Review of Sediment Removal

Appendix 3
Broads Lake Restoration Strategy


Sediment removal at Barnby Broad - Mike Page

Prepared by

Andrea Kelly Head of Conservation, Broads Authority

March 2008

1. Introduction

A lack of recovery of shallow lakes following reduction of the phosphorus (P) loading entering the system from external sources is often attributed to continued P release from the sediment (Marsden, 1989; Sondergaard *et al*, 1993). This internal P release in eutrophic shallow lakes, in the Norfolk and Suffolk Broads, is thought to have delayed the effect of reduction in external nutrient loading (Phillips *et al*, 1999, Moss, 1980) by maintaining high P concentrations in the lakes. This has promoted phytoplankton populations capable of out-competing submerged water plants, which are key to maintaining high aquatic species diversity.

Sediment removal has been used as a lake management tool for many years, with much information gathered on its environmental effects (Mueller *et al*, 1998, Sebetich and Ferriero, 1997, Klein, J. 1998, 1990, Bengtsson *et al*, 1975).

The Broads Authority restoration aims of sediment removal include the following objectives:

(1) Reduce internal phosphorus loading from relatively recently deposited nutrient rich sediment;
(2) Maintain or create sufficient water depth for submerged water plants;
(3) Increase water depth for navigation purposes; and
(4) Remove toxic substances associated with the sediment.

A discussion of results relating to sediment removal for the aforementioned aims forms the basis of this report.

Within the Broads attempts to reduce internal P recycling and to increase water depth for navigation and water plants have been made by employing several methods. These include dosing with chemicals and removal of nutrient rich sediment. This review focuses on the latter as chemical dosing, such as iron and finely divided chalk, has proved unsuccessful (George, 1992) and is therefore not an option for subsequent restoration projects within Broadland.

Over the past 20 years sediment removal, via mud pumping, has been widely practised in 11 small broads (each less than five ha.) and two larger broads (Hoveton Little 15 ha. and Barton, 75 ha.), (Kelly, 2008).

However, within similar timeframes these broads have also had one or more of the following additional restoration measures: nutrient reduction at sewage treatment works, isolation to reduce external loading, biomanipulation (removal of certain species of planktivorous and benthivorous fish) (Moss, 1986, Phillips *et al,* 1999, SCOPE, 1998), thus it proves difficult to determine the relative impacts of each restoration measure individually. For example isolation itself can in the short term increase nutrient loading by reducing the amount of flushing (Phillips *et al,* 1999) by allowing nutrients to be accumulated in water and sediment as well as allowing for more efficient uptake of nutrients by algae.

This document focuses on sediment removal in Barton Broad, describing the results of a three-year research project on sediment removal, the research undertaken by the Broads Authority in partnership with the Environment Agency, from 1996-2000, as part of the Clear Water 2000 project.

Mud pumping removes the surface nutrient rich silt, shown in Figure 1, and disposes this to land where it can be beneficially reused. Surface unconsolidated silt layer is made up of deposited algal matter and sediment from eroding riverbanks and land. The coarser structured marl has formed during the time water plants dominated the lakes and the underlying peat basin was laid down by decomposing wetland vegetation prior to the peat digging being excavated.

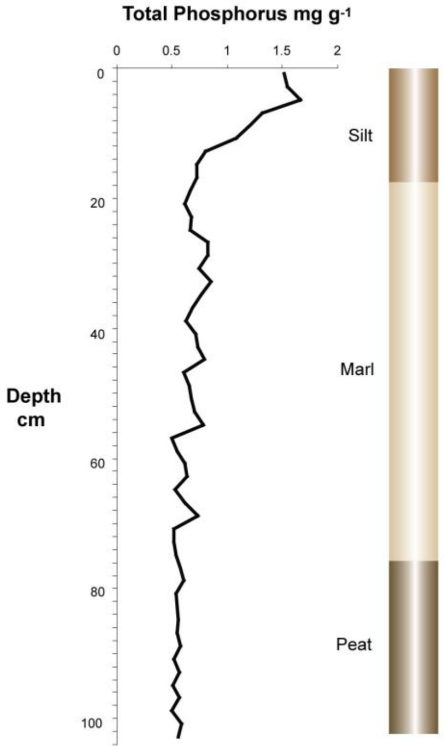

Figure 1. Sediment phosphorus and sediment profile in a one metre core taken from Barton Broad within the Turkey Broad area

Research aims for the Barton Broad dredging project included:

1. Evaluation of the effects of dredging on the total phosphorus (TP) concentration immediately after and five years after dredging
2. Investigation of the re-profiling of the P in the surface sediment
3. Monitoring any change in the partitioning and binding of P within the sediment
4. Investigating the effects of dredging on soluble reactive phosphorus (SRP) release and interstitial pore water SRP
5. Relating SRP release to the concentration of SRP in the interstitial pore water

1.1 Preparation and Feasibility Study

The Broads Sediment Management Strategy (Wakelin and Kelly, 2007) sets out the stages of investigation required prior to sediment removal. This decision-making framework is shown in Figure 2 with the addition of palaeolimnological study for lakes to gain information on the past ecology and target removal depth.

