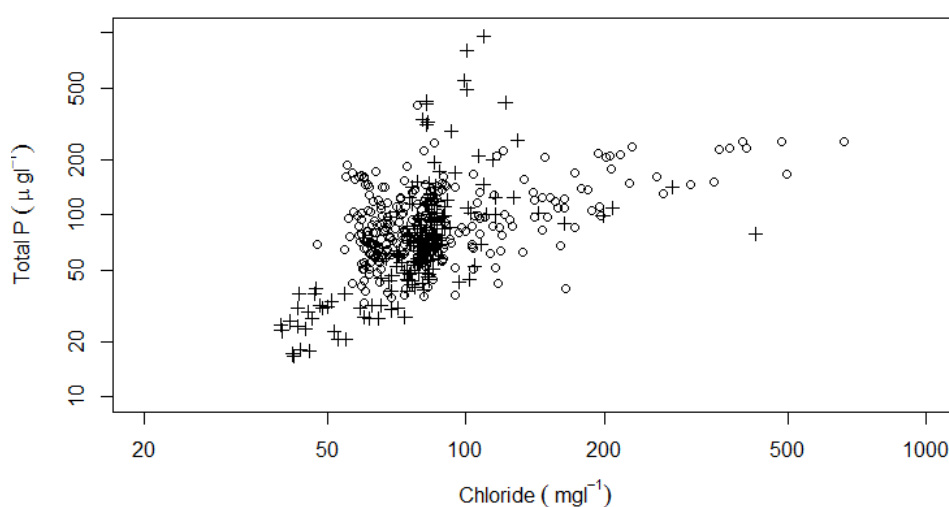


Figure 3.1.1-2 Relationship between mean annual total phosphorus and chloride concentration in the rivers Ant and Bure and their associated broads. Circles riverine broads, crosses isolated broads.

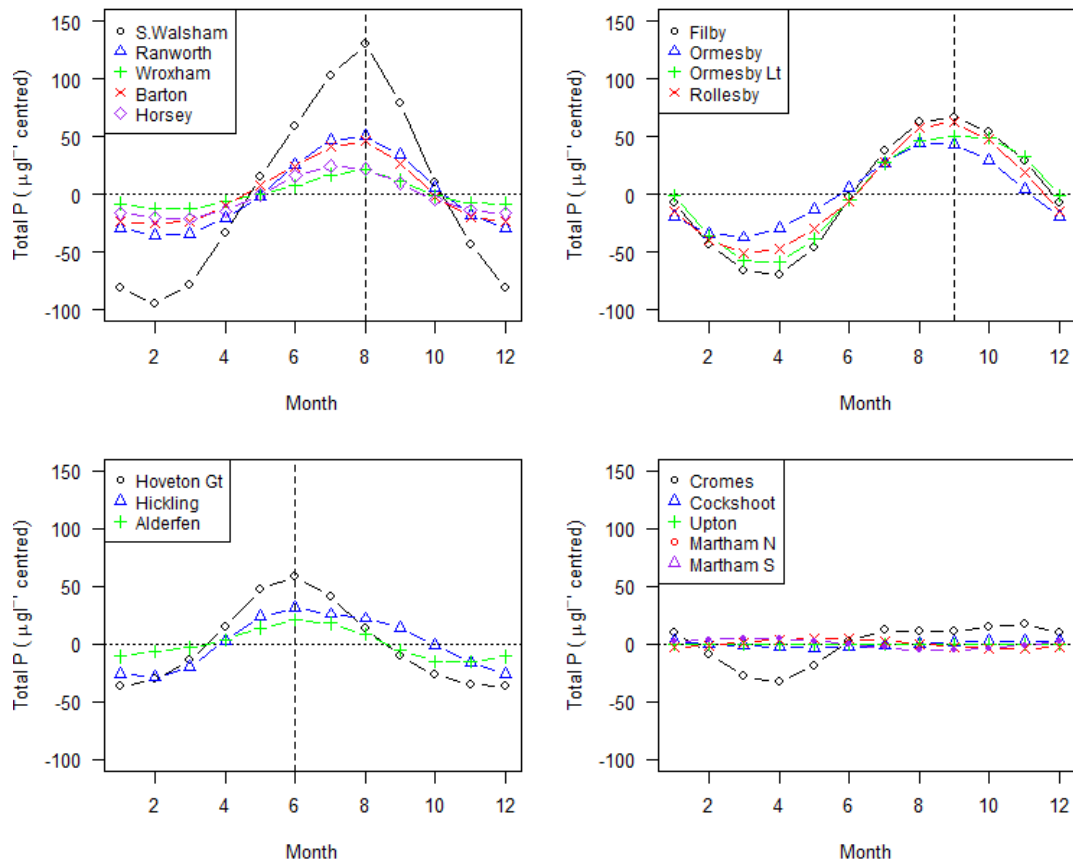


Figure 3.1.1-3 Seasonal pattern of total phosphorus for last decade, grouped by similarity. Lines are GAM smooths of monthly mean data, values expressed relative to the average for the period (2003-2012).

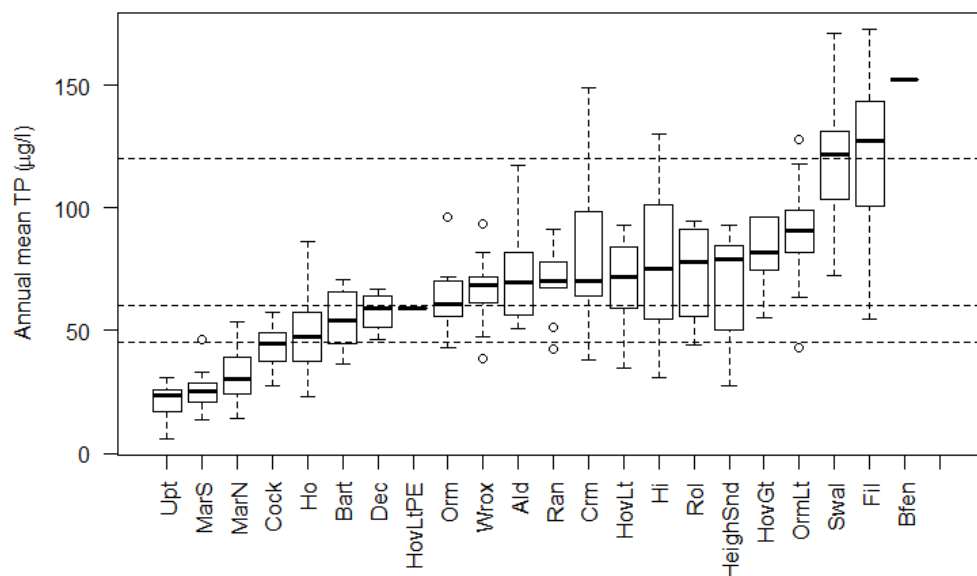


Figure 3.1.1-4 Range of annual geometric mean total phosphorus concentration ($\mu\text{g l}^{-1}$) for monitored broads over the decade (2003-2012). Horizontal lines mark Water Framework Directive boundary values (Moderate/Poor, Good/Moderate, High/Good).

3.1.1.2 Chloride

Chloride concentration in the Broads range from less than 50 mg l⁻¹ in Upton Broad to over 2000 mg l⁻¹ in Horsey Mere. This reflects their position relative to the influence of saline water from the tidal rivers and the influence of saline groundwater (Figure 3.1.1-5). All of the Thurne Broads have chloride concentration above 1000 mg l⁻¹ and are thus brackish in character, although the Martham broads have the lowest concentrations. The other broads typically have chloride concentrations around 100 mg l⁻¹ although higher values do occur in the broads positioned close to the middle or lower reaches of the rivers Ant and Bure as a result of tidal surges. These events were particularly marked during the low flow years of the mid 1990s and resulted in an elevation of chloride in the riverine broads of the Ant, Bure and Thurne, including the Martham Broads (Figure 3.1.1-6). The Trinity broads and Alderfen Broad are too far from the river to be influenced by surge tides, but all of these broads and Upton broad all show evidence of increases in average chloride concentrations during the years when river flow was low. The magnitude of these changes was small in the Trinity broads and in Upton Broad, but larger in Alderfen, Cockshoot and Cromes where, compared to the riverine broads, there is a lag in the chloride peak relative to the timing of low flows. The cause of this is unclear but may reflect temporary changes in the influence of groundwater.

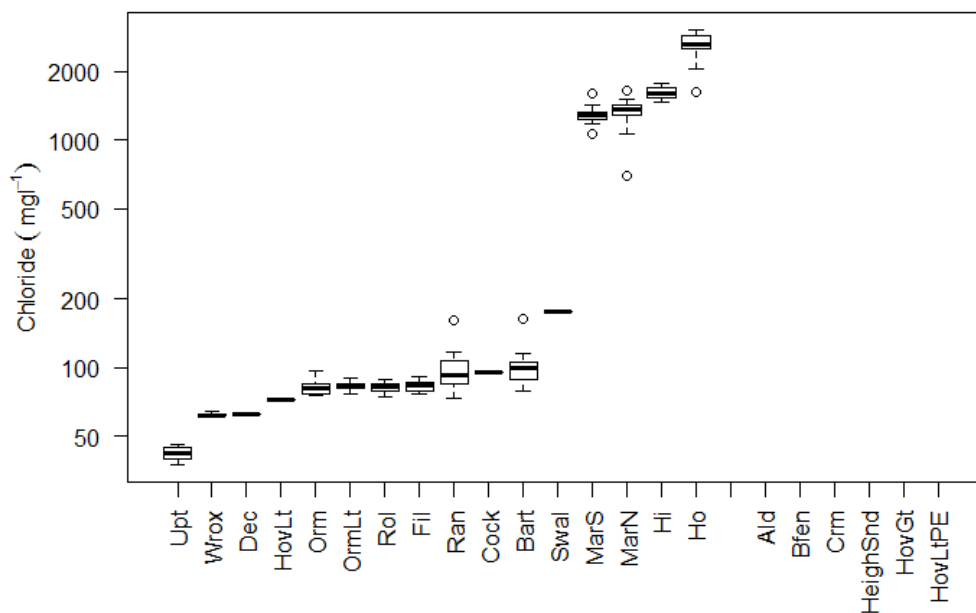


Figure 3.1.1-5 Range of annual geometric mean chloride concentration (mg l⁻¹) for monitored broads over the decade (2003-2012). The sites on the right lack sufficient data on chloride.

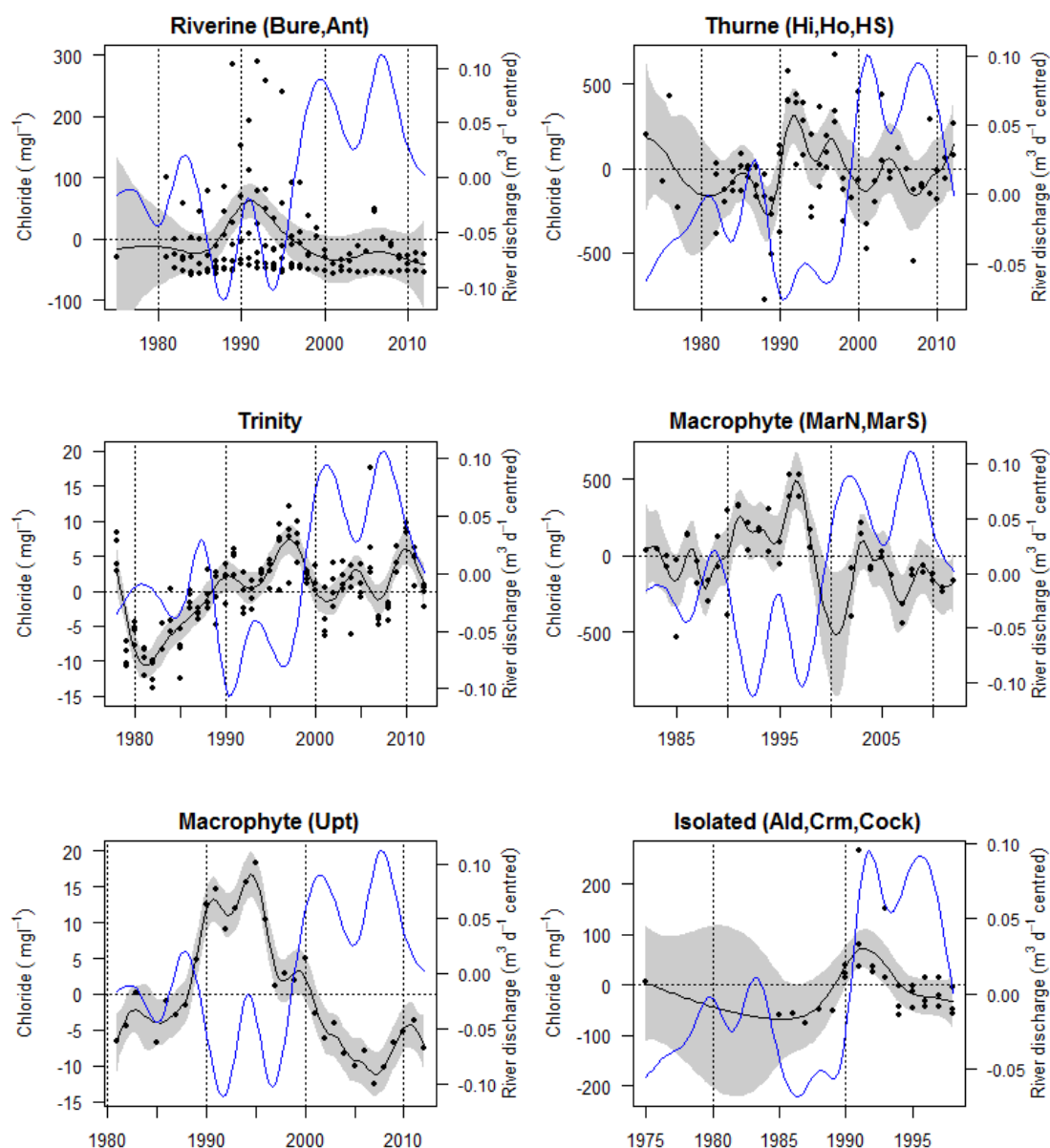


Figure 3.1.1-6 Trend of chloride concentration (black line) in different groups of broads compared to river discharge (blue line) a) riverine broads (Barton, Hoveton Great & Little, Ranworth South Walsham); b) Thurne broads (Hickling, Horsey, Heigham Sound); c) Trinity broads; e) Macrophyte dominated Thurne broads (Martham N & S), f) Small isolated broads (Alderfen, Cromes N, Cockshoot). Lines are GAM smooths of annual mean data, values expressed relative to the average for the period of data available for each broad. Black dots indicate yr x Broad annual mean chloride concentrations.

3.1.1.3 Nitrate (Total Oxidised Nitrogen Nitrate + Nitrite)

Although phosphorus is generally considered the most important nutrient responsible for eutrophication in freshwater, nitrogen is also an essential nutrient that can limit algal production and potentially macrophyte production. Available nitrogen is conveniently measured as Total Oxidised Nitrogen (TON), the sum of nitrate and nitrite, the latter nearly always only present in negligible amounts, as nitrite is rapidly oxidised to nitrate under aerobic conditions. Unfortunately TON has not been measured consistently in many of the broads and thus data available for assessing long-term changes are more limited, although clear general patterns can be seen.

In contrast to phosphorus, TON concentration in the Broads has undergone relatively minor change (Figure 3.1.1-5). In the riverine broads there has been a slight, but in most broads significant, increase in concentration. Fluctuations in the long-term trend are clearly related to river discharge, with increases in TON associated with increased river flows (Figure 3.1.1-5 blue line). Thus, as would be expected from other studies, it is clear that nitrate in the broads comes from diffuse rather than point sources as the nitrate concentration increases with run-off, rather than being diluted. As nitrate is rapidly assimilated by algae and plants during the growing season the best estimate of supply is provided by the mean winter (November – February) concentration (Figure 3.1.1-6). The highest median winter TON concentration (6.2 mgN l^{-1}) occurred in Wroxham Broad, a value significantly higher than the equivalent value for Barton Broad (2.2 mgN l^{-1}). This may be a result of different land use or simply reflect the larger catchment of the River Bure in comparison to the River Ant. The concentration of winter TON in the other broads reflects their proximity to the main river and size of catchments, with the riverine broads having concentrations around 2.0 mgN l^{-1} while the isolated broads have values less than 1.0 mgN l^{-1} .

Summer concentrations are less variable (Figure 3.1.1-7) with the majority of broads having a mean value of less than 0.1 mgN l^{-1} . Only the broads linked to the Rivers Ant and Bure have higher values, reflecting the influence of the nitrogen enriched river water.

In contrast to the Ant and Bure broads the Thurne, Trinity and macrophyte dominated broads (Marthams and Upton) had a decreasing long-term trend of TON (Figure 3.1.1-5d-e), although the influence of river discharge is still clear, with an increase of TON concentration during period of higher flow. Changes in mean TON concentration are a result of changes in supply and uptake. Thus the general decline in these broads is likely to reflect a greater change in uptake rate relative to supply, as they have smaller freshwater catchments, although from the limited temporal coverage of the TON data set it is difficult to identify reasons more fully.

However, it is clear that mean summer concentrations (June – October) are very low in many of the broads (Figure 3.1.1-7). The lowest summer concentrations are found in the broads not linked to the main river systems or in those at the bottom of the tidal river (South Walsham) or the tidal Thurne broads which have small freshwater catchments. This reflects the seasonal pattern of TON where concentrations reduce rapidly during the early spring (Figure 3.1.1-8). The average rate of reduction is similar in all of the broads, but as the riverine broads have a higher supply and thus higher winter concentration the lowest concentrations in these broads only occur for

a relatively short period around July when external inputs are lowest. In contrast the isolated broads or those with smaller catchments can have very low concentrations from as early as May, extending on until October or even November. These low concentrations can have important implications as nitrogen can potentially limit algal production, while macrophytes may still be able to exploit nitrogen via the sediment giving them a potential competitive advantage. Nitrate is also an important factor controlling the oxidation state of surface sediments. It acts as an oxidising agent and while present is likely to prevent the production of reduced iron compounds (iron sulphide) which in turn will keep phosphorus locked in surface sediments (Mitsch & Gosselink, 2000). Thus the potential for sediment P release increases as nitrate levels decrease. The increased TON observed in the riverine broads, particularly those nearer the top of the tidal river system will therefore have tended to decrease inputs of sediment derived P during the summer, while the decrease in TON in the isolated broads may have increased sediment P release.

Unfortunately the coverage of the TON data is insufficient to identify with certainty the factors influencing nitrate concentration. However, applying a mixed GAM model, which allows for both temporal dependency and spatial correlation, demonstrates that both river discharge and chlorophyll concentration are highly significant predictors of annual mean TON in a mixture of riverine and isolated broads (Figure 3.1.1-9 & Table 3.1.1-1)

The overall effect of the decrease in phosphorus and the increase in TON in the riverine broads has been to increase the N:P ratio, decreasing the likelihood of nutrient limitation by nitrogen. This has decreased the prevalence of cyanobacteria in the riverine broads, since cyanobacteria are at a competitive advantage in N-limited systems due to their ability to fix atmospheric nitrogen, while in the Thurne and isolated broads there is evidence of the opposite trend occurring, with a decreasing ratio and a greater likelihood of nitrogen limitation during the summer.

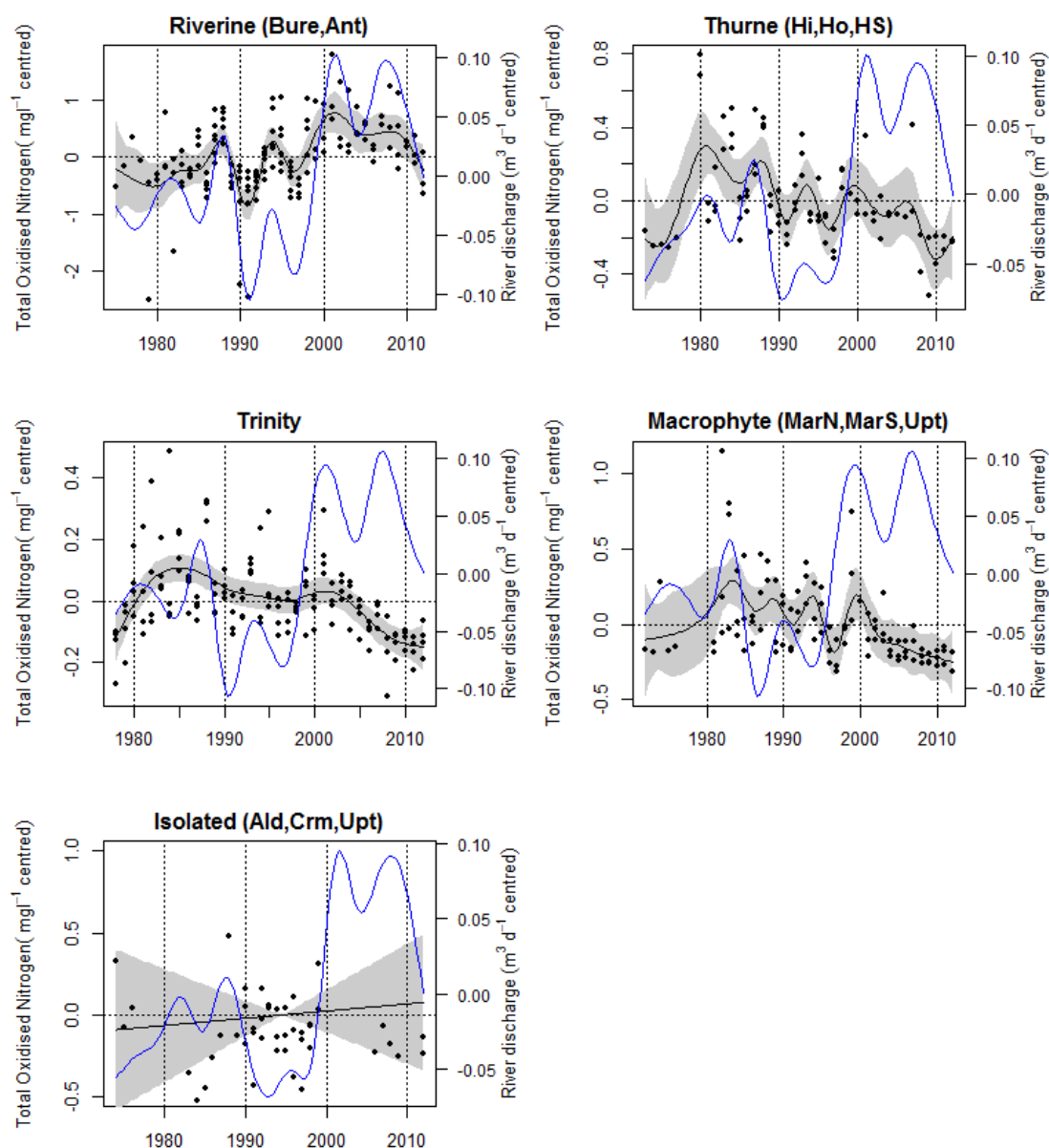


Figure 3.1.1-7 Trend of total oxidised nitrogen (black line) in different groups of broads compared to river discharge (blue line); a) riverine broads (Barton, Hoveton Great & Little, Ranworth South Walsham); b) Thurne broads (Hickling, Horsey, Heigham Sound); c) Trinity broads; d) Macrophyte dominated broads (Martham N & S, Upton); e) Small isolated broads (Alderfen, Cromes N, Cockshoot). Lines are GAM smooths of annual mean data, values expressed relative to the average for the period of data available for each broad.

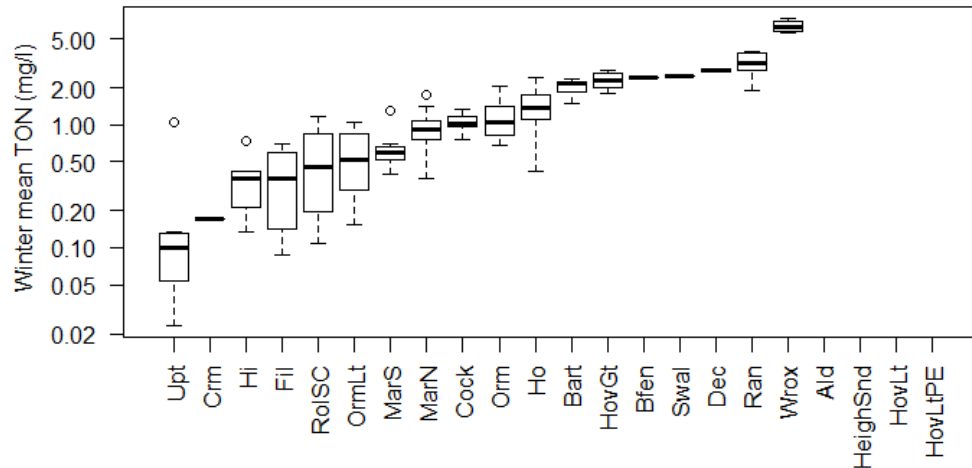


Figure 3.1.1-8 Range of winter (November – February) geometric mean total oxidised nitrogen concentration (mg l^{-1}) for monitored broads over the decade (2003-2012).

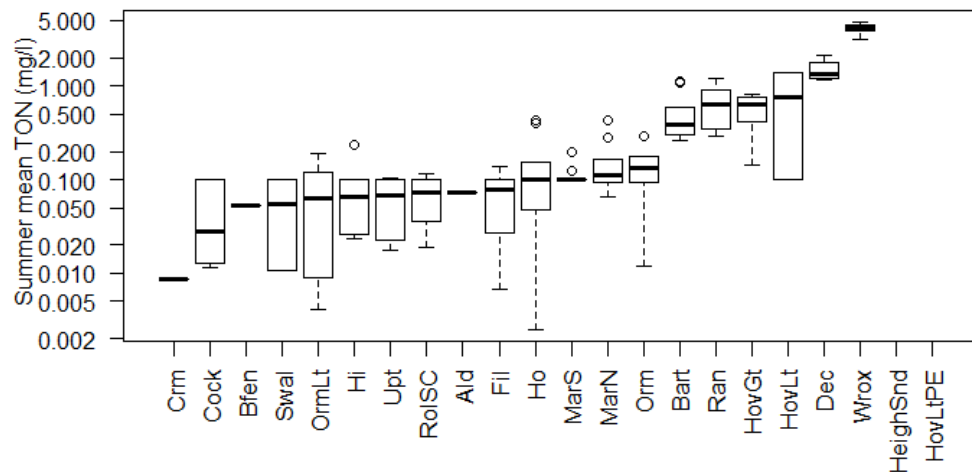


Figure 3.1.1-9 Range of summer (June-October) geometric mean total oxidised nitrogen concentration (mg l^{-1}) for monitored broads over the decade (2003-2012).

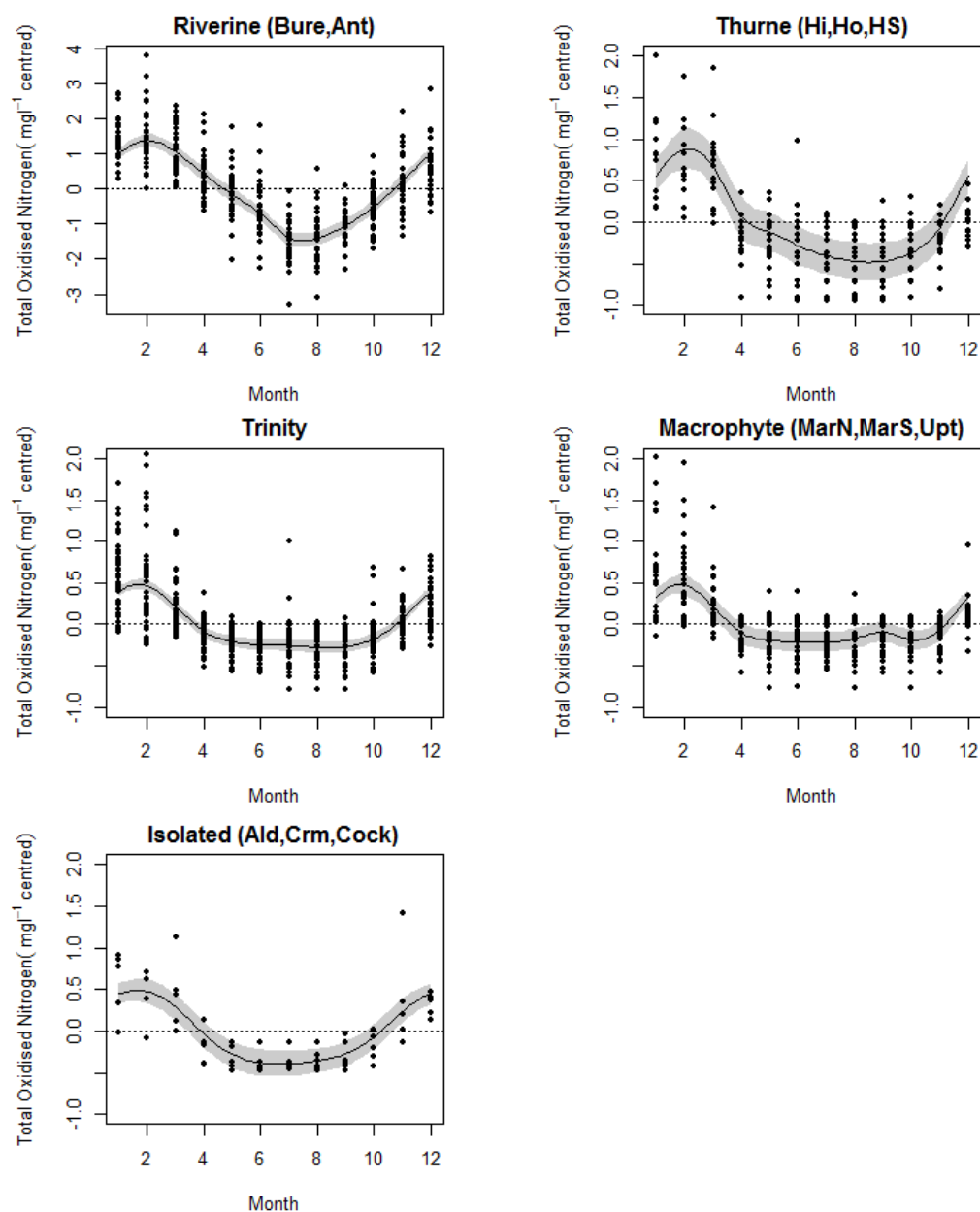


Figure 3.1.1-10 Seasonal pattern of total oxidised nitrogen for last decade, grouped by type of broad. Lines are GAM smooths of monthly mean data, values expressed relative to the average for the period (2003-2012).

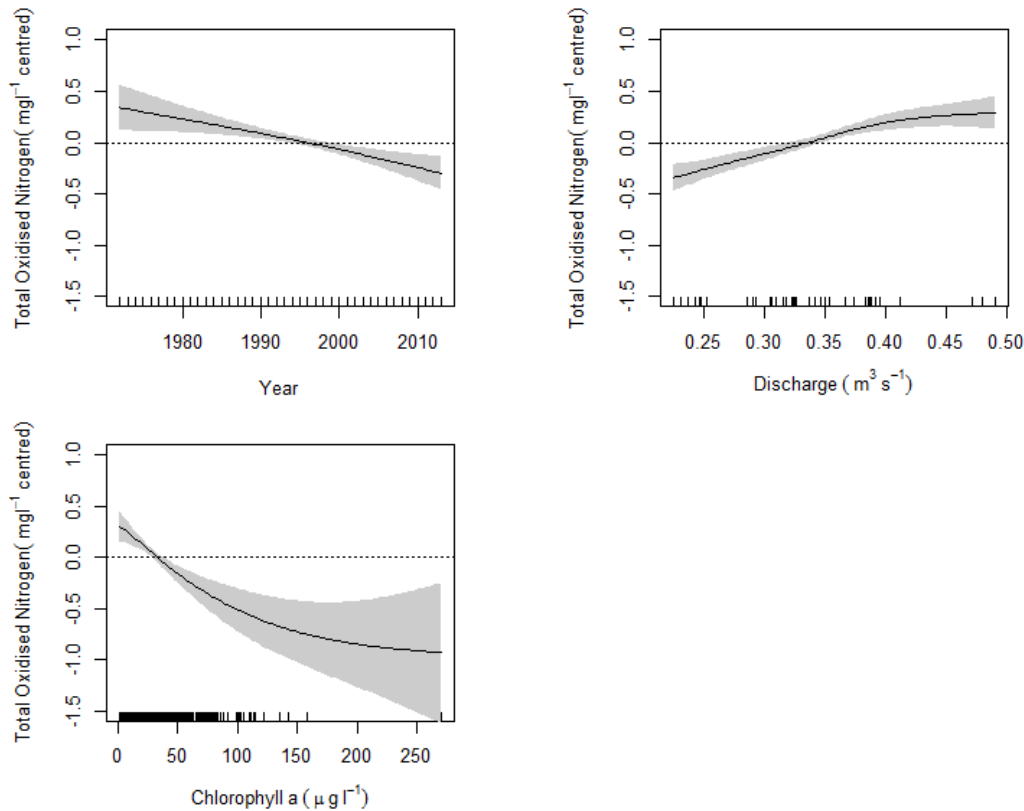


Figure 3.1.1-11 GAM model showing relationship of annual mean TON with Year, river discharge and mean annual chlorophyll a concentration for all broads with TON data covering the majority of the monitoring period. Values of TON expressed relative to the average for the period (2003-2012).

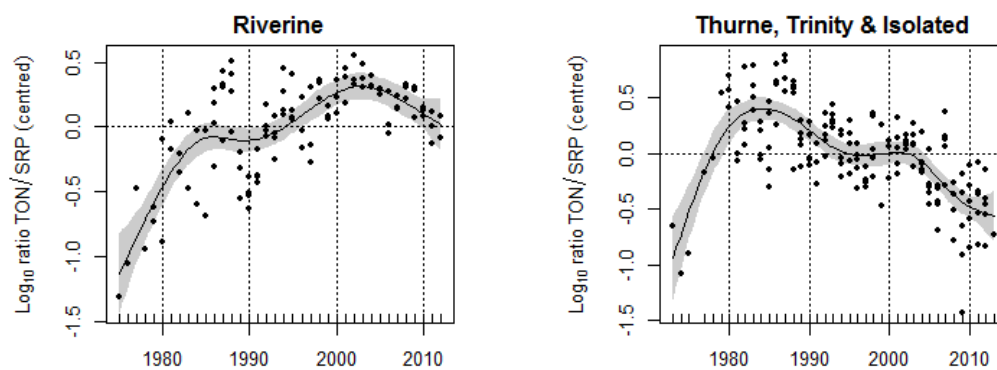


Figure 3.1.1-12 Trend of ratio of annual total oxidised nitrogen/soluble reactive phosphorus, grouped by riverine and isolated broads. Lines are GAM smooths of annual mean data, values expressed relative to the average for the period of data (1975-2012).

Table 3.1.1-2 Summary of GAM model showing relationship of annual mean TON with Year, river discharge and mean annual chlorophyll a concentration

Formula:

TON ~ s(Year, k = 9, bs = "cr") + s(AvgOfDisch_Hon, k = 7, bs = "cr") + s(Chl, k = 7, bs = "cr") + SHORT_NAME

Parametric coefficients:

	Estimate	Std. Error	t value	Pr(> t)	
(Intercept Bart)	1.1325	0.2406	4.707	3.74e-06	***
SHORT_NAMECock	-0.8647	0.3671	-2.356	0.019083	*
SHORT_NAMEHi	-0.6669	0.3337	-1.998	0.046521	*
SHORT_NAMEHo	-0.6020	0.3412	-1.764	0.078624	.
SHORT_NAMEHovGt	0.3470	0.3580	0.969	0.333108	
SHORT_NAMEMarN	-0.8187	0.3484	-2.350	0.019377	*
SHORT_NAMEMarS	-0.8640	0.3511	-2.461	0.014371	*
SHORT_NAMERan	0.2448	0.3428	0.714	0.475616	
SHORT_NAMETrin	-0.9449	0.3407	-2.773	0.005871	**
SHORT_NAMEUpt	-1.1985	0.3509	-3.416	0.000717	***
SHORT_NAMEWrox	3.5983	0.3392	10.609	< 2e-16	***

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value	
s(Year)	1.261	1.261	12.83	0.00017	***
s(AvgOfDisch_Hon)	2.460	2.460	14.57	1.63e-07	***
s(Chl)	2.222	2.222	11.90	5.66e-06	***

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

R-sq.(adj) = 0.91

Scale est. = 0.1893 n = 341

3.1.1.4 Chlorophyll a

As was the case for phosphorus, chlorophyll concentrations in the Broads over the last decade (2003-2012) have shown a wide range of variation (Figure 3.1.1-13) although the number of broads in each of the WFD classes is different from the distribution observed for TP. For chlorophyll only Upton Broad has had low enough chlorophyll to be consistently in High status, although the majority of isolated broads would have fallen into the Good status class for most of the last ten years. The riverine broads, including those in the River Thurne, were only moderate or poor status, with many close to the moderate/poor boundary, and those with the highest concentrations either at the lower end of the tidal river (South Walsham) or least influenced by riverine flushing (Hoveton Great, Hoveton Little, Hickling).

