Hickling Broad Dossier

Part of the review of lake restoration practices and their performance in the Broads National Park, 1980-2013

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1 Basic site characteristics

Hickling Broad is a large shallow brackish coastal lake and at 154 ha is the largest of the Broads. Together with Heigham Sound and Horsey Mere, Hickling forms part of a complex of broads on the Upper Thurne system (Figure 1). The water level in the Upper Thurne broads is higher than the surrounding land with water entering the system via land drainage pumps, the largest entering Horsey Mere. There is a much smaller water input to Hickling Broad via the Catfield land drainage pump with water leaving Hickling Broad via Candle Dyke where it mixes with water from Horsey. The Thurne broads also have a daily physical tide of 5-10 cm and a larger range over the spring/neap cycle that is caused by the confluence of the River Thurne with the tidal section of the River Bure 8km downstream. The main water flow in the system is through Horsey Mere, which receives water from the largest drainage pumps, but on the rising tide water from the Heigham Sound area is pushed back upstream thus entering Hickling Broad. The grazing marshes to the north of Horsey Mere have a naturally saline water table due to their proximity to the coast, but Horsey has become progressively more saline over the last century, and especially since the 1960s, as a result of pumped land drainage. This rising salinity has been transferred to Hickling where there is little moderation by the only significant freshwater inflow via Catfield Dyke.
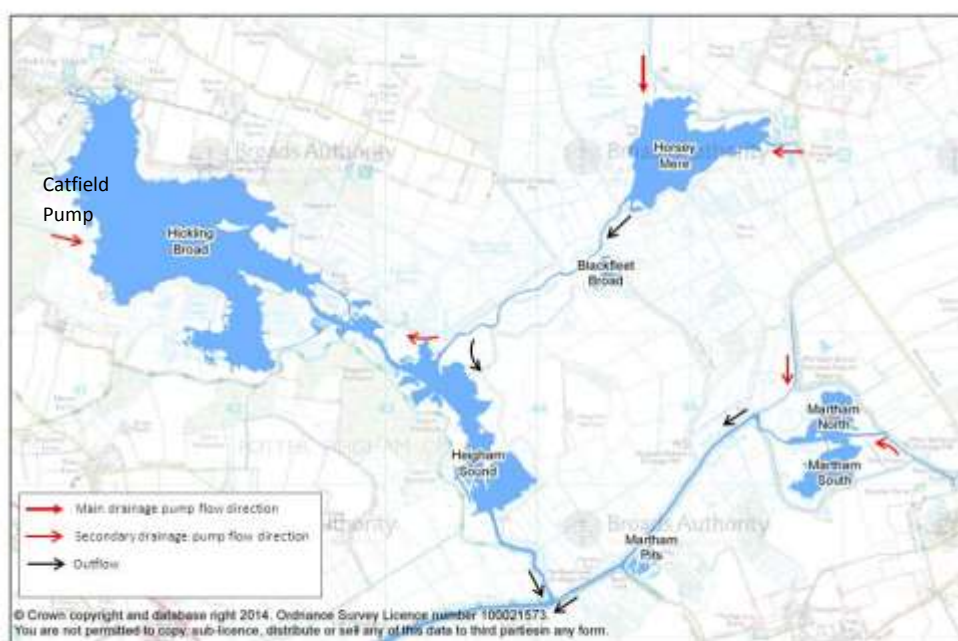

Figure 1 Geographical context of Hickling Broad

2   Basic site characteristics for Hickling Broad

Hickling Broad (UK Lake WBID 35640)

Location	TG419214
	(Easting 641988, Northing 321411)
Water body area (Ha)	153.8
Mean depth (m)	1.3
Mean alkalinity (mEq/l)	2.8
River system	River Thurne
Connectivity	Riverine
Navigation	Yes
Designations	Upper Thurne Broads and Marshes SSSI
	Broadland SPA and Ramsar
	The Broads SAC

Water Framework Directive water body GB30535640	Phosphorus boundary values
	High/Good: 44 μgl^{-1}
	Good/Moderate 62 μgl^{-1}
	Moderate/Poor 123μgl^{-1}
	Poor/Bad 247μgl-1
	Chlorophyll boundary values
	High/Good 9 μgl^{-1}
	Good/Moderate 20 μgl^{-1}
	Moderate/Poor 39 μgl-1
	Poor/Bad 118 μgl-1

Water Framework Directive Status (2012)	Poor Ecological Potential
	Phytoplankton: Poor
	Macrophytes: Good
	Phytobenthos: Good
	Invertebrates: Moderate
	Fish: N/A
	Phosphorus: Moderate
SSSI Condition:	Unfavourable declining

3   Palaeolimnology

Hickling Broad has been the subject of a number of palaeolimnological studies, namely; Moss (1978); Phillips *et al.* (1978); Liptrot (2002); Sayer *et al.* (2006); Hoare (2007) and Holmes *et al.* (2010). Plant and animal (cladocera, macroinvertebrates, bryozoans, molluscs & fish) macrofossils and diatom frustules have been examined from dated and undated sediment cores along with concentrations of TBT contamination. The majority of historic plant records for this review have been extracted from Bales *et al.* (1993) and the database of Madgwick (2009), with further data coming from Mason & Bryant (1975). Considerable historical and anecdotal

evidence exists for ecological change at Hickling Broad, which can be directly compared with the core data.

## 3.1 Sediment stratigraphy and dating

In 1975, Moss (1978) collected a number of sediment cores (including that presented in Phillips *et al.* (1978), which was undated but stratigraphically similar (Figure 6) from a quiet southerly bay, Heigham Corner, in Hickling Broad. This area is floored by a mixture of amorphous peat and clay, the latter deposited during a marine transgression about a thousand years before the basin was excavated. At 49 cm depth the core deposit was a grey clay with some intermixed fibrous peat. Between 49 and 34 cm it was of clay with an increasing organic content which was overlain at 30-34 cm with a black peaty layer. This seems to mark the bottom of the excavated basin. Similar deposits underlie Heigham Sound, just to the south of the coring site (Lambert *et al.* 1960). Within the lake sediment there is little differentiation in fresh appearance, the column being a uniform grey-brown colour. The ignited sediment, however, showed some marked changes, most notably, increasing organic content from ~14 cm to 6 cm and the highest carbonate (marl) content from 6 cm to 9 cm: (Figure 3) gives details of the appearance of a typical core and the age of the core layers determined by ^{210}Pb radiometric dating. Dates obtained were 1972 ± 6 yrs for 2 cm; 1955 ± 9 for 6 cm; 1962 ± 7 for 9 cm. Interpolation from the best fit curve (see Moss, 1978) gives a date of 1935 ± 9 for 15 cm. Figure 4 gives data on the relative rates of deposition of dry, inorganic and organic matter in Hickling Broad. Sedimentation rates, calculated from the datings, were very low in the first 500 years of the lake's existence if a nominal date of 1400 AD is taken as the date of flooding, and averaged 0.028 cm yr^{-1}. In the 1930s or 1940s the rate increased to 0.25 cm yr^{-1} and from the mid 1960s onwards it increased to its current rate of about 0.5 cm yr^{-1}. The later increase corresponds with the predominance of marl in the core, the earlier one with changes in organic content around 15 cm.

Figure 2 Location of the HICK1 sediment core.