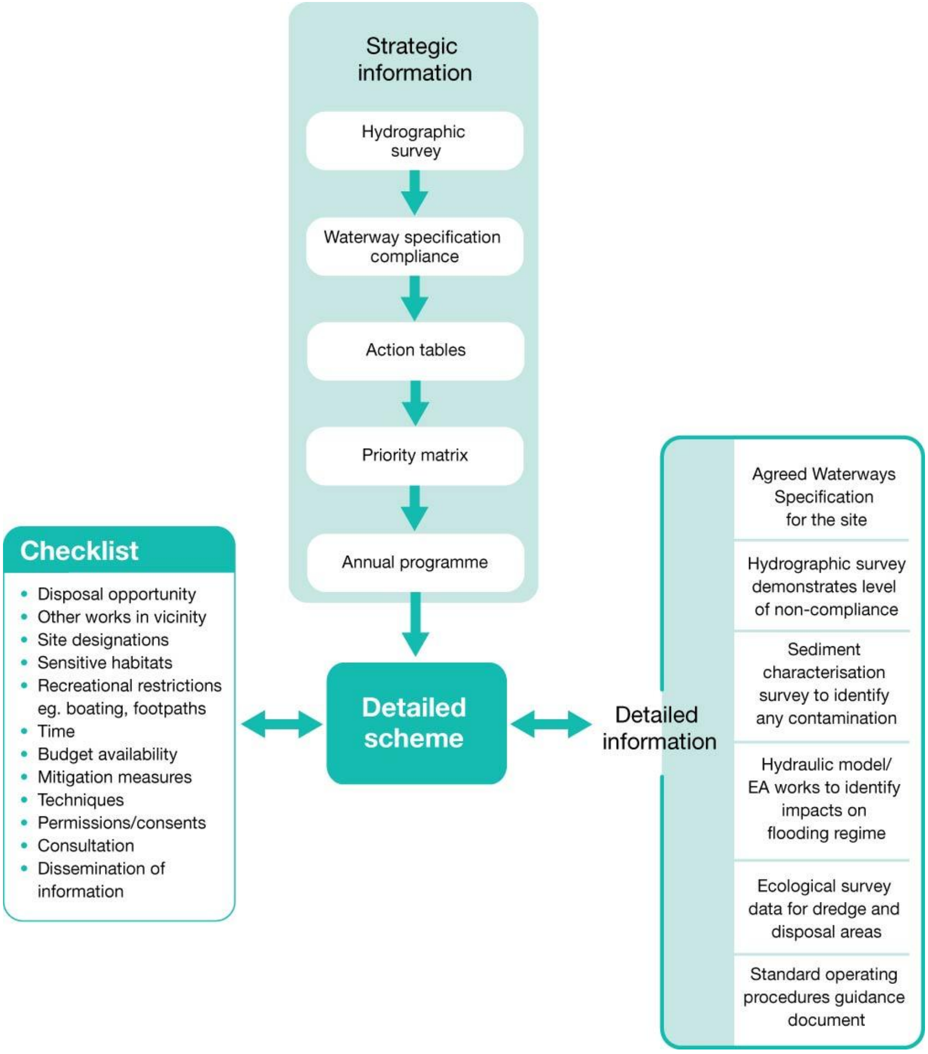

Figure 2. Sediment removal decision-making framework

Palaeolimnological studies provide information on sediment age, sediment accumulation rates, pollution histories and past ecology (by examining remains of algae, plants, zooplankton and to some extent the fish population). Palaeolimnological results for Barton Broad are presented in Figure 3.

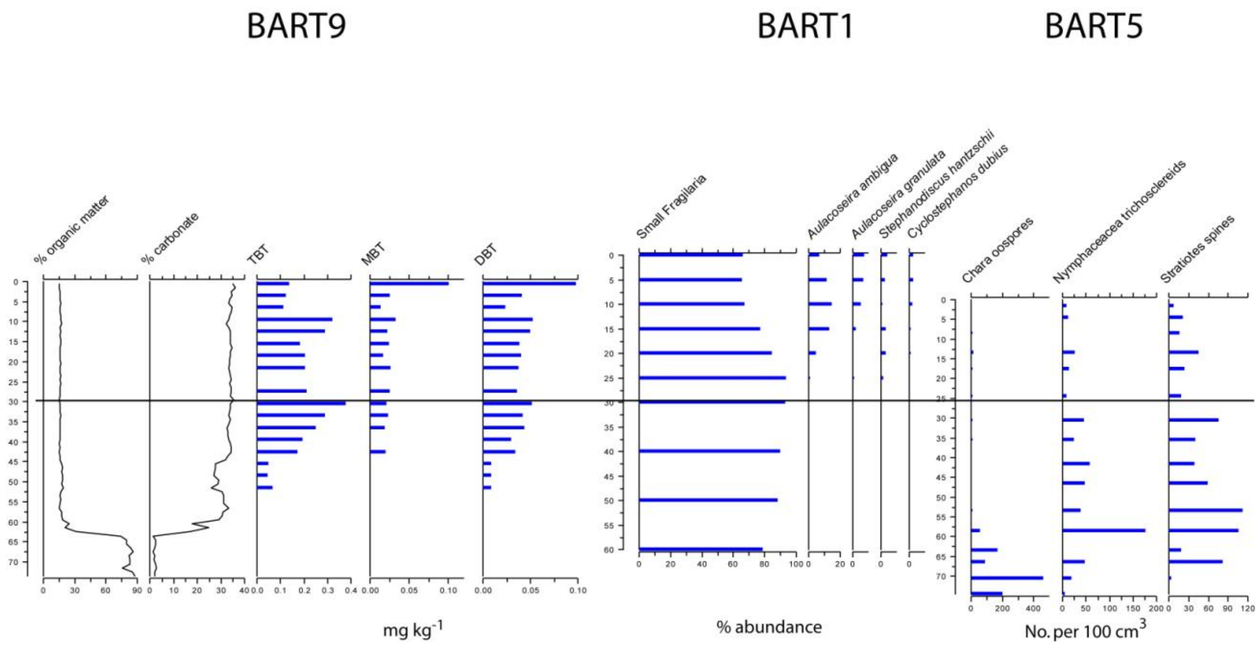

Figure 3. Sediment core data for Barton Broad
Source: (Emson, Sayer and Hoare, unpublished data)

2. Methods

2.1 Sediment removal

In the Broads silt removal projects in lakes generally give a final water depth of 100-175 cm, exposing the 'marl' layer. This layer, deposited at a time when the lakes were dominated by submerged aquatic vegetation (Moss, 1980), contains macrofossils such as seeds, leaves and oospores. Surveys using echo-sounding or lead-line are used to confirm the water depth and volume of sediment to be removed.

The Broads Sediment Management Strategy (Wakelin and Kelly, 2007) outlines the different techniques used for sediment removal in the Broads, with mud pumping (suction dredging), being the preferred option for lake restoration for several reasons. These include : low turbidity creation, achieving a precise and even bottom contour and enabling an efficient transfer of material from the lake bed to the disposal site. Mud pumping enables transfer of material, made up of around 90% water 10% solids, a distance of 1-2 kilometres from the lake to the disposal site. The disposal site either needs to have a sufficient area and infiltration capacity to cope with the quantity of sediment and water, or it needs to have constructed lagoons where material can be stored, dried and subsequently spread to land in accordance with the Waste Management Regulations. In accordance with the principles in the Sediment Management Strategy (2007) sediment should be recycled wherever possible, for example in flood banks, or reused on agricultural land.

2.2 Palaeolimnological analysis

Sampling and analysis was undertaken using the methods outlined in Appleby *et al* 1986; Zhao *et al*; 2006; Ayres *et al;* in press.

2.3 Sediment contaminants analysis

The determinands routinely tested for prior to each dredging operation are given in Appendix 1. The Broads Authority also holds a sediment quality database, with details of a complete study of the Broads waterways at 2 km intervals undertaken in 2004. This database also includes sediment data collected since this time as well as from isolated broads not included in the original study.

2.4 Surface sediment sampling for nutrients
Sediment cores were taken, at sites in Figure 4, using a pole corer and 500 mm long extruded acrylic tubes with a 69 mm internal diameter. The cores had 3 mm holes drilled at 10 mm intervals to allow samples of water and sediment to be taken, these holes were sealed with waterproof tape. Sediment height was 300 mm in all cores (±40 mm). Cores were sealed with rubber bungs at the top and the bottom. All the air was expelled from the overlying water to ensure that water movement was minimised within the core to avoid disturbance to the sediment surface during transportation. Cores were transported to the nearby laboratory in a cool box to retain in situ temperature and sampled along the core length or set up for experiment within four hours of collection.

2.5 Routine sampling at Barton Broad
Sampling of surface sediment was carried out at three monthly intervals from 1998 to 2000. Five replicate cores were taken from four sites as follows: 'West' dredged in August 1996, 'East' dredged in August 1997, 'Turkey dredged' in August 1998 and 'Turkey undredged' (which was subsequently dredged in October 1999) (Figure 4). Total phosphorus (TP) and total iron (TFe) were determined using an adaptation of the ignition method of Andersen (1976). Samples were taken from 1, 3, 5, 10 and 20 cm depths from the sediment surface.