Chlorophyll a concentration has generally decreased since the late 1970s (Figure 3.1.1-14). The reduction is most marked in the riverine broads of the Rivers Ant and Bure, where the greatest reduction in TP occurred (see 3.1.1.1), although significant reductions have also occurred in the Trinity Broads and the larger Thurne Broads which have direct connections to the river (Table 3.1.1-3). There have been much smaller changes in the isolated broads with no significant long-term change in Upton, Alderfen or Cromes broads. There has been a slight, but significant long term increase in concentration in Martham South and North broads. Over the last decade (2003-2012) most broads have shown an increase followed by a decrease with no significant general trend.

As reported for many lakes (Phillips *et al.*, 2008) chlorophyll and total phosphorus concentrations are significantly linearly related in the riverine Ant, Bure and Thurne broads ($r^2 = 0.63$ $p < 0.001$). Thus in these broads the decline in chlorophyll is a result of the reduction in phosphorus. In comparison to other European lakes the yield of chlorophyll per unit of phosphorus in these broads is relatively high with values close to the upper confidence limits of the European relationship (Figure 3.1.1-16ab). In contrast the chlorophyll concentration in the broads that have been consistently dominated by macrophytes (Upton, Martham North, Martham South) not only have lower absolute chlorophyll and TP concentrations but also have relatively lower chlorophyll in comparison to TP with values much closer to the regression predicted from European lakes (green points in Figure 3.1.1-16a). Fitting a non-linear model (GAM) to data from both the riverine and macrophyte dominated lakes suggests a more rapid decline of chlorophyll relative to phosphorus below a TP concentration of about $60 \mu\text{g l}^{-1}$ which is the typical WFD good/moderate boundary for the Broad. The relationship between TP and chlorophyll in the Trinity Broad and other isolated broads is clearly not significant (Figure 3.1.1-16) confirming that at least in some years phytoplankton yield in these systems is not controlled by phosphorus.

These differences can be seen clearly when the results are expressed as the ratio of chlorophyll/TP. All of the isolated broads have values below 0.4 (Figure 3.1.1-22) while the riverine and saline influenced broads of the Thurne have higher values. This is most likely a result of zooplankton grazing, as the isolated broads are all lakes where planktivorous fish density have been reduced, either as a result of natural fish kills or by management intervention (see 4.3.). The importance of zooplankton grazing is confirmed in the Trinity Broad, where good data on zooplankton are available and it is shown that the chlorophyll/TP relationship is significantly related to density of large bodied cladoceran zooplankton ($r^2 = 0.87$ $p < 0.001$ see Annex, Trinity dossier, section.10).

There have been no significant long-term trends in the ratio of chlorophyll/TP in the majority of the broads, the exception being the Trinity Broad where there was a marked decline in the late 1980s (Figure 3.1.1-23). The cause of this change is unknown, but given the significant relationship of this ratio with cladoceran grazers and that large cladocera are themselves influenced by planktivorous fish it is very likely that this was caused by a change in the fish population.

The seasonal pattern of chlorophyll in the Broad provides further evidence of differences between groups of broads. Using monthly mean data, averaged over the last decade (2003-2012) three distinct seasonal patterns were found (Figure 3.1.1-15). A unimodal summer peak, a bimodal spring and summer peak and a unimodal winter/early spring peak. The first pattern occurs in the riverine Ant and Bure broads and in Horsey Mere. These broads have their maximum chlorophyll concentration in the summer with a peak in August and have only a relatively small or undetectable spring peak (Figure 3.1.1-15ab). The second pattern is seen in the Trinity broads, which have a more typical bimodal chlorophyll distribution with similar magnitude spring and summer peaks occurring in February and August, separated by a clear water period during April and May. The third, was found in the small isolated broads, in Hickling and in the macrophyte dominated broads (Upton, Martham South, and Martham North). The phytoplankton in these broads lacked a

summer peak and the maximum chlorophyll occurred in early spring or winter. This then declined relatively slowly during spring and early summer, often not reaching the lowest chlorophyll concentration until July or August.

These quite distinct seasonal patterns reflect the rather different conditions found in these groups of broads. The riverine broads and Horsey Mere all experience significant hydraulic flushing during the winter and early spring which is likely to limit phytoplankton abundance during winter and potentially reduce the spring phytoplankton. In the other non-riverine broads discharge is much less likely to influence phytoplankton and thus they develop a very early spring phytoplankton which peaks in February or March. Modelling confirms the potential importance of river discharge in the riverine broads, although the relationship with this is weak in comparison to the other significant predictors: TP, chloride concentration and temperature (Figure 3.1.1-18 & Table 3.1.1-4; Figure 3.1.1-19 & Table 3.1.1-5). In the riverine broads chlorophyll concentration increases downstream as a result of increased effective water retention time and the influence of chloride is a reflection of upstream tidal water movement during surge tidal events. The model finding that spring chlorophyll declines significantly with increased temperature was unexpected as increased temperature is generally assumed to result in an increase in phytoplankton. In the broads the decrease is likely to be a result of increased grazing rates, as illustrated by the significant relationship ($r^2 = 0.76$ $p=0.01$) between spring chlorophyll minimum temperature and large cladocera abundance (see 6.26.10)

During the summer, chlorophyll was significantly related to TP in all of the groups of broads, but different other variables were also found to be significant. In the riverine broads temperature was marginally significant (Figure 3.1.1-19 & Table 3.1.1-6), in the isolated broads TON and river discharge were significant (Figure 3.1.1-20 & Table 3.1.1-7) and in the Trinity Broad non-algal plant cover was significant (Figure 3.1.1-21 & Table 3.1.1-8). These results suggest that during the summer zooplankton grazing is less significant in controlling phytoplankton abundance. In the riverine broads macrophyte cover is very low (Figure 3.1.2-1) and summer chlorophyll is high (Figure 3.1.1-15ab) controlled primarily by TP concentration. In the isolated broads, where macrophyte cover is relatively high (Figure 3.1.2-1) the variation in summer chlorophyll is more likely controlled by a combination of phosphorus, nitrogen availability and flushing. In the Trinity Broad where macrophyte abundance over the last decade has been more variable, summer chlorophyll is more likely to be influenced by phosphorus and macrophyte abundance (Figure 3.1.1-21).

Interpretation of these models has been complicated by the lack of consistent data for all the metrics that are likely to influence phytoplankton growth. However, in summary it is clear that the amount of phytoplankton in the riverine broads is primarily determined by phosphorus, with individual sites being influenced by water tidal water movement in some years. In the isolated lakes, zooplankton grazing has a significant influence on the spring phytoplankton and in warmer years this effect is greater. Thus in those broads where large cladocera have become abundant as a result of planktivorous fish manipulation the warmer conditions since 1989 have combined to further reduce spring chlorophyll (Figure 3.1.1-22). During the summer, zooplankton grazing appears less important, although increased macrophyte cover is

associated with lower phytoplankton and this may be due to zooplankton grazers within the macrophyte beds that are not adequately sampled during routine monitoring.

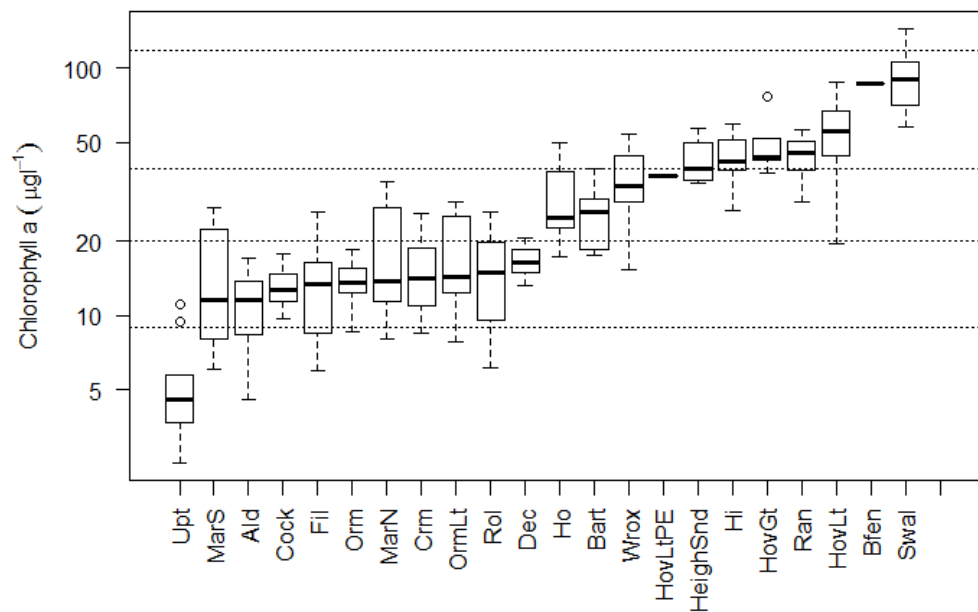


Figure 3.1.1-13 Range of annual geometric mean chlorophyll a concentration ($\mu\text{g l}^{-1}$) for monitored broads over the decade (2003-2012). Horizontal lines mark Water Framework Directive boundary values (Bad/Poor, Moderate/Poor, Good/Moderate, High/Good)

Table 3.1.1-3 Mann Kendall test for monotonic trend of mean annual chlorophyll a time series for monitored broads

	Mann Kendall Test			
	Tau	2-sided pvalue		
Barton	-0.70	0.000	***	1972-2012
Hoveton Great	-0.62	0.004	**	1983-2012
Hickling	-0.60	0.000	***	1972-2012
Filby	-0.58	0.000	***	1978-2012
Ranworth	-0.57	0.000	***	1983-2012
Hoveton Little	-0.54	0.001	***	1983-2012
Rollesby	-0.53	0.000	***	1978-2012
Ormesby Little	-0.47	0.000	***	1978-2012
Ormesby	-0.44	0.000	***	1978-2012
Cockshoot	-0.42	0.001	***	1979-2012
Wroxham	-0.40	0.001	***	1978-2012
S Walsham	-0.30	0.016	**	1978-2012
Cromes	-0.24	0.107	ns	1983-2012
Horsey	-0.06	0.609	ns	1978-2012
Alderfen	-0.06	0.595	ns	1971-2012
Upton	0.10	0.410	ns	1972-2012
Martham N	0.32	0.014	**	1981-2012
Marthan S	0.32	0.012	**	1982-2012

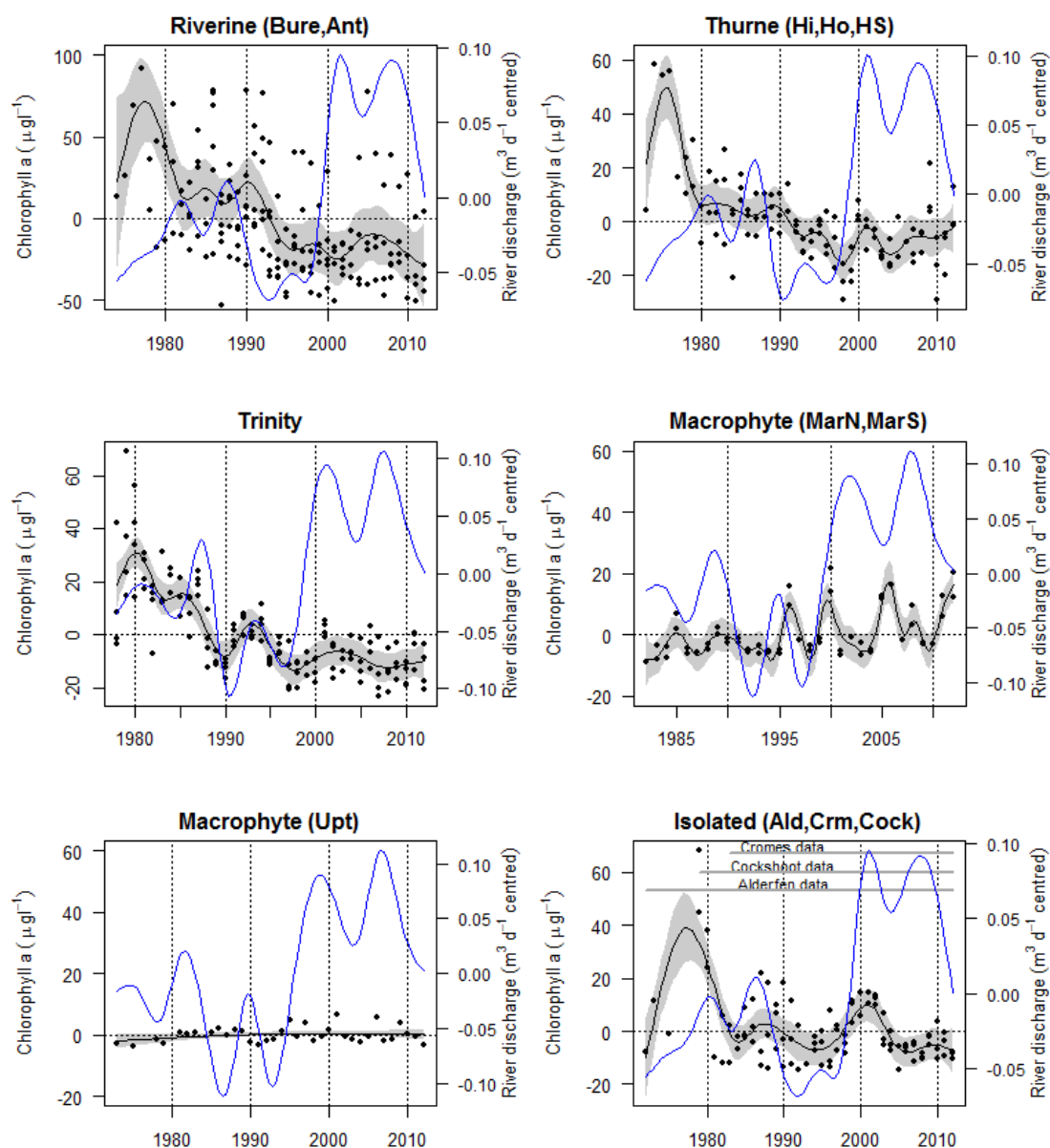


Figure 3.1.1-14 Trend of mean annual chlorophyll a concentration (black line) in different groups of broads compared to river discharge (blue line) a)riverine broads (Barton, Hoveton Great & Little, Ranworth South Walsham); b) Wroxham Broad, c) Thurne broads (Hickling, Horsey, Heigham Sound),d) Trinity broads; e) Macrophyte dominated broads (Martham N & S, Upton), f) Small isolated broads (Alderfen, Cromes N, Cockshott). Lines are GAM smooths of annual mean data, values expressed relative to the average for the period of data available for each broad.

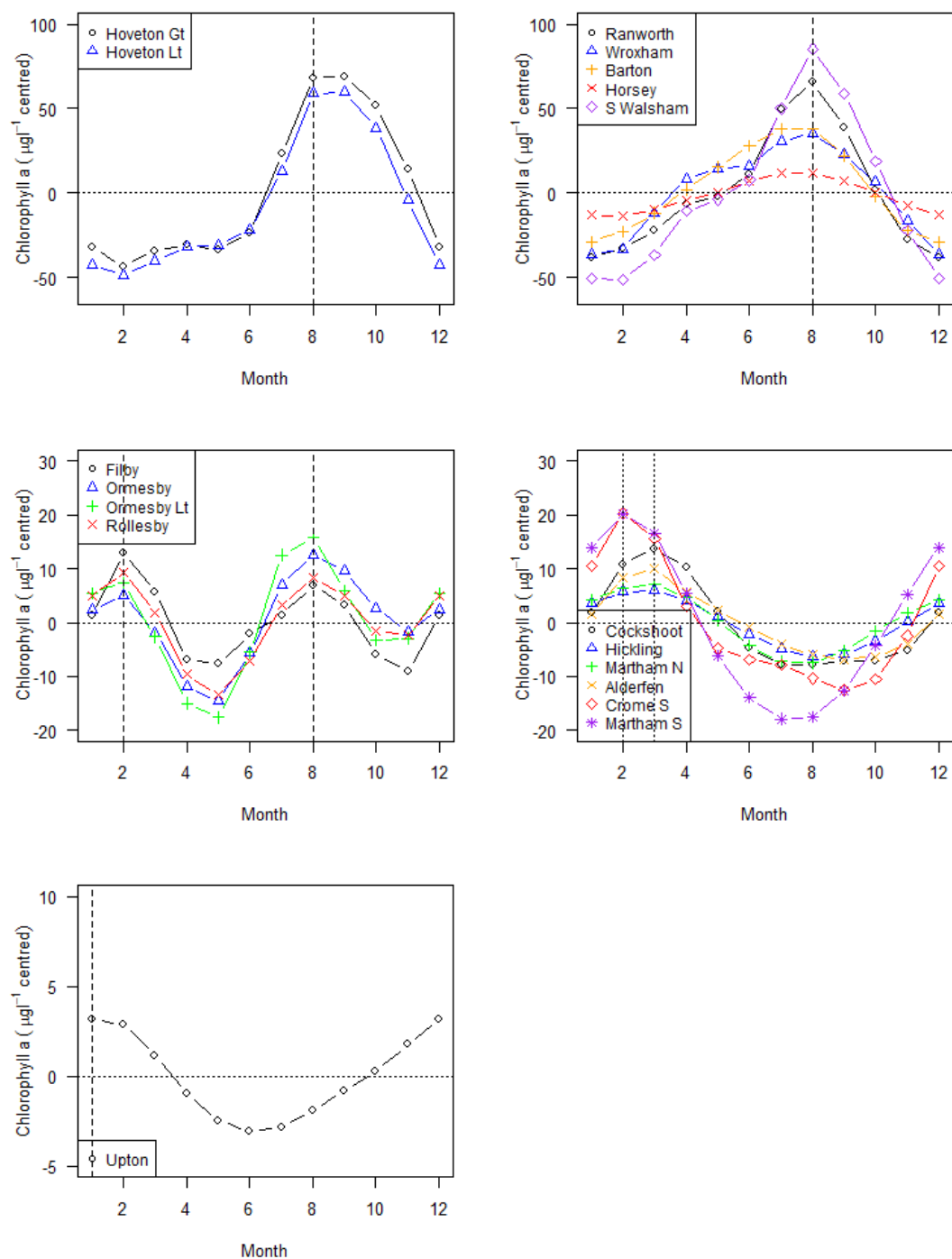


Figure 3.1.1-15 Seasonal pattern of chlorophyll a for last decade (2003-2012) (except Hoveton Great and Hoveton Little where all data used as too few samples in last decade), grouped by similarity. Lines are GAM smooths of monthly mean data, values expressed relative to the average for the period.

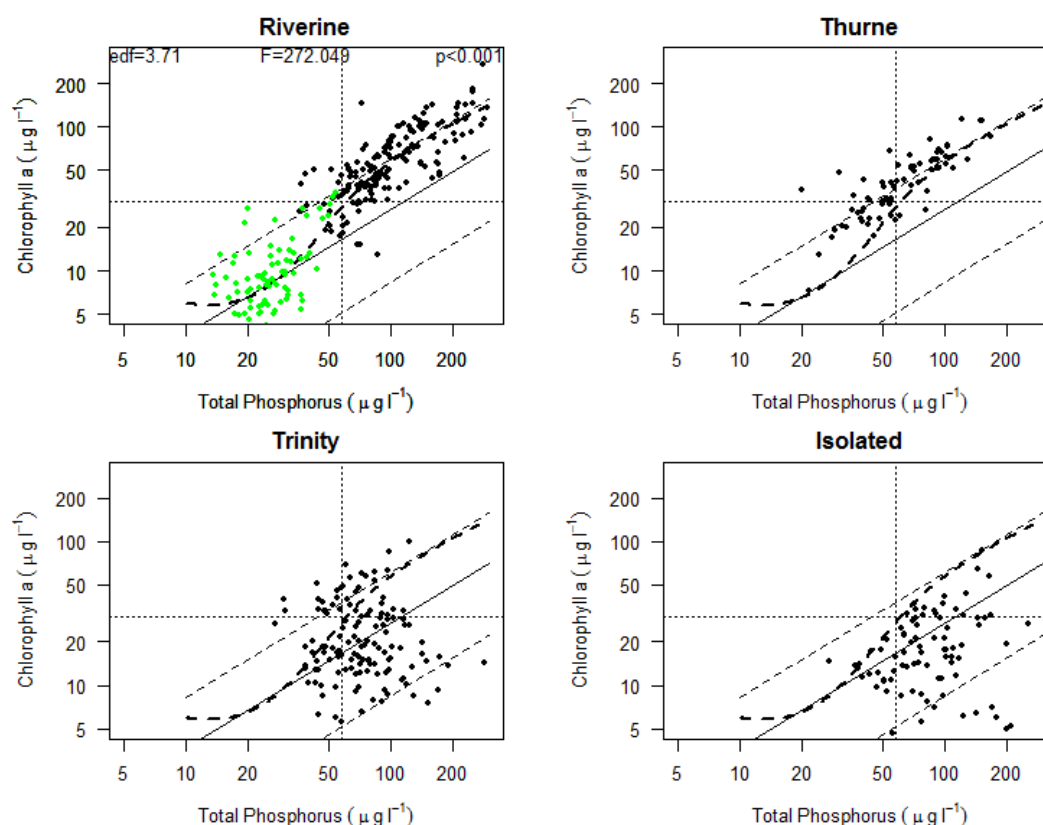


Figure 3.1.1-16 Relationship between mean annual chlorophyll a and total phosphorus concentration in a) Riverine broads (Barton, Wroxham, Hoveton Great, Hoveton Little, Ranworth, S Walsham) & macrophyte-dominated broads (Upton, Martham S Martham N); b) Thurne broads (Hickling, Horsey, Heigham Sound); c) Trinity broads (Ormesby, Rollesby, Ormesby Little, Filby); d) Isolated broads (Alderfen, Burntfen, Cockshoot, Cromes). Vertical and horizontal lines mark the WFD good/moderate boundaries for chlorophyll and total phosphorus, diagonal lines mark relationship from European Lakes (Phillips *et al.*, 2008), broken line shows GAM model fitted to riverine broads and macrophyte dominated broads $R^2=0.82$ $p<0.001$

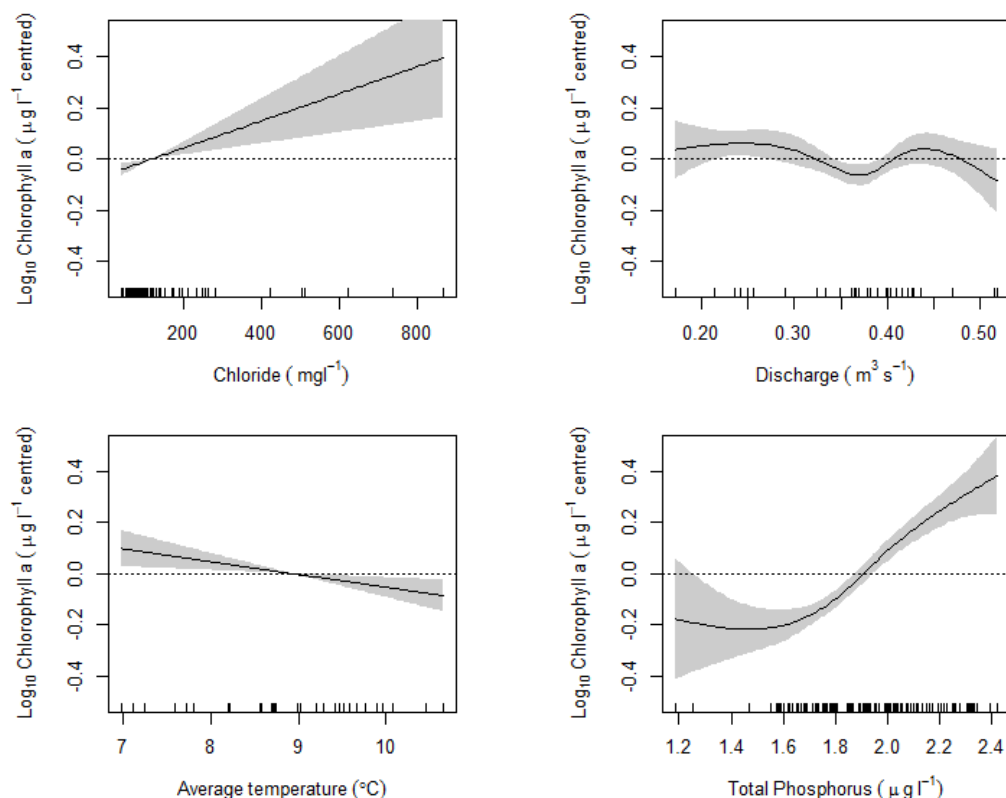


Figure 3.1.1-17 mixed GAM model showing relationship of spring mean chlorophyll a (\log_{10} transformed) with chloride concentration, river discharge, average monthly air temperature and mean total phosphorus concentration for riverine broads (Barton, Wroxham, Hoveton Great, Hoveton Little, Hoveton Great, S Walsham). Values of chlorophyll expressed relative to the average for the period (2003-2012)

Table 3.1.1-4 Detail of model shown in Figure 3.1.1-17

Parametric coefficients:

	Estimate	Std. Error	t value	Pr(> t)
(Intercept)	1.789283	0.097032	18.440	<2e-16 ***
SHORT_NAMEHovGt	-0.110583	0.144132	-0.767	0.445
SHORT_NAMEHovLt	-0.119178	0.140635	-0.847	0.399
SHORT_NAMERan	-0.005719	0.137924	-0.041	0.967
SHORT_NAMESwal	-0.034269	0.143855	-0.238	0.812
SHORT_NAMEWrox	0.032055	0.137361	0.233	0.816

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value
s(Cl)	1.000	1.000	11.482	0.000955 ***
s(Avg_DischHon_m3s)	3.605	3.605	3.453	0.013670 *
s(AvgOftavg)	1.000	1.000	7.796	0.006111 **
s(log10(TP))	3.186	3.186	24.808	1.65e-13 ***

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

R-sq.(adj) = 0.67

Scale est. = 0.026009 n = 130

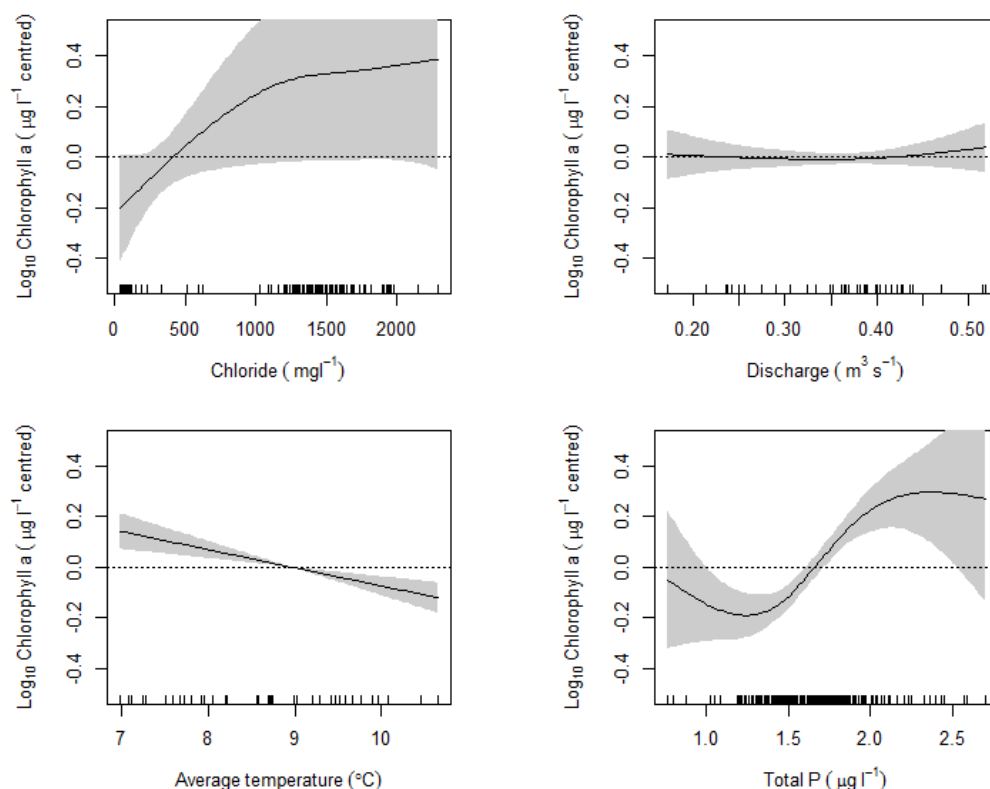


Figure 3.1.1-18 mixed GAM model showing relationship of spring mean chlorophyll a (log₁₀ transformed) with chloride concentration, river discharge, average monthly air temperature and mean total phosphorus concentration for Hickling, Trinity and isolated broads (Alderfen, Cockshoot, Cromes, Martham N & S, Upton). Values of chlorophyll expressed relative to the average for the period (2003-2012)

Table 3.1.1-5 Detail of model shown in Figure 3.1.1-18

Parametric coefficients:

	Estimate	Std. Error	t value	Pr(> t)	
(Intercept)	0.940801	0.218745	4.301	2.54e-05	***
SHORT_NAMECock	0.507783	0.267073	1.901	0.0585	.
SHORT_NAMECrn	0.522366	0.253969	2.057	0.0409	*
SHORT_NAMEHi	0.351481	0.358032	0.982	0.3273	
SHORT_NAMEMarN	0.029960	0.365907	0.082	0.9348	
SHORT_NAMEMarS	-0.004664	0.357727	-0.013	0.9896	
SHORT_NAMEOrm	0.288024	0.252091	1.143	0.2544	
SHORT_NAMEOrmLt	0.362801	0.252367	1.438	0.1519	
SHORT_NAMERol	0.342241	0.252036	1.358	0.1759	
SHORT_NAMEUpt	-0.060615	0.257278	-0.236	0.8140	

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value
s(Cl)	2.011	2.011	1.892	0.153
s(Avg_DischHon_m3s)	1.567	1.567	0.417	0.610
s(AvgOfavg)	1.000	1.000	16.153	7.90e-05 ***
s(log10(TP))	3.569	3.569	10.041	8.14e-07 ***

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

R-sq.(adj) = 0.655

Scale est. = 0.065879 n = 243

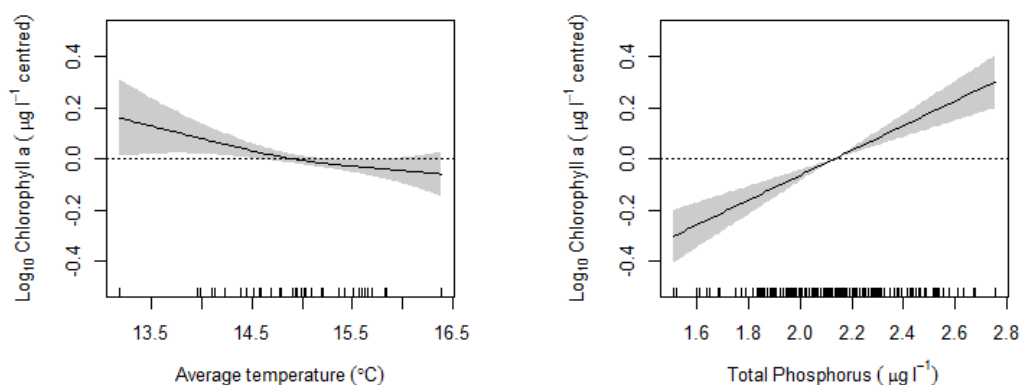


Figure 3.1.1-19 GAM model showing relationship of summer mean chlorophyll a (\log_{10} transformed) with average monthly air temperature and mean total phosphorus concentration for riverine broads (Barton, Wroxham, Hoveton Great, Hoveton Little, Hoveton Great, S Walsham). Values of chlorophyll expressed relative to the average for the period (2003-2012)

Table 3.1.1-6 Detail of model shown in Figure 3.1.1-19

Parametric coefficients:

	Estimate	Std. Error	t value	Pr(> t)
(Intercept)	1.98411	0.10572	18.767	<2e-16 ***
SHORT_NAMEHovGt	-0.08194	0.15974	-0.513	0.609
SHORT_NAMEHovLt	-0.01987	0.15209	-0.131	0.896
SHORT_NAMERan	0.02940	0.15260	0.193	0.847
SHORT_NAMESwal	0.04435	0.15203	0.292	0.771
SHORT_NAMEWrox	-0.12776	0.15073	-0.848	0.398

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value
s(AvgOftavg)	1.647	1.647	5.149	0.0116 *
s(log10(TP))	1.000	1.000	34.472	1.97e-08 ***

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

R-sq.(adj) = 0.465

Scale est. = 0.037926 n = 180

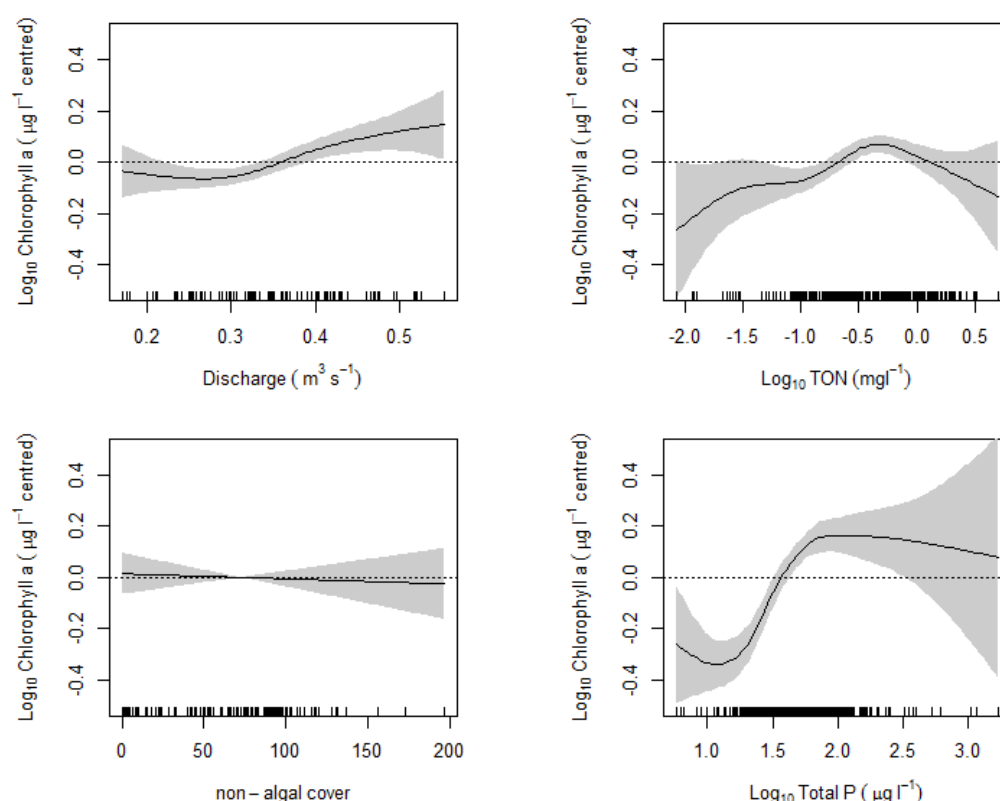


Figure 3.1.1-20 GAM model showing relationship of summer mean chlorophyll a (log₁₀ transformed) with river discharge, log₁₀ total oxidised nitrogen, non-algal cover and log₁₀ mean total phosphorus concentration for Hickling, Martham N & S and isolated broads (Alderfen, Cockshoot, Cromes). Values of chlorophyll expressed relative to the average for the period (2003-2012)

Table 3.1.1-7 Detail of model shown in Figure 3.1.1-20

Parametric coefficients:					
	Estimate	Std. Error	t value	Pr(> t)	
(Intercept)	0.8458	0.1760	4.804	2.37e-06	***
SHORT_NAMECock	0.1837	0.2046	0.898	0.369872	
SHORT_NAMECrn	0.3985	0.2020	1.973	0.049369	*
SHORT_NAMEHi	0.7314	0.2032	3.599	0.000368	***
SHORT_NAMEMarN	0.3754	0.2099	1.788	0.074626	.
SHORT_NAMEMarS	0.3001	0.2095	1.432	0.153002	

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1					
Approximate significance of smooth terms:					
	edf	Ref.df	F	p-value	
s(Avg_DischHon_m3s)	2.879	2.879	4.950	0.002761	**
s(log10(TON))	3.884	3.884	5.050	0.000711	***
s(sum.non.algal.cov)	1.000	1.000	0.132	0.717101	
s(log10(TP))	4.486	4.486	14.063	2.42e-11	***

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1					
R-sq.(adj) = 0.59					
Scale est. = 0.068308 n = 344					