Liptrot (2002) collected core HICK1 on 27/04/2002, also from the Heigham Sound area of Hickling Broad (52° 43.498'N, 1° 35.379 E), in the centre of the basin, approximately 500 m from the main navigation channel (see Figure 2). The top 37 cm of the core comprised dark brown non-laminated mud, representing the period of true lacustrine sedimentation subsequent to the earlier saline fen phase. Organic matter

is relatively stable (c. 22-25%) until around 13 cm where there is a small but sharp increase (to values >25%), followed by stable organic matter once again. Carbonate is low (<5%) below around 30 cm, above which there is a steady increase to a peak of ~12% at around 13 cm. Carbonate then declines to ~6% at 11 cm and thereafter remains broadly stable to the core top (Holmes *et al*. 2010). HICK1 was radiometrically dated using both ^{210}Pb and ^{137}Cs. Since AD 1950 the mean mass sediment accumulation rate has been 0.31 cm year^{-1}, only reaching a rate of 0.5 cm year^{-1} during the last decade. Sedimentation rates were significantly lower during the period AD1920-50, although because of low ^{210}Pb concentrations below 15 cm, values from this period have a large uncertainty. ^{137}Cs activity shows a relatively well-resolved peak between 13 and 16 cm that is thought to record the 1963 fallout maximum from the atmospheric testing of atomic weapons. 30 cm is dated at c.1845, with 14 cm dated at c.1968 (Hoare, 2007; Holmes *et a*l. 2010).
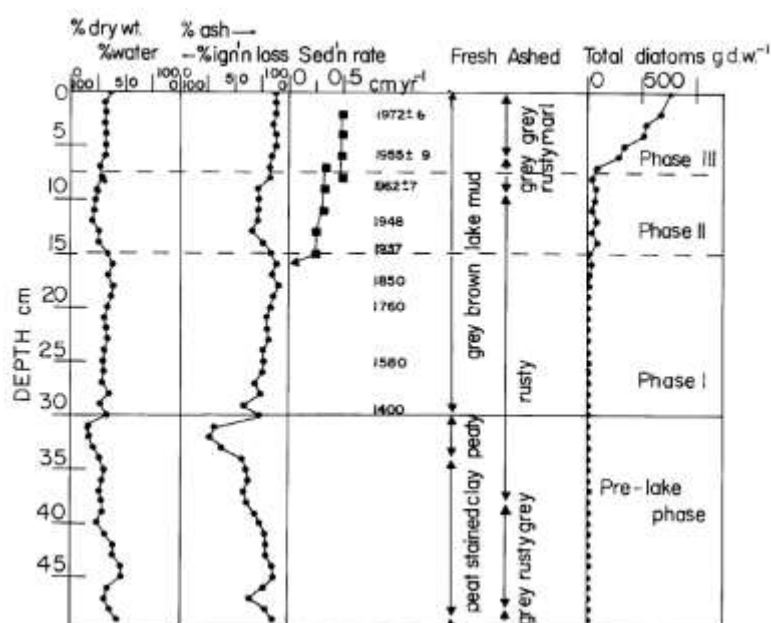
Figure 3 Physical characteristics, sedimentation rates and diatom content of the core taken by Moss (1978). Dates are deduced from ^{210}Pb datings.
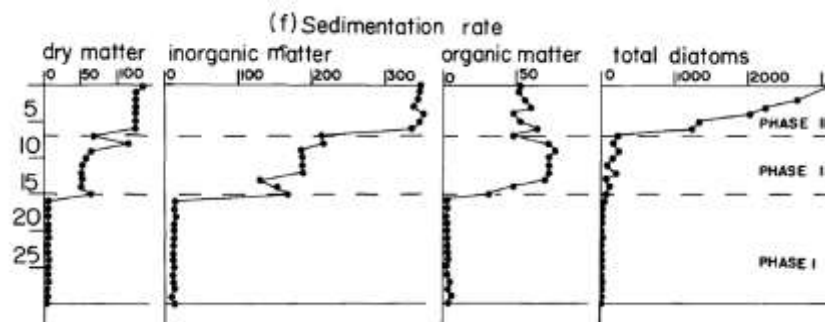
Figure 4 Sedimentation rates of dry, inorganic and organic matter and of total diatoms. Figures are relative and were derived by multiplication of the sedimentation rate (mm yr-1) at the time by, respectively, the percentage ash in dried sediment, the percentage loss on ignition at 500°C from dried sediment, and the diatom content per unit dry weight, for each 1 cm slice of the core (taken from Moss, 1978)
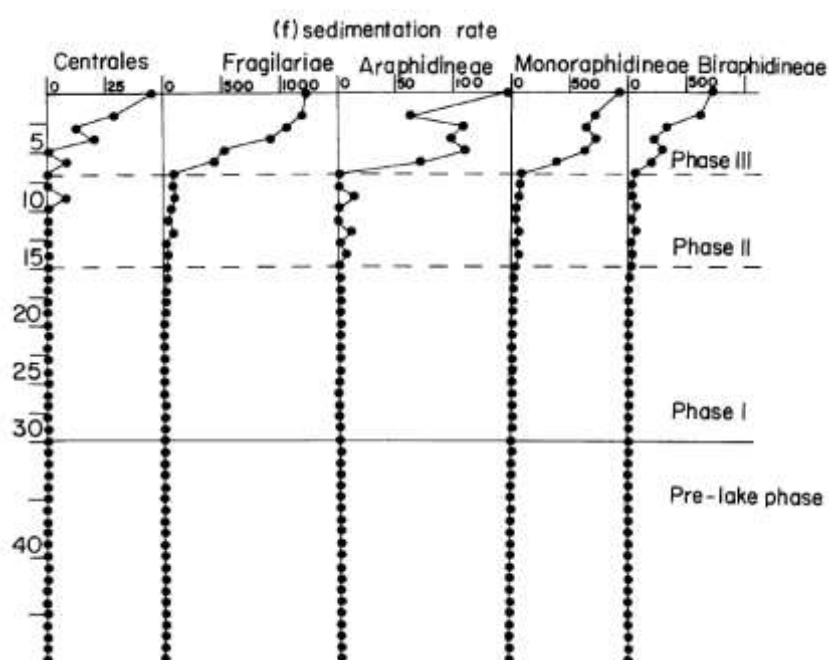
3.2 Diatoms

For the sediment core discussed by Moss (1978), Figure 3 - Figure 5 illustrate the results of diatom analysis. Figure 3 illustrates the total number of diatom frustules per unit dry weight of sediment. For the same core, Figure 4 displays relative rates of deposition of dry, inorganic and organic matter and of diatom frustules; Figure 5 illustrates relative sedimentation rates of major groups of diatoms and Figure 6 illustrates relative sedimentation rates of the most abundant diatom species and genera. Figure 7 taken from Phillips *et al.* (1978), illustrates changes in numbers per unit dry weight of sediment, of different diatom groups Figure 8 and Figure 6b (Hoare, 2007 & Holmes *et al.* 2010 respectively) illustrate dated stratigraphic changes in the percentage relative abundance of diatom functional groups for HICK1.

Figure 3 (Moss, 1978) and Figure 7 (Phillips *et al.* 1978) illustrate that below ~20 cm, diatoms were present in Hickling Broad, though scarce, probably signalling the beginning of lake sediment accumulation (Phase 1). The beginning of this phase appears equivalent to ~34 cm in core HICK1 (Figure 8 and Figure 10b). Holmes *et al.* (2010) reported remains of diatoms from 34 cm in core HICK1, although the continued occurrence of foraminifera until 30 cm and presence, albeit in extremely low abundances, of the marine diatom *Dimeregramma minor* at 31 cm, suggests continued marine influence and possibly some mixing of materials from the saline fen phase and accumulating lake muds.