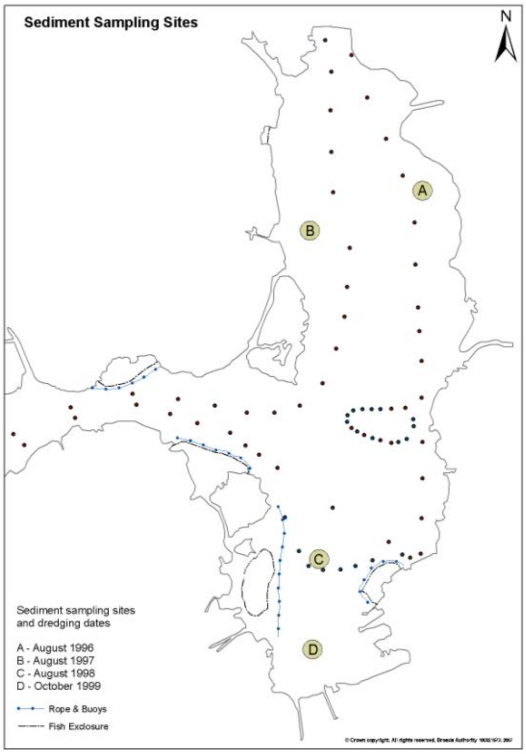

Figure 4. Routine sediment sampling locations from Barton Broad

2.6 Phosphorus fractionation
Published methods used are outlined in Pitt *et al* (1997), with the exception of Barton Broad with the following modifications: NaOH-P extraction (P bound to aluminium and metals in humic acids) used 0.1M rather than 1.0M NaOH to avoid precipitate formation during the molybdenum analysis. HCl-P (extraction of P bound to calcium as apatite) was undertaken using 0.1M HCl. The number of washings with 1M NaCl was reduced from 3 to 1 as repeated washing trials showed that no additional P was extracted.

2.7 Release and interstitial

In the laboratory the overlying water was drained off to just above the sediment surface and carefully replaced with standard water, to a depth of 10 cm using a peristaltic pump. This standard water closely resembled the major ion composition of 'typical' lowland rivers, omitting phosphorus, iron, manganese, nitrate and ammonium, since they could interfere with release. Using this zero phosphorus water for these experiments is representative of conditions in the broad, where mean SRP values were usually <0.004 mg l^{-1}. To sample the interstitial pore water without disturbance of the sediment a non-destructive method was employed. This involved inserting porous polymer tube ('Rhizon' soil moisture sampler) horizontally into the sediment at depth intervals (2, 6, 10, 21 cm). These depths were slightly different to those used for TP analysis and were chosen to avoid surface disturbance and to allow cores to fit into the controlled condition cabinet. Cores were placed into a dark growth cabinet at 20°C, representing ambient light and temperature conditions in the late summer. The overlying water was mixed and oxidised with gently bubbled atmospheric air.

At timed intervals 10 ml of the overlying water was extracted by attaching a 10 ml syringe to the 'Rhizon' and drawing off a sample under a vacuum. This was then filtered through 0.45 µm membrane filters pre-purged with nitrogen gas. Samples were analysed for soluble reactive phosphorus (SRP) using standard methods based on molybdenum blue formation (Murphy and Riley, 1962, as modified by Stephens, 1963). The overlying water was topped up to 10 cm above the sediment at all times using standard water. Release rates mg m^{-2} day^{-1} were calculated using the following equation: $C * H * 1000 * 24 / I$.

Where C = SRP concentration in the 'standard' overlying water in mg l^{-1}, H = height of the overlying water in m, I = incubation time in hours.

For Barton Broad five replicate cores were taken from undredged and dredged areas of sediment, (Fig. 1 areas A and B respectively), on four occasions between May and August 1999 and on two occasions in 2000 (June and August).

3. Results and conclusions

Barton Broad

3.1 Overall aim: To reduce internal phosphorus loading from relatively recently deposited nutrient rich sediment.

3.1.1 Research aims:

- **Evaluation of the effects of dredging on the total phosphorus (TP) concentration immediately after and five years after dredging**
- **Investigation of the re-profiling of the P in the surface sediment**

Removal of 284,000m^3 of wet sediment from Barton Broad removed approximately 50 tonnes of P from the lake. This represents approximately 20 years' worth of phosphorus loading from the River Ant, based on average loading recorded from 1987-95 from Environment Agency data.

Removal of the top 30-50 cm of silt uncovered the 'marl' layer, formed during the clear water macrophyte dominated period in the lake's history up to the mid 20th century (Figure 1). Before sediment removal, this marl layer had a TP concentration less than 50% of the uppermost fine organic sediment layer (Figure 5).

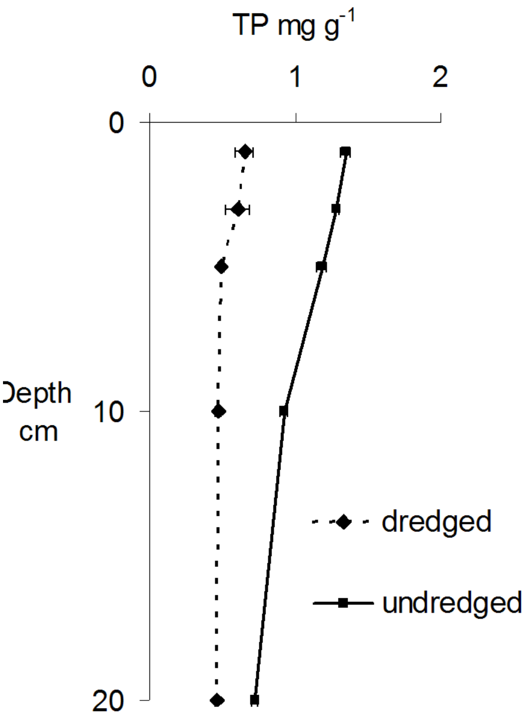

Figure 5. Total phosphorus concentration of the sediment before and after dredging

This surface layer is recorded as being deposited at a rate of 1.28 cm per year after the onset of eutrophication (Moss, 1980, Rose et al, 2005). This settling sediment was captured in tubes set regular heights in the water column for a period of 12 months. It contained almost twice the TP concentration as the dredged surface sediment as well as being substantially more than the undredged sediment (Figure 6).

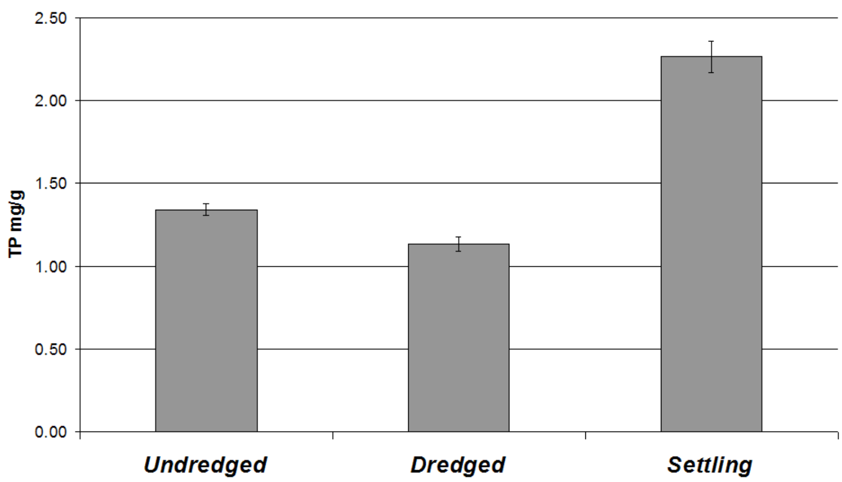

Figure 6. Mean total phosphorus at the sediment surface (1 cm) and settling particulate matter captured in sediment traps

Immediately after dredging, due to exposure of the marl the TP concentration of the new surface sediment was 50% lower (Figure 5). However, one month after dredging the newly exposed surface centimetre had increased to levels similar to that of the undredged silt (Figure 7).