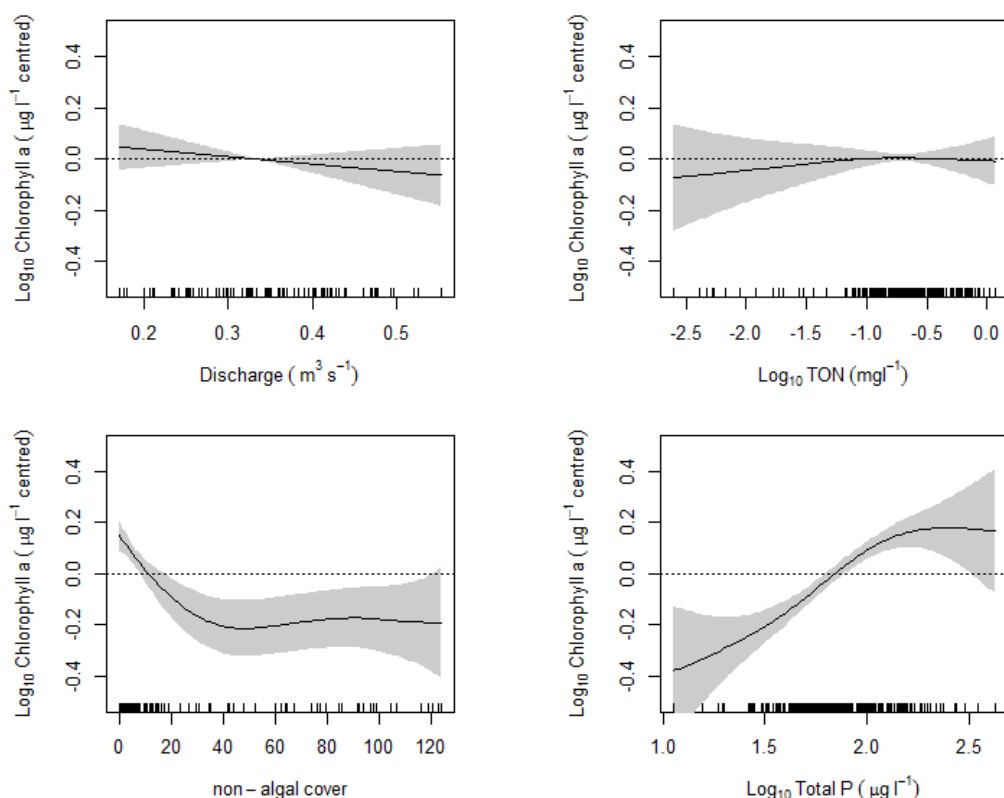


Figure 3.1.1-21 GAM model showing relationship of summer mean chlorophyll a (log₁₀ transformed) with river discharge, log₁₀ total oxidised nitrogen, non-algal plant cover and log₁₀ mean total phosphorus concentration for Trinity Broads. Values of chlorophyll expressed relative to the average for the period (2003-2012)

Table 3.1.1-8 Detail of model shown in Figure 3.1.1-21

Parametric coefficients:

	Estimate	Std. Error	t value	Pr(> t)
(Intercept)	1.1659	0.1064	10.958	<2e-16 ***
SHORT_NAMEorm	0.2776	0.1666	1.666	0.0969 .
SHORT_NAMEormLt	0.1098	0.1496	0.734	0.4638
SHORT_NAMEorl	0.1158	0.1503	0.770	0.4418

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value
s(Avg_DischHon_m3s)	1.000	1.000	1.116	0.292
s(log10(TON))	1.511	1.511	0.464	0.575
s(sum.non.algal.cov)	3.203	3.203	8.737	1.10e-05 ***
s(log10(TP))	2.747	2.747	19.067	3.34e-10 ***

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

R-sq.(adj) = 0.25

Scale est. = 0.078891 n = 254

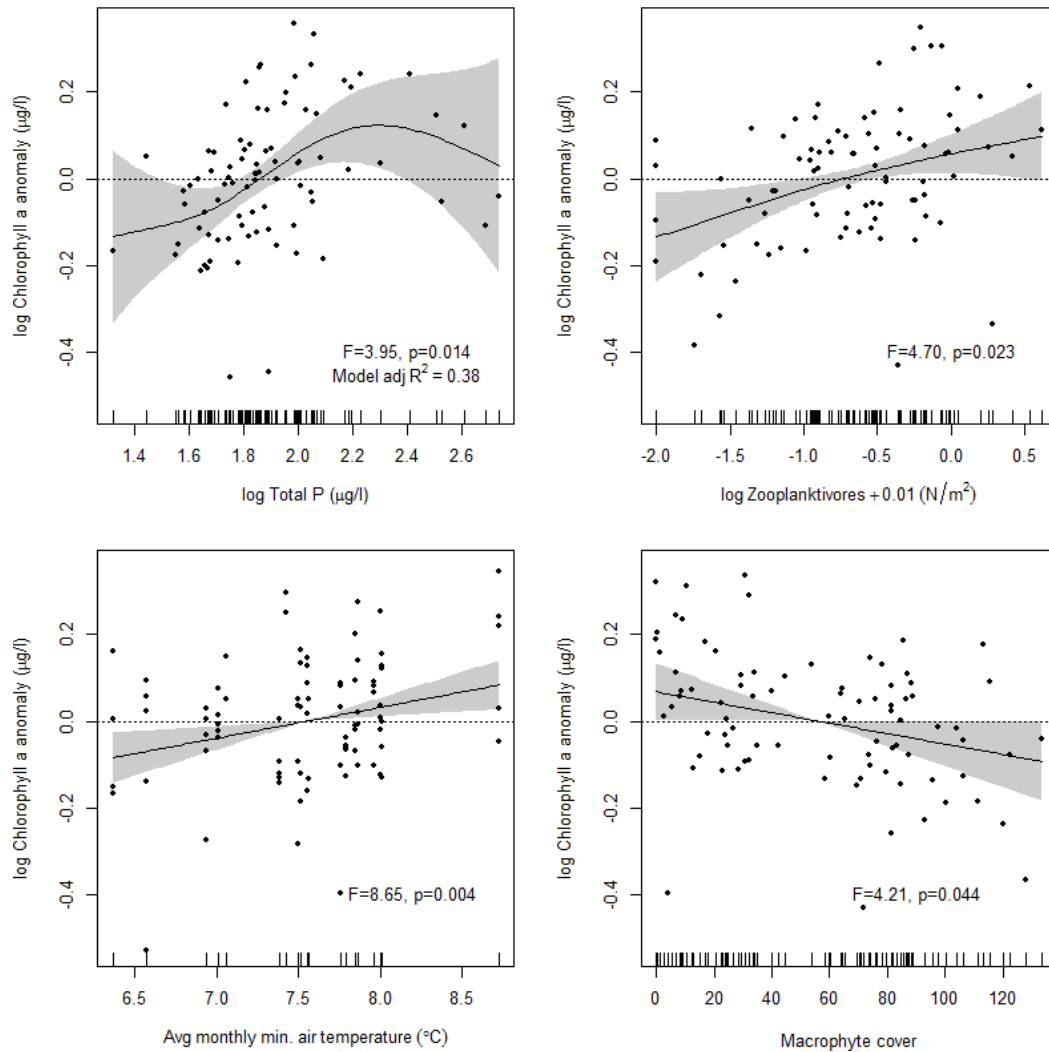


Figure 3.1.1-22 Relationship between annual mean chlorophyll a, total phosphorus, zooplanktivorous fish density, temperature and macrophyte cover in 6 isolated or semi-isolated broads.

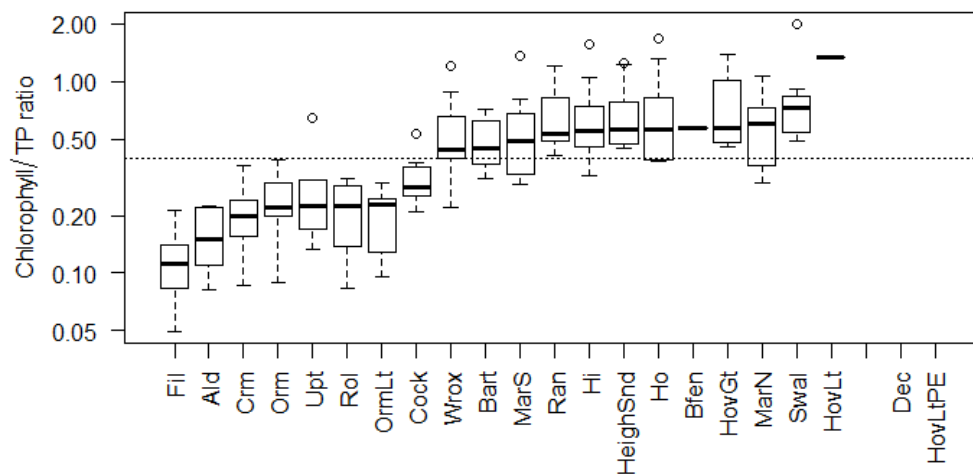


Figure 3.1.1-23 Range of ratio of annual mean chlorophyll a / total phosphorus concentration for monitored broads over the decade (2003-2012). Horizontal line at 0.4 indicates approximate separation of low yielding (probably top down controlled) from higher yielding (nutrient controlled) systems.

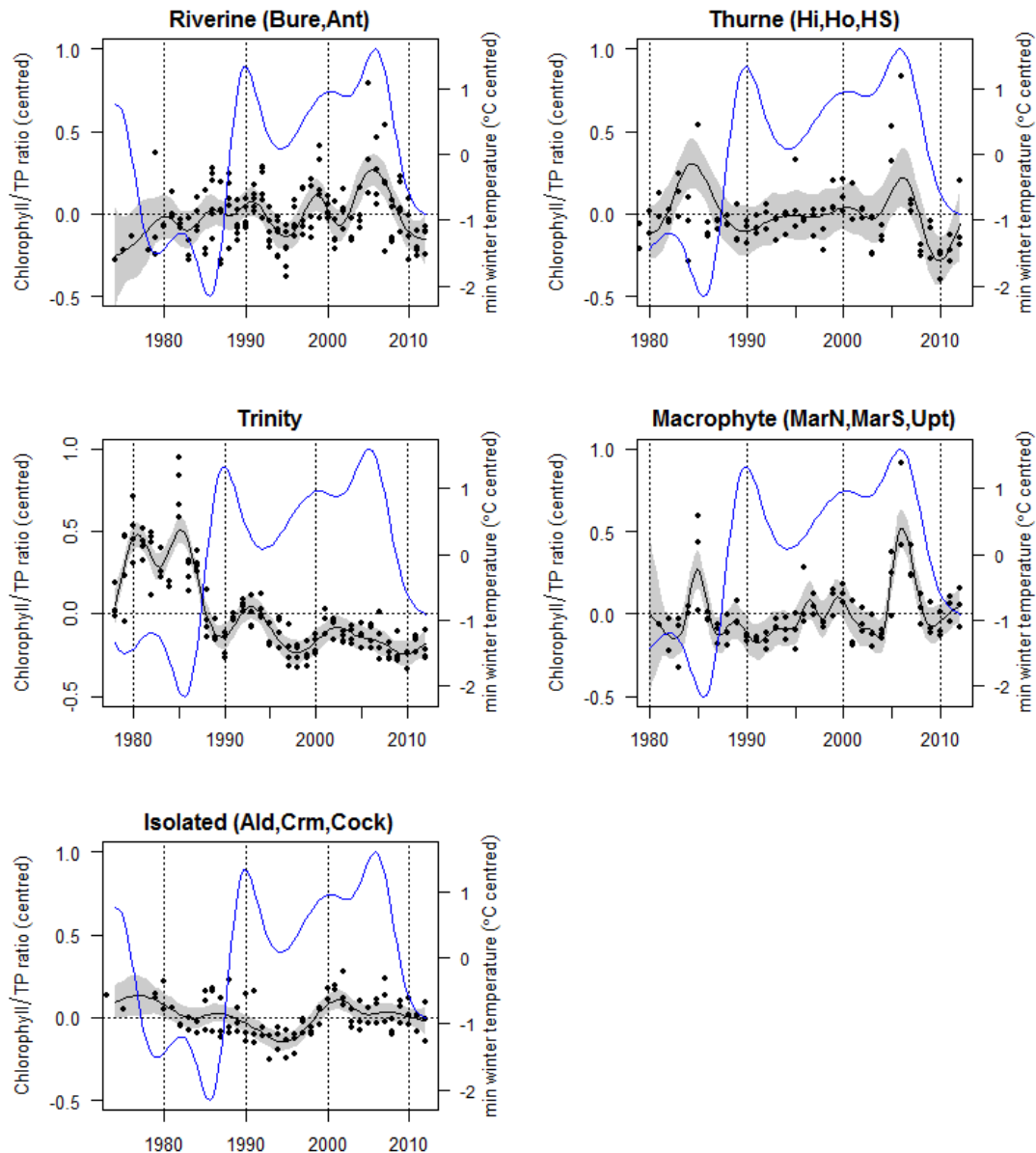


Figure 3.1.1-24 Trend of ratio of mean annual chlorophyll a/total phosphorus concentration (black line) in different groups of broads in relation to winter temperatures (blue line): a) riverine broads (Barton, Hoveton Great & Little, Ranworth South Walsham); b) Thurne broads (Hickling, Horsey, Heigham Sound); c) Trinity broads; d) macrophyte dominated broads (Martham N & S, Upton); e) small isolated broads (Alderfen, Cromes N, Cockshoot). Lines are GAM smooths of annual mean data, values expressed relative to the average for the period of data available for each broad.

3.1.2 Macrophytes

3.1.2.1 Macrophyte cover

Figure 3.1.2-1 summarises differences in macrophyte cover of the broads based on surveys undertaken during the last decade. For the purpose of presenting and discussing macrophyte cover we refer to non-algal cover (i.e. the cover of all

macrophytic species including charophytes, but excluding filamentous species such as *Cladophora*, *Enteromorpha* and *Hydrodictyon*). Various targets, ranging between 30-50% cover of macrophytes, have been proposed to achieve stable clear-water conditions and restore macrophyte-dependent ecosystem functions in shallow lakes (Hilt *et al.*, 2013; De Backer *et al.*, 2012). It is clear from Figure 3.1.2-1 that of the regularly-surveyed sites only a rather small number of isolated broads (e.g. Upton Little, Cromes South) and end of system broads (Martham North and South) consistently achieve this target. At the opposite extreme are a group dominated by riverine broads that never achieve this target (e.g. Wroxham and Barton). In between lie a range of broads covering a variety of types and characteristics that are either occasionally or regularly, but not permanently ‘macrophyte-dominated’.

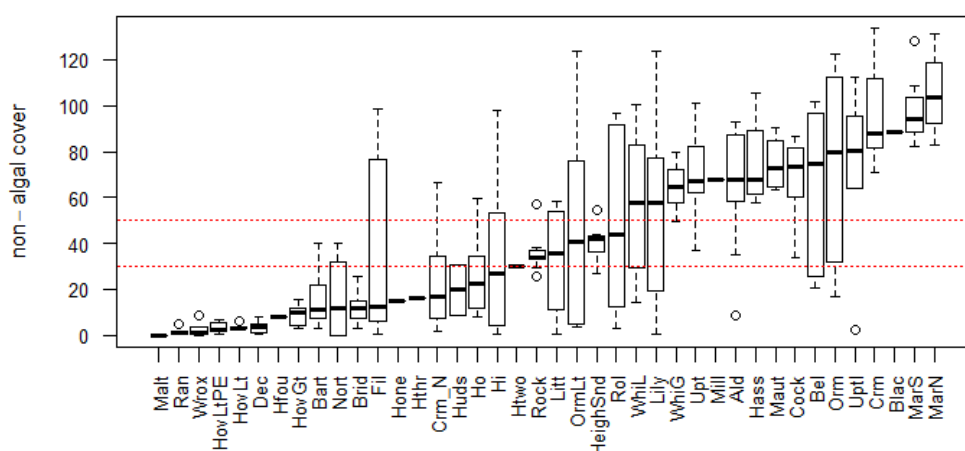


Figure 3.1.2-1 Range of non-algal macrophyte cover for surveyed broads over the decade (2003-2012). Red lines mark potential minimum critical cover for longer term stability

Over the Broad as a whole over the last 30 years there has been an upward trend in average cover (see section 3.2). However, this pattern differs strongly between different types of broads (Figure 3.1.2-2). Thus, in the isolated broads (including the Trinity Broad) cover has increased, with this increase being strongest since 2002. These are the sites most influenced by management in the form of sediment removal and biomanipulation, while the pattern of inter-annual variation in cover is suggestive of additional positive climatic influences on plant growth expressed through sunshine hours and, in dry years, lower lake levels. By contrast in the riverine broads cover has remained persistently low for the last 30 years, the major exception to this being the Thurne Broad. Here however, although cover has been consistently high in the Martham system there has been a major decline overall due to the collapse in cover of plants in Hickling Broad and to a lesser extent Heigham Sound since 2000.

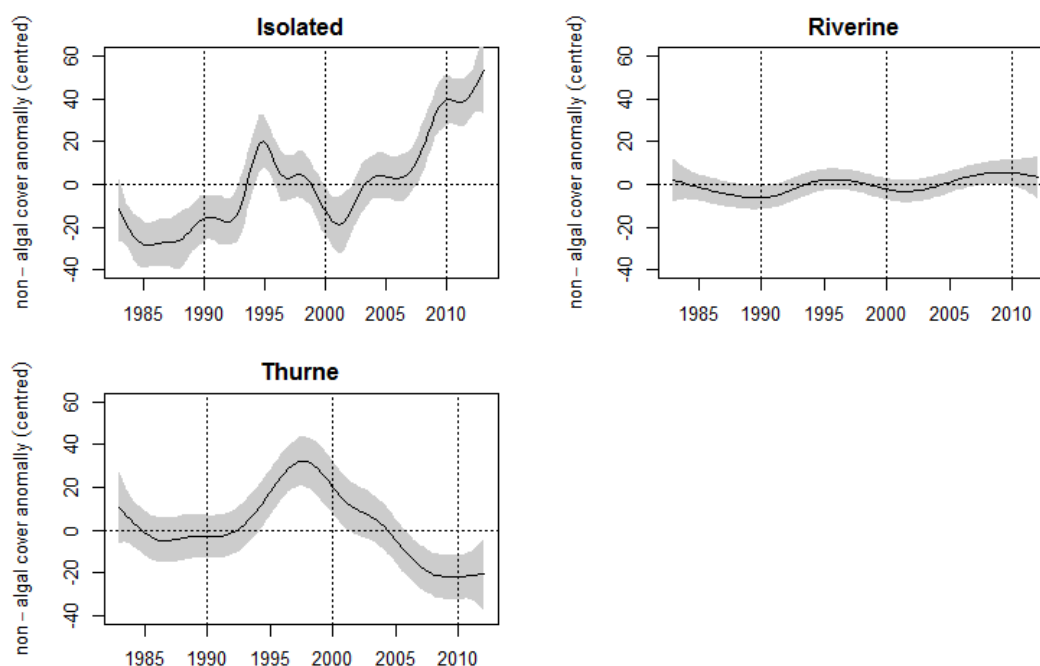


Figure 3.1.2-2 Trend of non-algal macrophyte cover in different groups of broads; a) Isolated broads, including Trinity Broad; b) Riverine Broad on Bure and Ant; c) Thurne Broad. Lines are GAM smooths with values expressed relative to the average for the period of data available for each group of broads.

3.1.2.2 *Macrophyte richness*

Figure 3.1.2-3 illustrates the range of variation in macrophyte richness in the broads over the past decade. A large number of mostly riverine broads, but also some small well-vegetated isolated broads, support very low macrophyte richness (<5 species). Conversely, a handful of sites have supported 10 or more species for at least 50% of years (Barton Hickling, Lily, Whitlingham Great, Ormesby Little, Martham North, Martham South and Ormesby) with Martham South standing out as the consistently most diverse site with 15-20 taxa.

Examining trends in macrophyte richness in different groups of broads over the last 30 years (Figure 3.1.2-4) several trends emerge, especially when considered in comparison with changes in cover over the same period (Figure 3.1.2-2). Firstly from 2001 onwards richness has increased steadily in the isolated broads, mirroring the trend of increasing plant cover. Secondly in the riverine broads richness has increased slowly but steadily despite a lack of change in plant cover over this period. Thirdly, the Thurne broads have largely conserved their high richness from the late 1980s onwards, despite a marked decline from the peak of plant cover in 1999.

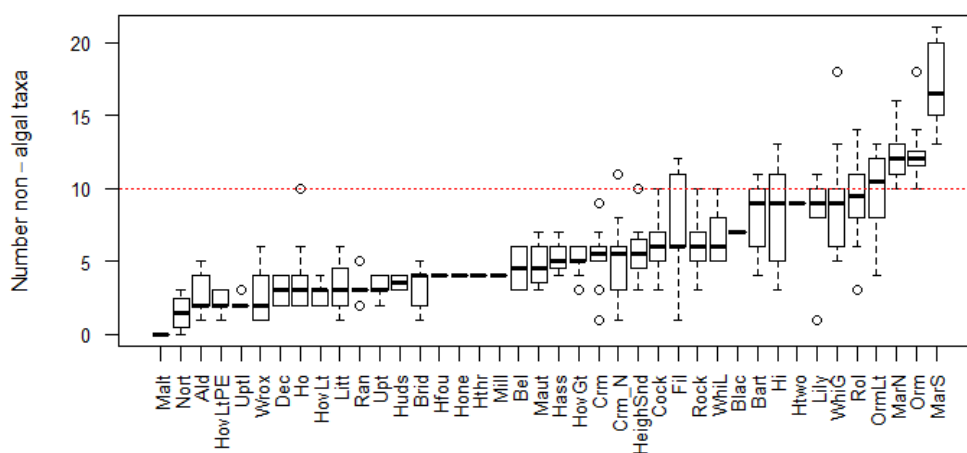


Figure 3.1.2-3 Range of recorded number of non-algal macrophyte taxa for surveyed broads over the decade (2003-2012)

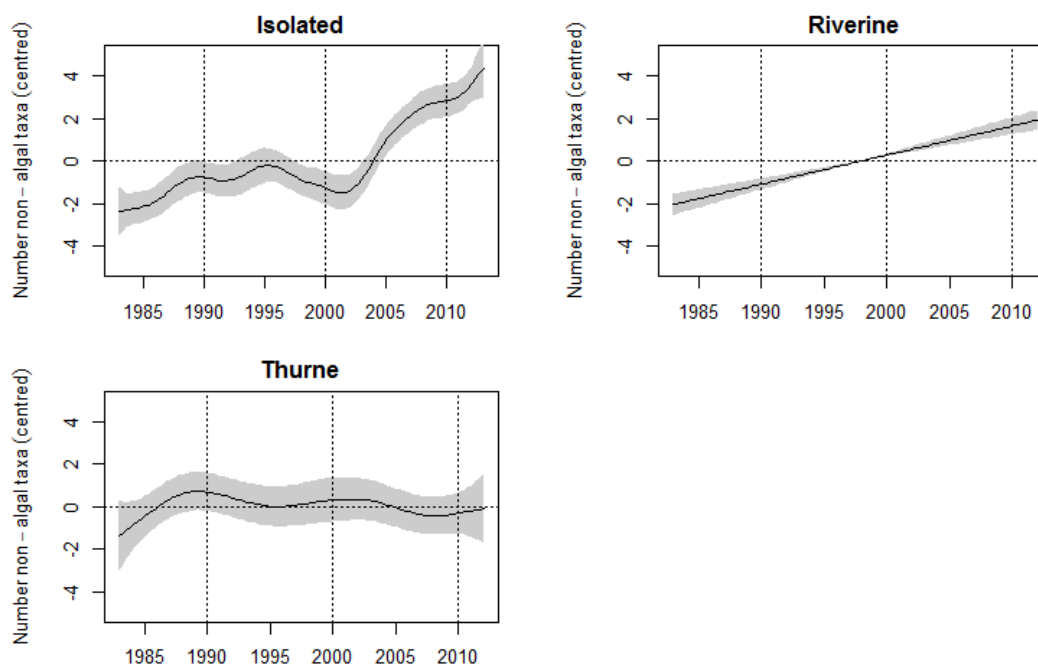


Figure 3.1.2-4 Trend for number of non algal macrophyte taxa in different groups of broads; a) Isolated broads, including Trinity Broad; b) Riverine Broad Bure and Ant; c) Thurne Broad. Lines are GAM smooths with values expressed relative to the average for the period of data available for each group of broads.

One of the key targets in lake restoration is to restore stable macrophyte cover in order to reinstate the buffering mechanisms that maintain a clear water state. Figure 3.1.2-5 demonstrates that inter-annual fluctuations in cover within a broad (i.e. between t and $t+1$ year) tend to be related to the number of species present at time t . When the number of species is large (>10) fluctuations in cover between successive years tend to be small, indicating a high level of stability of cover. On one hand, this relationship is a simple mathematical artefact of the principle that the greater the number of species a fixed amount of cover is distributed between the smaller will be the consequence for cover of a random fluctuation in the population of any one species. This does not however, rule out a possible underlying biological mechanism that promotes stability (for example the fact that a larger number of species is likely to incorporate a wider diversity of plant growth forms, plus a greater number of representatives of individual growth forms (i.e. greater redundancy) that can buffer changes in the performance of individual taxa through compensatory growth (the so called 'insurance hypothesis' of Yacchi & Loreau, 1999). On the other hand, it is of practical significance to know that achieving a high level of richness (by whatever means) will virtually assure a high level of stability of macrophyte cover. It should be noted that high stability of macrophyte cover *can* occur at low macrophyte richness but where this happens the identity of the species present becomes more influential. For example, higher stability of cover is associated with dominance by *Chara* spp. or *Najas marina* than is the case for *Ceratophyllum* or *Elodea* spp.

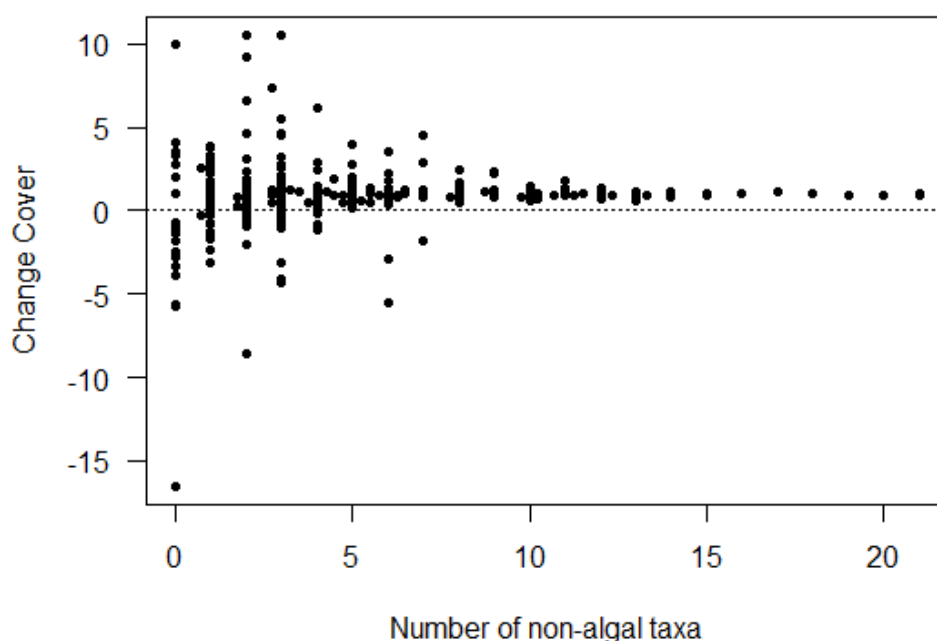


Figure 3.1.2-5 Year on change in non-algal cover compared to number of non-algal taxa present

3.1.2.3 Macrophyte composition

Two indicators of change in macrophyte composition have been used (i) the cover of *Chara* species and (ii) the cover of *Najas marina*. Reference to species optima (Figure 3.1.2-6) derived from an independent data set for lakes in central and NW Europe (Kolda *et al.*, 2014) shows that these indicators have lower TP optima than many of the taxa that currently dominate the aquatic vegetation of the broads, such as *Ceratophyllum demersum*, *Elodea* spp and *Potamogeton pectinatus*. An increase in the abundance and prevalence of species indicative of lower fertility therefore represents a positive change in vegetation composition.

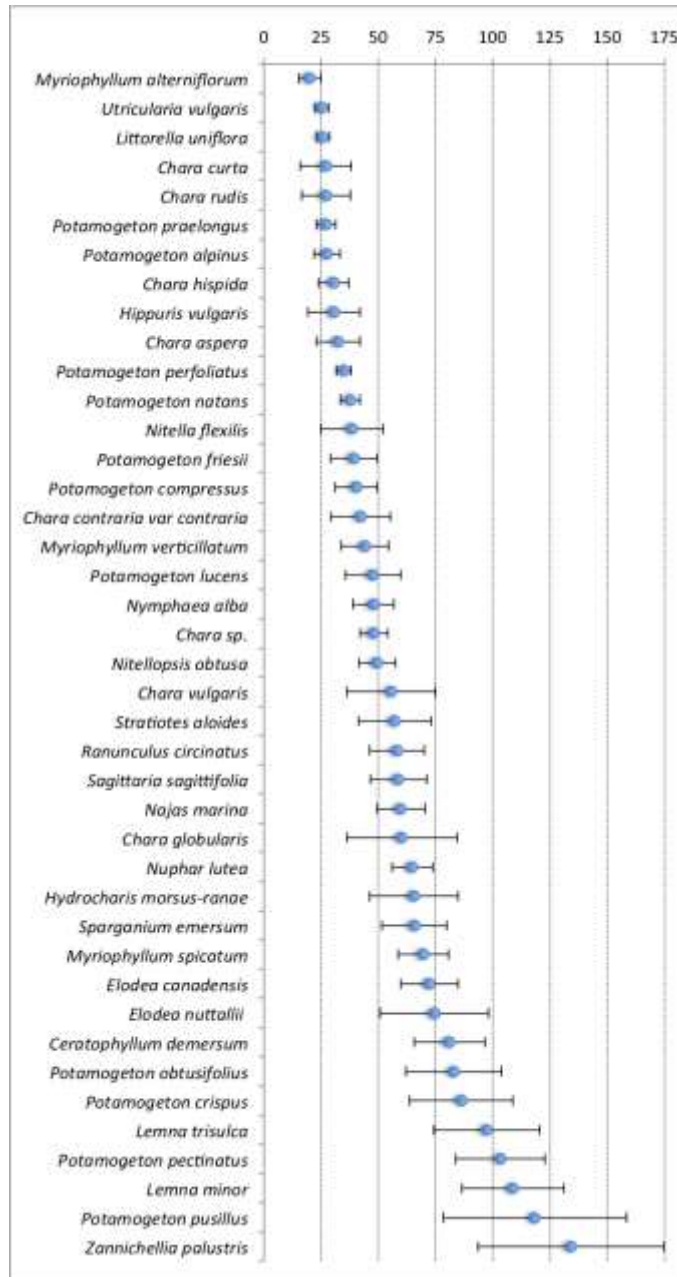


Figure 3.1.2-6 TP optima and standard error for broads macrophyte species found in contemporary surveys and historic records. Values based on a dataset of 1600 lakes in central and NW Europe.

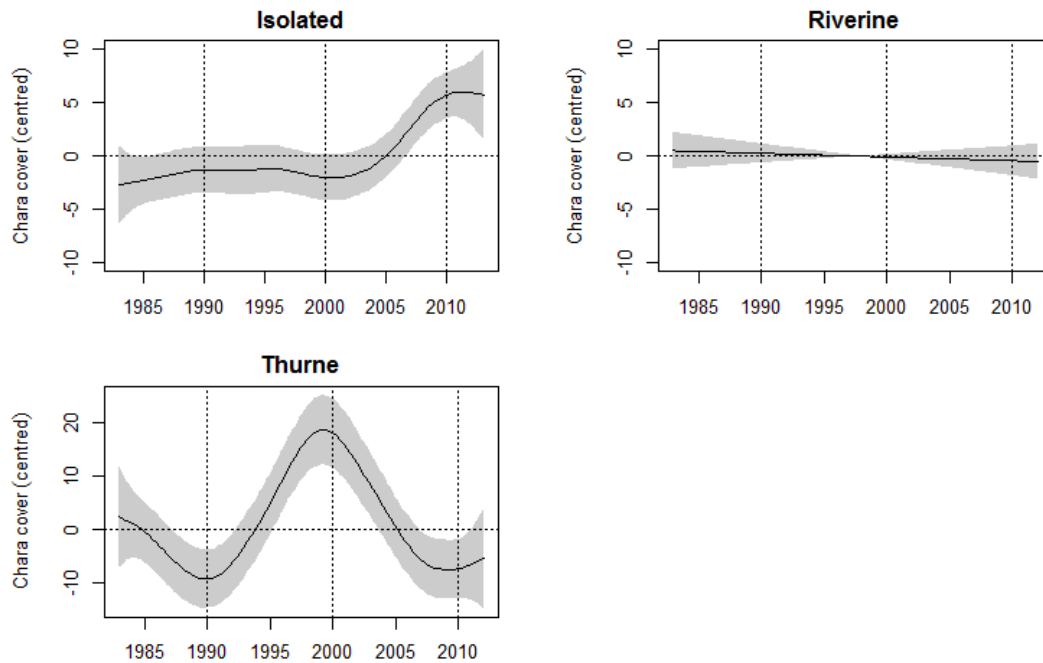


Figure 3.1.2-7 Trend for cover of *Chara* spp. in different groups of broads; a) Isolated broads, including Trinity Broad; b) Riverine Broad Bure and Ant, c) Thurne Broad. Lines are GAM smooths with values expressed relative to the average for the period of data available for each group of broads.

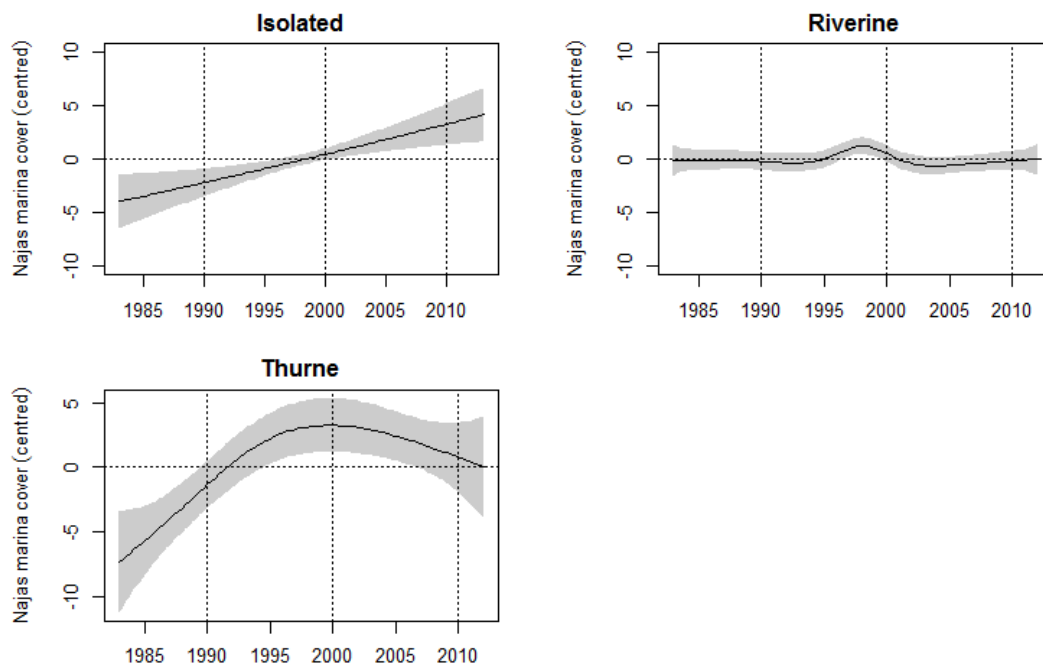


Figure 3.1.2-8 Trend for cover of *Najas marina* in different groups of broads; a) Isolated broads, including Trinity Broad; b) Riverine Broad Bure and Ant, c) Thurne Broad. Lines are GAM smooths with values expressed relative to the average for the period of data available for each group of broads.

In the isolated broads cover of *Chara* has increased consistently since 2002 (Figure 3.1.2-7, stabilising in recent years, whilst *Najas marina* has shown a gradual year on year increase over the last 30 years (Figure 3.1.2-8). In the Thurne broads, having increased dramatically through the 1990s, *Chara* cover crashed in 1999 (in Hickling Broad and Heigham Sound). Cover has since stabilised at a low base level and there are weak indications in recent years of an increase. *N. marina* cover peaked at the same time as *Chara* in the Thurne broads (Figure 3.1.2-7 and Figure 3.1.2-8), but since then has stabilised or declined only slightly. The riverine broads have proved immune to increases in cover of either indicator.

Although there is clear evidence of an increase in abundance of these indicators, especially in isolated lakes, it is notable that other formerly characteristic broads species with similar TP optima (e.g. *Stratiotes aloides*, *Sagittaria sagittifolia*, *Myriophyllum verticillatum*, *Potamogeton lucens*, *Ranunculus circinatus*) remain very scarce in the lakes themselves and some of the taxa with the lowest TP optima (e.g. *Littorella uniflora*, *Myriophyllum alterniflorum*) have been absent for over a century. A return of the Broad to the nutrient concentrations commonly associated with such taxa seems extremely improbable.

3.2 Interactions between macrophytes, light and nutrients: the eutrophication process

3.2.1 Effects of light and nutrients on macrophyte cover

The basis of our understanding of the eutrophication process in very shallow lakes is of competition between macrophytes and algae, mainly for light. As nutrients in the water column become more available it is likely that the faster-growing algal cells will take advantage of this and rapidly increase in abundance. These algae may be suspended in the water column, the phytoplankton, they may cover the sediment surface as filamentous algae or a thin film, or, if macrophytes are present, they will grow on the foliage as epiphytes. In all these situations shading by algae will reduce the light available to macrophytes and thus reduce or potentially even eliminate their growth. In the scenario of a lake recovering from excessive nutrients where macrophytes need to re-establish we would expect to see a relationship where macrophyte cover increases as chlorophyll *a* decreases. This is the case, especially once concentrations fall below 30 $\mu\text{g l}^{-1}$, although the scatter is high, such that a very low cover of macrophytes can still occur even at low concentrations of chlorophyll (Figure 3.2.1-1c). The relationship also shows that in the brackish Thurne broads a higher cover of macrophytes is likely at moderate concentrations ($<50 \mu\text{g l}^{-1}$) of chlorophyll. There are several possible explanations for this. Firstly much of Hickling Broad is particularly shallow ($<1\text{m}$ deep) and thus more light would be available at the bed for the same concentration of chlorophyll in the overlying water column. Secondly several of the species dominating the Thurne system, such as *Myriophyllum spicatum*, have a strongly apical growth form in which long almost leafless stems produce a dense canopy of leaves at the water surface.