Moss (1978) describes a gradual increase in diatom numbers from ~15 cm to 6 cm (Phase 2; 1930s to 1960s). Increases in Monoraphidineae, Biraphidindae, Araphidineae (almost entirely epi-benthic *Fragilaria*) and total epiphytes followed the trend in total diatoms. The approximate equivalent in HICK1 is zone 2 (29-13 cm; c. 1845-1968), where the diatom community is dominated by epi-benthic species,

especially *Achnanthes kuelbsii*, *Pseudostaurosira* (formerly *Fragilaria*) *brevistriata*, *Staurosirella* (formerly *Fragilaria*) *pinnata* and *Cocconeis placentula*.

Moss (1978) and Phillips *et al.* (1978) describe a marked increase in diatom numbers from 6-7 cm upwards, including increases in planktonic Centrales, predominantly *Cyclotella kutzingiana* (see Figure 6)*.* This phase is almost certainly equivalent to zone 3 (<13 cm (AD 1968 ± 4 years) in HICK1. Sayer *et al.* (2006) and Holmes *et al.* (2010) showed there to be distinct changes in the diatom flora (and other ecological groups) at ~13 cm, with a sharp decline in the percentage relative abundance of plant-attached (epiphytic) diatom species, notably *C. placentula*, and an increase in species within the *Staurosira-Staurosirella-Pseudostaurosira* complex (termed *Fragilaria* spp. in Figure 10b) and centric planktonic species, coinciding with observed increases in phytoplankton populations in Hickling Broad in the 1970s (Phillips *et al.* 1978). PCA of the diatom data suggests a sudden increase in species turnover at this time, coincident with the point of first TBT detection (Figure 8 and Figure 10b). Between 13 cm and the core top (AD 1970-2002), there was relative stability, with only a slight resurgence of epiphytic diatoms above 8 cm (1990s) – this phase is not seen in the earlier cores of Moss (1978) and Phillips *et al.* (1978).

Figure 5 Sedimentation rates (relative values) of major groups of diatoms in the core. Data for the pre-lake phase assume a similar sedimentation rate to that during phase 1 in the absence of definite dates for these layers (from Moss, 1978).

(f) sedimentation rate
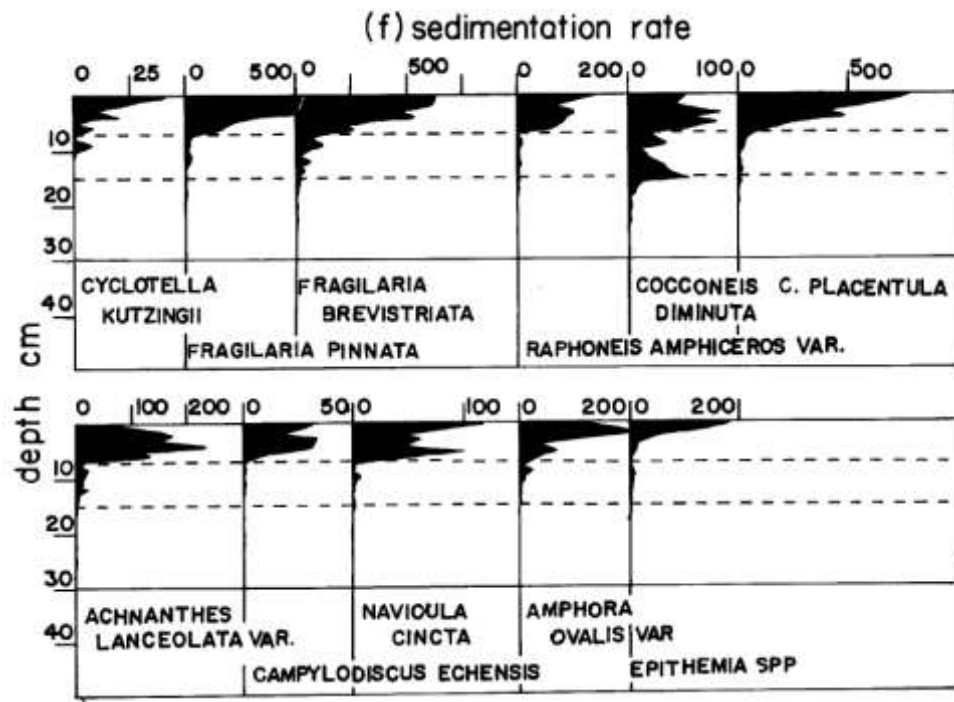
Figure 6 Sedimentation rates (relative values) for *Cyclotella kutzingiana* Thr. and the most common diatoms in the core – *Fragilaria pinnata* Ehr., *F. brevistriata* Grun., *Raphoneis amphiceros* var *rhomboides* Cl., *Cocconeis diminuta* Pant., *C. placentula* (Ehr.), *Achnanthes* lanceolata var *elliptica* Cl., *Campylodiscus echensis* Her., *Navicula cincta* Cl., *Amphora ovalis* var. pediculus* Kutz. and *Epithemia* spp. (from Moss, 1978).
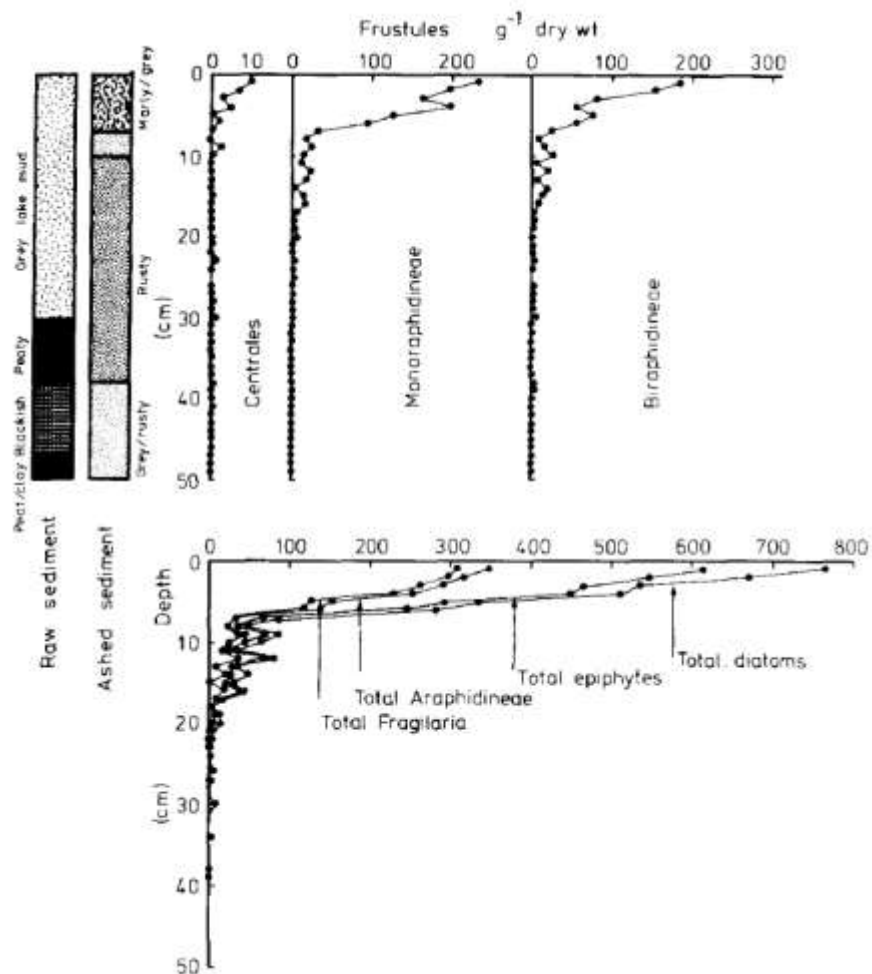
Figure 7 Changes in diatom numbers in a sediment core collected from Hickling Broad in the late 1970s. Numbers are given per unit dry weight of sediment (taken from Phillips *et al*. 1978).