Processes controlling this re-profiling of P after dredging may include P sorption, sediment redistribution and input of sediment from upstream and bank erosion (Reynolds and Davies, 2001).

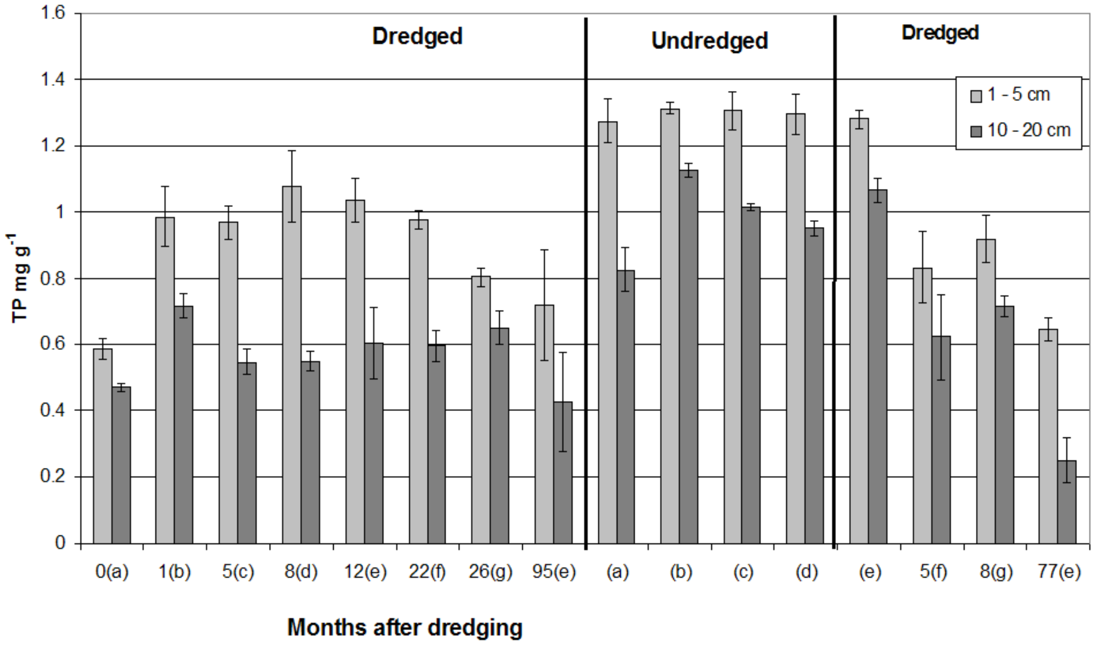

Figure 7. Mean TP concentration in undredged and dredged sediment in Turkey Broad in the top 1-5 cm and lower 10-20 cm depth up to eight years after dredging, with standard error bars.

The TP concentration of the lower layers in most dredged sites studied remained significantly lower up to five years after the dredging operation.

Despite addition of new nutrient rich settled material the dredged sediment had a significantly lower TP concentration at 3-20 cm depth (Table1) up to three years after dredging (Table 2). This lower TP concentration in the sediment is also retained eight and six years following dredging, shown as 95 and 77 months respectively on Figure 7.

	West Dredged	East Dredged	Turkey Undredged	Turkey Dredged
West Dredged		1-	3+,10+,20+	1-,20+
East Dredged			3+,5+,10+,20+	1-,20+
Turkey Undredged				1-,3-,10-,20-
Turkey Dredged				

Table 1. TP*Site. Significant difference in average total phosphorus concentration for each depth (1, 3, 5,10, 20 cm) for all dates sampled using General Linear Model, Tukey Simulation Tests. The depths that showed differences at the ≤0.05 significance level are given in this table; where no depths are written there is no significant difference.
+ Column factor mean > row factor mean
- Column factor mean < row factor mean

	West	East	Turkey Broad
Pre-dredge	1.22 ± 0.39 ('88 & '90)	N.D.	1.04 ± 0.32 (1996)
Post-dredge 1998	0.75 ± 0.39 ***	0.80 ± 0.35	0.86 ± 0.22 **
1999	0.91 ± 0.28 ***	0.82 ± 0.31	0.86 ± 0.28 *
2000	0.93 ± 0.34 ***	0.83 ± 0.30	0.79 ± 0.22 ***

Table 2. Mean total phosphorus concentration ± SD (n=≥25), for the top 1-20cm depth of sediment sampled in West, East and Turkey (sites A, B and C on Figure 4). Pre-dredge sample dates: 1990 in West and 1996 in Turkey, no data were available for East. Actual dredge dates for each site: West = 1996, East = 1997, Turkey = 1998. Mann Whitney test pre-dredged TP concentration v post-dredged *** p≤0.001; ** p≤0.01; * p≤0.05
N = between 30 and 100.

3.1.2 Research aim:

- **Monitoring any change in the partitioning and binding of P within the sediment**

The iron to phosphorus ratio (Fe:P) has been cited as affecting P release (Bostrom et al, 1991, Boers, 1991). Fe:P is a measure of the residual binding capacity, with higher ratios indicating that more free iron would be available for binding P in the oxic microlayer (the surface sediment layer which contains oxygen, often only a few mm thick in the broads). Mud pumping Barton Broad increased the sediment Fe:P ratio, no doubt contributing towards the reduced P release measured (Figure 8).

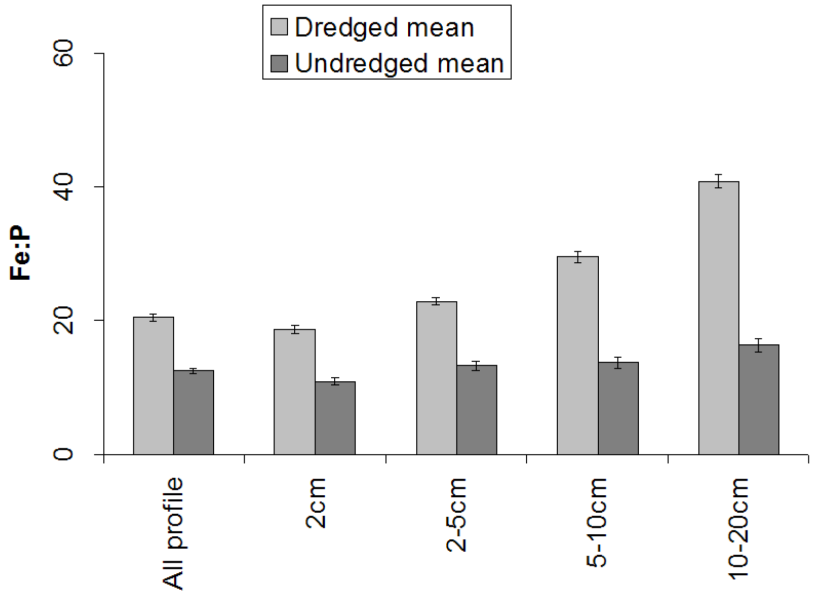

Figure 8. Iron:Phosphorus ratio in the dredged from 1996-2000 (n=) and the undredged (n=) sediment (standard error shown by lines) using the Mann-Whitney test (C.I.= ≤ 0.001, n=101 undredged, n=144 dredged).