The relationship between macrophyte cover and total phosphorus is much weaker (Figure 3.2.1-1d). This is not unexpected given the lack of relationships between

total phosphorus and chlorophyll in many of the isolated broads (see 3.1.1.4) and is largely consistent with alternative stable state theory (i.e. either clear water and macrophyte dominance or turbid water with algal dominance is possible over a relatively large range of TP concentrations). However, macrophytes dominate (cover >50%) fairly consistently once TP concentrations are less than 40 $\mu\text{g l}^{-1}$.

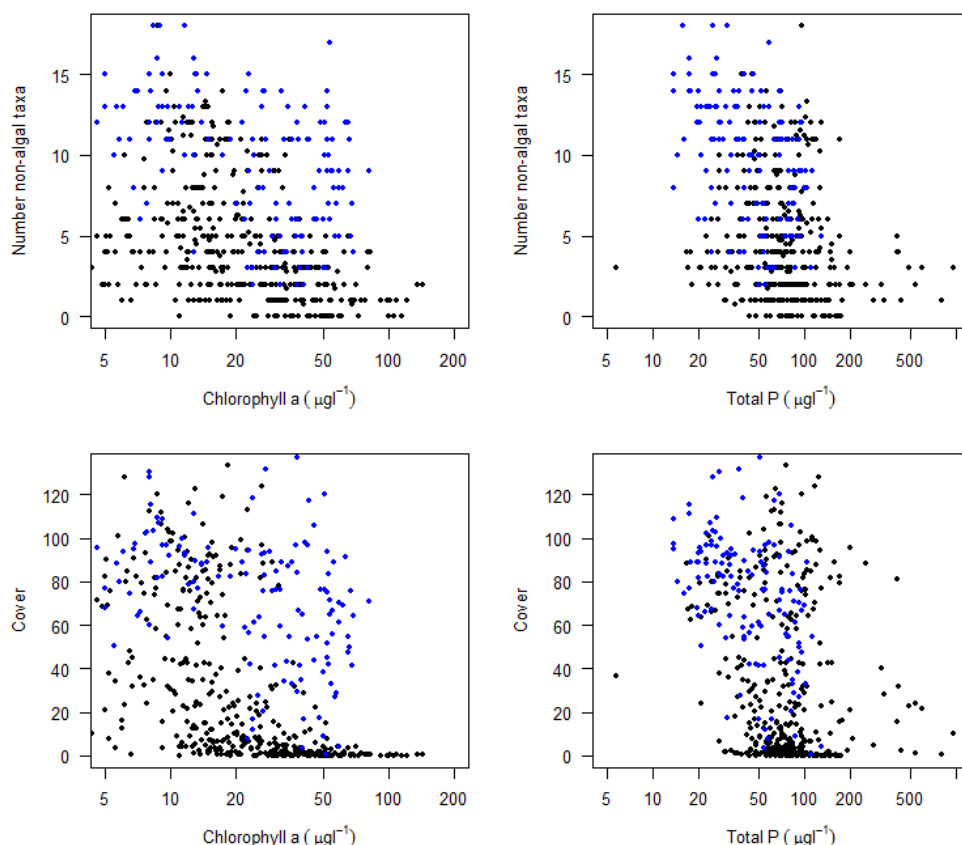


Figure 3.2.1-1 Relationship between a) number of non-algal macrophyte taxa and mean annual chlorophyll a; b) number of non-algal macrophyte taxa and total phosphorus concentration; c) non-algal macrophyte cover and mean annual chlorophyll a and d) non-algal macrophyte cover and total phosphorus concentration in monitored broads (1983-2012). Broad split into brackish Thurne broads (blue points) and other broads (black points)

3.2.2 Effects of light and nutrients on macrophyte composition

The relationship between cover of specific taxa and total phosphorus shows that different taxa tend to occur most abundantly in lakes with different concentrations of total phosphorus. Some taxa (*Chara* spp, *Najas marina*, *Nitellopsis obtusa*) are dominant at the lowest levels of TP, others (*Elodea* spp, *Potamogeton pectinatus*, *Myriophyllum spicatum*), have slightly higher TP optima, while *Ceratophyllum demersum*, filamentous algae and the free-floating lemniads are most abundant at the highest TP concentrations (Figure 3.2.2-1).

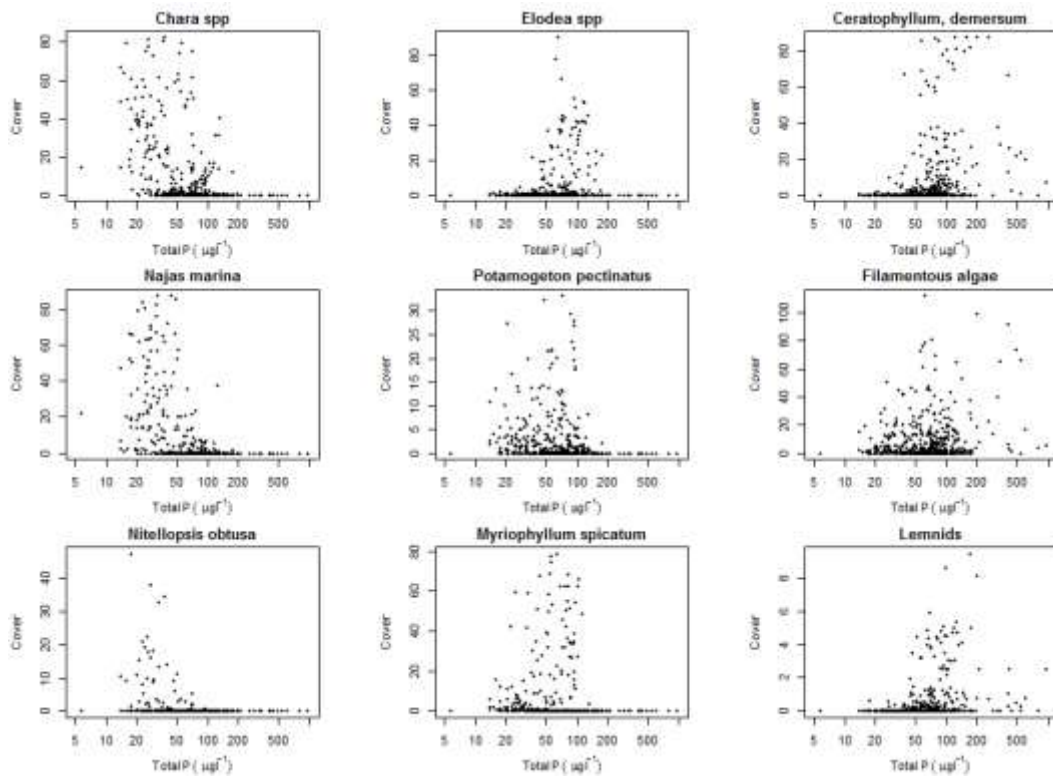


Figure 3.2.2-1 Relationship between cover of key macrophyte taxa and mean annual total phosphorus in all bays over the period of monitoring (1983-2012)

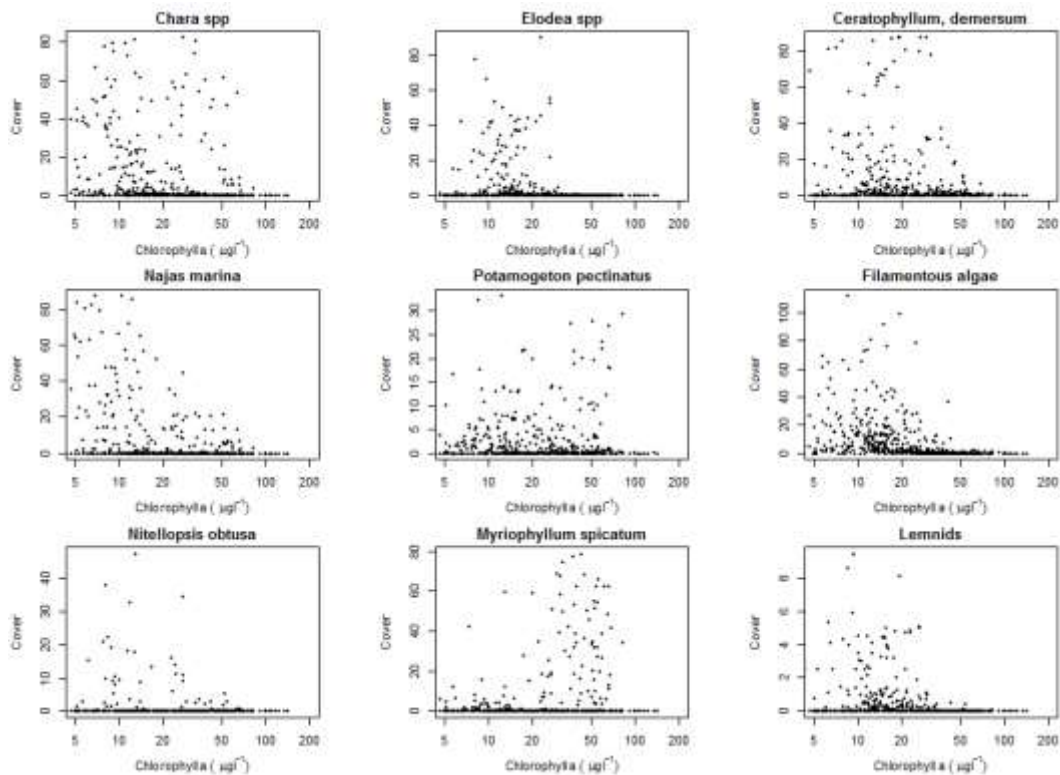


Figure 3.2.2-2 Relationship between cover of key macrophyte taxa and mean annual chlorophyll a in all bays over the period of monitoring (1983-2012)

The relationships between these taxa and chlorophyll are similar but the scatter is much greater suggesting that it is the phosphorus concentration, rather than the indirect influence of phosphorus on the light climate, that is more decisive. Given this it is not surprising that the largest number of macrophyte taxa are found in lakes with moderate TP concentrations (50-100 $\mu\text{g l}^{-1}$; Figure 3.2.1-1b). The important conclusion that can be drawn from these results is that while it is important to lower chlorophyll levels to ensure that sufficient light is available for macrophyte growth *per se*, the rather more sensitive taxa, such as *Chara* and *Najas marina*, also require TP concentrations of less than 50 $\mu\text{g l}^{-1}$, perhaps as low as 30 $\mu\text{g l}^{-1}$ in order to establish large stable populations. Nutrient uptake by prolific growth of these macrophytes may subsequently reduce TP concentrations further and reinforce low chlorophyll concentrations through competition with phytoplankton for nutrients. However, the initial establishment of the more P sensitive species is reliant on low water column P concentrations, and potentially, lower sediment fertility, although there is insufficient data to determine this. In reality the cause-effect relationship between P concentrations and the distribution of individual taxa is difficult to resolve and differences in physiology reflected in major growth form differences (and related access to different nutrient pools) are likely to be important. Phosphorus concentrations may also serve as a surrogate for a range of more proximal influences linked to plant physiology (e.g. toxicity of iron or ammonia (Smolders & Roelofs, 1996; Lambert & Davy, 2011) or competition for inorganic carbon) and the influence of other ecosystem drivers (e.g. herbivore densities and activity).

3.2.3 Integration of interannual fluctuations in growing conditions

There is a clear synchrony between changes in macrophyte cover in different broads, with increases in cover evident between 1992 and 1995 and again from 2002 to 2006. An example of this is shown in Figure 3.2.3-1 together with change in mean chlorophyll a. In all cases the first increase in macrophyte cover only occurs once the chlorophyll level has dropped below a value of approximately 30 $\mu\text{g l}^{-1}$. The synchrony between broads suggests that a local climatic effect may be involved and it is perhaps important that there was a marked increase in air temperature and hours of sun from 1988/1989. Temperature and irradiance are critical abiotic influences on macrophyte growth (Lacoul & Freedman, 2006), but in the common absence of long term standardised survey data the potential influence of climatic conditions on macrophytes is liable to be overlooked. Instead most evidence comes from experimental studies that indicate effects will be most pronounced during the developmental stage. For example, van den Berg *et al.*, (1998) found that sprouting of *Potamogeton pectinatus* from tubers and *Chara aspera* from oospores occurred in half the time at 16°C (4 and 15 days respectively) than it took at 10°C (9 and 27 days respectively).

A warm spring is therefore likely to favour an early onset of macrophyte growth, which is consistent with anecdotal observations made in the reports on Broad's Authority Annual Macrophyte Surveys. Comparing the mean monthly hours of sun with macrophyte cover in Upton Broad, a lake with a relatively stable population of *Najas marina* shows a clear similarity between peaks of cover and hours of sun (Figure 3.2.3-2).

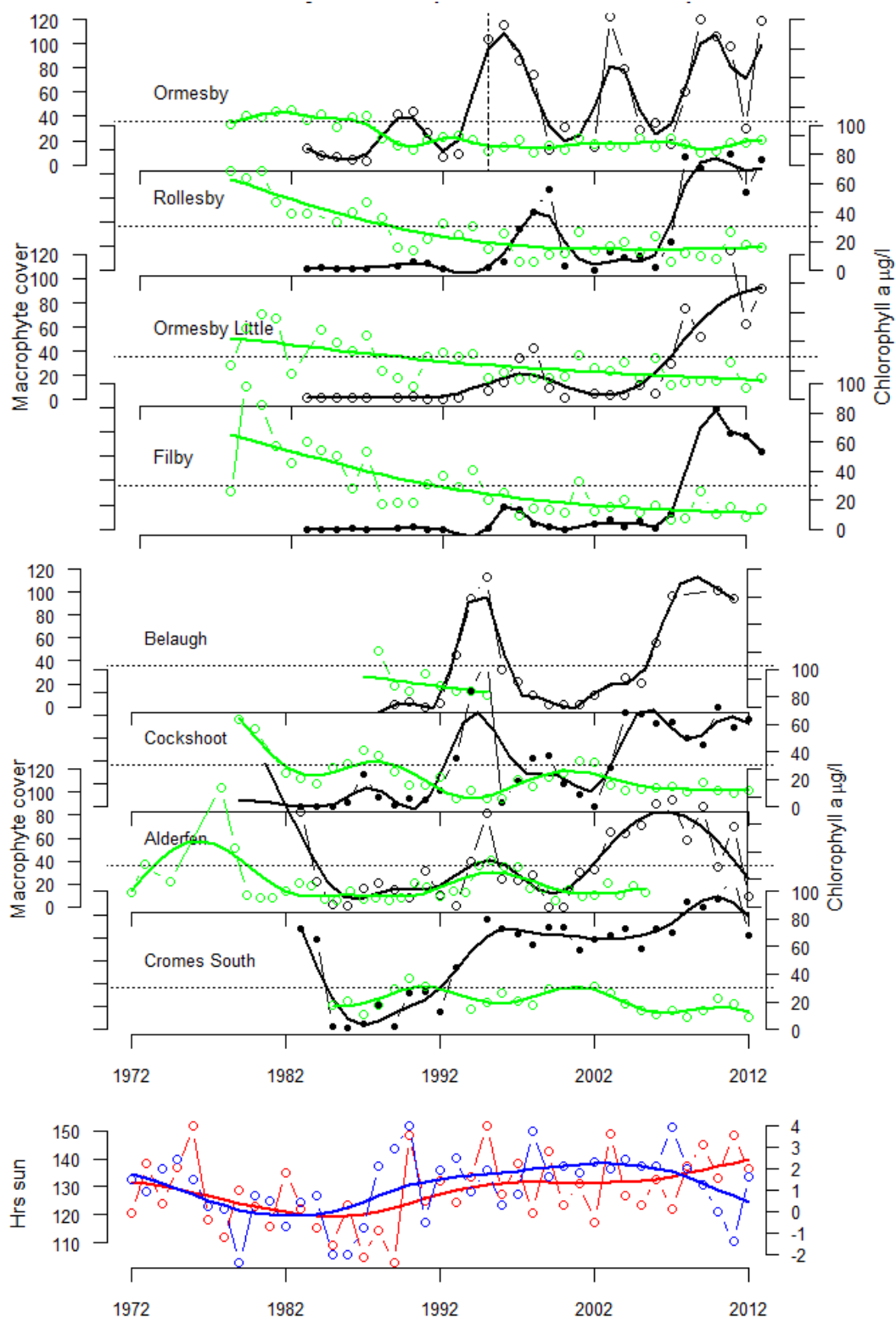


Figure 4.2.3-1 Pattern of annual mean chlorophyll a (green line) and macrophyte cover (black line) in isolated broads. Annual mean monthly hours of sun (red) and mean minimum winter temperature (blue) shown for comparison. Horizontal line shows value of $30\mu\text{g/l}^{-1}$ for chlorophyll as a reference point.

Thus it is clear that macrophytes typically perform better in warm sunny years, although this influence is likely to be greatly diminished in turbid lakes.

Taking data from the 8 lakes shown in Table 3.2.3-1 a non-linear (mixed GAM) model shows mean annual chlorophyll, hours of sun and rainfall are all significant predictors of macrophyte cover (Figure 3.2.3-4 and Table 3.2.3-1). The fact that Year was also a significant predictor, suggests that there is a general trend of increasing macrophyte cover, which is independent of the other predictors. Removing Year from the model only slightly increases the model uncertainty and also has the effect of increasing the significance of the term 'hours of sun'. Modelling

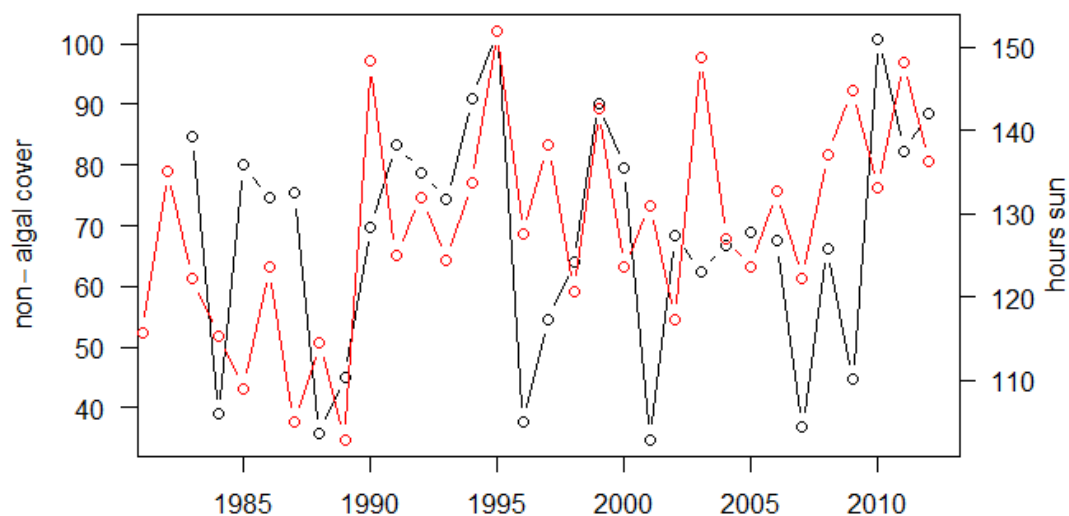


Figure 3.2.3-2 Comparison of changes in macrophyte cover in Upton Broad with annual average monthly hours of sunshine

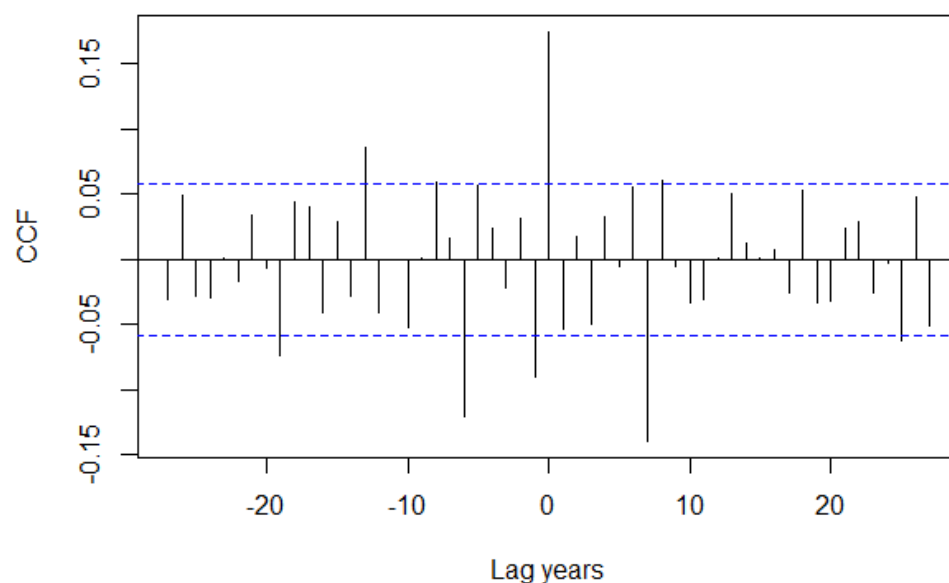


Figure 3.2.3-3 Cross correlation between detrended hours of sunshine and macrophyte cover for all broads (detrended by 1st order difference)

using a number of potentially correlated variables is a challenging task and the model presented here was selected after investigating several models and comparing their respective AIC values. All the significant models included chlorophyll and one or more climate variables. What emerges from this is that the recovery of aquatic vegetation is probably dependent on a number of factors. The most important is the improvement of the light climate through reduction of chlorophyll. Based on the inflection of the chlorophyll response shown in Figure 3.2.3-4 it is suggested that an annual mean concentration of $<25\mu\text{g l}^{-1}$ is required to enable macrophyte growth to occur. This is slightly higher than the WFD Good/Moderate boundary value of $20\mu\text{g l}^{-1}$ and the latter might be a more appropriate target concentration for chlorophyll in the Broads in order to increased the probability of sustained macrophyte growth.

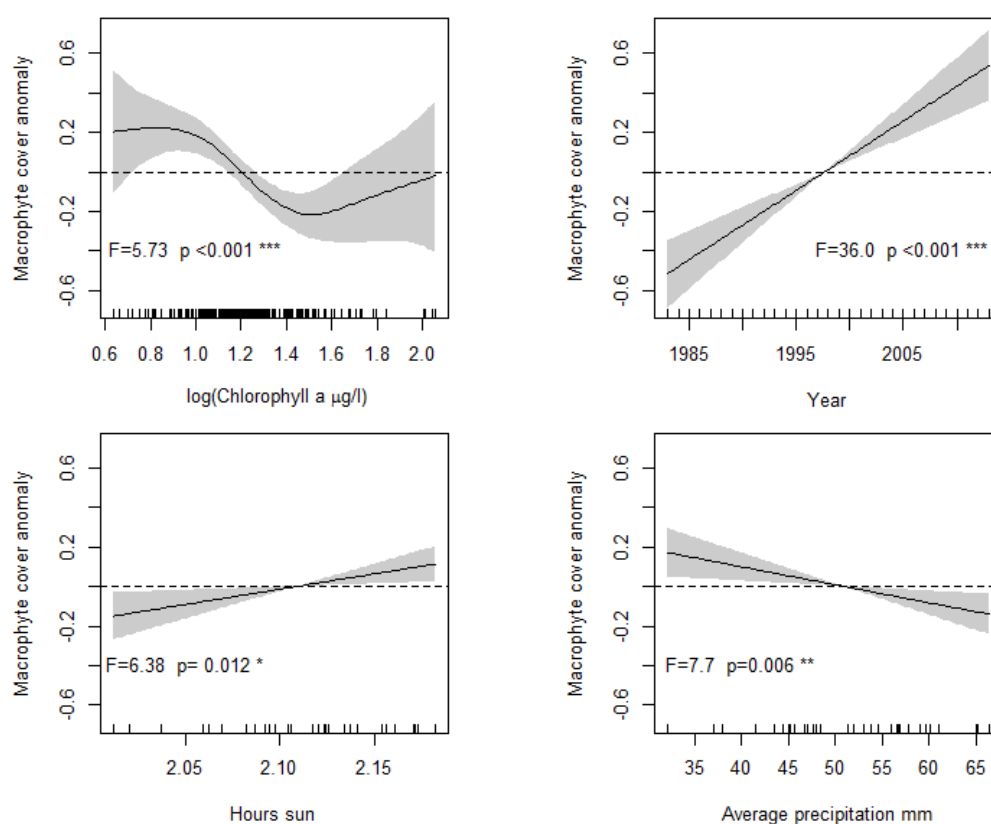


Figure 3.2.3-4 GAM model showing relationship of macrophyte cover (\log_{10} transformed) with \log_{10} annual mean chlorophyll a concentration, Year, \log_{10} average monthly hours of sun, and mean monthly rainfall phosphorus concentration for 8 isolated or semi-isolated broads (Alderfen, Belaugh, Cromes, Cockshoot, Ormesby, Rollesby, Ormesby Little and Filby). Values of macrophyte cover expressed relative to the average for the period (1983-2012)

Table 3.2.3-1 Detail of mixed GAM model shown in Figure 3.2.3-4

Parametric coefficients:

	Estimate	Std. Error	t value	Pr(> t)
(Intercept)	1.25721	0.26210	4.797	3.28e-06 ***
Belaugh	-0.11476	0.41614	-0.276	0.783
Cockshoot	0.04816	0.36944	0.130	0.896
Cromes	0.46971	0.37153	1.264	0.208
Filby	-0.59570	0.37052	-1.608	0.110
Ormesby	0.22407	0.36975	0.606	0.545
Ormesby Lt	-0.25476	0.37161	-0.686	0.494
Rollesby	-0.22576	0.37047	-0.609	0.543

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value
s(log10(chl))	3.699	3.698	5.726	0.000389 ***
s(Year)	1.000	1.000	35.999	8.97e-09 ***
s(log10(AvgOfsun_hr))	1.000	1.000	6.376	0.012378 *
s(AvgOfrain_mm)	1.000	1.000	7.678	0.006142 **

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

R-sq.(adj) = 0.61
Scale est. = 0.1852 n = 2022

4 Review of specific restoration techniques

4.1 External load reduction

4.1.1 Background

4.1.2 Point Source Control

By the 1970s it was well established that phosphorus enrichment, mainly from waste water treatment works, was the major cause of changes to lakes worldwide (Harper 1992; Sutcliffe and Jones 1992) and work carried out in the Broads during the 1970s confirmed eutrophication as the primary cause of the deleterious changes that had occurred in these shallow lakes (Moss 1983). While there was an initial reluctance to accept the need for controlling external nutrient loads evidence of the success of such action elsewhere, for example in Lake Washington (Edmondson 1996), combined with a detailed study of the nutrient status of Barton Broad (Osborne and Moss 1977) resulted in the installation of the then experimental reduction of phosphorus from the effluent from the waste water treatment plant (WWTP) that discharged just upstream of Barton Broad. Shortly after this effluent from a larger WWTP further upstream was diverted to a coastal outfall, removing a substantial amount of the phosphorus load to Barton Broad (Phillips *et al.*, 1999).

Reduction of phosphorus from effluent was subsequently extended to WWTPs discharging to the River Bure. While this initial work was considered experimental the growing recognition of the need to reduce phosphorus from waste water treatment and the requirements to do this under the Urban Waste Water Treatment Directive 91/271/EEC (Council Directive 1991) resulted in discharge consent criteria to restrict the concentration of phosphorus in the effluent being progressively introduced from 1992.

Details of these changes for individual broads are shown in the Annex, but essentially there has been control on the phosphorus discharged to the Rivers Ant from the autumn of 1980 and to the River Bure from autumn of 1986. The effectiveness of this control has progressively improved, with the introduction of 3 mg l⁻¹ TP standards to the larger discharges to the River Bure in 1992 and then the reduction of this standard to 1 mg l⁻¹ TP to these and other significant discharges, including two discharging to the lower tidal river, in 1997. The introduction of the 1 mg l⁻¹ TP standard was particularly important as to reliably achieve this required the introduction of additional filtration of the final effluent which reduced the effluent concentration well below the target of 1 mg l⁻¹. Further reductions in phosphorus load took place in 2003 with the introduction of 2 mg l⁻¹ standards to three smaller WWTP discharging to tributaries of the River Bure. Much of this investment in P removal was done in partnership and by negotiation with Anglian Water (and in one case the Crown Estates) and went beyond statutory requirements. As well as STWs phosphorus removal was also initiated at a food production facility at Westwick feeding into the R. Ant.

4.1.3 Isolation of lake

An alternative approach to reducing the external load, particularly for the smaller broads was to isolate them from their catchment, or at least the part of the catchment considered to be the primary source of enrichment. Examples of this are at Cockshoot Broad, which, prior to the construction of a dam that cut off the broad from the main River Bure in 1982, received phosphorus-rich water from the main river catchment. A similar example is Cromes Broad, although in this case water from the River Ant was periodically allowed to enter the broad to maintain water levels and to flush the broad when it was considered necessary. Cromes Broad is currently completely isolated from the River Ant via essential flood defence banks and a sluice board structure. Water flows in from the upland catchment, which still contain elevated nutrient levels from a handful of poorly functioning domestic waste water systems in properties along Sharpe Street. The water flows out of the broad via the dykes and through the marshes and eventually the Ludham drainage pump. Another more extreme example of isolation was carried out at Alderfen Broad. This broad was not directly connected to the main river system and thus only had a small local catchment. However studies carried out in the 1970s (Moss *et al.*, 1979; Phillips 1977) demonstrated that the broad was receiving substantially elevated phosphorus via its inflow stream and a decision was made to divert the inflow to a small ditch system that flowed along the perimeter of the broad, re-joining the outflow stream below the broad.

4.1.4 Evidence of effectiveness

4.1.4.1 Reduction of TP

There is little doubt that the introduction of phosphorus removal from the major WWTPs discharging upstream of the Broads resulted in a significant reduction in TP concentration in both the rivers upstream of the Broads, but also in the Broads themselves (Figure 4.1.4-1). There has been substantial discussion about the time course of recovery of very shallow lakes following point source reduction, with an emerging view that it takes about 15 years for a new equilibrium to be established with the phosphorus pool in the sediment needing to establish a new equilibrium with the overlying water (Jeppesen *et al.*, 2005). Nutrient budgets constructed during the recovery of Barton Broad support this finding as the broad moved through a period of net phosphorus release to net retention in a similar time scale (Phillips *et al.*, 2005). The time series for the River Bure suggests a similar trajectory of change, although in both cases it is difficult to determine the true behaviour of the system as progressive reductions in point source TP loads were made within this 15 year time period. What is, however, clear from these results is that the current TP concentration, which is close to the WFD good/moderate boundary has now stabilised at about 60 µg l⁻¹. This is very similar to the concentration of TP found in a variety of small tributary streams in the area lacking any WWT inputs and is probably typical of diffuse phosphorus loading for this part of Norfolk. It is thus unlikely that further control of point source phosphorus can ever reduce TP concentration below this value.

The fact that some broads (Upton, Martham N & S) which contain stable macrophyte communities have lower TP concentrations raises the question of how these are achieved. Do these sites benefit from catchment drainage with particularly low TP, or are there internal mechanisms that subsequently reduce TP concentrations. Further work is needed to determine this with certainty, but from the evidence available it is likely that the lower TP is a consequence of increased phosphorus retention in macrophyte tissue rather than particularly low phosphorus loading. One line of evidence for this comes from the time series at Hickling Broad, which showed a significant reduction in water column TP when a substantial biomass of *Chara* dominated the broad in 1997/1998. The sequestering of nutrients, especially nitrogen, in macrophyte biomass, is widely accepted to be a key step in the stabilisation of clear water conditions in biomanipulated lakes in the Netherlands (e.g. van Donk *et al.*, 1993). Some simple calculations illustrate the scale of potential nutrient storage in macrophytes in the Broads. Thus, Philipps (unpublished data), found that the P content of *Ceratophyllum* tissue was 0.48% (a typical concentration under fertile conditions). A measured plant biomass (dry weight) of 250g m⁻² (again a typical figure for a shallow lake), therefore stored 1.2mg P m⁻². This is an order of magnitude higher than would be found in the water column with a concentration of 0.1mg l⁻¹ assuming a water depth of 1 m. The NERC funded Biodiversity, Ecosystem Services and Sustainability programme includes a project on shallow lakes in which some of the authors of this report are collaborating. Part of this project is investigating the ecosystem service significance of CNP storage in macrophytes and the sensitivity of storage to macrophyte richness. Measurements of tissue nutrient storage will shortly be available for the Broads and Upper Lough Erne covering a wide range of sites and species. With this data it will be possible to account more fully for the contribution of macrophytes to lake level nutrient storage and any inhibitory effect on chlorophyll concentrations.

The success of isolation of broads from their catchment is not easy to assess as all of the monitored examples have had other management actions such as sediment removal and various fish manipulations undertaken at the same time. However, the general finding is that while the external loading can be reduced the internal load from sediment-derived phosphorus is likely to increase. This may reflect the re-establishment of a sediment-water equilibrium, or simply a reduction in export and shift to more deoxygenated conditions that favour sediment P release. The most obvious example comes from Alderfen Broad where isolation substantially reduced water throughput, which, over time, and in combination with a period of particularly low rainfall, resulted in a very substantial increase in TP concentration. Although the P concentration ultimately returned to the pre-isolation baseline the driver for this decline may have been increased growth of the dominant macrophyte *Ceratophyllum* following increased water clarity associated with biomanipulation of fish stocks.

4.1.4.2 Reduction of phytoplankton

It has been suggested that a chlorophyll a concentration of 30 µg l⁻¹ would be required to allow macrophytes to have a reasonable chance of developing. Taking the regression model between TP and Chlorophyll for the riverine lakes ($\text{Log}_{10} \text{Chl} = 0.862 \pm 0.084 * \text{Log}_{10}(\text{TP}) + 0.03 \pm 0.043$, $R^2 = 0.632$, $p < 0.001$) suggests that a mean TP

concentration of $60 \mu\text{g l}^{-1}$ (i.e. the approximate limit of what can be achieved by external load reduction though point source control) should achieve a mean chlorophyll of $37 \mu\text{g l}^{-1}$. While this is close to the $30 \mu\text{g l}^{-1}$ threshold required it is still significantly higher and this model implies that an average TP concentration of $48 \mu\text{g l}^{-1}$ would be required to achieve the desired chlorophyll concentration in the riverine lakes. Thus, the likelihood of achieving a sufficiently sustained reduction in chlorophyll by external load reduction alone is relatively low and improved diffuse pollution control would likely be needed to reduce TP concentrations below the $60 \mu\text{g l}^{-1}$ level. To achieve a Chlorophyll concentration of $30 \mu\text{g l}^{-1}$ or less would require a Chl/TP ratio of < 0.5 which is likely to require a shift in community structure that allows grazing zooplankton to become important, at least during the spring.. Once a stable macrophyte community is established however, the combined effects of the uptake of phosphorus by macrophytes, particularly species such as *Chara* which have the ability to sequester significant quantities of phosphorus (van Donk *et al.*, 1993), and a wider diversity of zooplankton and other invertebrate grazers, mean that the current level of external load reduction would likely to be adequate to maintain macrophyte dominance.

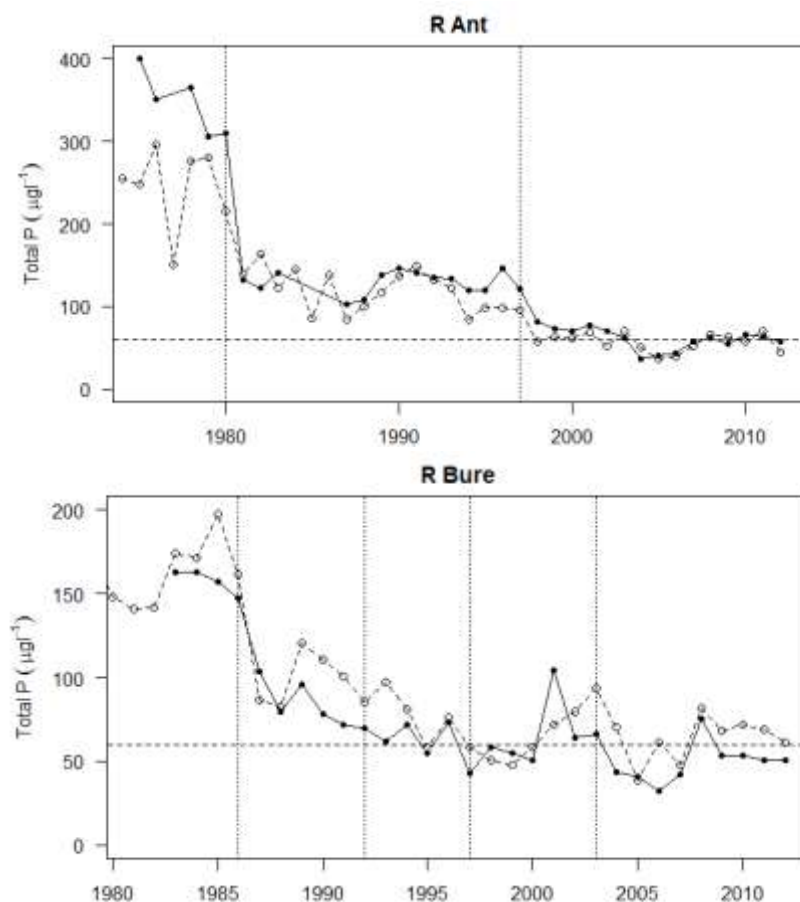


Figure 4.1.4-1 Changes in total phosphorus concentration in a) River Ant at Hunsett Mill, upstream of Barton Broad (solid line solid circles) and in Barton Broad (dotted line open circles), b) River Bure at Horstead Mill, tidal limit (solid line solid circles) and in Wroxham Broad (dotted line dotted circles). Broken horizontal line marks $60 \mu\text{g l}^{-1}$, a typical WFD good/moderate boundary for the Broads. Dotted vertical lines mark introduction of progressive improvements in control of TP from point sources in catchment.