3.3 Historic macrophyte records

Table 1 Records of aquatic vegetation in Hickling Broad and Heigham Sound from the 1800s to 1970s (from Bales et al., 1993).

Year	Record	Reference
	competes successfully with the larger *Chara* sp. and it does not appear to have altered its status there since Bennett first described it'.	(1952)
1954	Reports in the Eastern Daily Press state 'Improved methods of weed cutting will enable four times the amount of weed lifted in a year to be lifted by the early summer of 1955'. 'Total destruction of weed at Hickling is now regarded as likely in the foreseeable future.'	
1960	A weekend party from the Botanical Society of the British Isles recorded the charophytes: *Nitellopsis obtusa, Chara connivens, C. vulgaris, C. hispida, C. delicatula, C. aculeata, C. baltica, C. aculeata*.	Phillips (1963)
1956–65	*Chara hispida* var. *baltica* (Bruzz) Wood	Petch & Swann (1968)
July 1968	Cover by submerged aquatic plants 80%. Abundant: *Zannichellia palustris, Utricularia* sp., *Myriophyllum* sp., *Fontinalis, Chara* sp., *Cladophora*. Present: *Potamogeton pectinatus, Ceratophyllum demersum, Najas marina, Hottonia palustris, Hippuris vulgaris, Nitella* sp.	Morgan (1972)
1968	'The Thurne Broads still possess a remarkably rich assemblage of macrophytes. *Utricularia vulgaris, Myriophyllum alterniflorum, Ceratophyllum demersum, Potamogeton pectinatus* and *P. friesii* were all abundant and patches of *Najas marina* found in Hickling Broad.'	M. George (1970, unpublished manuscript, Nature Conservancy Council for England, Norwich)
1972	(Oct./Nov.) 'Water relatively turbid and aquatic macrophytes appeared to be sparse or absent in the greater part of the water area. The macrophytes were mainly confined to the margins of the open water. *Myriophyllum spicatum* and *Potamogeton pectinatus* plentiful (or locally plentiful). *Fontinalis antipyretica* and *Hippuris vulgaris* frequent (or locally frequent). *Chara* sp. and *Najas marina* rare.	R.J. Hornby (1972, unpublished report, Nature Conservancy Council for England, Norwich)
1972	(Nov.) '*Myriophyllum spicatum* and *Potamogeton pectinatus* were dominant in Hickling Broad with *Hippuris vulgaris* dominant in Heigham Sound. *Chara* and *Myriophyllum* – scattered patches. *Fontinalis antipyretica, Ceratophyllum demersum, Zannichellia palustris* and *Najas marina* very scarce.'	Mason & Bryant (1975)
1974	'Hardly any macrophytic plants found in the open water of the Broad. Patches of *Myriophyllum, Hippuris* and *Potamogeton pectinatus* were present sporadically at the edges and R. Driscoll (pers. comm) found traces of *Najas marina, Ceratophyllum demersum, Potamogeton pusillus, Zannichellia palustris, Fontinalis* and *Utricularia*'.	Phillips & Moss (1978)

There are numerous historic records for submerged and floating-leaved plants in Hickling Broad. Table 1, extracted from Bales *et al.* (1993), details anecdotal evidence of aquatic vegetation in Hickling Broad and Heigham Sound from the 1800s to 1970s. Most information is not quantitative and surveys were probably restricted in scope, however, a general pattern emerges of change from a clear-water '*Chara*-lake' in the late 19th century to one with abundant vascular plants until the late 1960s. Then there was a major decline in plant biomass, with an almost complete loss in the mid-1970s followed by a partial recovery in the 1980s albeit under turbid water conditions. Supporting records can also be found in the database of Madgwick (2009).

The earliest comments on Hickling's aquatic vegetation are from 1816-1850, when Lubbock (1879) reported 'shallow water abounding with pochard grass' (probably *Chara aspera*). The earliest record to mention a species name is from July 1883, when Bennett (1884) described '*Chara aspera* in profusion at Hickling Broad, covering the bottom to the exclusion of all other aquatic vegetation' and '*Najas marina* in Hickling and extending into Heigham Sound in fair quantity', 'accompanied

by *Chara stelligera, polycantha, aspera*'. *Potamogeton praelongus* was present in 1890 (Morgan, 1972)

There is much evidence of ecological change in Hickling Broad since the beginning of the 20th century: Pallis (1911) reported clear water with dominant *Chara* (*Chara aspera, C. hispida* & *C. polycantha*), *Cladophora aegagropila, Potamogeton interruptus [P. pectinatus]*, *Vaucheria dichotoma*, *Zygnema stellium* and *Spirogyra* sp.; abundant *Ranunculus circinatus*, *Myriophyllum spicatum* and *Elodea canadensis*; less abundant *Hippuris vulgaris*, *Stratiotes aloides*, *Lychnothamnus stelliger* (*Nittelopsis obtusa*?) and rare *Najas marina*. In the 1920s the *Cladophora* and *Potamogeton* was apparently so abundant that it was a nuisance to boats and had to be cleaned out (R. Gurney, diaries: 1922, 1925). *Cladophora* and *P. pectinatus* persisted until the late 1940s, when 1968, the water was still clear and *Zannichellia palustris*, *Utricularia* sp., *M. spicatum*, *Fontinalis antipyretica*, *Cladophora* and *Chara* were abundant, with six other species present (Morgan, 1972). *Elodea* had disappeared. In 1972, R.J. Hornby recorded seven species and *Potamogeton pectinatus* had become abundant, while *Z. palustris* was not found. A similar list was obtained on 6/11/1972 by Mason & Bryant (1975), when the water was very turbid. Phillips & Moss (1978) reported that by1974 there were 'hardly any macrophytic plants found in the open water of the Broad.

3.4 Historic Cladoceran records

Taken from Bales *et al*. (1993)

Records of Cladocera in Hickling Broad made by Robert Gurney between 1904 and 1920. Data were extracted from unpublished, handwritten notebooks held in the Castle Museum, Norwich. Gurney's visits were made sporadically and systematic long-term sampling was not undertaken. Details of samples are given at the foot of the appendix. Hi, Hickling Broad; HS, Heigham Sounds; Ho, Horsey Mere

Bosmina longirostris Hi, 3, 6, 10, 15; HS, 18
Ceriodaphnia laticaudata Hi, 8, 11, 13, 14; HS, 18
C. pulchella Hi, 1, 2, 7, 8, 10, 11, 14, 15, 16; HS, 18, 19; Ho, 21, 22, 23
C. quadrata Hi, 6, 11; HS, 20
Daphnia longispina Hi, 1, 2, 8, 13, 17; HS, 18
Diaphanosoma brachyurum Hi, 6, 8; HS, 19
Polyphemus pediculus Hi, 13, 15
Scapholeberis mucronatus Hi, 8
Sida crystallina Hi, 12, 13, 14, 15, 16; HS 19
Simocephalus expinosus Ho, 21
S. vetulus Hi, 1, 2, 3, 4, 8, 10, 12, 14, 16; HS, 19, 20; Ho, 21, 22
Acroperus harpae, Hi, 14
Alona affinis Hi, 13, 14, 15, 16; Ho, 23
Chydorus globosus Hi, 5, 7, 9, 10, 13, 14, 16; HS, 19, 20
C. sphaericus Hi, 1, 2, 3, 4, 5, 6, 8, 9, 10, 12, 14, 16, 17; HS, 19, 20; Ho, 22, 23
Eurycercus lamellatus Hi, 2, 4, 5, 12, 16; HS, 19; Ho, 23