Another factor that may have reduced P release is the significant reduction in organic bound P following dredging (Table 3). Organic P is loosely bound and therefore more available for release.

	1	*3*	*5*	*10*	*20*
Inorganic-P					+++
Organic-P			+++	+++	+++

Table 3. P-fractionation differences between the dredged and undredged sediment, undredged has significantly higher concentration where marked: +++p≤0.005; ++p≤0.01; +p≤0.05. 1-3 and 5-10cm n=14 undredged, 34 dredged, 1-10cm n=34 undredged, 26 dredged.

Results also showed that the newly dredged surface layer had more P bound to dithionite (iron-bound fraction) compared to the undredged sediment and the deeper sediment (5-10 cm depth), which had no regular contact with the overlying water. This supports the hypothesis that the relatively higher Fe concentration in the dredged sediment is actively binding P.

3.1.3 Research aims:

- **Investigating the effects of dredging on SRP release and interstitial pore water SRP**
- **Relating SRP release to the concentration of SRP in the interstitial pore water**

The ability of lake sediment to release P is dependent upon many factors such as the pore water SRP concentration, binding capacity, the activity of benthic invertebrates, pH, temperature, and frequent exposure to physical disturbance. It was beyond the scope of this three-year Barton Broad study to investigate all possible release mechanisms. SRP concentration in interstitial pore water in conjunction with measuring SRP release rates was selected to provide insight into how this interaction may affect the nature of release observed in the water quality data of Barton Broad (Phillips *et al*, 2005).

Figure 8 shows the potential P release rate measured from collected sediment cores over a 12-year period from 1987 to 1999. These show that sediment phosphorus release decreased continually over this period, especially in the early summer period (May – July), to become characterised by lower rates of late summer P release. This is supported by a decrease in chlorophyll and SRP in the lake water in early summer reported in Phillips *et al*, (2005).

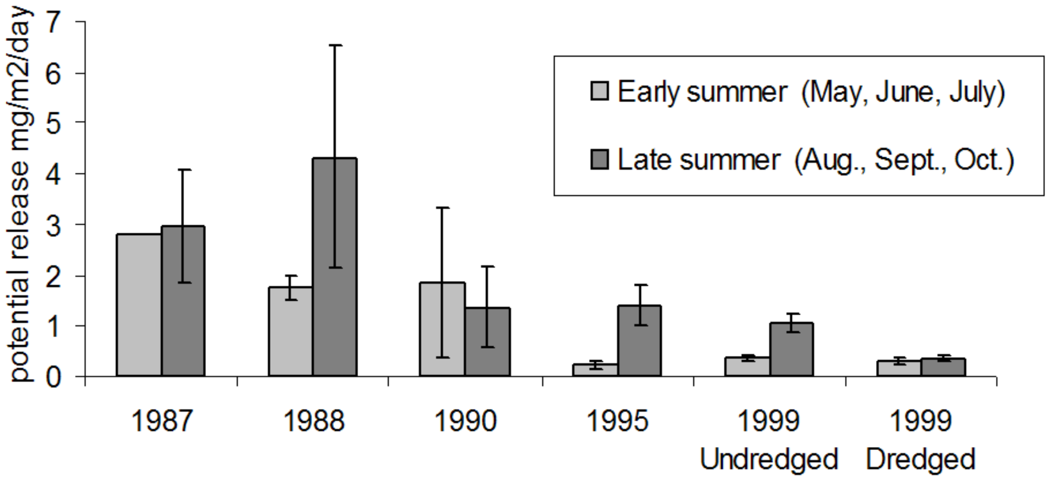

Figure 9. Mean potential SRP release measured from the overlying water within sediment cores for dredged and undredged sediment from 1987-1999. (standard error shown by lines)

Over the 12-year period SRP release had decreased and become less variable by 1995, prior to sediment removal, to a level comparable to the 1999 undredged sediment (Figure 9). This was probably a result of lower P entering from the inflow over the years changing the various water and sediment interaction processes facilitating net P retention within the lake (Phillips *et al*, 2005).

In addition to this gradual and significant decreasing internal P release, two years of sediment P release research confirms that dredging Barton Broad significantly further reduced sediment release of phosphates by approximately 60%, compared to the undredged sediment.

The interstitial pore water SRP concentration is almost always greater in the undredged sediment (Figure 10). This is likely to promote greater sediment release either through larger diffusion gradient or sediment disturbance in these shallow waters

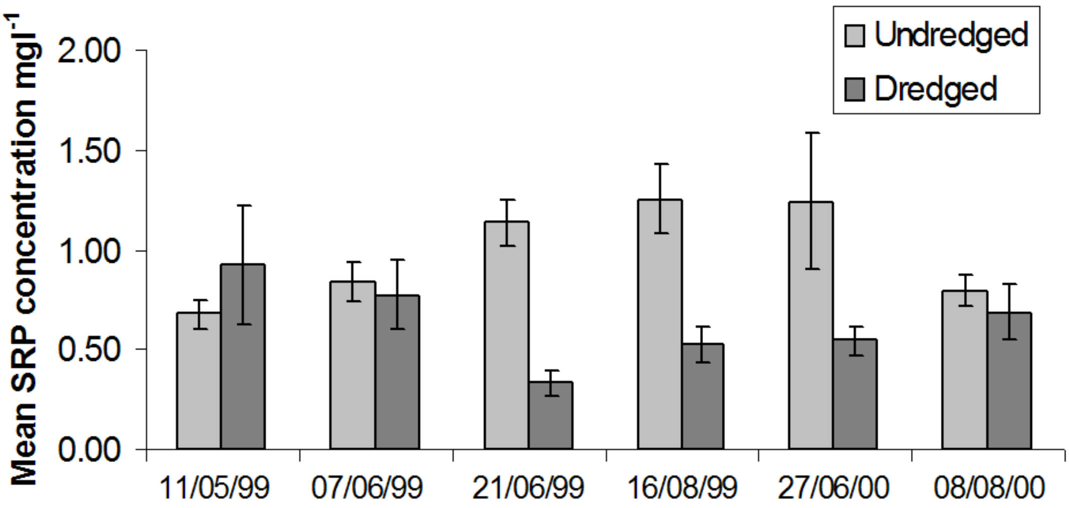

Figure 10. Mean SRP concentration recorded in the interstitial pore water of the dredged and undredged sediment

This research measured the SRP concentration in the sediment before and during the release experiment. The change in SRP concentration recorded in the sediment pore water during the experiments provided a better correlation with the initial sediment P release rate than the actual pore water SRP concentration recorded at the start of each experiment (Figure 11 and Table 4). The correlation coefficients calculated for percentage change in pore water SRP concentration and SRP release rates were below 0.60, supporting the fact that P release is not explained fully by change of sediment SRP concentration or actual concentration alone.