4.1.5 Factors constraining effectiveness

The early work carried out on the River Ant illustrated that the assumption that only the nearest WWTW was influencing conditions in Barton Broad was incorrect. It was only when an effluent discharging significantly further upstream was diverted that TP concentrations in the river system were significantly reduced (Bell 1981; Phillips *et al.*, 1999). Thus when implementing controls on point sources it is essential to consider all upstream sources, including point and diffuse sources.

Similarly, the broads system is tidal and while all the significant WWTWs discharging to the main rivers of the Northern Broads have had phosphorus removal introduced the impact of downstream sources still needs to be considered. It is clear that TP concentrations increase in a downstream direction in the River Bure with the highest current values found in South Walsham Broad. The influence of water from the lower tidal river in this broad is clear from the record of chloride and it is possible that the higher levels of TP still found in this broad are a result of WWTW effluents discharged to the tidal river further downstream. Alternatively, this broad received a direct discharge of effluent from a WWTWs until 1997 when the plant was closed and it may be that re-equilibration between the sediment and the overlying water is still occurring some 17 years later. This long recovery time may be a result of the hydrology of the broad which receives a relatively small direct freshwater input but is subject to significant tidal flushing via a side channel of the main river.

The effectiveness of isolation is likely to be limited by the resulting reduction in water exchange. Where the broad still has a significant freshwater input from unpolluted feeder streams, as it the case for Cockshoot, isolation may have a chance of being useful. However, it is likely that phosphorus release from the sediment will dominate the phosphorus budget and consideration would need to be given to the conditions of the sediment and the benefits of sediment removal, as has taken place at all of the broads where isolation has been used and monitoring data are available (Alderfen, Cromes, Cockshoot). Despite the isolated broads having relatively small catchments nutrient inputs from non-sewered domestic properties can be locally significant, such as for Cromes Broad currently and Alderfen Broad in the past (May *et al.*, 2015).

4.1.6 Wider ecosystem consequences

There are few wider consequences of external phosphorus control, although consideration may in the future need to be given to the most effective methods to ensure that the ferric products used to precipitate phosphorus do not introduce other contaminants, that the energy costs associated with improved effluent standards are minimised, and that suitable techniques can be introduced to improve recovery and re-use of P from effluents.

Isolation as a method of external load reduction in contrast has many potential dis-benefits. Apart from reducing water throughput and thus increasing the rate of accumulation of nutrient and biomass the isolation of water bodies reduces connectivity which is likely to be a key factor in the wider ecosystem recovery of the Broads system. With the reduction in nutrient status of the Broads and the progressive re-establishment of macrophytes it is expected that the fish community

will also restructure. To achieve this it is likely that the fish need access to back waters and the wider dyke networks and long term isolation of some of the flood plain wetlands would not support this ambition. However, this assumption needs testing as no work has been done to demonstrate the value of reconnecting the few fen dykes and the few broads (e.g. Cockshoot Broad) that are isolated to the wider waterways. The NERC funded Lake BESS project is currently assessing the role of hydrological connectivity in promoting resilience of shallow lakes to eutrophication, contrasting the Broads and the Upper Lough Erne system in Fermanagh where lakes undergo extensive reconnection during winter flooding. It is worth noting that in the Broads the majority of undefended and functioning floodplain is already connected over much of the year. Isolation, for example in the case of Alderfen, was a method of moving forward when the reduction of external load seemed an unlikely or at least a long-term ambition. While it still may have a role in nutrient reduction in some of the smaller broads located some distance from the main river system it is not thought to be the most effective technique in the longer term. It is however, important to note that we refer here to hydrological isolation, rather than the isolation required to manipulate fish which is dealt with below.

In summary the introduction of phosphorus removal as a method of reducing external load in the Broads has been extremely successful and has provided the foundation for recovery of the system. However, the evidence suggests that the current background phosphorus load, from both point and diffuse sources and following equilibration with sediments, is still slightly too high to reliably reduce chlorophyll concentration to the level where plant growth is likely. The gap is however relatively small and with time and the maintenance of the current controls on phosphorus in discharges the prospects for further improvements in the Broads must be good. However there is projected to be a 7% increase in human population in Norfolk from 2014-2022. To ensure progress it will be necessary for effluent standards to keep pace with future population growth in order to maintain an overall reduction in load.

4.2 Sediment removal

4.2.1 Background

Sediment removal has been commonly applied as an eutrophication management measure, both in the Broads and across the world. However, the specific objectives of the approach extend beyond the reduction of internal loading and vary across case studies. Peterson (1982) reviewed these objectives and those relevant to the Broads, with modifications, include (1) to deepen the lake for improved recreational use or navigation, (2) to prevent or reduce internal P loading through the removal of P-laden sediments, (3) to deepen the lake to improve growing conditions for macrophytes, and (4) to expose viable macrophyte seed banks in deeper sediment layers to encourage growth of native species. That the removal of sediment results in the deepening of a water body is not in question and so objective (1) will not be addressed here.

Jeppesen *et al.*, (2005) reviewed the recovery of 35 lakes following external P load reduction and concluded that internal P loading had delayed ecological recovery by

10 to 15 years. The duration and intensity of internal loading is affected by lake-specific factors such as water retention time, biological structure, pollution history, sediment P composition and concentrations, and lake morphology (Sas, 1989). Seasonal changes in water column P concentrations occur following the reduction of catchment P loading and this is reflected in observed changes in water-column TP concentrations across Danish lakes (Søndergaard *et al.*, 2007, 2012). In general, during the early years of recovery following successful reduction in catchment P loads, winter to spring and autumn water column TP concentrations decrease, and sediment P release during the summer to early autumn increases. As the magnitude of P release from the sediments decreases with time, summer in-lake TP concentrations also decrease, provided adequate flushing is maintained. This general recovery response has been confirmed through analyses of water quality time series data from, for example, Barton Broad (Phillips *et al.*, 2005) and Loch Leven (Sharpley *et al.*, 2014).

Where conditions in the lake support prolonged internal loading (e.g. long residence times, high sediment TP content, and reducing sediment conditions), even after effective catchment management, sediment removal has often been employed as a control measure. In order to successfully reduce internal P loading sediment removal should expose a 'new' sediment surface with a high P binding capacity and low 'release-sensitive' sediment P content (Søndergaard *et al.*, 2005). Unfortunately, very few studies are available with which the effectiveness of this approach can be assessed with confidence. For example, in a review of some 49 case studies Pierce (1970) reported that it was impossible to draw conclusions on the effectiveness of the sediment removal activities due mainly to data paucity or concurrent catchment or in-lake management events. This is a situation that is, unfortunately, common within the evidence available from the Broads' case studies, presented below. As such, there is little evidence from other case studies of a common set of responses against which to assess the observations from the Broads. To aid quantitative assessments of future sediment removal activities it is important that standard procedures are developed for operational design (e.g. Peterson 1982) coupled with the use of consistent sediment and water quality monitoring programmes.

Here we draw on results from meta-analyses of conditions supporting high internal loading (Sas, 1989) and on observations from long-term recovery responses in water column TP and chlorophyll a concentrations following catchment loading (e.g. seasonal changes as described above; Jeppesen *et al.*, 2005, Phillips *et al.*, 2005; Søndergaard *et al.*, 2005, 2012). Sas (1989) demonstrated that sediment TP concentration can be used as an indicator of the likely transient recovery time, as a result of sediment P release, following external load reduction. Sas (1989) classified the sediment TP concentration in the upper 15 cm of lake sediments in relation to the estimated recovery time following reduction of external P loading. At concentrations of less than 1 mg TP g⁻¹ dry weight, internal loading is expected to be negligible with moderate summer sediment-P release events. At concentrations between 1 mg TP g⁻¹ dw and 2.5 mg TP g⁻¹ dw, net annual sediment-P release will be high, initially, with recovery expected within a 5 year period; a high summer release event would be expected to occur that will be affected by pH, temperature, dissolved oxygen and microbial activity. At concentrations in excess of 2.5 mg TP g⁻¹ dw, net annual

sediment-P release will occur for more than 5 years; in this situation, sediment-P release is expected all year round and will be greatly influenced by pH, dissolved oxygen and microbial activity. Assessments of P release potential can also be made following analysis of sediment P 'fractions' (Boström *et al.*, 1992). Finally, direct measures of P flux in intact sediment core incubations provide a 'snap-shot' of the conditions that may be occurring in the lake, although laboratory conditions may not reflect conditions at the bottom of a lake.

In addition to reducing internal loading sediment removal may be carried out specifically to improve habitat for aquatic plants. Once water depths fall below 0.6m the likelihood of wind-induced sediment resuspension increases significantly and encroachment by emergent vegetation is likely to occur. However, in water bodies that have lost a significant proportion of their volume through sediment accumulation it is probably physicochemical factors associated with anoxic, highly fluid and organic-rich sediments (e.g. high concentrations of sulphide and reduced metals; ease of uprooting) that are the major constraints on macrophyte growth rather than lack of depth *per se*.

As well as increasing depth there is a common desire to expose propagule-rich layers of buried sediment as part of the de-silting process (Sayer *et al.*, 2012). Some groups of species such as charophytes typically form long-lived (centennial scale) propagule banks composed of very high densities of oospores (van den Berg *et al.*, 2001). Dominance of recently disturbed sites is characteristic of *Chara* spp (Wade 1990), and it is often found that dredging of ditches and canals is followed by rapid colonisation by charophytes (Wade & Edwards, 1980). In some sediment removal projects carried out in the Broads (e.g. Cromes Broad North, Upton Little and Little Broads) palaeolimnological investigations have been carried out in advance to determine the optimum depth for sediment removal by identifying the depth at which the highest concentrations of oospores occur (Sayer *et al.*, 2012).

4.2.2 Evidence of effectiveness shown

In this section we review the evidence of effectiveness of sediment removal for achieving the restoration objectives described in section 1.3. To achieve this we compare responses in a series of indicators across those broads in which sediment removal has been conducted. To date some 23 of the broads have been subjected to mud-pumping activities. These include the following sites: Alderfen Broad, Barnby Broad, Barton Broad, Buckenham Broad channel near house, Buckenham Broad, Belaugh Broad, Burntfen Broad, Catfield Broad, Calthorpe Broad, Cromes Broad South, Cromes Broad North, Hassingham Broad, Hoveton Little Broad, Little Broad, Mautby Decoy, Norton's Broad, Ormesby Great Broad - eastern arm, Pound End, Sotshole Broad, Strumpshaw Broad, Upton Little Broad, Wheatfen Broad. The earliest mud-pumping work was undertaken in the early 1980s and de-silting has continued up until the present-day with a recent spate of work over 2005-2010.

The availability of relevant data across the broads is variable (Table 4.2.2-1) and assessment of these indicators is confounded by various site-specific factors (Table

4.2.2-2). The indicators are categorised by the mechanisms of recovery associated with sediment removal and include:

1. Reductions in sediment phosphorus concentrations leading to reduced potential for sediment P release (Indicators: sediment TP concentration and composition and sediment P flux to the water column),
2. Reductions in TP and chlorophyll a concentrations in the water column (Indicators: summer:winter TP and chlorophyll ratios and annual mean concentrations)
3. Improvements in the physical conditions of the bed to favour macrophyte growth (Indicators: increased macrophyte cover and diversity)
4. The exposure of viable macrophyte seed banks to support desirable species recolonisation (Indicators: increased cover of *Chara* spp or other characteristic broadland species with moderate or low TP optima)

Conclusions are proposed based on assessments of responses across multiple broads and a confidence rating is provided based on the strength of the evidence available and the level of agreement between the report co-authors.

Table 4.2.2-1 Summary of sediment and water quality data used in the assessment of responses in Broads from which sediment has been removed.

Broad name	Year of dredging	Sediment TP		Sediment P composition		Sediment P flux		Summer/winter & annual water column TP		Summer/winter & annual water column Chl	
		B	A	B	A	B	A	B	A	B	A
Alderfen Broad	1993-1994	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
Cockshoot Broad ¹	1981	N	Y	N	Y	N	Y	Y	Y	Y	Y
Cromes Broad South ²	1988	N	Y	N	N	N	N	Y	Y	Y	Y
Cromes Broad North ³	2005	Y	N	N	N	N	N	N	N	N	N
Barton Broad ⁴	1996, 1997, 1998, 1999	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
Hoveton Little Broad	1989-1990	N	N	N	N	N	N	Y	Y	Y	Y
Hoveton Little Pound End	1989-1990	N	N	N	N	N	N	Y	Y	Y	Y
Ormesby Broad ^{5,6}	2010-2011	Y	Y	N	N	N	N	Y	Y	Y	Y
Upton Little Broad ⁶	2011	N	N	N	N	N	N	N	N	N	N

Note: TP – total phosphorus concentration; Chl- chlorophyll a concentration; ‘B’ – data available before sediment removal?; ‘A’ – data available after sediment removal?; ‘?’ – ¹ *The use of water quality data before and after dredging at Cockshoot Broad is confounded as isolation from the River Bure was conducted concurrently with dredging.* ² *Cromes Broad South was sampled from 1983- 1992 and 1994-2012. During 1983-1992 there was thought to be some connectivity to the River Ant and a sluice was repaired in 1982. It is unclear whether this sluice was linked directly to the North or South basins.* ³ *Cromes Broad North was sampled April 1993 – Dec 1993.* ⁴ *Determination of effects of sediment dredging complicated due to multiple dredging events.* ⁵ *Sediment dredging activity in Ormesby is expected to be of minor importance given the small surface area dredged.* ⁶ *The 1-2 years water quality data available for analysis following sediment dredging activities were deemed insufficient to support analysis.*

4.2.2.1 Sediment P composition and release

As a precursor to reductions in water column TP, SRP, and chlorophyll *a* concentrations and improvements in water clarity following sediment removal, conditions in the sediment should also change to indicate a reduced potential for internal loading. This should be evident by assessing sediment TP concentration, composition and the rate of release from sediments to the overlying water column.

Where sediment removal has been conducted in the Broads a decrease in sediment TP concentration has been reported (typically over the following 6-12 months). For the broads in which sediment removal has been conducted and for which sediment TP data were available both before and after sediment removal or in comparison to 'control' sites, this appeared to be the case. In Alderfen Broad sediment TP concentrations were reduced from 2.13 mg g⁻¹ dw to 0.68 mg g⁻¹ dw following sediment dredging. In Cromes Broad South, sediment TP concentrations following dredging appeared to be lower (1.08 mg g⁻¹ dw; 1995) than those reported for the 'control' Cromes Broad North (3.2 mg g⁻¹ dw; 1993). In Barton Broad, where the effects of multiple sediment removal events were pooled, an apparent reduction in sediment TP concentration was reported from 1.46 mg P g⁻¹ dw before sediment removal to 0.98 mg P g⁻¹ dw following sediment removal. In Ormesby Broad a reduction in sediment TP concentration from 1.5 mg TP g⁻¹ d.w before sediment removal to between 1.2 mg TP g⁻¹ dw to 0.6 mg TP g⁻¹ dw after sediment removal was reported. These reductions indicate a general decrease in the 'potential' for sediment P release (Sas, 1989) where sediment TP concentrations of less than 1.0 mg P g⁻¹ dw are indicative of very low sediment P release conditions.

In order to support improved conditions in the water column these reductions would be expected to persist at least for the period required to support macrophyte recovery. In Alderfen Broad sediment TP concentrations appeared to recover within about 1 year of sediment removal (1.73 mg TP g⁻¹ dw) with concentrations exceeding conditions before sediment removal when surveyed 288 days later (3.19 mg TP g⁻¹ dw). Although no data were available before sediment removal in Cockshoot Broad, surface sediment concentrations reported almost 10 years after removal were 2.6 mg TP g⁻¹ dw.

The sediment TP data from Barton Broad offer the most comprehensive assessment of recovery following sediment removal. TP concentrations were measured in surface sediment samples taken at various locations across the broad at various dates post desilting. When analysed as a range of sediment TP concentrations across multiple sites in which sediment removal was conducted it was apparent that the scale of reduction in values at the lower end of the range of observed concentrations was more pronounced than at the upper end of this range. Specifically, significant differences were observed only in the 25th percentile (i.e. the lower quartile) and 5th percentiles of the surface sediment TP range following sediment removal. These reductions persisted for about 1.5 years and 2.5 years for the 25th and 5th percentiles, respectively. After this time values were no longer significantly different from the values pre-sediment removal.

The relatively short term response in sediment TP concentration indicates the importance of in-lake processes in driving surface sediment conditions before and after sediment removal. These processes may include (1) the translocation of P rich sediments from areas of the broad in which sediment removal was incomplete to those being surveyed (i.e. sediment focussing; Björk *et al.*, 2010), (2) the persistence of organic P load (i.e. algal matter) to the

bed from the water column fuelled by persistent catchment P loading following sediment removal (Gulati *et al.*, 2008), or through the accumulation of upwardly migrating SRP within Fe-P complexes in newly exposed, and now oxidising, surface sediment layers (Boström *et al.*, 1992). The data does not allow us to identify which of these mechanisms are dominant. Further evidence of such processes acting to rapidly 're-set' internal P loading potential following sediment removal is available from sediment P composition data where 'release sensitive' pools include organic P and iron-bound-P. In Alderfen Broad, surprisingly little change in the P composition of the surface sediment following sediment removal was reported (before sediment removal – 'Organic' P 67%; 'Iron bound' P 13%; 'Aluminium/humic acid bound' P 10%; 'Calcium bound' P 10%; after dredging - 'Organic' P 67%; 'Iron bound' P 9%; 'Aluminium/humic acid bound' P 22%; 'Calcium bound' P 2%; Pitt *et al.*, 1997). In Cockshoot Broad, sediment P composition within 1 year of sediment removal also indicated P composition that would support persistent sediment P release (after dredging - 'Organic' P 60 % – 64%; 'Iron bound' P 9.4% - 12.1%; 'Aluminium/humic acid bound' P 17.0% - 19.9%; 'Calcium bound' P 8.4% - 10.4%; Pitt *et al.*, 1997). These observations are substantiated through the results of short-term intact sediment core assays in which direct measures of sediment P release indicated no significant reduction in P release rates from Alderfen Broad sediment or Cockshoot Broad following sediment removal (Pitt *et al.*, 1997).

Taken collectively these results indicate that although a significant reduction in the P content of surface sediments was consistently achieved following sediment removal the potential for P release from sediments was 're-set' within a few years. However, it should be noted that only a small proportion (mostly <5% over the range of sediment TP concentrations encountered) of the sediment P pool would be required to raise the water column TP concentration significantly in the broads. To provide a simple illustration of this we used sediment TP profile data collected in 2012 from Barton Broad to model the relationship between sediment TP concentration in the upper sediment layers, the mass of TP in the upper 10 cm of bed sediments, and the percentage of bed sediment TP in the upper 10 cm that would be required to affect an increase in water column TP concentration of 0.1 mg l⁻¹ (Figure 4.2.3-3).

Conclusion proposed [Confidence rating high]- Sediment removal will reduce the internal loading *potential* of bed sediments through reductions in sediment TP concentrations but these effects will be short lived (typically < about 3 years).

Strength of evidence [high] - moderate reductions in surface sediment TP were reported following most sediment removal events which corresponded to a decrease in the potential for internal loading. However, the effects of this reduction were short lived and varied within and between broads, most probably as a result of in-lake processes acting to 're-set' surface sediment TP conditions, and as a reflection of within and between lake variation in sediment characteristics.

Level of agreement between co-authors [high] - The effects observed were consistent across the treated broads and were short lived (i.e. < about 3 years).

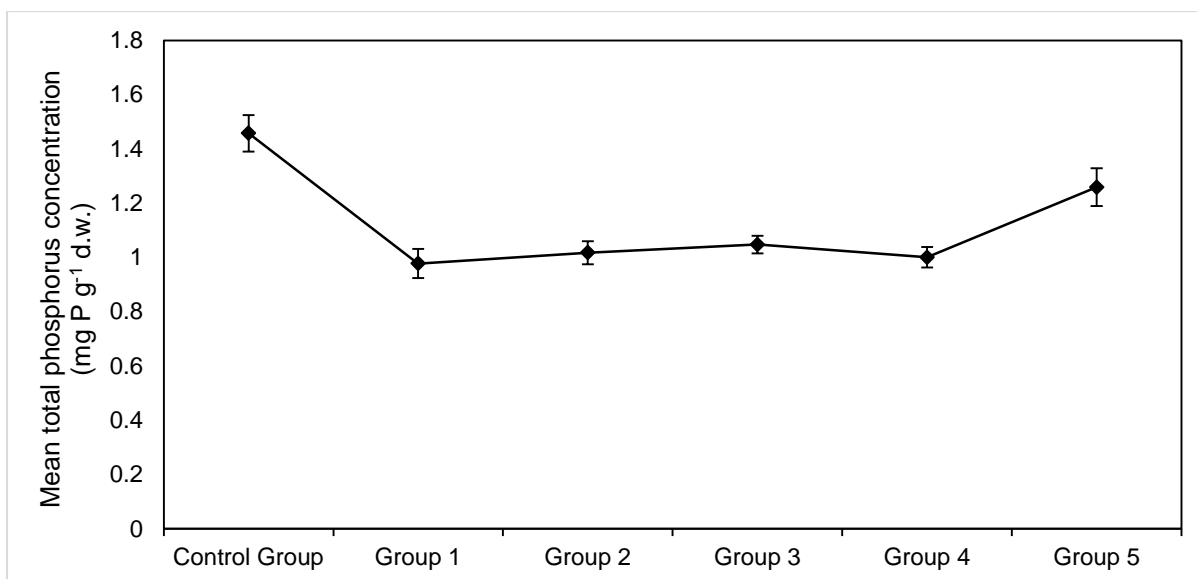


Figure 4.2.3-2 Mean sediment TP concentrations at sample depth 0-5 cm for each group (control group = 'untreated'; groups 1-5 = 'sediment removed'). Error bars show standard error of the mean with counts for each data point ranging from n = 29 to n = 60. Time after dredging is 0-5, 5-10, 10-17, 17-27, 27-37 months for groups 1-5 respectively.

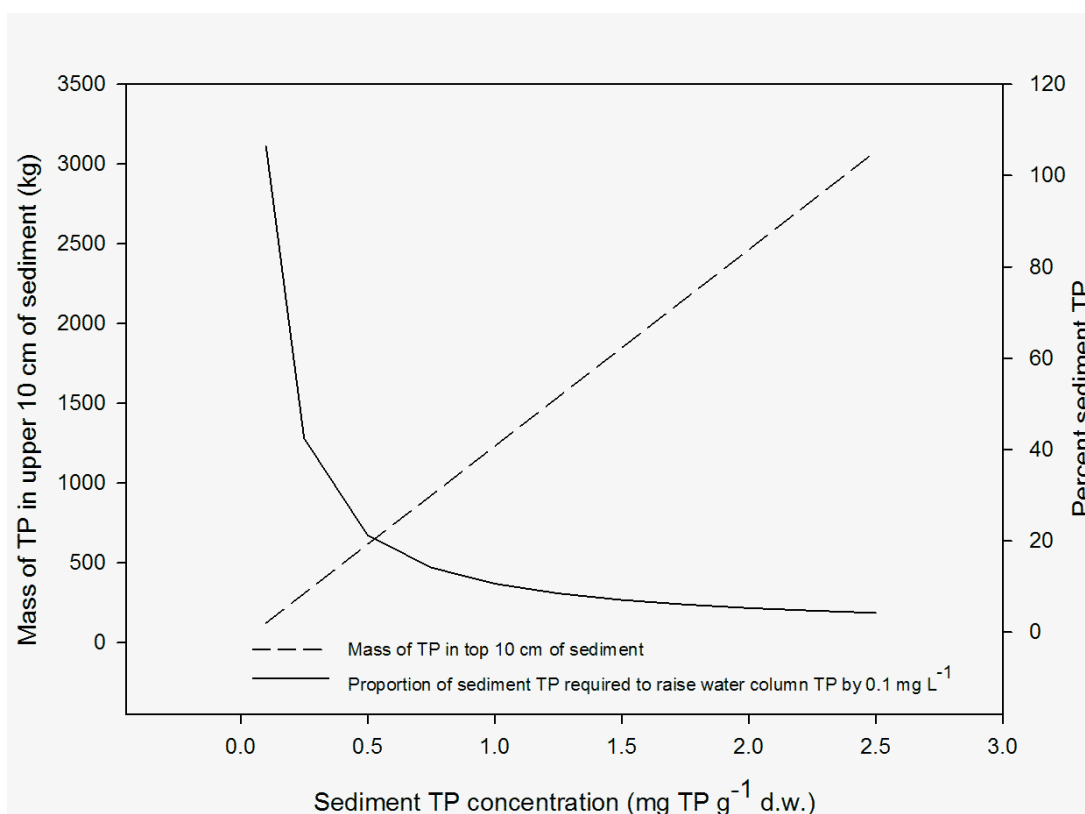


Figure 4.2.3-3 Modelled relationship between sediment total phosphorus concentration, the mass of sediment TP in the upper 10 cm of sediment, and the proportion of that mass that if released to the water column would result in an increase in water column TP concentration of 0.1 mg l⁻¹ (based on data from Barton Broad).

4.2.2.2 *Water column phosphorus reduction*

Reductions in annual mean TP concentrations were consistently reported following sediment removal in Barton Broad, Alderfen Broad, Cromes Broad, Hoveton Little Broad and Cockshoot Broad. However, these reductions should be viewed in the context of the many factors that complicate the determination of cause-effect (Table 4.2.2-2). The annual mean TP concentrations in these sites following sediment removal did not fall well below about $50 \mu\text{g L}^{-1}$ for a sustained period with the exception of a single year in Cromes Broad (i.e. 2007, 2 years following sediment removal activities in the North basin). Stable TP concentrations were only apparent within Alderfen Broad following sediment removal. However, the magnitude of change in Alderfen Broad (i.e. from about $900 \mu\text{g l}^{-1}$ at the time of sediment removal to about $100 \mu\text{g l}^{-1}$) was also significantly greater than at the other sites (in which TP concentrations prior to sediment removal were less than $150 \mu\text{g l}^{-1}$). Inter-annual variability in TP concentrations following sediment removal was similar and pronounced in all other broads. The order of the intensity of the response in annual mean TP concentration (based on concentration before to concentration in the three years after dredging), from weakest to strongest, was Cromes Broad South (1988; $100 \mu\text{g l}^{-1}$ in the year before to a minimum of to $75 \mu\text{g l}^{-1}$ after sediment removal) Barton Broad ($100 \mu\text{g l}^{-1}$ to $60 \mu\text{g l}^{-1}$), Hoveton Little Broad ($80 \mu\text{g l}^{-1}$ to $40 \mu\text{g l}^{-1}$); Cockshoot Broad ($160 \mu\text{g l}^{-1}$ to $60 \mu\text{g l}^{-1}$); Cromes Broad North (2005; $150 \mu\text{g l}^{-1}$ to $25 \mu\text{g l}^{-1}$); and Alderfen Broad ($900 \mu\text{g l}^{-1}$ to $75 \mu\text{g l}^{-1}$).

A general reduction in the ratio of summer to winter TP concentration is expected where a reduction the intensity of internal loading relative to catchment loading has been achieved following sediment removal. This was not obviously the case in Barton Broad where summer to winter TP ratio remained stable at about 2 following sediment removal indicating persistent internal loading (**Error! Reference source not found.**). However, reductions in summer to winter TP ratios were observed for a period of about 5 years following sediment removal in Alderfen Broad, Cockshoot Broad; Hoveton Little Broad and Pound End (after data only). Also, following dredging of Cromes Broad North in 2005, reductions in summer to winter TP concentrate ratio were observed in Cromes Broad South which is downstream of the North basin. We should clearly consider these responses in the context of the potentially confounding factors listed in Table 4.2.2-2. However, these results taken collectively indicate a reduction in the intensity of summer mean TP concentration relative to winter mean TP concentration and, therefore, a reduction in the relative intensity of internal loading following sediment removal for a period of about 5 years in these broads. In all cases, with the exception of Barton Broad, the summer to winter mean TP concentration ratio fell below 1 at some point during the 5 years following sediment removal. The order of the intensity of the response from weakest to strongest was Barton Broad (no obvious response), Cockshoot Broad (1.5 before to minimum of 0.8 after sediment removal) Hoveton Little Broad (2 to 0.95), Alderfen Broad (2 to 0.5), Cromes Broad North (2005; 2 to 0.4); and Cromes Broad South (1988; 2 to 0.4).

Although an apparent reduction in the intensity of internal loading was reported using summer to winter mean TP concentration ratios across most sites from which sediments were removed the absolute annual mean TP concentrations did not fall to values that would support full ecological recovery.

Table 4.2.2-2 Evidence for and against drawing conclusions of a reduction in annual mean TP concentration for broads from which sediment has been removed. The apparent time-lag from sediment removal to the onset of annual mean TP reduction is reported in brackets.

Broad	Evidence for	Evidence against
Barton (1-3)	Reduction in mean annual TP following period of sediment removal	<p>TP trend in broad was not completely stable prior to sediment removal and summer: winter TP ratio unchanged following management. Other catchment actions taken over same period to reduce P inputs</p> <p>Similar change of TP concentration in river upstream of the broad suggesting catchment actions were significant</p>
Alderfen (4-5)	<p>Trend of TP shifts to a downward trajectory following P removal</p> <p>TP has been low and relatively constant since 2000</p>	<p>This response may have been caused by the influence of low water table which probably continued until the marked increase in river flows in 1998/1999. Although any potential mechanisms are unclear CI is elevated during this period & fluctuates in a similar way to TP. Elevated CI in peat from Catfield Fen has been reported (Giller and Wheeler 1986) and these peaks probably indicate continued low water levels which might have allowed for continued P release from oxidising sediment exposed following sediment removal.</p>
Cromes (1-3)	TP in Cromes South significantly lower after mud removal from North Broad. As North flows into the South this could be caused by the sediment removal	<p>Increase of TP following initial sediment removal from South Broad in 1988. However, broad still occasionally linked to R. Ant until 1992. Original management allowed some flushing of broad from R. Ant which could have reduced P levels prior to isolation in 1992 and thus account for subsequent increase in TP, particularly as S. broad is influenced by N. broad. Change in loading from catchment including domestic waste has not been considered in detail.</p>
Hoveton Little (2)	TP lower following P removal	<p>Reduced TP at least as likely to be a result of a decrease already established by catchment control of P, particularly as trajectory of reduction of TP does not show any clear evidence of change.</p> <p>Paired comparison of summer TP levels in Hoveton Great and Hoveton Little broads before and after sediment removal in the latter show no evidence of effect (see Annex: Hoveton Broads dossiers)</p>
Cockshoot (1)	TP reduced following sediment removal	<p>Impossible to determine if change is a result of isolation or sediment removal. Additionally input of river water from tidal flooding may confuse effect. Episodes of catchment loading from arable land not considered.</p>

Conclusion proposed [confidence rating medium - Sediment removal will not result in a reduction in water column TP to a point at which macrophytes would be expected to respond positively (i.e. $< 50 \mu\text{g l}^{-1}$). The beneficial effect on water column TP is least likely to be seen in well flushed broads linked to the river system as these systems will tend to reach a new equilibrium between internal and external load relatively rapidly and will also be influenced by interactions between sediment and water in the wider river system.

Strength of evidence [low] - Although a reduction in TP often followed sediment removal, there was a lack of consistency in the forms of the responses observed and responses rarely ended in stable reduced TP conditions. Other management activities or climate differences may have confounded response detection in some cases. However, a reduction in summer to winter mean TP concentrations indicating a reduction in the intensity of internal loading following sediment removal was most obvious in isolated broads, although it also occurred across most treated broads.

Level of agreement between co-authors [high] - Even if the reported reductions in TP were a direct response of sediment removal, the minimum TP concentrations following dredging were rarely lower than $50 \mu\text{g l}^{-1}$ for consecutive years and in most cases TP concentrations increased again within 5 years.

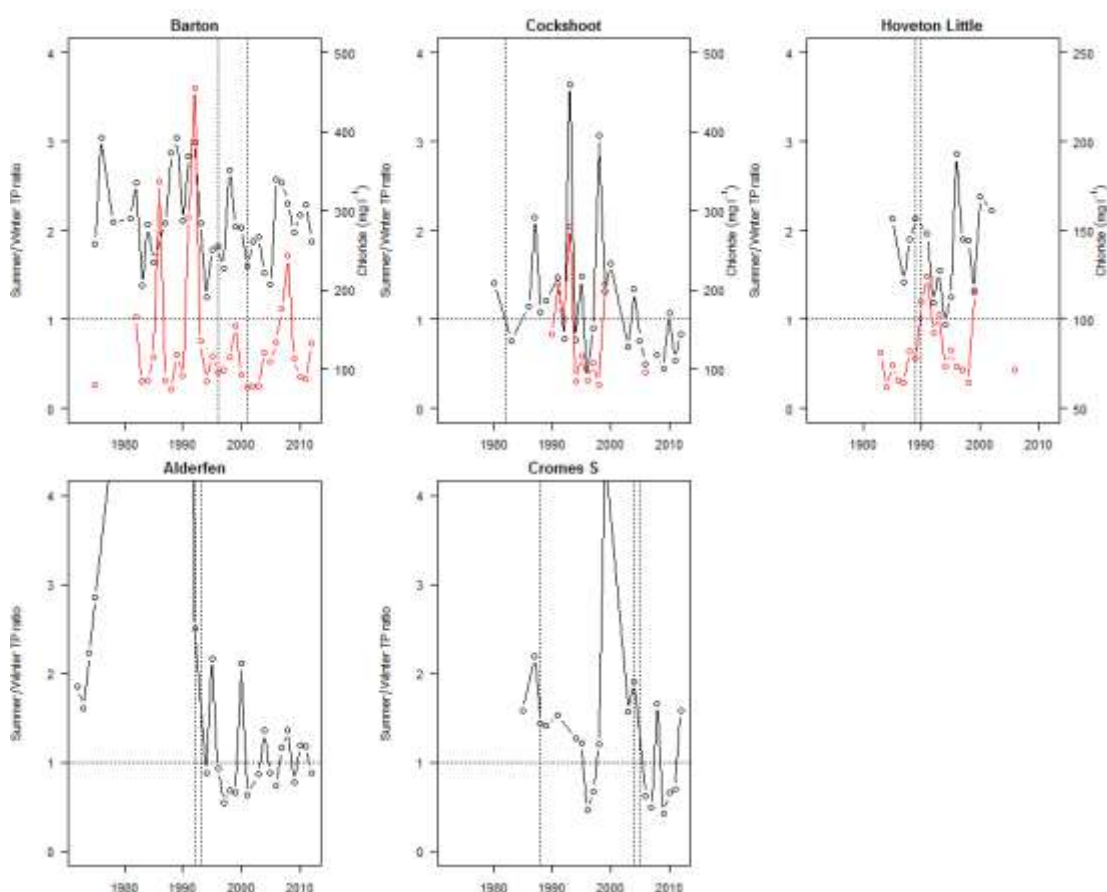


Figure 4.2.2-1 Changes in the ratio of mean summer/winter total phosphorus in broads that have had sediment removed. Vertical lines mark period of sediment removal. For riverine broads chloride concentration also shown (red line) to indicate periods when downstream river water enriched with phosphorus is likely to have increased winter TP concentration.

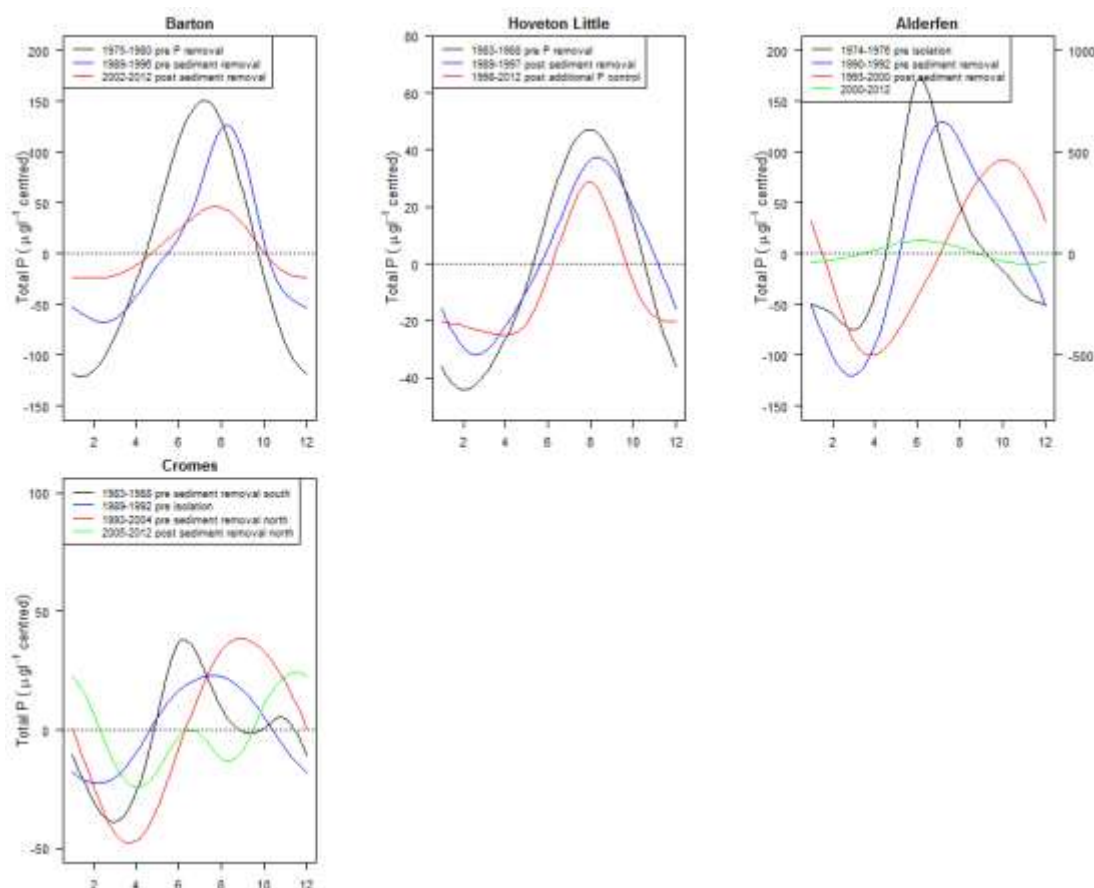


Figure 4.2.2-2 Seasonal pattern of total phosphorus concentration in broads that have had sediment removal in addition to other restoration actions. Lines are GAM smooths ,values of TP expressed relative to the average for the period (Note Alderfen 1990-1992 is plotted on scale shown on right).