Graptoleberis testudinaria Hi, 15; Ho, 23
Ilyocryptus sp. Hi, 3
Lathonura rectirostris Hi, 8, 10; Ho, 23
Lynceus affinis Hi, 3, 4, 5, 8, 10; HS, 18, 20; Ho, 21
L. tenuicaudis Hi, 7, 9
Pleuroxus aduncus Hi, 1, 3, 4, 5, 8, 9, 10, 12, 14, 15; HS, 19; Ho, 21, 23
P. laevis Ho, 21

Sample details:
1. Ditch leading into Hickling Broad, 11 June 1904; 2. Near Hickling Staithe with *Potamogeton pectinatus*, 27 Sept. 1904; 3. Broad, 31 May 1905; 4. Swim Coots Bay, 31 May 1905; 5. Broad, 1 June 1905; 6. Broad, 21 Aug. 1905; 7. In *Chara*, Hickling Broad, 21 Aug. 1905; 8. Broad, 8 Sept. 1905; 9. In *Chara*, Hickling Broad, 18 Sept. 1905; 10. In *Chara* and *Potamogeton*, Hickling Broad, 1 Sept. 1906; 11. Channel, Hickling Broad, 11 June 1909; 12. Hickling Broad, by Pleasure Island, 18 Sept. 1910; 13. Broad, Oct. 1913; 14. Hickling Broad in *Chara*, 14 July 1919; 15. Broad, 23 July 1919; 16. In *Chara*, Hickling Broad, 13 July 1919; 17. Broad, 3 March 1920; 18. Heigham Sounds, 1 Jan. 1905; 19. In weed, Heigham Sounds, 22 Aug. 1905; 20. Deep Go Dyke and Heigham Sounds, 11 June 1909; 21. In *Potamogeton pectinatus* & *Chara*, Horsey Mere, 28 Sept. 1904; 22. Horsey Mere, 30 May 1905; 23. Horsey Mere, 14 July 1919

3.5 Plant and animal macrofossils

When HICK1 was collected in 2002, sediment samples were frozen and sufficient sediment remained to perform plant and animal macrofossil analyses at a later date (Hoare, 2007 & Holmes *et al.* 2010). Figure 9 displays the variation in the flux of macrofossil remains through core HICK1. A relatively low number of macrofossil taxa were found, which may be due to the smaller volume of sedimentary material available for macrofossil analysis at each depth level and/or Hickling's brackish waters excluding saline intolerant species. It has also historically been a charophyte-dominated lake, which will have acted to exclude other macrophyte species. However, despite the reduced number of taxa present, zonation of macrofossil results in core HICK1 display clear temporal changes in the remains.

The lowest zones 1a and 1b (Figure 9) were characterised by abundant *Chara* sp. oospores and also frequent detection of *Nitella* sp. oospores, dated as being present in the early 1900s through to their decline the late 1960s. Phillips (1963) provides direct evidence of charophytes in Hickling Broad into the early 1960s. Of the seven mollusc species found through HICK1, all were found in the lower two zones (Figure 9). The most abundant species was the mud snail, *Potamopyrgus antipodarum*, which was found mostly as immature individuals <2 mm. However, the depth profiles of adult and immature stages of this species followed a similar pattern throughout the core, suggesting a continuously high reproduction rate, but with relatively few individuals reaching the adult stage. *Gyraulus crista* was also found frequently, with

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greatest numbers occurring in zone 1a. Ephippia of four cladoceran taxa were also found in this lower part of the core. The most abundant ephippia were of *Ceriodaphnia* sp., which were found continuously through zones 1a and 1b. In the lower zones bryozoan statoblasts were also found sporadically and in low numbers (Figure 9).

Zone 2 appears to represent a transition in abundances of the HICK1 macrofossil remains dated as occurring in the late 1960s. *Chara* sp. oospores progressively increase up core and reach their maximum at 14 cm, as do orabatid mites and *Ceriodaphnia* sp. ephippia. These three proxies, however, decline sharply above 12 cm (1968 ± 4 years), with *Ceriodaphnia* sp. completely disappearing from the core in zone 3, consistent with a reduction in the availability of plant refuges. *Nitella* sp. oospores were absent from zone 2 and above. Conversely, the mud snail *Potamopygus antipodarum* - a species closely associated with open sediment surfaces as opposed to submerged plant beds - numbers suddenly increased at 14 cm, as did *Plumatella* sp. statoblasts. The peak abundance of these two proxies was greatest within zone 3. The only other mollusc species present relatively frequently in zone 3 was *Gyraulus crista*, but in lower numbers than in the bottom section of the core. In zone 1, the cladocerans *Simocephalus* sp. and *Leydigia* sp. were only found sporadically and had never been common lower down in the core. *Plumatella* sp. statoblasts persisted through zone 1 to the core top.

The division in the biostratigraphic data between zones 2 and 3 occurred at the same depth as the proportion of carbonate suddenly decreased (13 cm). The carbonate profile matches most closely that of the *Chara* sp. oospores, suggesting precipitates from charophytes have been the dominant source of carbonate to the sediment within Hickling Broad. The mollusc PCA profile suggests that this community was changing gradually over time, up until around 12 cm depth, after which a relatively stable period of low total abundance and diversity ensued. The first detection of TBT (and DBT) in the core was at 12 cm depth. This occurs at around the same time as the sharp decline in the abundance of charophyte oospores and daphnid ephippia.
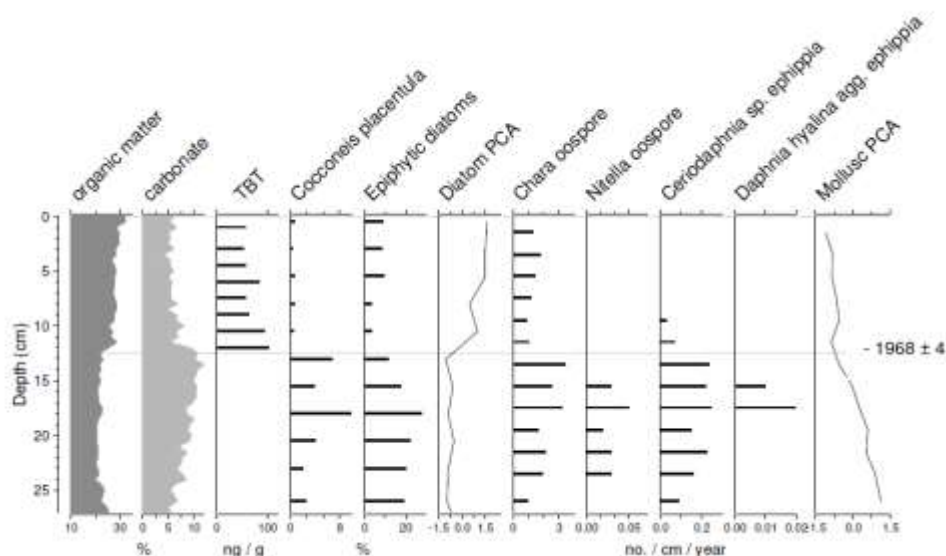
Figure 8 Summary of HICK1 palaeolimnological results (taken from Hoare, 2007).
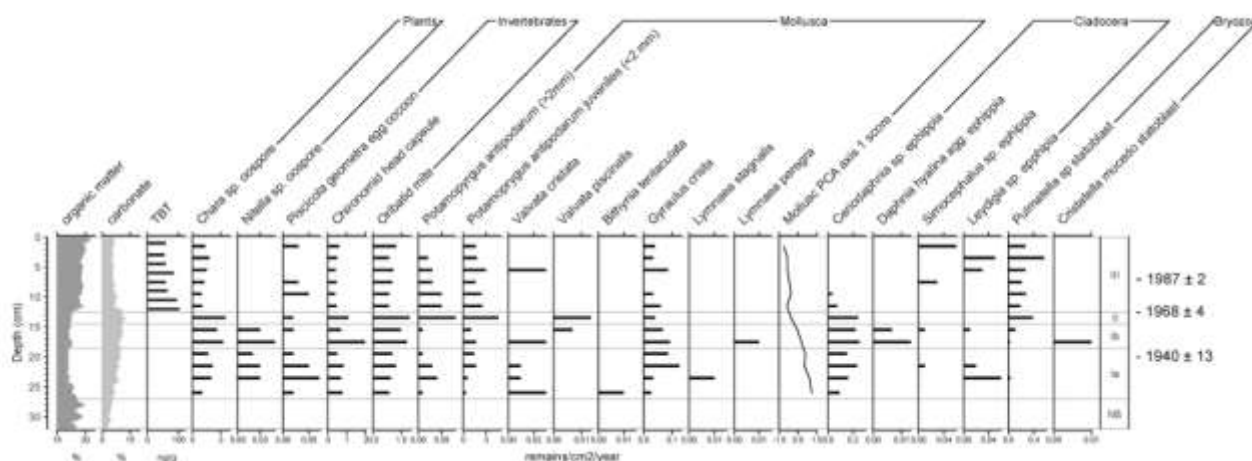
Figure 9 Macrofossil remains from core HICK1 (taken from Hoare, 2007).