This indicates that release is not simply a function of sediment SRP concentration; it is also a result of other factors such as bacteria, P binding, pH and redox (Bostrom, 1982) some of which were not specifically investigated here.

The timing of sediment P release within algal dominated Barton Broad appears to be correlated with late summer peak algal biomass in the water. It is probable that settlement of algal matter onto the sediment surface increases the organic content of the surface sediment, contributing to increased P release through breakdown of the oxic microzone. In addition the underwater light climate, presence of benthic algae and potentially water temperature due to increased solar energy absorption are all influenced by this water turbidity. All of these factors influence the conditions in the surface sediment. In addition the overall amount of SRP release from the sediment is likely to be moderated by the SRP concentration as shown by potential release rates in Figure 9, calculated from sediment SRP concentrations.

	Undredged pore water concentrations				Dredged pore water concentrations			
	% change 2-21cm (1)	% change 2cm (2)	SRP 2-21cm (3)	SRP 2cm (4)	% change 2-21cm (1)	% change 2cm (2)	SRP 2-21cm (3)	SRP 2cm (4)
0-20 hours release rate	0.46 **	0.52 ***	0.54 ***	0.49 **	0.49 **	0.58 ***	-0.30	-0.10

Table 4. Spearman rank correlation for the relationship between the SRP release rate (0-20 hours) from all six experiments in 1999 and 2000 and the (1) % change in interstitial pore water phosphorus concentration over 2-21 cm depth, (2) % change at 2 cm only, (3) the mean SRP concentration over 2-21 cm depth recorded at 0 or 18 hours of the experiments and (4) the SRP concentration at 2 cm only.

***p≤0.005; **p≤0.01; *p≤0.05

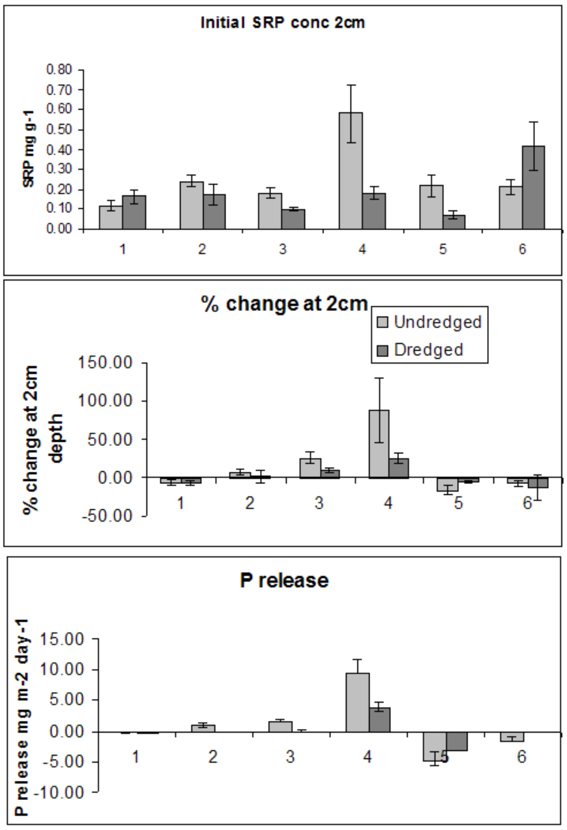

Figure 11. Interstitial pore water concentration at 2 cm at the start of the experiment, percentage change from the start to 20 hours and P release recorded in the overlying water 20 hours after the start of the experiment.

Dredging Barton Broad increased the Fe:P ratio compared to the undredged sediment (Figure 8), this is likely to have contributed towards the reduced P release measured. Another factor that may have reduced P release was that after dredging the reduction in P concentration was mainly within the organic bound P, which is loosely sorbed and therefore more available for release.

Overall the sediment P release within Barton Broad, calculated from mean initial P release four experiments in 1999, contributed 1.50 g P m^{-2} yr^{-1} tonnes of P into the water column from the undredged and 0.52 g P m^{-2} yr^{-1} from the dredged sediment. This loading is within the same order of magnitude as external catchment loading, which following phosphorus control on effluents discharging to the River Ant at the end of the 1970s, was halved by the year 2000, falling from over 10 g P m^{-2} yr^{-1} to less than 5 g P m^{-2} yr^{-1} (Phillips *et al*, 2005).

From the evidence presented here, Barton Broad has switched from a major nutrient source originating from the sediment in the 1980s, to a seasonally limited nutrient source and potential overall nutrient sink in the late 1990s. However sediment remains a significant source contributing over 10% of the phosphorus entering the lake. This results from a combination of changing conditions in the water and sediment, partly as a result of dredging. Thus in combination with external nutrient control dredging in Barton Broad has proved effective for nutrient reduction in the water of the broad and assisted with achieving Water Framework Directive Targets.

3.2 All Broads

Overall aim: To reduce internal phosphorus loading from relatively recently deposited nutrient rich sediment.

A total of 13 broads have been subject to sediment removal, via mud pumping, as part of the Broads lake restoration programme (Kelly, 2008). Some of these have been monitored on a regular basis for water quality parameters.

3.2.1 Crome's Broad

Crome's Broad has two basins, north and south, divided by a reed swamp. The north basin was mud pumped in 2005 and the south basin in 1988. The former dredging was accompanied by only partial isolation due to a leaking sluice and isolation was completed in 1992.

Internal phosphorus loading from the nutrient rich sediment (TP over 3 mg g^{-1} before dredging) has resulted in high total phosphorus concentrations in the water (0.1-0.25 mgl^{-1}) for over two decades. However in 2005 and 2006, following the Broads Authority removing sediment in the north of the broad the TP concentrations in the water are well below 0.1 mgl^{-1} for the first time for two consecutive years.

Crome's Broad has been subject to fish kills, the most notable in 1999, potentially affecting the nutrient cycling via benthic invertebrates and the presence of dense beds of water plants.

3.2.2 Cockshoot Broad

The isolation and suction dredging of Cockshoot in 1981 has been well studied (Moss, 1986). This original restoration was initially successful resulting in increased water depth capable of supporting submerged plant growth. However ingress of fish around the barriers has resulted in loss of water clarity and water plants at times. The general trend in improving water quality in Cockshoot is also shown in the adjacent River Bure at Horning, demonstrating a potential hydrological connection with the river or similar recovery period. However TP concentrations remain slightly higher in the broad compared to the river and thus it is likely that sediment nutrient release remains an influence.

Since 2004 leaks to the river barriers have been repaired, roach and bream have not been found in significant numbers in the broad (Hoare, 2007) and TP concentrations in the water have not exceeded 0.05mg l^{-1}.

3.2.3 Hoveton Little

Hoveton Little is connected to the main river and its decreasing TP concentration since 1990 is in step with other river connected Bure broads, such as Wroxham and Hoveton Great that have not been mud pumped. Thus the decrease is likely to be more of a result of decreased external input from sewage treatment works upstream rather than as a direct result of sediment removal.

However the reservoir of sediment P has been reduced, helping limit the potential for nutrient release. Without specific research it is difficult to identify the contribution of sediment removal to overall restoration in these connected well-mixed broads that are also subject to significant external nutrient reduction.