4.2.2.3 Water column chlorophyll a concentration

In broads where the phytoplankton community biomass is constrained predominantly by P-limitation, chlorophyll a concentrations should reduce in a manner similar to that described for TP above following the reduction in intensity of internal loading. Again, this should correspond with a reduction in summer to winter chlorophyll a concentration ratios and an overall reduction in the annual mean chlorophyll a concentration, where internal loading was the dominant source of P before sediment removal.

A general reduction in annual mean chlorophyll a concentration was observed in Barton Broad, Cockshoot Broad, Hoveton Little Broad, and Hoveton Little Broad 'Pound End' whereas a general increase was observed in Alderfen Broad, Cromes Broad South and Cromes Broad North following sediment removal. Where annual mean chlorophyll a concentration decreased, the minimum concentrations reached following sediment removal ranged between about 10 $\mu\text{g l}^{-1}$ to 40 $\mu\text{g l}^{-1}$. The apparent form of the response varied across the lakes where a short term (i.e. return to conditions before sediment removal within about 5 years) reduction was reported in Cockshoot Broad, a moderate term reduction was reported in Hoveton Little Broad (i.e. return within about 15 years) and apparently stable

moderate to long-term reductions were reported in Barton Broad (i.e. up to 12 years) and Hoveton Little Broad 'Pound End' (i.e. up to 20 years). However, the response in Barton Broad is perhaps atypical given annual mean chlorophyll a concentrations were steadily decreasing for the duration of the time series, that is, there was no obvious change in the long-term trend following sediment removal. The order of the intensity of the response from strongest increase to strongest reduction (comparing the annual mean concentration in the year prior to sediment removal, to the annual mean in the 3 years after sediment removal) was Cromes Broad South (1988; 10 $\mu\text{g l}^{-1}$ before to minimum of 35 $\mu\text{g l}^{-1}$ after sediment removal), Alderfen Broad (10 $\mu\text{g l}^{-1}$ to 20 $\mu\text{g l}^{-1}$), Cromes Broad North (2005; 15 $\mu\text{g l}^{-1}$ to 25 $\mu\text{g l}^{-1}$), Hoveton Little Broad (75 $\mu\text{g l}^{-1}$ to 40 $\mu\text{g l}^{-1}$), Barton Broad (50 $\mu\text{g l}^{-1}$ to 20 $\mu\text{g l}^{-1}$), Cockshoot Broad (60 $\mu\text{g l}^{-1}$ to 18 $\mu\text{g l}^{-1}$), and Hoveton Little Broad 'Pound End' (75 $\mu\text{g l}^{-1}$ in year 1 following sediment removal to 10 $\mu\text{g l}^{-1}$ the following year).

In those sites where a decrease in annual mean chlorophyll a concentration was reported above a decrease in the intensity of chlorophyll a concentrations between July and October (i.e. the period when internal loading is expected to be most intense) were also observed. This is with the exception of Cromes Broad North where the reduction occurred predominantly in concentrations between February and May (i.e. clear water phase).

Conclusion proposed [confidence rating low] - Sediment removal will not result in a reduction in water column chlorophyll a concentration to a point at which macrophytes would be expected to respond positively (i.e. < 30 $\mu\text{g l}^{-1}$).

Strength of evidence [low] – Although a reduction in chlorophyll a concentration was observed in some broads to below 30 $\mu\text{g l}^{-1}$ following sediment removal, increases were observed in others. In those broads where chlorophyll a concentration was reduced there was inconsistency in the forms of the responses observed where reductions occurred in both summer-autumn periods and spring. Responses rarely ended in stable reduced chlorophyll a concentrations. The periods over which chlorophyll a concentrations remained low were, however, longer than those reported for sediment and water column TP. Other management activities (catchment nutrient load reduction and fish community manipulation, in particular,) or climate differences, as well as variation in the efficiency of sediment removal between sites, may have confounded response detection in some cases. Section 4.2.2.4 suggests that relatively significant reductions in chlorophyll concentrations will be required to compensate for the effect of increasing depth on the submerged light climate. Confidence that sediment removal alone can generate reductions of chlorophyll concentrations on the scale required is therefore low.

Level of agreement between co-authors [high] – The variation in responses makes it extremely difficult to conclude that consistent reductions in chlorophyll a concentration were achieved as a direct result of sediment removal.

4.2.2.4 Increase in macrophyte cover

It is extremely difficult to demonstrate categorically that changes in macrophytes pre- and post-sediment removal are (or are not) a direct response to this intervention because many of the sites that have been dredged have also been subject at the same time, or subsequently to biomanipulation or fish kills, while there have been ongoing reductions in TP

concentrations due to a declining external load. Adding to the problem of identifying cause and effect comparable untreated sites that are suitable as controls rarely exist, while climatic differences in the years immediately pre- and post-sediment removal material may have delayed the timing or magnitude of the macrophyte response.

An examination of changes in macrophyte assemblages pre- and post-sediment removal in the smaller broads and ponds may prove revealing as such sites are less prone to confounding effects (e.g. through being hydrologically isolated or either lacking fish altogether or having fish communities that have not been biomanipulated), though some comparisons are frustrated by the lack of baseline survey data. Data is summarised in Table 4.2.2.3. Mautby, Barnby, Calthorpe and Little Broads all appear to have experienced an increased abundance of macrophytes in the post relative to the pre-dredging period. However, as many broads (Burntfen, Nortons, Wheatfen and Strumpshaw) have shown no obvious increase in macrophyte abundance.

An increase in filamentous algae is seemingly a common response to sediment removal having occurred in at least half of the broads treated. Filamentous algae are highly opportunist and are very effective at exploiting shallow, potentially clear water conditions before macrophytes can establish (e.g. due to delays imposed by lack of propagule bank). Filamentous algal growth might also be favoured in recently dredged sites due to a temporary reduction in herbivory caused by the depletion of benthic invertebrates (and possibly also fish). It is unclear from the available data if filamentous algae decline over time due to shading by canopy forming macrophytes, or nutrient competition, or as a result of re-establishment of grazer populations.

One of the obvious and desired consequences of sediment removal is to increase water depth. The range of depth increases that have been achieved in the broads by sediment removal are typically from depths of 0.6-0.7m up to 1.0-1.2m. In an environment where macrophytes are potentially light-limited due to shading by phytoplankton, increases in water depth will have adverse consequences for macrophyte growth if the new lake depth lies deeper than the compensation depth as defined by the light extinction coefficient. If the lake depth pre-dredging is already close to the compensation depth this implies that light transparency post-dredging must be increased to preserve net macrophyte growth (or the potential for macrophyte growth if none previously occurred).

To make an assessment of the potential impact of sediment removal on light availability the relationship between the Extinction Coefficient E and chlorophyll a has been derived using available data from four broads (Figure 4.2.2-3). The relationship is significant, although relatively uncertain as non-algal material (including suspended inorganic material and pigments), will influence light attenuation. However, it allows an estimate to be made of the influence that chlorophyll has on the loss of light with depth.

Macrophytes are generally assumed to require a minimum of 4.5% of surface irradiation for growth (Søndergaard *et al.*, 2013). Using the relationship in Figure 4.2.2-3 the maximum chlorophyll concentration that on average would provide 4.5% of surface light in the Broads is shown in Figure 4.2.2-4. For a typical Broad with a mean depth of 1.5 m the predicted value is $35 \mu\text{g l}^{-1}$ which is similar to the critical annual mean value suggested in section 3.2.

Table 4.2.2-3 Key information on mud-pumping activities and the response of aquatic macrophytes for less intensively monitored broads.

Site	Size (ha.)	Mud- pumping date	Mud- removal extent	Connectivity	TP before mud- removal (\square g/L)	Macrophytes Annual average species richness/pool/ abundance BEFORE	Dominant plants BEFORE	Macrophytes Annual average species richness/pool/ abundance AFTER	Dominant plants AFTER	Succe ss?	Notes
Barnby Broad	2.51	1990 (south $\frac{1}{2}$) & April 2007 (north $\frac{1}{2}$)	all area by 2007	Isolated	145 *1	rich 1.5, pool 3, ab moderate (no. yrs=4)	<i>Ceratophyllum demersum</i> in previously pumped area south (2003-2006)	rich 7, pool 9, ab high (no. yrs=4)	<i>Ceratophyllum demersum</i> , <i>Chara</i> spp., filamentous algae (2007, 2009)	YES	Plants still absent in shallow margins, but high abundance of <i>Ceratophyllum demersum</i>
Buckenham Broad	1.78	1980	all area	Isolated	No data	No data	No data	rich 6.45, pool 15, ab moderate (no. yrs=15)	Filamentous algae + <i>Ceratophyllum demersum</i> (1998, 2001-2014), <i>Myriophyllum verticillatum</i> (2010- 2014)	YES	TP 75 \square g/L for inflow to Buckenham system 2005-2006 (n=11 - BA data), bream removed late 1990s
Buckenham channel near house	0.4	1980- check	all area	Isolated	No data Before but 87 in 1999/2000 *2	No data	No data	rich 9, pool 17, ab high (no. yrs=16)	Filamentous algae (1998-2013), <i>Ceratophyllum demersum</i> (2006- 2012), <i>Myriophyllum verticillatum</i> (2009- 2014)	YES	TP 75 \square g/L for inflow to Buckenham system (2005-2006 n=11), macrophyte community dynamic between years, although filamentous algae often abundant,

											broad quite deep c. 2 m, bream removed late 1990s
Burntfen Broad	4.63	Oct. 2009	all area	Isolated	179 * 4 79 * 5 116 * 1	rich 4, pool 6, ab low (no. yrs=2)	<i>Nuphar lutea</i> fringe (1998, 2006)	rich 3, pool 5, ab low (no. yrs=2)	<i>N. lutea</i> fringe (2010, 2013)	NO	Turbid, largely plant-free lake pre- and post-pumping, with high TP - 152 µg/L for 2012 - EA data)
Calthorpe Broad	1.1	Feb. 2009	Partial not north end	Isolated	108 * 6	rich 1, pool 1, ab high (no. yrs=2)	<i>Nupha lutea</i> covering all broad (1998, 2003)	rich 5.75, pool 12, ab moderate (no. yrs=4)	<i>Chara hispida</i> (2010-11), <i>Potamogeton natans</i> (2012, 2014)	YES	Broad no longer a <i>Nuphar lutea</i> monoculture which had been response to 1970s acidification events – No fish currently, TP 52 µg/L for 2010-2011, n=13 (NE data)
Catfield Broad	1.27	Unknown	Unknown	Isolated	67 * 1	rich 6, pool 8, ab low (no. yrs=2)	Filamentous algae, <i>Lemna minor</i> , <i>Lemna minuta</i> (1998, 2004)	rich 7, pool 7, ab low (no. yrs=1)	Filamentous algae + <i>Ceratophyllum demersum</i> (2009)	NO?	More chemical and macrophyte monitoring urgently needed to detect longer term response
Cromes North Broad	2.3	Nov. 2005 (South 1998)	all area	Isolated	141 * 7 249 * 2	rich 3.5, pool 4, ab moderate (no. yrs=2)	Filamentous algae, <i>Ceratophyllum demersum</i> (1998, 1999)	rich 7.8, pool 16, ab high (no. yrs=5)	Filamentous algae + <i>Ceratophyllum demersum</i> (2009-2013), <i>Utricularia vulgaris</i> (2013)	YES	Diversity increased. Data not separated from South in most years, only North data presented. Phoslock applied March 2013
Hassingham	2.72	2000	all area	Isolated	113 * 3	rich 2, pool 2, ab	Filamentous algae	rich 5.46, pool	Filamentous algae (2001-2013),	YES	Excellent response with consistent <i>Chara</i> -

Broad						low (no. yrs=1)	(1998)	15, ab high (no. yrs=15)	<i>Ceratophyllum demersum</i> (2001-2012, 2014), <i>Chara globularis</i> (2006-2013)		dominance starting 6 years after sediment removal, although filamentous algae abundant
Little Broad	1.3	Mar. 2008	All area	Isolated	No data	rich 1, pool 1, ab low (no. yrs=1)	Filamentous algae (2005)	rich 3.25, pool 8, ab high (no. yrs=4)	<i>Chara hispida</i> (2009-2010), filamentous algae (2013)	YES?	<i>Chara</i> -dominance 2009-2010. More chemical and macrophyte monitoring urgently needed to detect longer term response – concern over filamentous algae dominance and accelerated siltation in 2013
Mautby Decoy	2.5	Dec. 2008	Partial not east end	Isolated	89 *1	rich 6, pool 9, ab moderate, macrophytes confined to deeper eastern end (no. yrs=2)	<i>Ceratophyllum demersum</i> , <i>Chara globularis</i> , filamentous algae (2007-2008)	rich 3, pool 4, ab moderate (no. yrs=2)	Filamentous algae + <i>Chara globularis</i> (2009-2010)	NO	Low diversity and only moderate macrophyte abundance. More chemical and macrophyte monitoring urgently needed to detect longer term response
Ormesby Broad East Arm	c. 2	Dec. 2010	Substantial portion of E. arm (rest of broad untouched)	Isolated	57	rich 11, pool 11, ab high (no. yrs=1)	<i>Zannichellia palustris</i> , <i>Potamogeton pusillus</i> (2010),	Rich 13, pool 13, ab high (no. yrs=1)	Filamentous algae, <i>Potamogeton pusillus</i> , <i>Potamogeton pectinatus</i> (2011)	YES	Quick recovery and expansion of species pool – bringing more species into Ormesby system (e.g. <i>Myriophyllum verticillatum</i>)
Norton's	0.67	Oct. 2008	All area	Connected	53 *1	rich 0.5, pool 1,	Filamentous algae	rich 2, pool 4,	Filamentous algae,	NO	Very poor macrophyte colonisation, low

Broad						ab low (no. yrs=2)	(1998, 2004)	ab low (no. yrs=3)	(2009-2011), <i>Elodea</i> spp. (2010-2011), <i>Callitriche</i> sp. (2009, 2011)		abundance
Sotshole Broad	0.77	2010/11	All area	Connected	310 *1	rich 2, pool 2, ab high (no. yrs=1)	<i>Nuphar lutea</i> (1998)	No data	No data	No data	
Strumpshaw Broad	3.71	1983/ 1984	All area	Isolated but tidal influence	174 *2	no data	No data	rich 6.44, pool 18, ab moderate (no. yrs=9)	Filamentous algae, <i>Utricularia vulgaris</i> , <i>Najas marina</i> (1999, 2005-2008), Filamentous algae (2010-2013)	NO	Currently dominated by filamentous algae - might be response to saline incursion in 2010
Upton Little Broad	1.37	Sept. 2011	All area	Isolated	Can assume low, neighbouring Upton Great Broad <40	rich 2.4, pool 4, ab variable (high-low) (no. yrs=5)	<i>Najas marina</i> (2005-2007, 2009- 2010)	rich 3, pool 5, ab low (no. yrs=3)	<i>Chara vulgaris</i> . <i>Chara contraria</i> (2012), <i>Chara</i> <i>hispida</i> (2013-2014)	YES	Despite low diversity, site dominated by <i>Chara</i> and continued presence of <i>Najas</i> <i>marina</i>
Wheatfen Broad	4.16	1995 (main broad), 2002 (Mystery Pool), 2009 (channels)	All area In stages	Isolated but tidal influence	1150 *1	no data	no data	rich 6.8, pool 13, ab low (no. yrs=5)	Filamentous algae (1998), <i>Ceratophyllum</i> <i>demersum</i> + <i>Sparganium</i> <i>emersum</i> (2004- 2007, 2012)	NO	Now heavily silted and submerged macrophytes declining. Maybe could be judged as successful over 2004- 2007. More recent data required.

Average species richness excludes filamentous algae. Legend: rich = average annual macrophyte species richness for period (excluding filamentous algae), pool = total aquatic plant species pool for period, no. yrs = number of years represented by the data, *1 = data point for August, 1998 (see Sayer *et al.*, 2008), *2 = average data for 1999/2000 (Sayer *et al.*, 2010), *3 = C. Sayer unpublished data point for 4/11/2000, *4 = EA average data for 1993, *5 = EA average data for 1994, *6 = Natural England data for 15/12/2003-15/12/2004 (n=13), *7 = EA data for 1993-1994 (n=20).

The relationship can also be used to calculate the decrease in chlorophyll required to maintain 4.5% of surface irradiation at the sediment if the depth of the broad is increased (Figure 4.2.2-5). This suggests that if the broad depth was increased on average by 0.3 m from a starting depth of 1.0m the chlorophyll would need to be reduced on average by 23 $\mu\text{g l}^{-1}$ to maintain a minimum of 4.5% surface light.

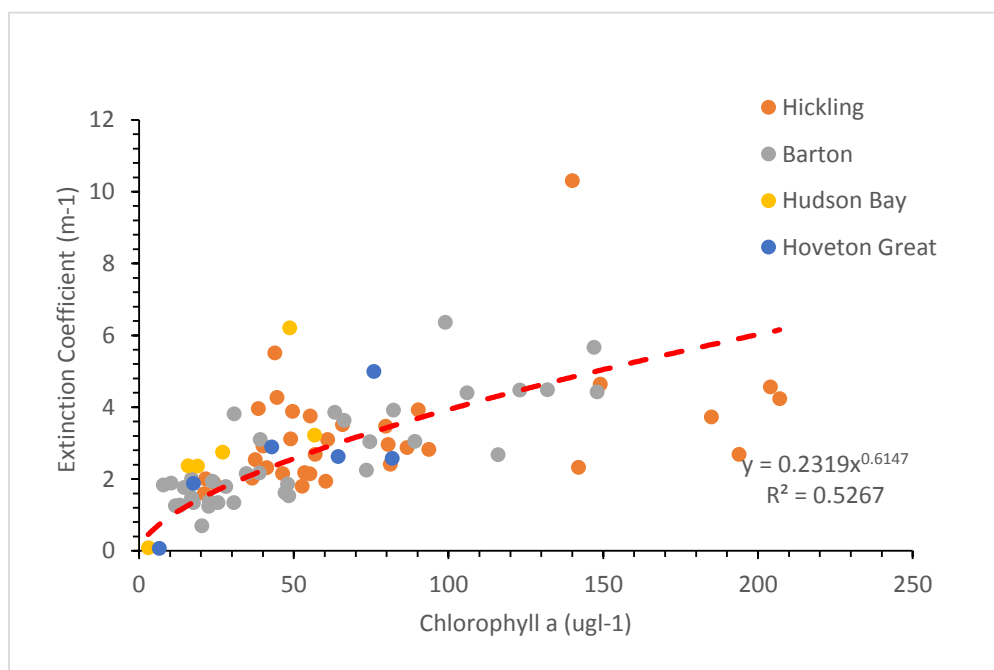


Figure 4.2.2-3 Relationship between Extinction Coefficient (m^{-1}) and Chlorophyll a concentration for selected broads

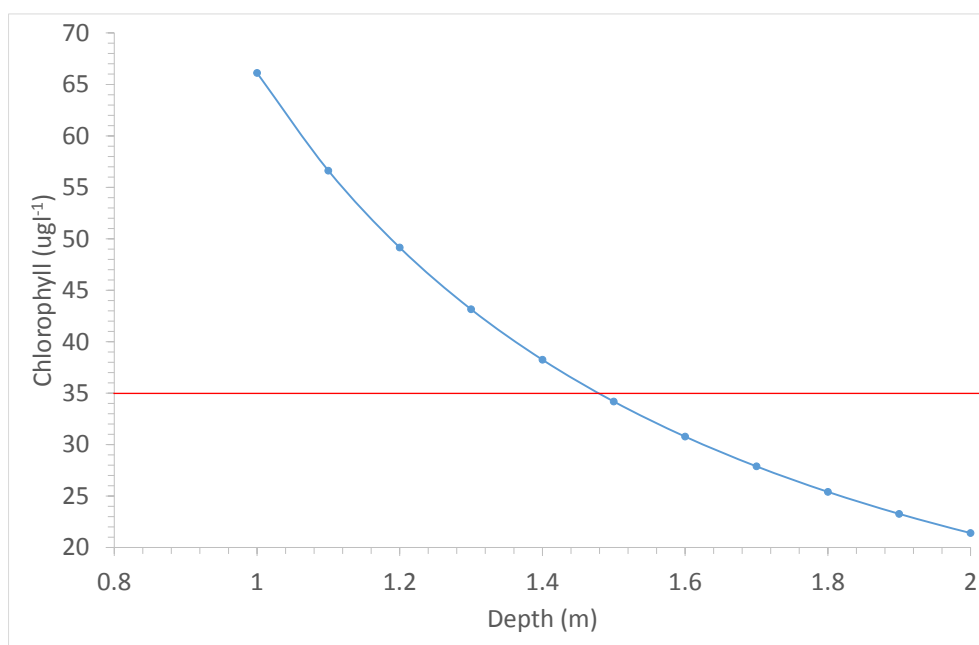


Figure 4.2.2-4 Relationship between maximum chlorophyll concentration which would provide 4.5% of surface irradiation at sediment surface for different depths. Derived using the relationship between Chlorophyll and Extinction Coefficient shown in Figure 4.2.2-3

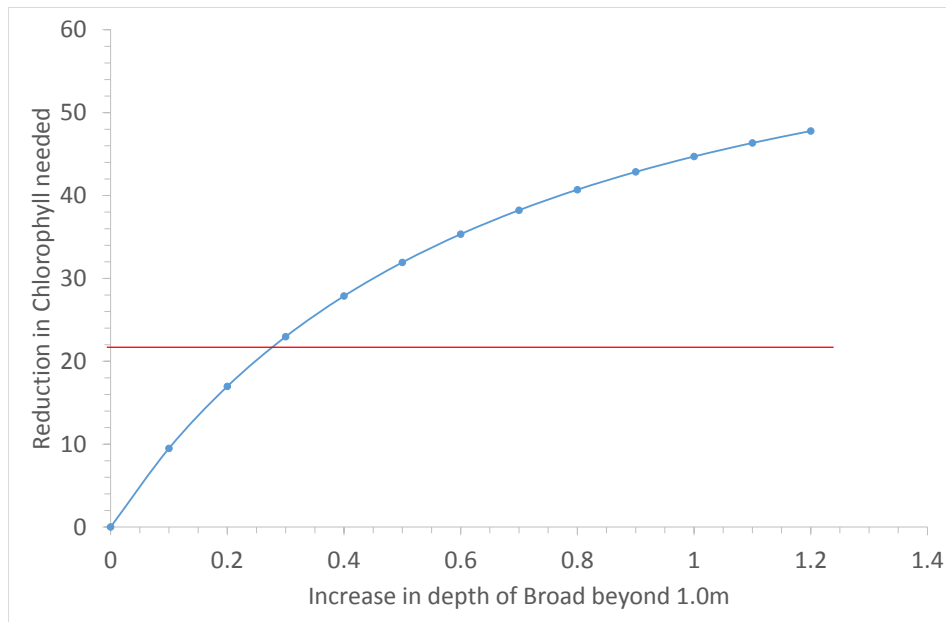


Figure 4.2.2-5 Relationship between the reduction in chlorophyll a needed to maintain 4.5% of surface irradiation at bottom of broad, assuming initial depth was 1.0m. Derived using the relationship between Chlorophyll and Extinction Coefficient shown in Figure 4.2.2-3

Table 4.2.2-4 Calculated percent surface irradiance at different depths in Hoveton Great Broad based on light extinction data collected in 2014 by Stephen Lambert

Date	28.05	16.07	14.08	22.09	22.1	27.11	8.12
Extinctn.	2.09	3.06	5.6	2.66	2.49	2.13	0.08
Depth(m)	Percentage Surface Irradiation						
0	95.0	95.0	95.0	95.0	95.0	95.0	95.0
0.1	77.1	70.0	54.3	72.8	74.1	76.8	94.2
0.2	62.5	51.5	31.0	55.8	57.7	62.0	93.5
0.3	50.7	37.9	17.7	42.8	45.0	50.1	92.7
0.4	41.2	27.9	10.1	32.8	35.1	40.5	92.0
0.5	33.4	20.6	5.8	25.1	27.4	32.7	91.3
0.6	27.1	15.1	3.3	19.3	21.3	26.5	90.5
0.7	22.0	11.2	1.9	14.8	16.6	21.4	89.8
0.8	17.8	8.2	1.1	11.3	13.0	17.3	89.1
0.9	14.5	6.0	0.6	8.7	10.1	14.0	88.4
1.0	11.8	4.5	0.4	6.6	7.9	11.3	87.7
1.1	9.5	3.3	0.2	5.1	6.1	9.1	87.0
1.2	7.7	2.4	0.1	3.9	4.8	7.4	86.3
1.3	6.3	1.8	0.1	3.0	3.7	6.0	85.6
1.4	5.1	1.3	0.0	2.3	2.9	4.8	84.9
1.5	4.1	1.0	0.0	1.8	2.3	3.9	84.3
1.6	3.4	0.7	0.0	1.3	1.8	3.1	83.6
1.7	2.7	0.5	0.0	1.0	1.4	2.5	82.9

Red cells highlight dates and depths where % irradiance <4.5% indicating lake depth lies below the compensation depth.

This demonstrates the influence of depth on light availability and highlights a potentially undesirable side effect of sediment removal from broads where depth is already at least a metre.

An illustration of the effect of deepening Hoveton Great Broad on the light received at the bed, assuming no parallel reduction in chlorophyll concentrations is given in Table 4.2.2-4 based on measured extinction coefficients. This highlights the fact that under the current light climate depths of >0.9m are likely to be outside the growing range of rooted macrophytes for most of the growing season (unless they are species with a strongly apical growth form which establish earlier in the growing season when the light climate is more favourable). For relatively low growing species such as charophytes a significant increase in water depth would need to be accompanied by an increase in water clarity in order to maintain the potential for growth in deeper water.

One conclusion of this analysis is that sediment removal should only be carried out where:

1. Pre-dredging chlorophyll concentrations are already low enough to compensate for any effect of increasing water depth, or,
2. There is a high level of confidence that internal load reduction will translate to a sufficient reduction in TP and thus chlorophyll to conserve the light climate, or,
3. If biomanipulation is going to be carried out in parallel to restore grazing of phytoplankton and thus increase light penetration to the bed.

4.2.2.5 Increase in macrophyte diversity

Based on the sparsely monitored sites where pre- and post-dredging survey data is available we can say with reasonable confidence that macrophyte diversity increased in Barnby, Calthorpe, Hassingham and Little Broads. Of the more regularly monitored sites increases also appear to have occurred in Cromes North and South and Ormesby (Eastern arm) broads. In some cases the sediment removal may simply have acted as a form of physical disturbance of the pre-existing vegetation and any subsequent increase in diversity may simply be a reflection of this disturbance rather than the dredging *per se* (i.e. vegetation management alone would have had roughly the same effect). For example, prior to sediment removal, Calthorpe Broad was dominated by *Nuphar lutea*. Sediment removal combined with plant removal served to reduce the dominance of *Nuphar* by uprooting and removing rhizomes. *Chara hispida* and *Potamogeton natans* were amongst the beneficiaries. It is not possible to separate the effects of *Nuphar* removal from sediment removal.

4.2.2.6 Macrophyte community composition

An increased cover of charophytes is a common objective of sediment removal. High *Chara* cover is desirable, partly because it is more indicative of a less nutrient-impacted system (Blindow, 1992), but also because large scale, synchronous and prolific growth of *Chara* is an effective method of nutrient sequestration and will therefore impede phytoplankton growth (possibly reinforced by allelopathic effects on some algal taxa). The following broads all showed an increase in the cover and

diversity of *Chara* species following sediment removal (Mautby, Barnby, Calthorpe, Upton Little, Little). In these instances it is assumed that dredging was adequate to at least partly expose a layer of oospore-rich sediment corresponding to an earlier phase of *Chara* dominance. In some cases other annual species or those with high seed production, such as *Zannichellia palustris*, fine-leaved pondweeds or *Najas marina*, have recolonized successfully after dredging. With the exception of the Pound End basin of Hoveton Little Broad we are not aware of any specific cases where *Najas marina* has appeared in the immediate aftermath of sediment removal having previously been absent (it also now occurs regularly in Alderfen Broad, but only since 2006). *Ceratophyllum demersum* is often one of the commonest species in the post-dredging community. This may in part simply reflect its abundance pre-dredging since *Ceratophyllum* is the most abundant macrophyte in the broads as a whole. However, it may also be that as a species that overwinters via dormant buds or turions, these structures are resuspended by dredging and then resettle at high density on the newly exposed sediment. Other less common species with a similar overwintering strategy that may have benefitted in the same way include *Myriophyllum verticillatum* (in Buckenham Broad, Ormesby Broad) and *Utricularia vulgaris* (in Cromes Broad and Strumpshaw Broad). In one case (Wheatfen Broad) it appears that *Sparganium emersum* increased significantly following dredging. This species often dominates in upper parts of all Broadland rivers, UK canals and weed cut and periodically dredged rivers in Denmark (Baattrup-Pedersen *et al.*, 2002) and the closely related *S. angustifolium* commonly increased in abundance following dredging of ditch systems on the Insh Marshes, Scotland (Willby pers. obs). In Wheatfen broad partial connectivity with the river may account for the increased abundance of *S. emersum* which has increased significantly in all Broadland rivers around the head of the navigation over the past five to ten years (A. Kelly pers. comm.).

4.2.3 Factors constraining effectiveness of sediment removal

Given the many factors likely to regulate internal loading in lakes it is no surprise that the evidence of sediment and water quality responses in the broads suggests lake specific responses following sediment removal. However, identifying the drivers of this variation is difficult. We are faced with a set of broads from which sediment has been removed during periods of significant environmental and management change, some of which have been summarised in Table 4.2.2-2 and sections 2 and 4.3. This is a common issue across similar case studies. The factors likely to confound effectiveness of sediment removal, many of which are relevant to the broads, are discussed in detail by Peterson (1982), Gulati *et al.*, (2008) and Søndergaard *et al.*, (2007). These factors are discussed below in the context of responses reported above for the broads.

Persistent catchment loading is a common factor limiting the duration and magnitude of water quality improvements following sediment removal. In our analysis of the broads, we considered evidence of persistent catchment P loading using the winter to summer mean TP concentration. Where winter conditions were higher than, or similar to, summer conditions preceding sediment removal there is a higher likelihood that the main source of TP to the water column is from the catchment, not the

sediment. However, perhaps with the exception of Cockshoot Broad, summer mean TP concentrations were consistently higher than winter mean TP concentrations preceding sediment removal indicating the sediment as being the main source of TP to the water column in the majority of cases. This, of course, comes with caveats, for example, during periods of low water levels when TP concentrations may be atypically high or where in-lake conditions mirror closely these conditions in inflowing rivers. In addition, significant changes in the P loads to the broads before and after sediment removal further confounds our ability to detect cause-effect of sediment removal (e.g. Barton Broad).

Insufficient removal of surface sediment can result in a failure to effectively control internal loading. Two factors are important here. The first is the extent of sediment removal across the surface area of the bed where areas that have not been sufficiently dredged can contaminate those that have (Van der Does *et al.*, 1992). Anecdotal evidence from experiences of the co-authors indicates that, in the Broad, inconsistencies were apparent in the surface area of sediments removed. For example, in Cockshoot Broad it is estimated that about 60% of the surface area of the bed was suction dredged. Similarly, at Alderfen Broad much of the edge sediment was left in place and to a lesser extent, Hoveton Little Broad and its basin 'Pound End' were incompletely dredged. As such, the efficiency of sediment removal and the manner in which it was done may account for some of the variation in the responses observed. These operational issues are not uncommon and have been addressed through the development of precision dredging techniques and post dredging monitoring programmes to ensure sufficient sediment removal, as demonstrated, for example, in Lake Trumen (Bjork *et al.*, 2010). The second issue of sediment removal is concerned with the appropriate depth of sediment to remove to ensure a reduction in internal loading. Ideally, this would be based on vertical horizons of sediment P composition where 'release-sensitive' sediment P pools decline with sediment depth. The point at which no further decrease in sediment TP concentration is observed is a useful indicator of the depth below which sediment P release potential will be low. Although this depth varies across lakes, evidence from the comprehensive surveys of Jackson (1991, 1992) and Pitt *et al.*, (1997) indicate that this 'compensation point' occurred at sediment depths of above about 20 cm across the surveyed Broad. In Barton Broad, for example, the sediment depth at which TP concentrations stabilised was 30cm, equivalent to the depth of the marl layer. An average of 43cm of sediment were removed across the entire broad (A. Kelly, pers.comm.). The conservation partners that have undertaken sediment removal in the Broad (mainly the Broad Authority over the past 30 years) have sought to remove all the surface nutrient-rich sediment, using the evidence of the compensation point combined with site specific data to ensure that the historic peat basin remains intact, and damage the littoral margin is avoided. Evidence that the sediment removal efforts were sufficient in this respect is available in the consistent reduction of sediment TP concentration to levels that would indicate a reduction in sediment release potential (i.e. to $< 1 \text{ mg TP g}^{-1} \text{ dw}$) in the broads for which data were available.

The importance of flushing rate in regulating the effectiveness of sediment removal should not be underestimated. Van der Does *et al* (1992) highlighted the lack of

responses in well-flushed lakes following sediment removal. In the Broads, isolation from inflowing rivers had been conducted in Alderfen Broad, Cockshoot Broad and Cromes Broad to some extent prior to sediment removal, although in the case of Cromes the broad had been disconnected from the river for many decades prior to sediment removal, for reasons of flood defence. The simple indicators of the intensity of responses following sediment removal applied to the broads indicated that these isolated broads consistently exhibited the strongest reductions in water column annual mean TP concentration and in the ratio of summer mean to winter mean TP concentration. However, although the largest decrease in annual mean chlorophyll a concentration was observed in Cockshoot Broad, this was not the case in Cromes Broad or Alderfen Broad. This lack of response in chlorophyll a concentration in Alderfen Broad and Cromes Broad is explained through very low chlorophyll a to TP concentration ratios (Chl:TP of about <0.1 before sediment removal), relative to Cockshoot Broad (Chl:TP of about 0.4 before sediment removal). Where chlorophyll a to TP concentration ratios are very low it is unlikely that reductions in TP concentrations in the water column will result in similar reductions in chlorophyll a concentrations. Instead, something other than P is limiting phytoplankton biomass in these broads (e.g. grazing by zooplankton, nitrogen-limitation). The complex nature of the interactions between isolation, biomanipulation and sediment removal processes and their effects on the chemical and ecological responses in Cockshoot Broad are discussed in detail by Moss *et al.*, (1996). It should however, be recognised when comparing these three isolated broads that only Cockshoot was radically altered by isolation due to it being disconnected from the main river.

Some 16 non-monitored broads have been subjected to sediment removal. A lack of data for many of the sites (especially pre-pumping) precludes firm conclusions in many instances. For nine of the broads, based on increases in macrophyte species diversity, abundance and in particular increases in the abundance of characteristic broads species such as *Chara* spp., *Myriophyllum verticillatum* and *Utricularia vulgaris*, sediment removal can be judged as a success; although in many cases further monitoring is needed to confirm these conclusions (especially at Catfield Broad and Little Broad). For six sites, based on a general lack of plant colonisation (low diversity and abundance) after sediment removal, restoration can be judged as unsuccessful. For one site (Sotshole Broad) insufficient data exists to make any kind of a conclusion.

In general sediment removal has been most successful at isolated sites with lower nutrient levels (e.g. Buckenham Broad channel near house, Buckenham Broad, Hassingham, Barnby Broad, Little Broad, Upton Little Broad) and has been generally unsuccessful at sites which were highly eutrophic prior to sediment removal (e.g. Burntfen Broad, Norton's Broad (although TP was low in single available measurement at $53 \mu\text{g l}^{-1}$), Strumpshaw Broad, Wheatfen Broad). In particular Norton's Broad (connected to the Bure) and Burntfen Broad became turbid, phytoplankton-dominated lakes after sediment removal. In Strumpshaw and Wheatfen Broad tidal influence and saline incursions likely also negatively impacted on freshwater aquatic macrophytes. In 2010 it is suspected that an incursion led to a deterioration in macrophytes in Strumpshaw Broad by promoting filamentous algae dominance (Williams, 2011 – Annual macrophyte survey report for 2010).