3.6 Ostracoda (Crustacea) and foraminifera

The ostracod faunal assemblage of core HICK1 is illustrated in Figure 10. Despite large changes in the salinity of Hickling Broad over time and the supposed salinity sensitivity of ostracods, the fossil ostracod assemblages do not clearly reflect the salinity trends inferred from other independent data. Instead, it appears that a complex series of changes has occurred in the lake over the past 200 years and factors other than salinity, including eutrophication, toxicity and associated complex alterations in habitat availability have probably influenced ostracod assemblages (Holmes *et al.* 2010).

The zonation of the foraminifera (see Figure 10b) matches exactly zones defined for the ostracod data. Zone 1 (36-29 cm) is dominated by three species: *Jadammina macrescens*, *Trochammina inflata* and *Miliammina fusca*, all of which are found in highly saline environments, but which can tolerate salinity values less than that of seawater.

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Figure 10 Palaeolimnological data from core HICK1; (a) ostracod faunal assemblages; (b) physical sedimentology, toxic pollutant and other palaeoecological data. *Fragilaria* spp. includes several species within the *Staurosira-Staurositella-Pseudostaurosira* complex. TBT and DBT are tributyltin and dibutyltin, respectively. (Taken from Holmes *et al*. 2010).

3.7 Summary of changes based on palaeolimnological record

Ecological, natural history and studies and palaeolimnological data can be combined to infer a succession of ecological changes in Hickling Broad over the last two centuries. Historical data allude to a clear-water, relatively nutrient-poor lake during the late 19th and early 20th centuries, with luxuriant beds of charophytes (particularly *Chara aspera*), broad-leaved pondweeds (e.g. *Potamogeton praelongus*) and water milfoils (*Myriophyllum verticillatum* and *M. spicatum*) amongst other species (Jackson, 1978; Bales *et al*., 1993). The palaeolimnological record confirms this by way of extremely high densities of charophyte oospores (*Chara* and *Nitella* spp.), abundant remains of plant-associated snails (*G. crista*, *L. peregra* and *V. cristata*) and a high representation of epiphytic diatoms (e.g. *C. placentula*), low plankton populations and a low sedimentation rate.

From around 1920 to the late 1960s, sedimentation rates increased and both historic records and palaeolimnological data show a decline in Hickling Broad's charophyte abundance, together with a decline in snail shell abundance. Epiphytic diatoms were abundant. The local naturalist Robert Gurney reported 'overgrowths' of the

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filamentous algae *Cladophora* sp. and an abundance of *Potamogeton pectinatus*, a clear indication of early eutrophication probably linked to increased fertiliser use in the catchment (Moss, 1978).

A major change in the palaeolimnological record occurred around 1968 ± 4 years, where there was a relatively sharp decline in charophyte remains (Hoare, 2007), disappearance of *Ceriodaphnia* (Hoare, 2007) and a decrease in the proportion of epiphytic diatoms (Sayer *et al.*, 2006; Hoare, 2007; Holmes *et al.*, 2010). This is when Hickling Broad changed from a clear-water, macrophyte-dominated lake to a turbid, phytoplankton-dominated lake, with almost the complete loss of aquatic macrophytes by the mid-1970s (Phillips & Moss, 1978). The timing of this change was coincident with the onset of the use of highly toxic TBT-based antifoulants (Sayer *et al.*, 2006); increasing eutrophication - including guanotrophication by black-headed gulls (Moss & Leah, 1982); increased salinity due to intensified agricultural drainage in the catchment (Bales *et al.*, 1993 and Barker *et al.* (2008) and a large fish kill in 1969 due to toxicity resulting from blooms of the alga *Prymnesium parvum* (Holdway *et al.*, 1978).

Historic records indicate partial recovery in the 1980s, despite turbid waters.

4   Chronology of management activities and key events

Table 2 Chronology of management activities and key events for Hickling Broad (those marked with * shown on figures below)

Date	Description
1968 & 1974*	Dredging of main channel
1994 – 1999*	Cutting of *M.spicatum* and *Potamogeton* spp in 1/3 of broad
1999	Experimental cutting of small (c7ha) plots
2002*	Dredging Catfield dyke
2003	Experimental cutting of small (c7ha) plots

5   Water quality
Data prior to 1980 taken from UEA studies

5.1   Total Phosphorus
Total phosphorus in Hickling Broad has reduced significantly (Mann Kendall tau= -0.409 p<0.001) since the late 1970s although it is still relatively high, with a median concentration for the last decade of 69 µgP.l^{-1}, similar to several other broads in the Bure catchment and higher than the other broads in the Thurne system (Figure 11). The broad is currently classified at Moderate status for phosphorus under the Water Framework Directive. The decline of phosphorus is likely to be a result of the

disappearance of a very large black-headed gull roost (Bales et al. 1993). There is an indication that since 2005 phosphorus levels have been rising again, although this is not statistically significant (Mann Kendall tau= 0.429 p=0.115).

Total phosphorus concentration in Catfield pump, which discharges into Hickling Broad, has a similar trend to that of Hickling Broad and shows a similar increase in phosphorus concentration since 2005 that is just significant (Mann Kendall tau = 0.571 p=0.031). However, the fluctuations in concentration in Catfield pump are negatively correlated with rainfall (r^2 = 0.42 p<0.001) suggesting that there has been no increase in TP load from this pump and that the decreases in concentration are a result of increased dilution. Without details of pump volumes, it is difficult to draw conclusions about the sources of phosphorus to Hickling, but a multiple regression for data since 1984, when reasonably consistent pump TP values are available, shows that a model with TP concentration in Horsey Mere, Stubb Mill Pump, Catfield Pump and rainfall are all significant predictors of TP in Hickling (Table 3). Unfortunately pump data are not available during the period of high macrophyte cover from 1997-2000 and it is thus impossible to include the period of high macrophyte cover in the model. However, there is a marked dip in TP concentration during this period and it is very likely that this is caused by macrophyte uptake. Comparing GAM smoothers for TP in Hickling, Horsey, Catfield and Stubb Mill pumps with rainfall and macrophyte cover, using best sub-set regression, shows that the optimum model (selected using BIC) of the smoothed trend for TP in Hickling includes all of these smoothers, thus confirming the importance of macrophytes in changing the TP concentration of Hickling Broad.