3.2.4 Barton Broad

The extensive evidence for the role of sediment removal in Barton is pulled together in section 3.1. Looking at the water quality data alone dredging appears to cause a step change in TP concentration in the water (Figure 12). However during the sediment removal starting in 1996further effluent treatment was introduced at an upstream sewage treatment works, meaning that it is difficult to tease out the relative impact of these two restoration measures.

The extensive research on Barton has however concluded that sediment removal has made a significant further reduction in P input from the sediment.

Following both sediment removal and external nutrient measures annual mean TP has consistently been below 0.075 mg l^{-1}.

3.2.5 Alderfen Broad

Alderfen received high nutrient concentrations from the septic tank discharge and treatment plant discharge draining into its inflow stream prior to isolation in 1979. Following dredging large internal P release continued up until 1999, resulting in maximum water concentrations of 2 mg l^{-1} compared with concentrations less than 1 mg l^{-1} prior to sediment removal. However since 2000 TP has reached lows of 0.1 mg l^{-1} in the broad. Over recent years a high biomass of submerged plants has also appeared on a regular annual basis, with several species recorded in recent years.

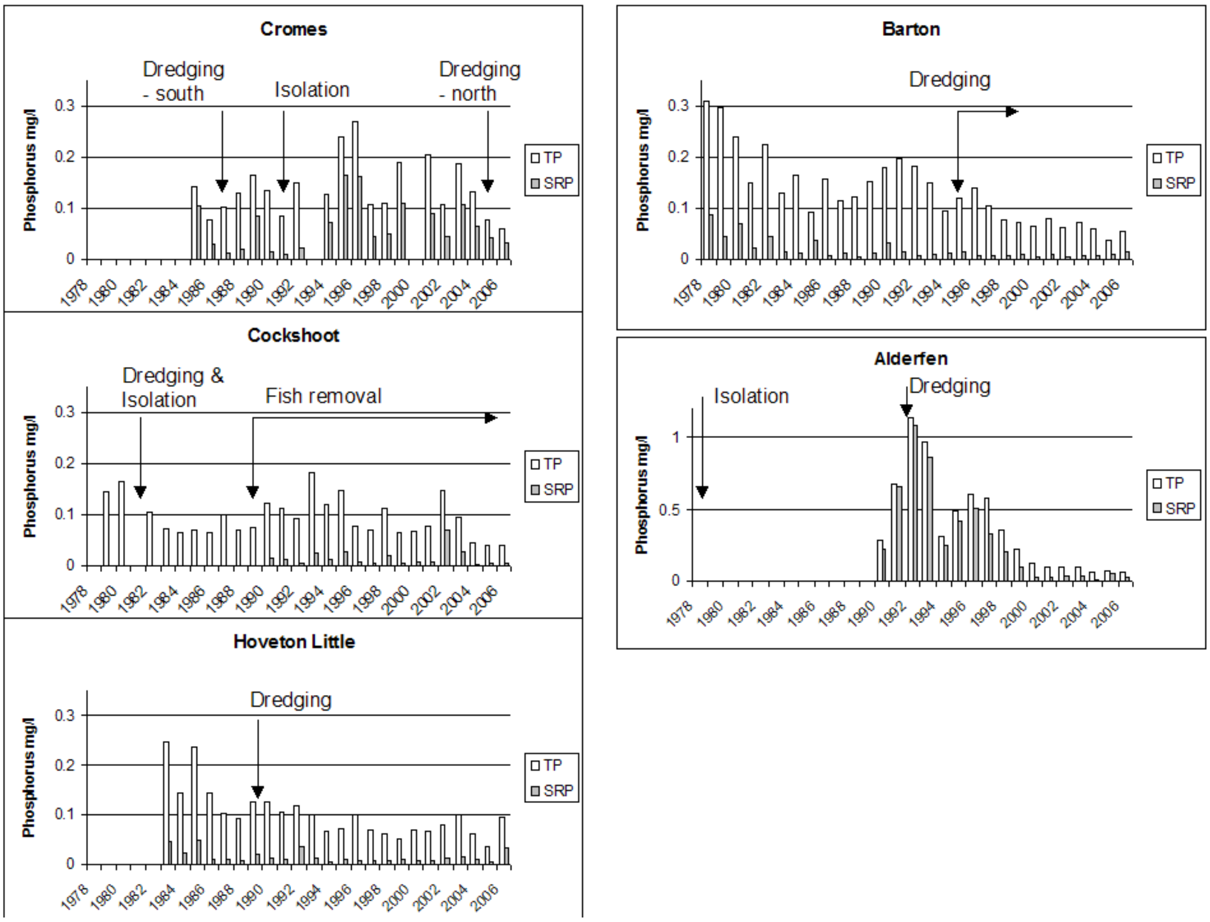

Figure 12. Changes in the water quality parameters in broads subject to sediment removal. Figures are annual means.

3.3 Overall aim:

- **Maintain open water for submerged water plants**

Submerged plants find it hard to grow in less than 50 cm water depth, especially in places such as the Broads with mobile fine organic silt providing difficult rooting conditions and populations of grazing water birds, such as coot and swans.

Sediment removal in the Broads aims to remove the nutrient rich silt without deepening waters beyond their original basin, marked by peat, gravels or clay (Wakelin & Kelly, 2007). Several silted-up broads, which were less than 50 cm deep, have had sediment removal to restore the depth to 1-1.5 m depth and remove the recently deposited silt for both navigation and conservation reasons. These include Alderfen, Cockshoot, Pound End, Cromes, Barton, Barnby, Buckenham, Hassingham, Catfield and Belaugh. In all of these broads water plants have returned, with the exception of Pound End and Hoveton Little broads, where the water remains too turbid for plants to recolonise.

3.4 Overall aim:

- **Increase water depth for navigation purposes**

The Broads is a unique recreational resource, providing safe inland navigation over 125 miles of lock free waterways. It is home to approx 13,000 recreational vessels, both motor and sail, with up to 1000 vessel movements per day during the summer in certain reaches. Boats provide an excellent way to explore this extensive wetland, which is internationally recognised, with large areas of its waterways designated under both the Birds and Habitats Directive for the ecology and wildlife it supports.

The Broads Authority carries out maintenance dredging[1] on a routine basis with the main objective being to secure a reasonable depth for navigation. These ideal depths are identified through user-agreed Waterway Specifications (Wakelin & Kelly, 2007). Maintenance dredging may contribute in a small way to removal of nutrients from the Broads; however this has never been investigated. Maintenance dredging in the waterways of the Broads is principally for navigational purposes.

3.5 Overall aim:

- **To remove toxic substances**

The Sediment characterisation survey undertaken as part of the Sediment Management Strategy (Wakelin & Kelly, 2007), shows only a small proportion of the sediment in the Broads is contaminated.

These sediments are located in the Yare from Norwich to upstream of Cantley and are contaminated with mercury and copper discharged from the Whitlingham sewage treatment works, Norwich, from 1964 to 1973. An Environment Agency policy provides a clear framework for decision-making relating to the treatment of dredgings in the Yare and concludes that the bulk of material needs to be disposed of to licensed landfills with site-specific risk assessment for all other disposal routes. The vast quantities of potentially contaminated sediment in the River Yare preclude removal of all this contamination.