Some lakes with high TP before mud-pumping also improved following dredging, these being Calthorpe Broad and Cromes North Broad. Calthorpe Broad, which was dredged in 2009, had a high TP of $108 \mu\text{g l}^{-1}$ in 2003-2004. It is a currently fishless lake following fish kills in the 1970s and this likely explains its positive response (transition from a *Nuphar lutea* monoculture to *Chara-Potamogeton natans* dominance), although, it should be noted that TP was also low post-dredging ($52 \mu\text{g l}^{-1}$ in 2010-2011). In addition Cromes North Broad recovered relatively well following sediment removal in 2005, with increases in macrophyte diversity which may have been aided by a significant fish kill in 1999 and limited recolonisation.

Various other factors are likely to constrain the response of macrophytes to sediment removal. The first maybe as simple as 'overdeepening' of a lake relative to light availability, although if this has occurred it is likely that it would only be expected to affect part of a lake. In other cases propagule supply from the sediments may restrict recovery. Although freshwaters often contain a large and diverse sediment propagule bank, elements of which are viable for over 100 years, germination trials suggest that submerged taxa other than charophytes tend to occur very sparingly in the propagule bank (Abernethy & Willby, 1999; Boedeltje *et al.*, 2003).

Connectivity with alternative reservoirs of propagules (e.g. ditch systems), or a high density of dispersal vectors (i.e. waterbirds) may be critical in the re-establishment of aquatic vegetation, both following de-silting or in cases where macrophytes have been lost for decadal time scales. Thus, for example, Ormesby Great Broad eastern arm recovered well likely due to an abundant adjacent species pool in the main broad. In addition the Buckenham-Hassingham system is connected to a well-maintained (de-silted) species-rich dyke system, from which species such as *Myriophyllum verticillatum*, *Hydrocharis morsus-ranae* and *Stratiotes aloides* have been observed to spread into the lakes (Sayer, unpublished data). In Cromes Broad North re-opening of an interconnecting channel with the well-vegetated south basin may have been instrumental in accelerating recolonisation.

Sediment removal is likely to improve the physical and chemical properties of the residual sediment in terms of macrophyte growth because sediment density and cohesion should increase, whilst organic matter content, sulphide and iron, all of which can potentially limit macrophyte growth (Barko & Smart, 1986; Smolders & Roeloffs, 1993), are likely to decrease. However, in some cases concentrations may remain sufficiently high to impede germination of long-lived propagules or their subsequent establishment. Without data on sediment metal or sulphide concentrations pre and post dredging it is difficult to evaluate the relevance of chemical barriers to germination. Bird grazing may limit response of macrophytes to sediment removal but the evidence for this is equivocal. Several authors, including Moss *et al.*, (1996) and Sondergaard *et al.*, (1998) conclude, based on enclosure studies, that waterfowl grazing reduces macrophyte biomass. However, such studies have mostly been based on transplants and only observed significant impacts from the peak of the growing season onwards after which macrophytes start to senesce in parallel with a gradual build up in bird numbers. Thus, Perrow *et al.*, (1997) question the importance of bird grazing during the colonisation and establishment phase of macrophytes, noting that impacts of coot grazing during this phase of the macrophyte growth cycle were negligible compared to late summer and autumn. They also note

that macrophyte establishment occurred successfully in Cockshoot Broad following sediment removal, in the absence of any bird exclosures, whilst in Alderfen and Pound End Broad, macrophyte development was minimal during the study period, regardless of the presence of bird protection. Other studies have reported strong synergistic effects of bird and fish herbivory on macrophytes, with especially marked effects of exotic species such as grass carp and signal crayfish (Rodrigo *et al.*, 2013). The extent to which invertebrate grazers, such as snails, may influence plant establishment, whether by direct herbivory or periphyton grazing, is poorly known. Mesocosm studies (Elger *et al.*, 2006) suggest that snails selectively consume emerging seedlings of the most palatable species that results in the development of a species poor community dominated by less palatable *Ceratophyllum* and *Nuphar*.

4.3 Biomanipulation

4.3.1 Background

Fish are a particularly important stabilising (or de-stabilising) influence on lake trophic structure and functioning through a range of top-down and bottom-up interactions that may cascade through different trophic levels. Important trophic interactions include: i) the selective predation of phytoplankton-grazing large-bodied cladoceran zooplankton by small zooplanktivorous fish (e.g. roach *Rutilus rutilus*); ii) the turnover of bottom sediments by large benthivorous fish (e.g. bream *Abramis brama*) as they forage, thereby releasing nutrients available for algal uptake, re-suspending fine lake sediments and enhancing turbidity or uprooting submerged plants and iii) predation (or prospective predation as predation risk) of zooplanktivorous/benthivorous species by piscivorous fish (e.g. northern pike *Esox lucius* or Eurasian perch *Perca fluviatilis*).

The effects of fish are generally thought to be more intense in shallow rather than deep lakes as fish may reach higher numerical or biomass density per lake volume in shallow lakes where resources are concentrated in the photic zone and fish may easily exploit both benthic and pelagic resources without the need for vertical migration (Jeppesen *et al.*, 1997). In shallow systems particularly, zooplanktivorous/benthivorous fish may prevent the beneficial effects of any nutrient reduction being expressed, especially where nutrient levels remain at a level at which clear macrophyte-dominated or turbid phytoplankton-dominated states exist as alternative stable alternatives. Thus, the manipulation of fish communities, typically termed biomanipulation (although this term originally encompassed a wider range of biological techniques) has been widely used as a restoration tool over the last 30 years. Biomanipulation arguably originated in the US (e.g. Carpenter & Kitchell 1993) before being widely applied particularly in 'cold' lakes in several countries in north-western Europe (see Perrow *et al.*, 1997, Jeppesen & Sammalkorpi 2002, Søndergaard *et al.*, 2007, 2008), and more latterly in 'warm' lakes in tropical China and South America (Jeppesen *et al.*, 2012). The difference between cold and warm lakes is that favourable effects in warm lakes occur via bottom-up processes such as reduced sediment suspension and nutrient release from sediments rather than via top-down control of phytoplankton by zooplankton.

Although often used as a supportive technique to nutrient control of one form or another, such is the pervasive power of well-performed biomanipulation, that Jeppesen & Sammalkorpi (2002) regarded it as the principal and most cost-effective means of shallow lake restoration. Their case studies include favourable effects from small to very large lakes such as Lake Vesijärvi in Finland at 109 km², where fish removal was undertaken at a commercial scale.

A common misconception of biomanipulation is that fish are simply removed, preferably to extinction and are thus perceived as the 'enemy'. In fact, the aim is generally to precipitate a shift to a desirable state, typically one dominated by submerged macrophytes that supports an alternative fish community, that may reach equivalent or higher numerical or biomass density than that originally present. This is theoretically possible as a result of the wider range of available ecological niches in a structured system that may also support higher biomass of a wider range of invertebrates consumed by fish. However, the balance is a fine one in that even fish such as tench (*Tinca tinca*) that are seen to be intrinsically linked to macrophytes, may generate negative effects by consuming molluscan grazers that would otherwise suppress the development of filamentous epiphytic algae that in turn may compete with submerged macrophytes (Brönmark 1985).

Clearly then, it is important to have defined targets of which species are to be removed to what levels, whether this is to be accompanied by stocking of piscivorous species to help maintain pressure on undesirable species, and to have a clear 'road-map' of how the fish community may develop. In the Broads, although there has been considerable research into the fish>zooplankton>phytoplankton interaction (see Townsend *et al.*, 1986, Phillips *et al.*, 1996, Moss *et al.*, 1996, Perrow *et al.*, 1999a), particularly in relation to roach, there is virtually no knowledge of the effects of benthivorous bream. Data from both the Broads and structurally and functionally similar estate lakes ($n=28$ lakes) does show however that benthivorous fish including bream and Common carp *Cyprinus carpio* form a distinct fish trophic guild end group associated with turbid systems without macrophytes and high nutrient concentrations especially nitrate-N (Zambrano *et al.*, 2006).

Thus, in the absence of a thorough understanding of exactly which components of the fish stock are responsible for undesirable conditions, and accepting that fish readily shift diet and foraging strategy, potentially creating a problem where they did not previously, the aim of removal has been to remove the entire population of all species and age groups that conceivably contribute to the undesirable algal-dominated state. Only the principal predatory species such as pike and European eel *Anguilla anguilla* and those species that would hopefully predominate in the future community of the restored lake, such as tench and rudd *Scardinius erythrophthalmus*, have generally been retained. Perch has been problematic in that whilst young fish are often zooplanktivorous (and therefore generally undesirable), older fish are piscivorous and through cannibalism may control recruitment of young zooplanktivorous fish. The realisation that perch may be functionally very important, particularly as an 'open-water chasing piscivore' (as opposed to the 'sit-and wait' ambush tactic of pike, restricting them to submerged vegetation and littoral margins) led to a change of policy for this species (see example of Alderfen below) and in any

subsequent removals not even young perch have been removed in the hope of building an age-structured population of predatory perch.

The removal strategy for biomanipulated Broad has typically involved a combination of: 1) a massive initial removal, 2) manipulation of the remaining potential spawners and 3) subsequent 'top-up' removals especially of young-of-the-year (YOY) recruits or invaders. Manipulation of undesirable species was to continue until self-sustaining populations of appropriate fish species, probably at equivalent biomass to the populations removed, had developed. In fact, the last crucial step has not actually been achieved generally because the attainment of a clear water state with submerged macrophytes has generally taken longer to realise than hoped, thereby falling out of the reach of relatively short-term projects. As a result, subsequent fish manipulation or management has been continued wherever possible, but in a more *ad hoc* fashion where resources allow and where suggested by annual surveys. Moreover, in one notable case, a subtle shift in the political objectives of partner organisations led to fish manipulation being discontinued, risking the successful outcome of the project. The final step of the development of a fish community compatible with the macrophyte-dominated state has thus largely relied on natural processes, which may depend heavily on time, particularly as the generation time for some long-lived fish species may be several years or even decades (see *Factors limiting effectiveness* below).

Full-scale biomanipulation has been undertaken in three whole-lakes: 1) Cockshoot Broad, a 5.5 ha lake isolated from the River Bure with manipulation from 1989-2008); 2) Pound End a 5.1 ha basin isolated from Hoveton Little Broad with manipulation from 1990-1998), and; 3) Ormesby Broad, a 55 ha basin isolated from the wider Trinity system with manipulation from 1995-2008). Partial targeted biomanipulation of Alderfen Broad (a 5.1 ha isolated lake) involving some fish removal and some stocking was undertaken from 1994-2000 inclusive. A number of smaller scale manipulations have been undertaken in experimental systems as demonstrations of the technique for potential application to the wider lake system. These include 1) Hoveton Great Broad, in a 1 ha fish enclosure from 1992-1995; 2) Barton Broad in two fish enclosures to 2.4 ha from 2000 (smaller one ongoing) with a further 2 enclosures to 0.8 ha from 2003 (ongoing), and; 3) Ranworth Broad, a <0.1 ha enclosure manipulated in 2013.

In addition, large-scale fish kill events were recorded at Alderfen Broad in 1991, Cromes Broad north basin (2.3 ha) in 1999 and Upton Broad (5.6 ha) in 1995. Large-scale fish kill is effectively a natural form of biomanipulation and as such, may be a valuable demonstration of the potential effectiveness of the technique and, in particular, how the fish community subsequently responds when left to recover naturally.

4.3.2 Evidence of effectiveness shown

4.3.2.1 Reduction in P

The removal of fish through biomanipulation leads to some reduction in total in-lake P as a result of the loss of P retained in fish bodies. Penczak & Tatrai (1985) calculated that P constitutes 2.7% of the biomass of bream and it seems likely that a similar proportion is present in other fish species. Considering that 15.3 g m^{-2} (Moss *et al.*, 1996–Annex: Cockshoot dossier), 20.3 g m^{-2} (Tomlinson *et al.*, 2002–Annex: Ormesby dossier) and 17.5 g m^{-2} (Phillips *et al.*, 1996–Annex: Pound End dossier) were removed in Cockshoot (from 1989 & 1990), Ormesby (from 1995 & 1996) and Pound End (from 1995-1998) respectively, suggests export of $0.37\text{-}0.55 \text{ g m}^{-2}$ P in fish bodies. Such values appear to be relatively inconsequential in terms of total P load potentially removed by suction dredging or simply present in the water column. Since no removal of sediment was initially undertaken in Ormesby Broad, which also retained lower P concentrations, the contribution of P within the bodies of the fish removed was proportionally higher in this lake than the other two.

Following the major fish kills in Alderfen (1991) and Cromes (1999) there is a suggestion of relatively short-lived increases in P concentration in the water column, although the scale of the increase would suggest that this is more than just the result of the P release from decaying fish bodies. There are no data prior to biomanipulation to observe a P response in Pound End. However, increases in P were also observed in the biomanipulated Cockshoot and Ormesby Broads and these collective experiences suggest the possibility of bioturbation, the pumping of interstitial P in the sediments in the water column, as a result of an increase in chironomid populations following a massive reduction in predation pressure from fish. Monitoring of the Hoveton Great Broad fish enclosure relative to the main lake basin provides some evidence of this mechanism (Phillips *et al.*, 1994).

Overall, evidence from both biomanipulated and fish-kill lakes suggest that a short-term reduction in P is unlikely, with in fact the opposite being observed. In the medium term, it is plausible that P sampled in the water column reduced as P became bound with the biomass of recovering fish and especially macrophyte populations. But, it seems unlikely that concentrations reached have been low enough (i.e. $<50 \mu\text{g l}^{-1}$) to guarantee stability of the desirable macrophyte-dominated and macrophyte diverse state. Thus, in the longer term, if the fish stock continues to recover, the conditions leading to high TP concentrations in the water column could well be re-activated, as was observed in 27 Danish lakes subject to effective biomanipulation (removal of $>200 \text{ kg ha}^{-1}$), Søndergaard *et al.*, (2008). Here, recovery of similar TP concentrations to those before treatment generally occurred after five years.

4.3.2.2 Reduction in chlorophyll a

A large reduction in chlorophyll a and corresponding increase in Secchi depth (often to the lake bed) providing 'gin-clear' conditions appears to be guaranteed in biomanipulated lakes and enclosures in the Broads. This is the commonly observed response to major ($>75\%$) reductions in fish stocks reported elsewhere (Søndergaard

et al., 2007). A particularly compelling demonstration of the reduction of chlorophyll *a* following biomanipulation occurred in Pound End relative to the unmanipulated, but connected (through a fish-proof barrier with 1 mm steel mesh) Hoveton Little Broad sharing the same water supply, thereby providing an experimental 'control'. Immediately following successful wholesale fish removal in the previous year, during 1993 Pound End supported annual chlorophyll concentrations $<35 \mu\text{g l}^{-1}$, reducing to values $\sim 5 \mu\text{g l}^{-1}$ from April to July. In contrast, concentrations in Hoveton Little Broad peaked at $>150 \mu\text{g l}^{-1}$, with values mostly $>30 \mu\text{g l}^{-1}$ between April and July (Philips *et al.*, 1996). A similar enclosure in Hoveton Great Broad showed equally conclusive results, albeit at generally higher chlorophyll concentrations (up to $40 \mu\text{g l}^{-1}$ in the fish enclosure from March-July with a peak around $60 \mu\text{g l}^{-1}$ in August), whereas the concentration in main lake climbed steadily from $\sim 50 \mu\text{g l}^{-1}$ in March to a peak of $250 \mu\text{g l}^{-1}$ in August (Phillips *et al.*, 1994)

In Cockshoot Broad, any effect was less immediate and therefore less obviously attributable to fish reduction, although the decline to annual mean chlorophyll concentrations of $\sim 10 \mu\text{g l}^{-1}$ by 5-6 years after fish removal could not be easily linked to other factors. A subsequent increase to similar levels before biomanipulation after a further 8 years or so appeared to be in response to an increase in the zooplanktivorous fish stock (roach), demanding further removal. Low chlorophyll concentrations then resumed within three years adding weight to the notion of a direct link between chlorophyll *a* and zooplanktivorous fish through the mechanism of size-selective predation of zooplankton. It remains unclear whether the effective extinction of large benthivorous bream by 2000 has had any role in the longer-term attainment of low chlorophyll in Cockshoot Broad.

At Cromes Broad however, a loss of large bream through a fish-kill in 1999 was coincident with the subsequent decline in annual mean chlorophyll of $\sim 30 \mu\text{g l}^{-1}$ to values between $10\text{-}20 \mu\text{g l}^{-1}$. This could represent the 'switch-off' of normal mechanisms whereby P becomes available for algal uptake by transfer to the water column via mobilisation of sediments by bottom-feeding fish (Breukelaar *et al.*, 1994) or by processing through fish bodies. In contrast, a similar fish-kill at Alderfen did not provide any supporting evidence for a meaningful contribution of benthivores to these potential pathways. Here, chlorophyll increased in step with the recovery of zooplanktivorous roach and rudd and only declined to low levels during the last decade (annual median of $12 \mu\text{g l}^{-1}$) as the fish community shifted to rudd, tench and pike in the presence of macrophytes.

In Ormesby Broad, the trend of chlorophyll concentration over time shows no demonstrable effect of biomanipulation. In fact chlorophyll declined substantially prior to fish removal in around 1989, which ironically, may have been caused by a poorly documented fish-kill event. Moreover, all the Trinity Broad's showed reduced chlorophyll before and after the main biomanipulation event and installation of the fish barrier between Ormesby and Rollesby in 1995. Although some fish were transferred from Ormesby to Rollesby, large numbers were also transported to other sites by the Environment Agency leading to a net loss from the system. This was exacerbated by the accidental loss of what was thought to be large numbers of fish over the Muck Fleet sluice around this time, as well as in other years subsequently.

What appears to be a generally biomanipulated system of lakes has thus maintained low chlorophyll (median annual means of 13-15 $\mu\text{g l}^{-1}$ in each basin). There is thus no obvious negative effect of the retention of the benthivorous fish stock in the wider system as a result of the transfer of some bream from Ormesby to Rollesby Broad. Whether the removal of the remaining bream (at least half of the stock was thought to remain in the early 2000s) would lead to a further reduction in chlorophyll in Ormesby relative to the other lake basins, remains uncertain.

4.3.2.3 Increase in macrophyte cover

Following a reduction in chlorophyll concentration via biomanipulation, an increase in macrophyte cover will almost invariably follow, although the extent of cover and the timescale over which this occurs is likely to be variable depending on other factors (e.g. propagule bank, source of colonists, climatic conditions, herbivory by waterbirds; Bakker *et al.*, 2013). Thus delays of 2-8 years and responses varying from zero to dramatic or stable to highly inter-annually variable have been reported (e.g. Lauridsen *et al.*, 2003; Pot & ter Heerdt, 2014). The most rapid response was observed in the Heron's Carr North (now called Neatishead Arm South) fish enclosure at Barton, where dense coverage of *Chara* spp. and especially *Ranunculus circinatus* was achieved within a year, contrasting sharply with the main broad, where few plants grew. The scale and rapidity of plant establishment implies the exposure of a significant long-lived propagule bank. Meanwhile a nearby enclosure remained clear as a result of effective fish removal but macrophytes were absent. This may have been caused by the lack of a suitable propagule bank, or one which was not exposed as a result of less efficient dredging.

At Pound End, benthic filamentous algae responded immediately in the clear waters promoted by biomanipulation, although after 4-5 years of dominance, a not insubstantial presence of macrophytes (>25% cover) had developed by 1997 (mostly *Najas marina*). This cover was largely lost after 1998 when the biomanipulation was abandoned.

In Cockshoot Broad, macrophyte cover also increased dramatically after a period of around 5 years after biomanipulation. However, whilst this was only possible in the clear waters generated by fish removal the trigger for the dramatic increase seems most likely to be climatic (probably high summer sunshine hours), considering the observed increases, albeit at different scales, in other unmanipulated lakes in the same year (1995). The subsequent decline of macrophytes in Cockshoot coincided with an initial increase in fish populations, whereas recovery of macrophyte cover from 2003 onwards with subsequent stability at values around 70%, followed renewed fish removal effort in the spring of 2003. As benthivorous bream had been functionally extinct since 2000, any fish-induced increase in macrophyte cover appeared to be through the top-down control of phytoplankton through grazing by zooplankton, rather than through any potential benthic mechanism.

In Ormesby Broad and the wider Trinity system, low chlorophyll and high transparency led to an increase in macrophyte cover to values >100 in Ormesby in some years, with lower values in the other basins. Notably, the initial increase in

macrophyte cover in Rollesby coincides more closely with biomanipulation than it does in Ormesby, where the initial increase began in the warm summer of 1995 prior to biomanipulation. Nevertheless, the lack of fish still seems likely to have had a beneficial effect and by 2001, fine-leaved pondweeds dominated the water column in spring over virtually all of Ormesby, although this is not reflected in the macrophyte data due to the late summer timing of the surveys. The upward trend in cover continues in all basins, although with annual fluctuation especially in Ormesby.

In contrast to the biomanipulated lakes, the patterns in Alderfen and Cromes subject to fish-kill show no clear response of macrophytes, or rather the dominant *Ceratophyllum demersum*. In both lakes, macrophyte cover fluctuated over the entire monitoring period, with this being broadly cyclical in Alderfen for reasons that remain poorly understood. Although fish-induced reduction of chlorophyll may have had a role in promoting macrophyte cover, it is not a clear driver of any increase in either lake, nor is there any good evidence of the reduction of benthivorous fish leading to an increase in macrophyte cover. It must be noted however that any effect of benthivorous fish would be best reflected in rooted species, whereas *Ceratophyllum* is largely a semi-buoyant or floating non-rooted species that may be little affected by fish.

4.3.2.4 Increase in macrophyte diversity

An increase in macrophyte diversity would logically occur over time, except where perhaps a diverse propagule bank is immediately available. Thus, the relatively short-lived biomanipulation of Pound End only led to three species being recorded after 9 years, although it is notable that species richness was still the same a decade later following fluctuation in the intervening years and despite the general loss of cover. It is interesting to speculate if richness would have increased with cover if the biomanipulation effort had been maintained in Pound End.

Broadly continuing the same trajectory, species richness reached the highest values yet recorded some 15-20 years after biomanipulation in both Cockshoot and Ormesby Broads, with 7-10 species in Cockshoot and 10-17 species in Ormesby. It is important to note that continued presence of benthivorous bream in the latter compared to the former has not been obviously detrimental to the development of a diverse plant community, including of rooted species that may otherwise have been vulnerable to uprooting by benthivores. It may simply be that the large size of Ormesby allows effective segregation between benthivorous fish and macrophytes.

In Cromes and Alderfen where the stock of benthivorous bream was largely lost through fish-kill, there has been no obvious response in terms of macrophyte diversity with both lakes remaining as virtual monocultures of *Ceratophyllum*, although there are recent signs of increase in other species (e.g. *Najas* in Alderfen). This may simply represent a resumption of the *status quo* for an adaptable macrophyte species that had previously dominated in both lakes. An alternative explanation is that fish kills propagate dominance by *Ceratophyllum* (or nymphaeids) due to an increase in snail populations following release from fish predation. Elger et al, (2006) found in mesocosm studies that moderate densities of snails (*Lymnaea*

stagnalis) resulted in dominance by *Ceratophyllum* and *Nuphar* due to selective grazing of the more palatable seedlings of other species. Dominance by *Nuphar* is a common feature of previously acidified fishless lakes (including Calthorpe Broad). Recent studies in the US where rudd is invasive, also, however, suggest that selective herbivory by rudd on other macrophytes with poor chemical defences could promote dominance by *Ceratophyllum* which appears to be sparsely grazed owing to high foliar concentrations of oxalic acid (Kapuscinski *et al.*, 2014).

In summary, an increase in species richness of macrophytes may occur following biomanipulation, but this is likely to take a relatively long time to achieve (15-20 years) and it is certainly not guaranteed. A range of circumstances and factors are likely to be required to allow colonisation of new species or re-colonisation of species that formerly occurred. At present these factors are understood more in principle than in practice but an ongoing decline in phosphorus concentrations rather than simply an increase in water clarity is likely to be critical.

4.3.3 Factors constraining effectiveness

Fish removal is technically demanding and it is rather easy to fail to remove enough fish to make a difference. For example, Søndergaard *et al.*, (2008) showed that biomanipulation in nutrient-rich lakes in which <200 kg ha⁻¹ of fish were removed in the first three years effectively failed to deliver the expected clearwater conditions, with a significant effect on chlorophyll only in the first year after removal. Removal rates of 75-80% are commonly referred to in the context of successful biomanipulation.

Effective fish removal requires a clear strategy, but with the ability to adapt and innovate should this not quickly deliver the expected results. Only on one occasion, at Cockshoot Broad, was a complete 'fish-out' attempted by systematically fishing sections of the lakes created between stop-nets across the entire lake basin. Otherwise, the strategy has been to target concentrations of fish to maximise catch per unit effort (cost). Typically, the initial removal is undertaken in winter when fish are most aggregated and low water temperatures reduce swimming speed and escape ability and enhance survival of fish subject to capture and transport. At Pound End, small fish accumulated in bird exclosures that offered some protection from piscivorous birds and large piscivorous pike. In Ormesby, the dykes and input stream around the lake that represented the most important overwintering sites for small fish were the initial focus of attention. Following this, the target became the large bream in open water. These fish were captured as they aggregated to spawn in early summer at artificial substrate placed in the margins. Using artificial substrate had the added advantage of allowing eggs to be removed which helps limit recruitment of young-of-the-year. In Cockshoot, input of artificial substrate was repeated over several years in an attempt to reduce recruitment of any remaining roach.

A range of methods have been employed to capture fish, including standard survey methods such as electric fishing and seine netting, as well as more traditional (local) means of fish capture such as fyke nets, but always with adaptation to the

circumstances. For example, in Ormesby, electric fishing has been used to drive small fish already concentrated in their winter quarters in ditches and streams outside of the main lake, over considerable distance (hundreds of metres and perhaps up to 1 km) into seine nets set at convenient locations for subsequent transport. Further, in the open water zone a 'scare line' comprised of a rope with hundreds of weighted coloured plastic tails that was strung between two boats with a third patrolling back and forth behind the line was used to drive large bream into waiting seine nets. Just three runs of the scare line in the summer of 1995 provided 55% of the 690 large bream ultimately removed.

In simple terms, enough fish have always been removed in Broads biomanipulations to generate clear water conditions in the short-term. In Ormesby, surveys indicated that 95% of fish had been removed in the first year. Similarly, in Cockshoot, following the initial removal of 1.35 ind.m^{-2} equating to 15.3 g m^{-2} , subsequent removal efforts over the next few years suggested a residual stock of around 1 g m^{-2} was likely to remain, thereby suggesting removal of at least 94% of fish. It is of note that the standing biomass of fish in degraded Broads appears to generally be $<200 \text{ kg ha}^{-1}$ and provided that $>150 \text{ kg ha}^{-1}$ of fish are removed the outcome may be immediately successful. This contrasts with many lakes in Denmark where a higher initial fish biomass requires a higher removal (removal of $>200 \text{ kg ha}^{-1}$ - Søndergaard *et al.*, 2008). Targets should however be lake specific and governed by previous surveys, which also provides essential information on the distribution of fish in the system.

A further key to success in the Broads has been the use of repeat interventions within an adaptive management framework informed by annual surveys. Nearly 20 years ago, Perrow *et al.*, (1997) clearly outlined that the target of biomanipulation is to generate a period of clear water of sufficient length to allow macrophytes to establish, and that stabilising the system needs to be undertaken as the final critical phase. In simple terms, 'walking away' after a single event in nutrient-rich lakes is almost certainly doomed to failure in the longer term. This is borne out by the experiences in 70 Danish and Dutch lakes described by Søndergaard *et al.*, 2007 (probably containing many, if it not all, of the 36 Danish lakes documented by Søndergaard *et al.*, 2008). Here, biomanipulation was still commonly used as a single treatment. Longer-term data from some lakes indicated a return to turbid conditions within 10 years or less in most cases.

There are three fundamental issues that lead to repeat intervention: 1) failure of fish barriers in connected systems, 2) incursion of fish in high water levels and, 3) recruitment from a residual stock. Poor barrier design was the ultimate problem for Pound End and the lake was effectively biomanipulated almost annually from 1990 to 1998, although large bream continued to be excluded following their removal in 1993 after improvements to the barrier. Despite these issues, a favourable response was seen and in the end, continued manipulation could well have seen the establishment of macrophytes over much of the Broad. In Ormesby, the initial gabion barrier proved impassable to fish, although the ramp over it to allow boat access proved to be impractical. Two further barrier designs were then tried with the first incorporating brushes and the second being a plastic curtain. The second successful iteration of the latter design involved mounting the curtain onto a frame attached to the road

bridge structure. In the period of barrier trial, huge numbers of fish accessed the broad. The following spring massive numbers of fish amassed at the barrier attempting to reach the wider system. Some 470 kg of 1+ roach (90%) and bream were captured and released over the barrier. The estimated total of 122,000 individuals represented around 40% of the initial number of fish removed in the first winter.

Despite attempts to manipulate spawning (see above), recruitment of young fish may be considerable. Based on Danish experience Hansson *et al.*, (1998) considered the rapid expansion of YOY fish in the years immediately following fish reduction to be the single most significant cause of biomanipulation failures. In Ormesby Broad for example, an estimated 1,447,000 young-of-the-year bream were transferred over the barrier following a single recruitment event in the summer of 1995. This represented around 4.8-fold more fish than had been initially removed. Clearly the potential for immediate failure is high without the ability for interventionist management. In Ormesby, it should be noted that a shift in policy of one of the project partners led to abandonment of the plan to complete the removal of adult bream that had begun in 1995. Periodic recruitment of young-of-the-year has threatened the continued development of favourable conditions for macrophytes on several occasions over the years, and fortunately the overwinter survival of young bream has proved to be very poor.

In lakes such as Cockshoot, although the key fish barrier has remained intact, fish have been thought to access the broad on high river water levels, especially in autumn or late spring, necessitating their removal. In spring 2003, some 10,000 fish (almost entirely roach) were removed as barriers in the surrounding dyke network were reinstated. The low-lying nature of the Broads and connection with drainage ditches and fen dykes means that some incursion of fish is inevitable, but this may be at a low level that does not threaten the restoration attempt and may often be of species that are either beneficial (e.g. pike or eel) or benign.

Perrow *et al.*, (1999b) stressed that further manipulation of the fish community to attempt to increase the prospects of achieving the desirable end community in balance with the macrophyte-dominated state should be part of any management strategy. To this aim, a 'seed stock' of 5,100 rudd was introduced to Ormesby from Alderfen in 2000. Rudd is now one of the numerically dominant species in the system. Equally, stocking of piscivores to attempt to achieve a high piscivore:planktivore ratio, which is a feature of macrophyte-dominated systems, can be a useful strategy. In Denmark, stocking of piscivores (typically young pike) has often been attempted (in at least 47 lakes - Søndergaard *et al.*, 2007), but has rarely been successful and only when stocking is at high density ($>0.1 \text{ ind m}^{-2}$) and where it has been repeated annually until clear water is established (Søndergaard *et al.*, 2000). Nonetheless, stocking of fingerling pike may lead to the establishment of older fish in later years, which may have a beneficial impact upon lower trophic levels (Skov *et al.*, 2002). Unfortunately, the single attempt to stock piscivores in Alderfen, in the form of stocking some 10,900 perch in 2000, failed as a result of poor survival that stemmed from poor capture and transport practice by the commercial supplier used.

There has been no recent management of the fish community in either the biomanipulated Cockshoot or Ormesby Broads, with the last intervention occurring in 2008 in both lakes. In Alderfen, no further manipulation has been undertaken since 2000 and in Cromes no manipulation has ever been undertaken since the fish kill in 1999. The fish communities of all lakes are in what is thought to be a more or less favourable configuration when comparing amongst functional groups i.e. rudd often more numerous than roach (apart from very recently in Cockshoot), tench more numerous than bream (bream effectively extinct in all lakes but Ormesby) and perch and pike prevalent, especially by biomass in the case of the latter. In all lakes, absolute numerical density is low at $<0.2 \text{ ind m}^{-2}$ (apart from Alderfen where it can reach 0.5 in m^{-2}) with biomass density also low at $<5 \text{ g m}^{-2}$ (50 kg ha^{-1}).

A key factor limiting future biomanipulation is good knowledge of the trajectory and speed of restoration, generally more appropriately termed rehabilitation with the lack of a clear historical blueprint. With few equivalent examples of fish communities in long-established macrophyte-dominated lakes in the Broads - apart from Upton which is rather unusual in many respects - and a lack of precise measures from the paleolimnological record, there is considerable uncertainty about what is expected from a 'restored' fish community that is in harmony with the macrophyte-dominated state. Upton does at least provide assurance that at low nutrient concentrations even if the fish community is perturbed either by natural or anthropogenic means it is likely to return to a stable state. To illustrate, periodic attempts to establish both roach and rudd in Upton by the angling club in the past were unsuccessful. More importantly, the loss of many pike in fish-kill conditions in 1995 ultimately allowed the development of a moderate density of three-spined sticklebacks by 2005, a species hitherto recorded only occasionally. Pike re-established in 2007 and sticklebacks all but disappeared. In this situation, any predation by sticklebacks upon zooplankton appeared to be of little consequence for chlorophyll concentrations constrained by low nutrient concentrations.

The fear is that if nutrients are maintained above the critical level guaranteeing low chlorophyll ($\sim 35 \mu\text{g/L TP}$) then any unfavourable shifts in the fish community could quickly lead to resumption of turbid conditions. It seems likely that maintenance of macrophyte-dominance probably drives the species composition of the fish community (e.g. through modifying the efficiency of different foraging or predator avoidance strategies; Winfield, 1986), rather than the other way around, and in some species, such as perch, will also be a major regulator of age class structure (Hargeby *et al.*, 2005). In turn, should macrophytes fail for any reason, then some fish species, especially roach, but probably also bream, may be released from the habitat constraints that naturally impose competition and predation from other species. This will precipitate a rapid shift to phytoplankton dominance. It would appear that high macrophyte diversity with a range of growth strategies that promote seasonal succession and habitat partitioning is critical to buffer against catastrophic collapse of cover.

If we remain uncertain of the boundaries of longer-term stability and this limits our effectiveness in reaching our goal of stable macrophyte-dominated conditions, it is clear that even should a managed lake revert to unfavourable condition, it is

relatively straightforward to re-set the restoration trajectory through refined biomanipulation (i.e. targeting particular species) in order to resume the journey.

The brackish broads on the Thurne system deserve a final mention, as there has been no direct experience in the Broads of attempting to biomanipulate such lakes. Typically the zooplankton is dominated by small salinity tolerant copepods with limited grazing potential (Barker *et al.* 2007). Therefore the normal top down mechanisms of release of cladoceran zooplankton from zooplanktivory by fish are unlikely to apply in such lakes. Periodic fish kills in Hickling Broad associated with blooms of the alga *Prymnesium parvum* testify to the lack of an effect on the zooplankton community of effective fish removal. In Hickling Broad the mysid shrimp, *Neomysis integer*, is also a predator of zooplankton and forms an intermediary in the food chain. As such it may benefit from the removal of fish predation, thereby increasing the loss of zooplankton. The available evidence thus suggests that biomanipulation of brackish lakes will at best be of limited effectiveness.

4.3.4 Wider ecosystem consequences

Biomanipulation seeks to return lakes to a plant-dominated state that is ultimately compatible with a self-sustaining mixed fish community. In this process several additional trophic cascades are likely to be set in motion that will potentially reinforce a macrophyte-dominated state and will promote macrophyte stability through increasing diversity. Arguably, the stronger these interactions are the quicker a stable macrophyte-dominated system will be achieved. Additional interactions are likely to include;

(i) an increased abundance and diversity of plant-associated macroinvertebrates (Leppa *et al.*, 2003; Boll *et al.*, 2012) which in turn will improve control of benthic and epiphytic algae by grazing (Jones & Sayer, 2003) thus favouring dominance by rooted macrophytes;

(ii) an increased abundance of infaunal invertebrates (principally chironomids and oligochaetes; Leppa *et al.*, 2003) which is likely to increase transfer of sediment P to the water column through bioturbation (Fukuhara & Sakamoto, 1987). This particular response may be strongest following selective removal of benthivores and some studies have even reported a decline in chironomids following fish removal which they attribute to a decline in planktonic production following a shift to macrophyte dominance (Jyvasjarvi *et al.*, 2013).

(iii) an increased abundance of overwintering water birds, especially coot and diving duck (e.g. tufted duck and pochard), which will benefit from the increased availability of submerged macrophytes and benthic and large plant-associated macroinvertebrate prey (Nordhuiss *et al.*, 2002). The Trinity Broads appear to provide a particularly clear example of this given the increases in wintering coot and diving duck observed there in recent decades (see Annex: Trinity dossier). However, for reasons unknown, the benefits have not extended to breeding wildfowl, a number of which have shown significant declines. Winfield & Winfield (1994) found that dietary overlap in Lough Neagh was greatest between tufted duck and roach due to a

shared reliance on molluscan prey, implying that tufted duck might benefit from a reduction in roach numbers. However, positive responses of water birds to increased clarity are not universal. Thus Tomankova *et al.* (2014) document a 65% decline in benthic macroinvertebrates (mainly chironomids) in Lough Neagh over a 12 year period during which annual mean chlorophyll concentrations halved from a peak of 70ug/L due to a combination of point source nutrient control and reduced diffuse loading. Major declines in overwintering water birds in Lough Neagh that rely on infaunal benthic invertebrates were attributed to this decline in phytoplankton production and increased water clarity. Diving duck may also play a role in controlling densities of small fish and all water birds will act as dispersal vectors, thereby increasing the supply and diversity of macrophyte propagules from external sources (Figueroa & Green, 2002).