The increase in TP since 2005 is similar to an increase in both suspended solids (Figure 26) and total iron (Figure 24), suggesting that some of the increase in TP may be associated with an increase in suspended sediment, potentially caused by an increase in the effect of wind-induced resuspension of sediment following the loss of macrophytes. However, over the last decade the TP concentrations have also risen substantially in Horsey Mill pump and to a lesser extent in Catfield pump. In the former case this is thought to have been caused by recent deeper drainage and conversion to arable in the Horsey Mill catchment. This may be contributing to increased TP, Fe and suspended solids in Hickling.

Table 3 Linear regression for annual mean TP in Hickling and TP in Horsey Mere, Catfield Pump and Stubb Mill Pump

```
Coefficients:
            Estimate Std. Error t value Pr(>|t|)
(Intercept) -56.3838    28.9206  -1.950 0.066978 .
TP_Ho         1.0414     0.2454   4.244 0.000488 ***
TP_Catf       0.2431     0.0795   3.058 0.006775 **
TP_Stubb_p    0.4678     0.2388   1.959 0.065821 .
AvgOfrain_mm  0.8340     0.3930   2.122 0.047972 *
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 12.44 on 18 degrees of freedom
Multiple R-squared:  0.7597,    Adjusted R-squared:  0.7063
F-statistic: 14.23 on 4 and 18 DF,  p-value: 2.093e-05F-statistic:
15.64 on 3 and 20 DF,  p-value: 1.794e-05
```
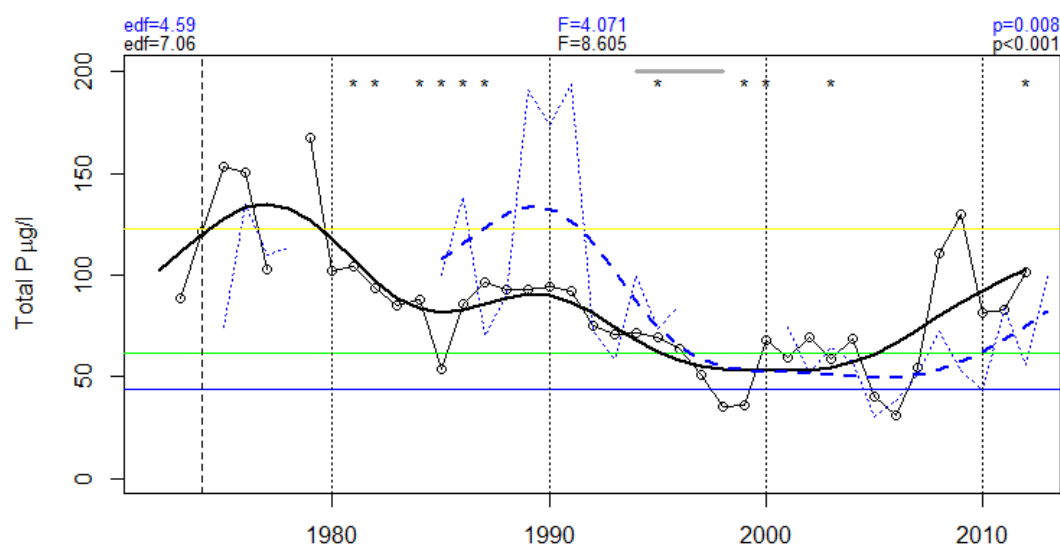

Figure 11 Trend in annual mean TP for Hickling Broad (solid black line), compared with trend in Catfield Pump (broken blue line), showing GAM smoothers. Horizontal lines mark WFD boundary values, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting.

The seasonal pattern of TP in Hickling Broad is similar to many shallow lakes with maximum values occurring in July-August (Figure 12b). This summer peak was markedly less obvious during the period of significant macrophyte growth (1995-2005) and the recent increase in TP in Hickling has been most pronounced in the May - August period (Figure 12c) when macrophyte uptake would normally be at its most active.
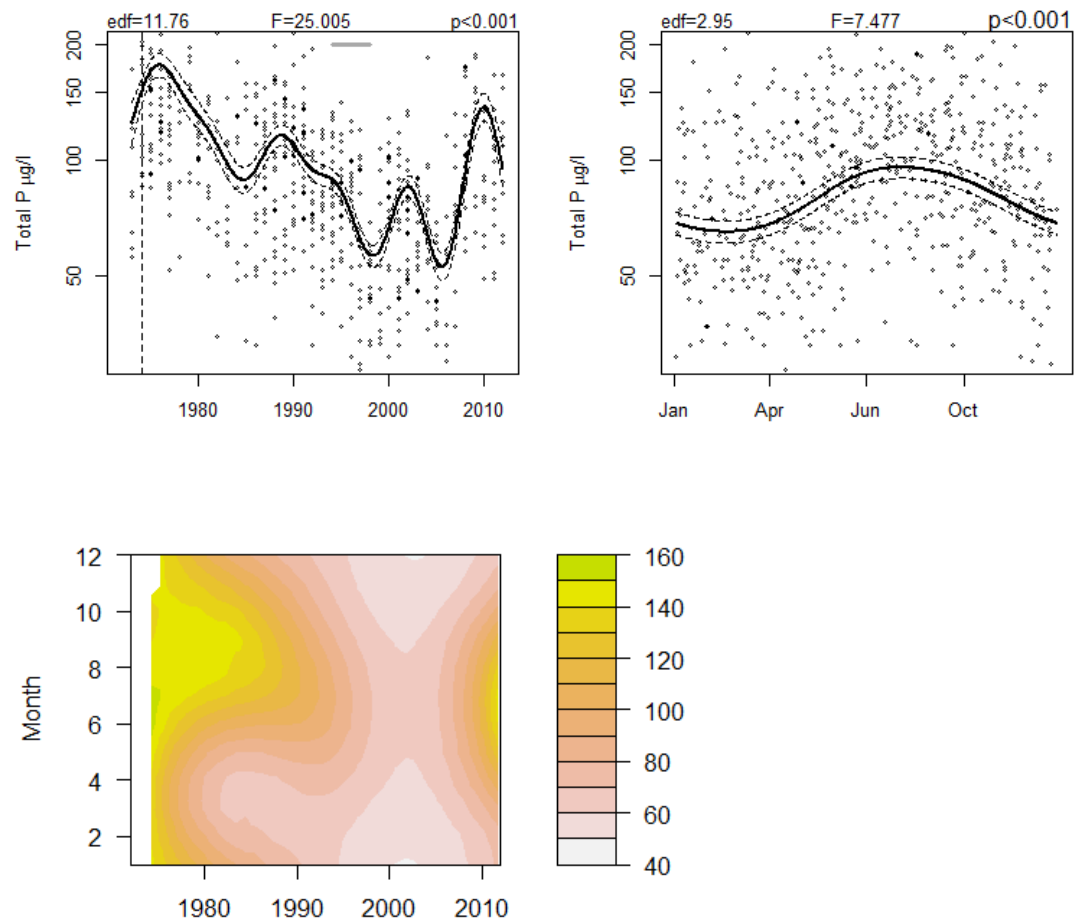
Figure 12 Changes in TP for Hickling Broad: a) Long term trend (Annual); b) Seasonal trend; c) contour plot showing changes in seasonality with time. (Trends extracted using GAM model, contour plot using a bi-variate GAM model)