Other contaminants include the now banned antifoulant tributyltin (TBT) polycyclic aromatic hydrocarbons (PAH's), which are present predominantly in boatyard sediments and sites affected by industry but can also be found in many river connected broads (Sayer *et al,* 2006). The latter are principally derived from the combustion of fossil fuels, such as in boat engines, as well as from oil and fuel spillages.

To date minimising inputs of these contaminants has been the focus of legislation such as that under the EU Water Framework Directive (WFD). Article 16 (1) of the WFD requires that member states put in place measures to effect "the cessation or phasing-out of discharges, emissions and losses" of priority hazardous substances such as mercury, TBT and PAHs. The approach to priority hazardous substances being within or lost to overlying water from contaminated sediments will be confirmed, when Environmental Quality Standards (EQS) for sediment-bound contaminants are agreed.

Generally due to the low levels of contamination in the lakes of the Broads, dredging to remove contaminants is not required. However for contaminated hotspots, such as boatyards and urban areas, a risk-based, cost benefit, approach is required to assess the impact of removal and disposal or retention within the aquatic system with the inherent risk of further release and continued impact on aquatic organisms. The outcome of these decisions can have considerable economic implications.

References

Andersen, J.M. (1976) an ignition method for determination of total phosphorus in lake sediments. Wat. Res., 10, 329-331.

Appleby, P.G., Nolan, P.J., Gifford, D.W., Godfrey, M.J., Oldfield, F., Anderson, N.J. & Batterbee, R.W. (1986) ^{210}Pb dating by low background gamma counting. Hydrobiologia 143, 21-27.

Ayres, K.R., Sayer, C.D., Skeate, E.R., Perrow, M.R. (In Press). Palaeolimnology as a tool to inform shallow lake management: an example from Upton Great Broad, Norfolk, UK. Biodiver. Conserv.

Bengtsson, L., Fleischer, S., Lindmark, G. & Ripl, W. (1975) Lake Trummen restoration project 1. Water and sediment chemistry. Verh. Internat. Verin. Limnol 19 1080-1087.

Boers, P.C.M. (1991) The release of dissolved phosphorus from lake sediment. Thesis.

Boström, B., Jansson, M. & Forsberg, C. (1982) Phosphorus release from sediments. Arch. Hydrobiol. Beih. Ergebn. Limnol. 18: 5-59.

George, M. (1992) The land use, ecology and conservation of Broadland. Packard Publishing Limited.

Hoare, D. (2007) Biomanipulation Review - Broads Case Studies, Broads Authority Report.

Kelly, A (2008) Lake Restoration Strategy – Appendix 5 History of lake restoration measures, Broads Authority Report.

Klein, J. (1990) Sediment dredging and macrophyte harvest as lake restoration. Land and Water 42 (3) 10-12.

Klein, J. (1998) Sediment dredging and macrophyte harvest as lake restoration techniques. Land and Water 42 (3) 10-12.

Marsden, M.W. (1989) Lake restoration by reducing external phosphorus loading: the influence of sediment phosphorus release. Freshwat. Biol. 21: 139-162.

Moss, B. (1980) Further studies on the palaelimnology and changes in the phosphorus budget of Barton Broad, Norfolk. Freshwat. Biol. 10: 261-279.

Moss B., Balls H., Irvine K., Stansfield J. (1986) Restoration of two lowland lakes by isolation from nutrient-rich water sources with and without removal of sediment. J. App. Ecol. 23 391-414.

Mueller, D., Pfitzner, S., Wunderlich, M. (1998) Impact of dredged material relocation on the oxygen and nutrient budget. Wasser und Boden. 50(10) 26-32.

Peterson, S.A. (1982) Lake restoration by sediment removal. Wat. Res. Bull. 18 (3) 423-435.

Phillips G.L., Bramwell, A., Pitt, J., Stansfield, J. & Perrow, M. (1999) Practical application of 25 years research in the management of shallow lakes. Hydrobiologia, 395/396, 61-76.

Phillips, G.L., Kelly, A., Pitt, J-A., Sanderson, R., Taylor, E.D (2005) The recovery of a very shallow eutrophic lake, 20 years after the control of effluent derived phosphorus. Freshwater Biology 50 (10) , 1628–1638.

Pitt, J-A., Kelly, A and Phillips, G.L. (1997) Control of nutrient release from sediments. Broads Authority report.

Reynolds, C.S., Davies, P.S. (2001) Sources and bioavailability of phosphorus fractions in freshwaters: a British perspective. Biol. Rev. 76 27-64.

Rose, N. L., Appleby, P.G., Sayer, C.D. & Bennion H. (2005) Sediment accumulation in the Broads, Broads Authority Report.

Sayer C.D., Hoare D. J., Simpson G. L., Henderson A. C. G. & Liptrot, E. R. (2006) Environ. Sci. Technol., 40 (17), 5269 -5275.

SCOPE, (1998) Shallow lakes biomanipulation and eutrophication, October, issue 21.

Sebetich, M.J. and Ferriero, N. (1997) Lake restoration by sediment dredging. Verh. Internat. Verin. Limnol. 26 776-781.

Sondergaard, M., Kristensen, P. & Jeppesen, E. (1993) Eight years of internal phosphorus loading and changes in the sediment phosphorus of lake Sobygaard, Denmark. Hydrobiologia 253: 345-356.

Tomlinson, M., Perrow M. (2006) Summary of fisheries work undertaken at Cockshoot and Barton Broads: spring 2006. Broads Authority Report.

Wakelin, T. & Kelly, A. (2007) Sediment Management Strategy for the Broads. Broads Authority Report.

Zhao, Y., Sayer, C.D., Birks, H.H., Hughes, M. & Peglar, S.M. (2006) Spatial representation of aquatic vegetation by macrofossils and pollen in a small and shallow lake.

Appendix 1.

BROADS AUTHORITY STANDARD ANALYTICAL SUITE

Purpose: Results of sediment survey, across navigable waterways in Broads Authority executive area, will form the basis of the Sediment Management Strategy

Chemical determinands mg kg^{-1}

1. Cadmium (Total)
2. Chromium (Total)
3. Copper (Total)
4. Nickel (Total)
5. Lead (Total)
6. Zinc (Total)
7. Arsenic (Total)
8. Mercury (Total)
9. Selenium (Total)
10. Boron (Total)
11. Tin (Total)
12. Tri-butyl Tin (TBT)
13. Total Polycyclic Aromatic Hydrocarbons (PAH) and speciated where required
14. Phenols *(Total monohydric)*
15. Potassium
16. Total phosphorus
17. Magnesium
18. Ammonia
19. Total Nitrogen
20. Nitrate
21. pH
22. Chloride

Leachate test for selected determinands as required

Physical determinands

Loss of Ignition
Organic Matter Content
Air-dried solids (at 30^oC)
Sample description - *Colour, sand/silt/clay, content, smell*

Additional determinants for high risk sites or specific study

Particle Size Distribution
Total Sulphide
Silver (Total)
Cyanide (Total)
Antimony (Total)
Barium (Total)
Beryllium (Total)
Boron (Available)
Cobalt (Total)
Molybdenum (Total)
Thallium (Total)
Tungsten (Total)
Vanadium (Total)