(iv) increased recruitment of macrophytes from the sediment propagule bank or successful establishment of external colonists in gaps created through sustainable selective herbivory by fish and water birds.

4.4 Emerging techniques

A number of so called 'ecotechnological methods' have been proposed and trialled in recent years, some in the Broads and others to date only elsewhere. Apart from the use of chemical-isolation methods, such as the application of P-sorbing materials, these methods generally seek to improve growing conditions for macrophytes or remove the bottleneck to macrophyte establishment.

4.4.1 Chemical isolation approaches in lakes

4.4.1.1 Background

Sediment removal has been widely used in the Broads as a means of deepening water level, exposing viable seed banks and improving water quality. However, the long-term effects of the approach are questioned elsewhere in this report for water quality improvement. One risk of sediment removal for water quality improvement is the potential disturbance of macrophyte communities of high conservation status. The cost of sediment removal is also commonly perceived to be high, relative to other in-lake management measures. In fact the cost of sediment removal estimated by the Broads Authority for Upton Little Broad (2014) was £20.53 per m³ with 4500 m³ being removed in 2011. This represents a cost of about £67,434 per ha assuming sediment was removed from the complete surface area of Upton Little Broad (1.37 ha). Upton Little Broad may, however, have been a relatively costly site to treat. In comparison, Spears *et al.*, (2013) estimated costs of sediment treatment with alum or lanthanum-bentonite clay of any lake in the EU at between £2229 and £5943 per ha. In fact the total cost of treating the 5ha Cromes Broad in 2013 was £45,000, a cost of £9000 per ha. It should be noted, however, that the cost estimates above are not directly comparable and are likely to vary significantly across sites and with currency exchange rates. Given the large difference in cost it is, therefore, important that both measures are assessed fully and comprehensively with respect to their ability to support management objectives.

A range of P-sorbing materials have been applied to lakes for the control of sediment P release (e.g. modified clays, industrial by-products, flocculants and physical barriers). These are designed to strip P from the water column and, after settling to the lake bed, reduce P release from the sediments (Hickey and Gibbs, 2009). The use of these techniques in combination with external load reduction and other management methods (e.g. biomanipulation) have resulted in the rapid recovery of some shallow lakes in the short term (Mehner *et al.*, 2008). However, the long-term responses appear to be regulated by the accuracy of the applied dose and the magnitude of continued external loading (Meis *et al.*, 2013). A discussion of some of the successes and failures of the approach are provided by Mackay *et al.*, (2014) and recommendations for improving its efficacy by Spears *et al.*, (2014).

A number of P-sorbing materials have been reported in the literature and are considered to be commercially available (Table 4.4.1-1). In addition, a range of industrial waste materials are being investigated for use in the UK as P-capping products (Spears *et al.*, 2013).

Field scale assessments of the chemical and ecological responses following application of such materials to lakes have not been completed at the Norfolk Broads. In general, long term assessments of ecological responses at the ecosystem scale are scarce. Gunn *et al.*, (2014) report no apparent change in the macrophyte community composition within Loch Flemington two years following an application of lanthanum-bentonite despite a reduction in water column annual mean TP concentrations from 60 $\mu\text{g l}^{-1}$ to 27 $\mu\text{g l}^{-1}$ (2 years following the application). However, Novak and Chambers (2014) demonstrate the use of lanthanum-bentonite and macrophyte planting to encourage a state shift from phytoplankton dominance to macrophyte dominance by reducing TP concentrations to around 100 $\mu\text{g l}^{-1}$ and chlorophyll a concentrations to around 30 $\mu\text{g l}^{-1}$ in the Canning River, south-western Australia. As is the case in most in-lake restoration techniques, these effects may be short lived if they are not conducted in combination with catchment nutrient load reduction. One perennial issue appears to be the slow speed of recovery of the macrophyte community following application of materials. In addition, the operational performance of the materials can vary with the chemistry of the receiving water and so site specific assessments are required prior to material selection and application (Lüring *et al.*, 2014). Clearly, where the in-lake management objective involves deepening of the water level then application of such materials should not be considered until after sediment removal has been conducted.

4.4.1.2 Available Evidence

In the last few decades there have been a number of experimental attempts to treat nutrient-related problems in the broads through chemical isolation methods. These are catalogued by Kelly (2013). For example in Ranworth Broad in 1992, 1ha of sediment was dosed with ferric chloride to increase binding of phosphorus in the silt. However, within the mobile sediment the iron was rapidly lost and technique was not successful at binding phosphorus within the sediment. The same treatment was similarly ineffective at Alderfen in 1995. On the basis of this experience it was concluded that iron dosing was not a viable method for isolating phosphorus in the Broads.

Table 4.4.1-1 Summary of previously used materials to control sediment P release at the whole lake scale, with references to documented examples of their application.

Method	Mechanism	Reference
Physical		
Discing	RIPLOX, manual injection of chemicals (FeCl, Ca(NO ₃) ₂ , CaCO ₃ (including Siltex and related products) into bed sediments to manipulate P sorbing capacity and/or denitrification rate through altered pH.	Ripl, 1976
Capping – Passive		
Sand/gravel	Adding sand/gravel/clay (> 5 cm layer) to existing bed sediments to reduce diffusion of nutrients from sediment to water column.	Cooke <i>et al.</i> , 2005
Capping-active		
Alum	Buffered (>6.5 pH) alum addition to sorb phosphate in water column and settle out to lake bed; also acts as a flocculent, but potentially toxic Al ³⁺ release if low pH conditions are not correctly buffered.	Cooke <i>et al.</i> , 2005
Modified Zeolite	Al-zeolite clay with high binding affinity for NH ₄ -N and moderate binding affinity for PO ₄ -P; applied as slurry to the water column, where it binds dissolved nutrients, before settling to the lake bed, where it caps sediment nutrient release when applied at the correct dose.	Hickey & Gibbs, 2010
Fe-Gypsum	Fe enriched Ca-gypsum formed as a by product of various industrial processes, including mining; applied as slurry or granules to the water column to acts as (a) a physical barrier, (b) a P binding agent and (c) a method of promoting sulphur reducing bacteria over methanogens; most useful in deep lakes with anoxic hypolimnion.	Salonen <i>et al.</i> , 2001
Ferric sulphate	Ferric sulphate dosed and contained within a bund in an attempt to reduce P concentrations at the inflow; applied in Alton Water reservoir; P sorption capacity increased in the treated bund relative to untreated areas.	Perkins & Underwood, 2001
Phoslock®	Lanthanum-bentonite clay with high binding capacity for PO ₄ -P; effective over a wide range of pH (4.5-8.5); solubility product with PO ₄ -P is very low, so it forms a very stable La-P complex in the sediment; concerns about La toxicity still being investigated.	Hickey & Gibbs, 2010

In 2013, Cromes Broad was treated (18.9 tonnes across both basins; 18th March 2013) with the lanthanum-modified bentonite product (Phoslock) for chemical inactivation of P in the bed sediments. This treatment followed up a treatment with Siltex (CaCO₃: dosage of 1tonneha⁻¹) in 1988. Similar treatment at Barton Broad

carried out at the same time proved ineffective. No water quality data were available for assessment in the current study. However, sediment phosphorus composition data are presented in section 5.12.5.3 and indicate an increase in bed sediment TP concentrations, apparently associated with total lanthanum profiles in the bed sediment. No data were available with which the chemical or ecological responses following material application could be quantified in this report. However, these data will become available in the coming years.

There have been other attempts to treat water quality related problems in the past at Calthorpe Broad, but in this case management has been designed to reduce the effects of acidity associated with oligotrophication. Since the late 1970s chalk dust has been applied here on a number of occasions to counter the effect of increasing acidity. Bark chips have also been used in the surrounding dykes to precipitate iron ochre which has reached high concentrations due to oxidation of iron following deep drainage of the surrounding catchment.

4.4.2 Physical habitat engineering

4.4.2.1 Background

Various approaches have been proposed including wave barriers, such as spits or islands (Gulati *et al.*, 2008), excavating deep narrow pits in a lake bed, or dressing fine sediment with sand (Penning *et al.*, 2013) as a means of reducing wind-induced resuspension of sediment and to restrict macrophyte losses through uprooting. Although macrophyte development in lakes is typically greatest in sheltered areas with short fetch and low turbidity the true effectiveness of these techniques is difficult to gauge as presently they are mostly evaluated through hydrodynamic and sediment transport modelling. Best *et al.*, (2008) concluded based on such models that a hypothetical levee designed to reduce fetch and sediment resuspension would have neutral or detrimental effects depending on the plant species concerned. The overall ecological significance of sediment resuspension in lake recovery has also been contested. Thus, Jeppesen *et al.*, (2003) conclude that recovery of shallow lakes after reduction of nutrient loading or biomanipulation is not significantly delayed by resuspension of organic or inorganic matter accumulated in the sediment during eutrophication. There are already advanced plans to use hydraulic engineering on a visionary scale in the restoration of very large Dutch lakes (e.g. Marker Wadden) but with the possible exception of Hickling Broad it is difficult to see a niche for options on this scale in the broads.

In large shallow wind-exposed lakes physical shelter and shoreline reconfiguration may prove beneficial in promoting establishment of macrophytes if nutrient concentrations have already been successfully reduced and there is historical evidence of greater shoreline complexity. Re-grading eroded sections of shoreline to reduce the gradient of the bank-water interface may also assist establishment of emergent vegetation which will promote habitat complexity for macrophytes and fish and increase nutrient sequestering. Increasing water level fluctuations has also been proposed as a mechanism to stimulate expansion of emergent vegetation, although this would not be feasible or desirable in the Broad.

4.4.2.2 Experience

There is limited relevant experience of these techniques in the Broads. The closest parallels to date come from island creation in Barton Broad for landscaping purposes but which anecdotally had little ecological benefit. The other most relevant example is from Heigham Sound where dredging from the navigation channel was used to re-create the Duck Broad spit. In 2014 there was clear water and increased macrophyte growth in this area which presumably reflects the more sheltered conditions (Broads Authority Committee reports).

In general, with the exception of Hickling Broad, the Broads do not contain very large open expanses of water with a high abundance of perennial submerged rooted plants. Hydraulic engineering techniques, developed in the Netherlands to deal with lakes of this type may therefore have limited potential in the Broads. If it can be combined with management of navigation there may be some opportunities for imaginative disposal of dredgings that has benefits for species of shallow water or low energy habitat. The risk that islands or spits form roost sites for gulls or geese, thus leading to eutrophication or increased herbivory, would need to be guarded against.

4.4.3 Transplanting macrophytes

4.4.3.1 Background

Having established clear water conditions it is usually considered critical that macrophyte establishment occurs rapidly in order to sustain this phase. In reality lags of several years are often common before this occurs. Various factors have been implicated in the bottleneck in plant establishment, including waterfowl grazing, over growth by filamentous algae, or a physically or chemically inhospitable substrate (Irfanullah & Moss, 2004; Lauridsen *et al.*, 2003; Moore *et al.*, 2010). However, lack of a suitable propagule supply is commonly invoked (Bakker, 2013), and the direct transplanting of macrophytes is hence sometimes considered as a means of relieving this problem (Lauridsen *et al.*, 2003).

In general aquatic plants will naturally colonise newly created water bodies fairly rapidly (within a few years at most), suggesting that prolonged delays in the colonisation of a lake are likely to be more indicative of other limiting factors, such as sediment chemistry (Smolders *et al.*, 2003). The most likely exception to this is lakes that have recently been sediment dredged, where propagule limitation might be accentuated if the dredging has inadvertently removed most of the viable propagule bank. However, connectivity with other water bodies or surrounding wetlands is often common in lakes, while exposure to suitable propagule vectors in the form of water birds and humans (and their recreational equipment) is frequently high. Indeed, one of the factors that obstructs understanding of the delays in plant colonisation of restored lakes is a lack of knowledge of where colonising plants have actually come from (Bakker *et al.*, 2003).

4.4.3.2 Experience

Various attempts have been to introduce submerged or floating-leaved vegetation deliberately, often as part of experiments on bird herbivory. Species such as *Ceratophyllum demersum* and *Elodea canadensis* were found to establish readily

and to expand, especially given additional protection from bird grazing (Perrow *et al.*, 1997). Lilies (both *Nymphaea alba* and *Nuphar lutea*) established well in various broads, including Belaugh and Ranworth, in some cases benefitting from bird exclosures (e.g. at Pound End and Hoveton Great broads). In general *Nuphar* persisted less well than *Nymphaea*, which might reflect greater palatability of the soft submerged leaves of *Nuphar*. Attempts to establish *Stratiotes aloides* have proved almost universally unsuccessful, although this species has occasionally appeared spontaneously in areas exclosed by fish barriers. Despite its ability to expand rapidly to nuisance proportions in some ponds and canals following deliberate introduction *Stratiotes* has proved notoriously difficult to establish elsewhere in Europe at sites in which it formerly occupied. The species is known to be highly sensitive to sulphide and nitrate concentrations in sediment porewater (Smolders *et al.*, 2003). While nitrate concentrations in the broads are comparatively low and unlikely to be restrictive to *Stratiotes* the influence of sediment anoxia may be critical.

It is generally considered desirable to increase the extent of reedswamp habitat in the broads there having been a significant loss of this habitat over the last 30 years (Boorman & Fuller, 1981). Emergent macrophytes, including *Phragmites*, *Typha angustifolia* and *Schoenoplectus lacustris* have been introduced to various sites via pre-planted floating wooden pallets. Usually some form of protection has been incorporated at the time of introduction to discourage grazing by feral geese. The success of emergent vegetation planting in the broads has been limited, probably in part for the reasons alluded to above, but also as a result of low water levels in some sites in drought years. However, attempts to re-establish emergent vegetation via transplants commonly appear to be unsuccessful (Vanderbosch & Galatowitsch, 2011). In general, without a fuller understanding of the causes of reedswamp failure across the broads as a whole it is difficult to identify the reasons that transplants fail to establish.

4.4.4 Artificial plant beds

4.4.4.1 Background

Since the seminal work of Crowder & Cooper (1982) on the role of structural complexity in buffering predator-prey interactions in freshwaters there has been considerable interest in the use of artificial plant beds (APBs) as daytime refugia to increase survival of the large bodied zooplankton that are selectively grazed by planktivorous fish. The rationale is that if phytoplankton grazing by zooplankton can be increased the shift to a clear water phase and macrophyte dominance following nutrient load reduction should be accelerated. However, APBs have met with variable success. Recent work by Schou *et al.*, (2009) suggest that dense APBs perform best but that zooplankton utilise these structures most effectively when clear water conditions already persist, whilst under turbid conditions, APBs are either utilised directly by fish, or zooplankton do not derive sufficient additional benefit from aggregation in APBs due to the visual refuge effect of turbidity in open water. Boll *et al.*, (2012) used APBs to successfully increase habitat for macroinvertebrates as part of a biomanipulation project in a Danish lake. The objective was to bridge the gap

between zooplanktivorous fish removal and macrophyte re-establishment and to provide a food resource for small perch that would assist the transition to piscivory.

4.4.4.2 Experience

In the broads, experience with APBs extends to the use of cobweb brushes in two sites, Alderfen (2000) and Barton (2001) Broad. Prior to their removal, in 2007 and 2008 respectively, cobweb brushes were considered to have provided excellent habitat for invertebrates but subsequently became overgrown with sponges. In the absence of a firm substrate and with limited macrophyte surface area cobweb brushes in Barton also provided extensive habitat for periphyton (estimated at 70ha) and therefore probably contributed locally to nutrient sequestration and reduced phytoplankton densities as demonstrated by a mesocosm trial in Barton Broad (Kelly pers com).

In general such structures are probably only of temporary and local value and their refugia value would be quickly redundant if littoral reedswamp could be effectively established.

4.4.5 Grazing exclosures

4.4.5.1 Background

On the basis of experimental studies using grazing exclosure it is often concluded that grazing by waterbirds (most notably swans and coot) is a common cause of the failure of aquatic vegetation to establish in restored lakes (e.g. Moss *et al.*, 1996; Søndergaard *et al.*, 1998). However, many studies have relied on transplants that arguably lack realism as they bypass most of the natural barriers to plant establishment (Perrow *et al.*, 1997). There is little doubt, based on direct observations of diet and behaviour, that herbivory by birds *can* result in the loss of large quantities of aquatic vegetation, especially in late summer and autumn. However, it is equally the case that extensive macrophyte cover and large numbers of water birds can coexist for long periods and that macrophytes will establish in restored (i.e. clear water lakes) that are well used by birds. Whether or not grazing is the decisive factor in determining if macrophytes will establish is therefore difficult to say. However, it seems unlikely that poor growth of macrophytes in a clear water lake over prolonged periods will be caused exclusively by grazing. A more critical consideration is the possibility that herbivory does not prevent macrophyte establishment but in the years following management it may create instability and delay the achievement of a macrophyte-dominated state (Bakker *et al.*, 2013) in which nutrient sequestration within macrophytes reinforces macrophyte dominance through a positive feedback.

4.4.5.2 Experience

In the broads grazing exclosures of two basic types have been utilised (i) cages to protect submerged rooted plants from grazing by birds (mainly coot) or fish, and (ii) barriers or floating pallets to provide a media for planting and protect emergent plant beds from grazing or other damage by geese.

In the former case exclosures were found to have species-specific effects (probably related to the palatability and ease of handling of different plant species). These effects were also most evident in the latter part of the growing season (Perrow *et al.*, 1997), consistent with other work (Lauridsen *et al.*, 1993). However, effects were generally weak and not sufficient to prevent plant establishment. In carousel-type exclosures (used in Hoveton Great, Cockshoot and Pound End), the prevention of grazing often merely served to delay the decline and loss of taxa such as *Stratiotes*. Establishment of submerged plants in carousels was often superior in the following year, suggesting that overwinter losses of turions or other vegetative structures may have been higher in the presence of grazing with impacts on plant abundance in the following season.

Protection of emergent vegetation from goose damage has been widely used in the Thurne broads since the late 1990s, reportedly to good effect. The fencing used ranges from alder poles to which chicken mesh is attached, through to floating fences and bundles of brushwood faggots or floating coir rolls. In Hickling and Heigham Sound 2km of protection was established in the period 2000-2003. In general regrowth behind protective fences has been found to be good throughout the Broads although fences must be adequately maintained. Presumably emergent vegetation sheltered by fences also benefits from reduced wave exposure, with submerged macrophytes growing behind emergent planting and silt curtain structures in Rockland Broad. Although re-growth in some instances has been vigorous the loss of littoral reedswamp in the broads since the 1940s remains a significant but unresolved concern.

5 Conclusions and future perspectives

5.1 Main achievements and outstanding challenges

5.1.1 Achievements

A number of significant achievements in the ecological condition of the broads can be recognised that can be traced, at the very least in part, to restoration initiatives undertaken over the past 30 years. These include

- A significant decline in TP concentrations that is primarily attributable to external load reduction and secondarily, in isolated water bodies, to internal load reduction through sediment removal
- A significant decline in chlorophyll and associated increase in transparency caused mainly by reduced TP and, in biomanipulated broads, by increased grazing of phytoplankton by zooplankton.
- Macrophyte cover has increased significantly, mainly in response to increased water clarity
- Macrophyte richness has increased significantly, probably mainly through the reduction in TP concentrations and parallel mechanisms that promote coexistence
- There has been an increase in abundance of *Chara* spp and *Najas marina* that are both associated with lower TP concentrations.
- The abundance and occupancy of different sites by plant-associated fish species such as rudd and tench has increased.

5.1.2 Challenges

Although there has been a major overall improvement in the state of the system a number of outstanding problems remain. These include:

- Persistently high TP concentrations over parts of the system. Although much reduced concentrations commonly still exceed $50\mu\text{g l}^{-1}$.
- Limited recovery of most riverine broads – turbid sites more or less devoid of plants are still common on the Bure and Ant. Most examples of successful restoration are based on techniques that rely on isolation and are thus not appropriate for riverine broads.
- Decline in condition of Hickling Broad and Heigham Sound and limited recovery of Barton Broad.
- Lack of any major shift towards the pre-1900 flora characteristic of the broads. Even the increase in cover and incidence of *Najas marina*, while a welcome improvement in the status of a nationally rare species, is more characteristic of the Broad in the 1930-1950 period than the pre-1900 phase. As a species with a predominantly southern European distribution *Najas marina* in the UK may also be benefitting from climate change.
- *Ceratophyllum demersum* remains the dominant macrophyte in the broads system. This species is prone to marked fluctuations in cover which are likely to have a destabilizing effect in species-poor lakes.
- No large scale re-establishment of the structurally complex littoral margin that once characterised many broads
- Increased prevalence of invasive species, some of which may provide opportunities for lake restoration (e.g. zebra mussel) but which are likely to be increasingly joined by other rapidly expanding species in the near future (e.g. signal crayfish, quagga mussel, killer shrimp). These will potentially threaten ecosystem stability and some have already established a foothold in the

broads. Although zebra mussels are treated in some situations as an ecotechnological solution to lake restoration due to their potential to generate clear water conditions (Gulati *et al.* 2008) through filtering, they may also induce food web instability and have negative effects on native molluscs (Maguire *et al.* 2003).

5.2 Rules for restoring shallow lakes

Figure 5.1.2-1 provides a simple schematic summarising the relationship between chlorophyll and phosphorus concentrations and where the major restoration options lie in this framework as far as promoting macrophyte dominance is concerned. Although not illustrated in this figure catchment level initiatives should be considered an integral part of external load reduction and the management of diffuse sources will become increasingly important as point sources come under tighter regulation.

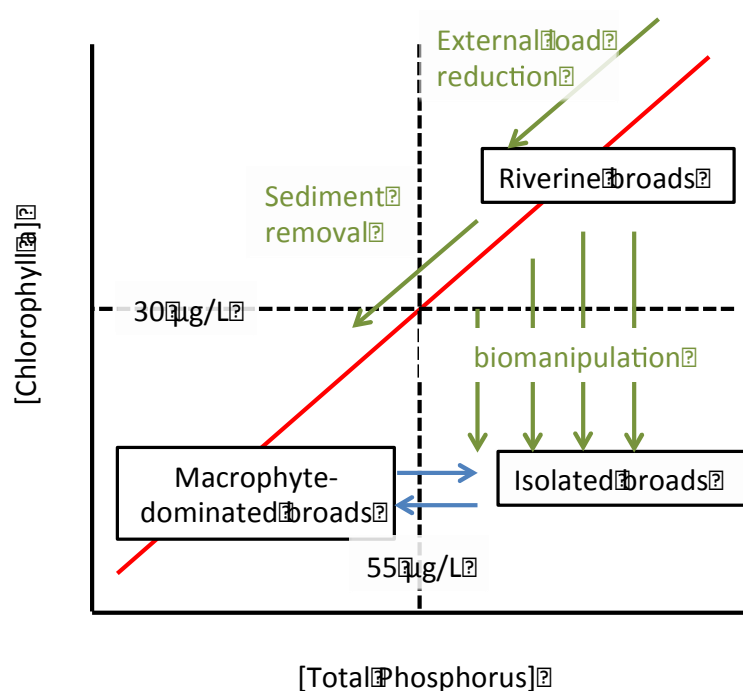


Figure 5.1.2-1 Framework summarising the relationship (in red) between annual mean chlorophyll and TP concentrations ($\mu\text{g/L}$) in the Broads and the role of the major restoration options in driving macrophyte-dominance (green). Blue arrows signify feedback mechanisms that may affect plant dominance.

There are a number of key rules as far as lake restoration is concerned. These are detailed below with some justification.

- Significant reduction of external nutrient loading to eutrophic shallow lakes is essential to create a light climate in which dominance and long term stability of macrophyte cover is possible and buffering mechanisms to retain plant

dominance are re-instated. Annual mean total Phosphorus concentrations of $<55\mu\text{g l}^{-1}$ are likely to be required.

- If such concentrations cannot be achieved in isolated lakes by external load reduction then sediment removal or chemical isolation may be considered as a means of reducing the internal load caused by P release from sediments. The success rate of these techniques in restoring macrophyte cover is highly variable and is likely to depend on the post treatment substrate conditions (both physical and chemical), the new water depth, the level of P reduction achieved, the presence of a viable sediment propagule bank or connectivity to external propagule sources, the extent of any filamentous algal growth, and whether sediment removal was carried out in conjunction with biomanipulation.
- Clear water (chl a concentrations $<\sim 30\mu\text{g l}^{-1}$) is essential for aquatic plant establishment but, equally, is not a guarantee that this will occur (Bakker *et al.*, 2013). Effective biomanipulation will virtually assure clear water conditions due to a reduction in zooplanktivory. Failure of aquatic plants to respond quickly to clear water conditions at moderate to high P is likely to result in a system reverting to a turbid water phase if zooplankton populations crash due to starvation and there is no nutrient competition from rooted plants to prevent high chl a (Romo *et al.*, 1996). Biomanipulation will therefore be most sustainable over the long term once external nutrient loading has been significantly reduced, ideally to a point where annual water column TP is about $40\mu\text{g l}^{-1}$ and certainly $<100\mu\text{g l}^{-1}$ (Jeppesen *et al.*, 2012).
- Plants are essential to the stability and resilience of shallow lake ecosystems because they compete with phytoplankton for nutrients, provide alternative food sources for grazers and buffer interactions between predators and prey. By strengthening linkages between different trophic levels macrophytes will also promote ecosystem stability.
- Macrophytes are vulnerable to direct biotic interactions (e.g. periphyton grazing by invertebrates, and direct grazing of foliage by birds or fish) that are likely to increase nutrient turnover and favour phytoplankton, thus increasing the dynamics of macrophyte beds. The likelihood of reversion from plant dominance to phytoplankton dominance increases with TP concentrations. Although high TP concentrations may simply represent a syndrome of undesirable disturbances, rather than the precise cause of macrophyte failure.
- Restoring shallow lakes such as the broads to a pre-1900s baseline predating the onset of large scale eutrophication is not realistic; the P concentrations required are probably unachievable due to diffuse loading and sources of the necessary mesotrophic species (e.g. *Littorella uniflora*, *Myriophyllum alterniflorum*) have been lost from the landscape. The next best option is a diverse (>10 species) and stable high cover ($>50\%$) of macrophytes in which diversity promotes stability. If high diversity cannot be attained the identities of the species present becomes a more critical influence on stability and lakes with high macrophyte cover that is supported by a low number of species must be considered vulnerable to future perturbations leading to crashes in macrophyte cover.

- The ultimate target for biomanipulation is a low biomass mixed fish community in which piscivores (pike and large perch) and plant-associated species, such as rudd and tench, coexist. A complete absence of fish is not the desired endpoint of biomanipulation and indeed could be considered a vacuum in which other perturbations, such as invasions, become more likely.
- Within the restoration process the success in restoring a system to plant dominance will be influenced by climatic conditions, with irradiance and water temperature during the spring perhaps being most critical. Current climate change appears to have supported macrophyte dominance, at least in the isolated lakes, and rising temperatures may have served to stimulate consumer activity more strongly than that of producers (Gilbert *et al.* 2014), thus favouring control of phytoplankton by grazing. However, warmer water, perhaps coupled with salinity episodes, might yet trigger a shift to phytoplankton dominance, even after several decades of effective reduction in availability of P.
- Lake restoration takes time. Many apparent successes have turned into failures a few years later and it may take 5-10 years or more (supported by suitable data collection and, potentially, repeat interventions) before a particular approach can be judged a success. Pot & ter Heerdt (2014), for example, report a six year gap following biomanipulation before macrophyte populations stabilised. It has to be recognised that there are few quick wins in lake restoration and certainly not durable ones.
- There are many general principles in shallow lake ecology and which form the basis for restoration actions. However, lakes are individuals that behave differently and will respond differently to management according to their morphology, hydrology and history (Valiere & Gulati, 1992). Resolving the lake specific factors that cause differences in response rate and magnitude in small datasets is problematic yet it is these very factors that are likely to determine whether a restoration measure is judged as being successful.
- Monitoring is essential to judge the effectiveness of restoration measures and to learn from experience. In some instances there is no or insufficient baseline data from which to assess the influence of management, or an intervention has not been followed up by monitoring until several years later. Ideally baseline data should be available for the 3 years prior to management and the 5 years following to accommodate interannual variations in baseline conditions and to give management a chance to take effect. As a minimum, major nutrient and chlorophyll concentrations need to be monitored monthly in managed water bodies alongside annual surveys of plant cover and composition. Sediment nutrient concentrations, fish community structure, zooplankton populations and information on the condition of the littoral margin would also be advantageous, depending on the form of management undertaken.

5.3 Outstanding questions

The work presented here represents the first detailed attempt since the mid 1990s to assess the full diversity of restoration practices undertaken across the broads and to

link these to long term changes in water chemistry, macrophytes, fish and, to a lesser extent, waterbirds, against a backdrop of significant climate change. Although a number of significant conclusions can be made there are limitations in the data available and the way treatments have been applied and some of our analyses throw up as many questions as they appear to answer. Of these questions the following are perhaps the most important:

What are the key factors that regulate macrophyte diversity in base-rich shallow lakes?

Base-rich shallow lakes in their natural state are renowned for their high plant diversity. Shallow lakes are, as a rule rather, spatially simple environments, with little variation in depth or substrate to form a basis for habitat partitioning. Several other factors are therefore likely to be of key importance in maintaining diversity: (i) moderate to low nutrient availability (winter TP 20-30 $\mu\text{g l}^{-1}$) which prevents dominance by a small number of canopy forming elodeids and ensures that rooted plants that obtain most of their nutrients and/or inorganic carbon from the sediment have an advantage over floating or semi-floating species. In a well populated and intensive agricultural region such as Norfolk it is unrealistic to expect to return lakes to this level of fertility and given the long absence of a suitable regional propagule pool the re-establishment of some of the species characteristic of this nutrient regime (e.g. *Littorella uniflora*, *Myriophyllum alterniflorum*, *Potamogeton coloratus*, *Utricularia intermedia*) seems very doubtful. Maintaining richness at higher nutrient concentrations (TP of 30-50 $\mu\text{g l}^{-1}$) is likely to depend increasingly on three supporting factors: (ii) the spatial heterogeneity afforded by the vegetation itself, especially low energy patches created by lilies and emergent species, such as *Schoenoplectus lacustris* and *Typha angustifolia*, which form a forest of low density stems in deeper water; (iii) a long macrophyte growing season which is not compressed between spring and autumn phytoplankton peaks (Sayer *et al.*, 2010). This promotes temporal partitioning, with richness (and most importantly plant cover) being maintained via a succession from overwintering and spring peak species through to late season peak species, such as *Najas marina*; (iv) low level disturbance associated with grazing by fish (especially rudd) and water birds (especially coot). This is likely to be important in suppressing the dominance of canopy forming plant species. Other biotic influences are likely to include gap creation by mute swans or turnover of the sediment within these patches by benthivorous fish, which may promote recruitment from the propagule bank. It seems likely that as P concentrations increase the influence of grazers will first become neutral and ultimately negative, by either reducing the level of nutrient immobilization within higher plants or promoting a shift to dominance by *Ceratophyllum demersum* which is chemically defended and relatively unpalatable, but is prone to marked population fluctuations.

How much does connectivity with fen dyke systems influence lake vegetation compared to the sediment propagule bank?

Some fen dyke systems within Broadland continue to support some of the plant species that were characteristic of the vegetation of the broads themselves prior to their decline in the 1950s (e.g. *Stratiotes aloides*, *Utricularia vulgaris*, *Hydrocharis morsus-ranae*, *Myriophyllum verticillatum*, *Hottonia palustris*, *Potamogeton compressus*). The level of interchange of propagules between these dykes and the broads is unclear and it would be instructive to assess the level of similarity between both established vegetation and propagule banks in the broads and the dykes with which they potentially interact. Considering such data alongside the Broads Authority Annual Macrophyte Surveys would be a useful exercise to determine whether connectivity with fen dyke systems is a potential constraint on the re-establishment of a diverse aquatic flora in the broads. However, although connectivity is a potentially important influence on propagule supply to lakes recent work on shallow lakes in the US (Nolby *et al.*, 2015) suggests that size and connectivity are comparatively minor influences on lake macrophyte richness with biotic interactions exerting the main effect.

What factors constrain the recovery of a diverse littoral margin?

The spatial heterogeneity afforded by a diverse littoral marginal habitat is a key factor in habitat provision for some fish species (e.g. tench), increased refugia for zooplankton and the low energy habitat favours the persistence of some of the more fragile or poorly anchored plant species. A complex littoral margin was historically a characteristic feature of the broads (Pallis, 1911). Erosion due to wind action and boat wash, goose grazing, manual harvesting in the past and stem thinning caused by high nitrate loading have all been implicated in the loss or dieback of reedswamp in the Broads and elsewhere in Europe (Boar *et al.*, 1989). However, in reviewing the evidence for and causes of reedswamp decline in broadland, Boorman & Fuller (1981) conclude that grazing by feral coypu was primarily to blame. We suggest that such grazing was also a significant and grossly underestimated contributor to loss of associated aquatic vegetation in the broads from the 1950s until coypu were eradicated in the late 1980s. There is significant evidence from the southern US and Italy where coypu are also invasive that their diet is dominated by aquatic vegetation (Wilsey *et al.*, 1991; Prigioni *et al.*, 2005) and studies of a captive coypu population at Calthorpe Broad confirm the level of reliance on aquatic vegetation of invasive coypu in the UK (Gosling 1974). Even low densities of a native aquatic rodent, the beaver, have been found to exert a significant impact on deeper water swamp species such as *Cladium mariscus* and *Schoenoplectus lacustris* in Scotland (Willby *et al.*, 2014). Given that the carrying capacity of beavers in Norfolk has been estimated at 18-40 territories (<200 animals) (South *et al.*, 2001), while numbers of coypu in Norfolk may have been as high as 20,000 in the mid 1970s (Gosling 1974) it seems highly likely that coypu exerted an unsustainable level of grazing pressure on littoral marginal vegetation and associated macrophytes. Aquatic rodents such as beaver and coypu are destructive feeders, preferring fleshy rhizomatous plants and often uprooting the whole plant to access the starch-rich stem base. At their natural carrying capacity the effects of beavers on aquatic vegetation are likely to be sustainable, since animals abandon territories after 5 years or so, moving to other sites, during which vegetation recovery in former territories is likely. By contrast the impact of a very large invasive

population of coypu is unlikely to have been sustainable. It is likely that the foraging activity of coypu, combined with direct grazing also impacted on aquatic vegetation associated with reedswamp. The impacts of coypu would have been considerably greater at the time than those of an expanding feral goose population which is likely to have affected emergent plant stands more by grazing of growing tips or physical damage caused by roosting. However, the lack of a large scale recovery of littoral emergent vegetation following coypu eradication points to ongoing constraints on re-establishment of littoral marginal vegetation.

How will climate change affect lake responses to management?

A shift to warmer and sunnier summers and milder winters and springs has likely contributed to an improvement in growing conditions for macrophytes and therefore exacerbated the effects of reductions in P and improvements in water clarity and potentially reduced the lag phase in macrophyte re-establishment. Although IPCC scenarios would suggest a prolonged shift to cold dry winters in the UK is now very unlikely it is still unclear if the system in its present condition is sensitive to such conditions or if these could shift the balance in favour of more responsive phytoplankton in lakes where nutrient concentrations remain high. A recent review of climate change implications for freshwaters in the UK (Watts *et al.*, 2015) suggests that warmer wetter conditions could ultimately stimulate phytoplankton growth at the expense of macrophytes, or favour rapid population growth of planktivorous fish and increased prevalence of invasive species, whilst increased river flows are likely to favour nitrogen-limited species.

What role do water birds play in the stability of shallow lakes?

In either larger or biomanipulated lakes it seems likely that water birds will increasingly replace fish in the food chain. While macrophytes essentially retard turnover of the limiting nutrient this turnover is stimulated by planktivorous fish. If bird numbers, especially of swans or coot, increase in response to increased abundance of aquatic vegetation it is possible that excretion could accelerate turnover of plant-derived N and P. While macrophyte diversity remains high and there is a persistent cover of overwintering species this may not be critical but if high levels of autumn grazing by wildfowl are followed by cold spring conditions and a delayed onset of macrophyte growth small celled phytoplankton may be well placed to exploit an increase in nutrient availability.

How will/are invasive species influencing the system and are sites with high plant diversity any more resistant to any destabilizing effects of invasion

Increasing connectivity (both hydrological or to human and animal vectors) is likely to increase vulnerability to invasion. However, connectivity is also likely to promote exchange of propagules between water bodies and by increasing richness may

increase resilience of some water bodies to the effects of invasion. Well vegetated sites with a high diversity of growth forms and effective nutrient sequestering in plant biomass may prove more effective at containing some invasions. The NERC Lake BESS project that is currently in progress will provide some answers to this question.

How long term stable are biomanipulated lakes: if fish re-entry is permitted will habitat filtering in clear water, plant-dominated lakes favour the retention of a mixed piscivore/rudd/tench assemblage without further intervention?

The reconnection of 'restored' lakes to the main river system is arguably the ultimate acid test of internal stabilizing mechanisms. Can these persist and maintain macrophyte dominance once open access to fish is possible? Will habitat filtering favour the existing plant-associated fish assemblage over planktivores and benthivores? The planned restoration of Hoveton Great Broad, which will see the Broad temporarily isolated from the main river for several years, before being reconnected, is probably the best opportunity in the near future to determine the answer to this question.

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