There is a slight but significant increasing trend in the ratio of summer to winter TP (Mann Kendall tau= 0.239 p=0.031). While this ratio is an indicator of the significance of sediment-derived phosphorus it is also influenced by the uptake of phosphorus by macrophytes. In Hickling the much higher ratio values occurring since 2006 are associated with the lowest levels of macrophyte cover (Figure 14) and it is suggested that changes in this ratio are further evidence of macrophytes taking up phosphorus during the summer period.
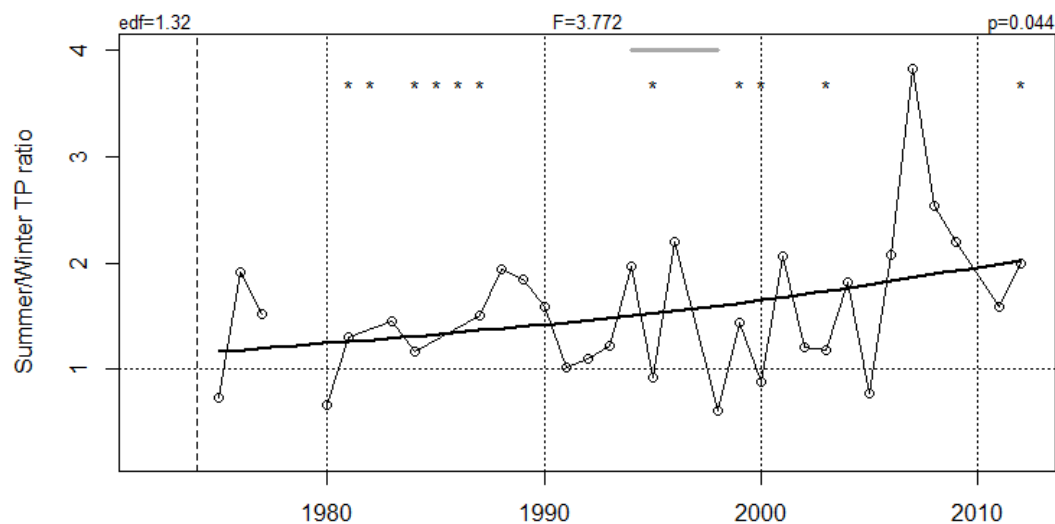
Figure 13 Trend in ratio of summer/winter mean TP for Hickling Broad, horizontal line marks 1:1 ratio, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting, line GAM model.
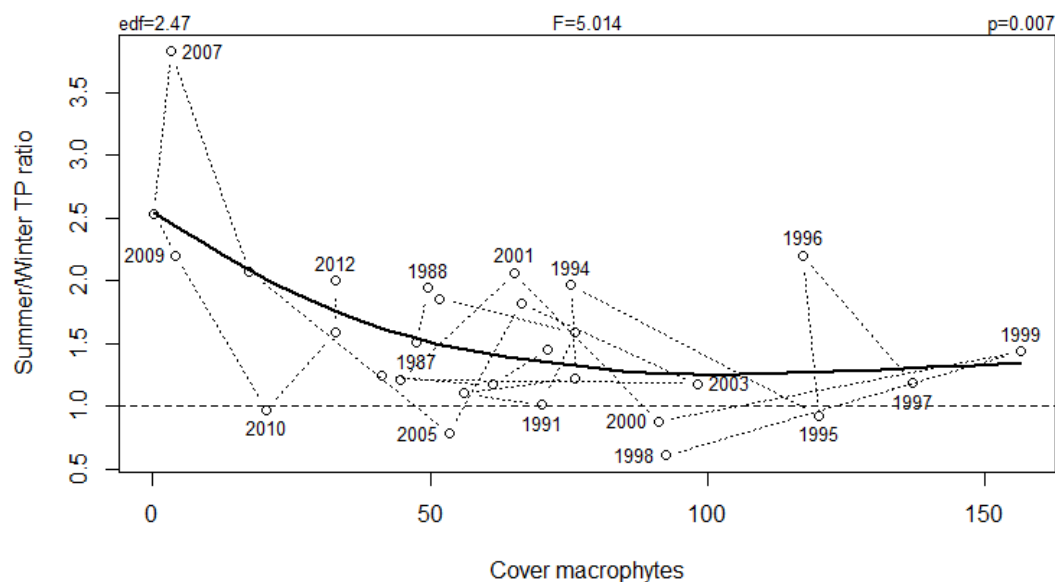
Figure 14 Relationship between summer/winter TP for Hickling Broad and cover of macrophytes showing trajectory of change with time, line GAM model

5.2 Soluble Reactive Phosphorus

Soluble Reactive Phosphorus (SRP) is extremely low in Hickling Broad (Figure 15) with no significant trend (Mann Kendall tau= 0.007 p=0.480). There is no seasonal

variation of SRP (Figure 16b & c) with no evidence of summer peaks of soluble phosphorus, in contrast to most other broads. This suggests that there is no sudden summer release of sediment-derived phosphorus and that all available phosphorus is rapidly taken up by either phytoplankton or macrophytes.

In contrast to the broad, the SRP levels in Catfield Pump have decreased significantly (Mann Kendall tau= -0.35 p=0.004) and while the peaks of SRP in Catfield are similar to some of the peaks in Hickling the significantly lower SRP in Hickling than in Catfield confirms the rapid biological removal of phosphorus in the broad.
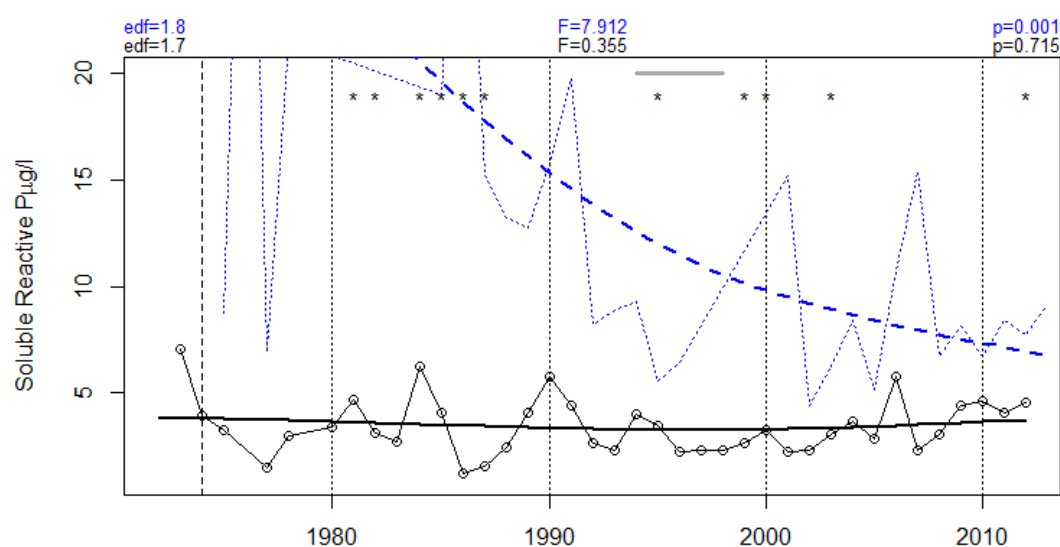
Figure 15 Trend in annual mean SRP for Hickling Broad, compared with trend in Catfield Pump (broken blue line), stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting.
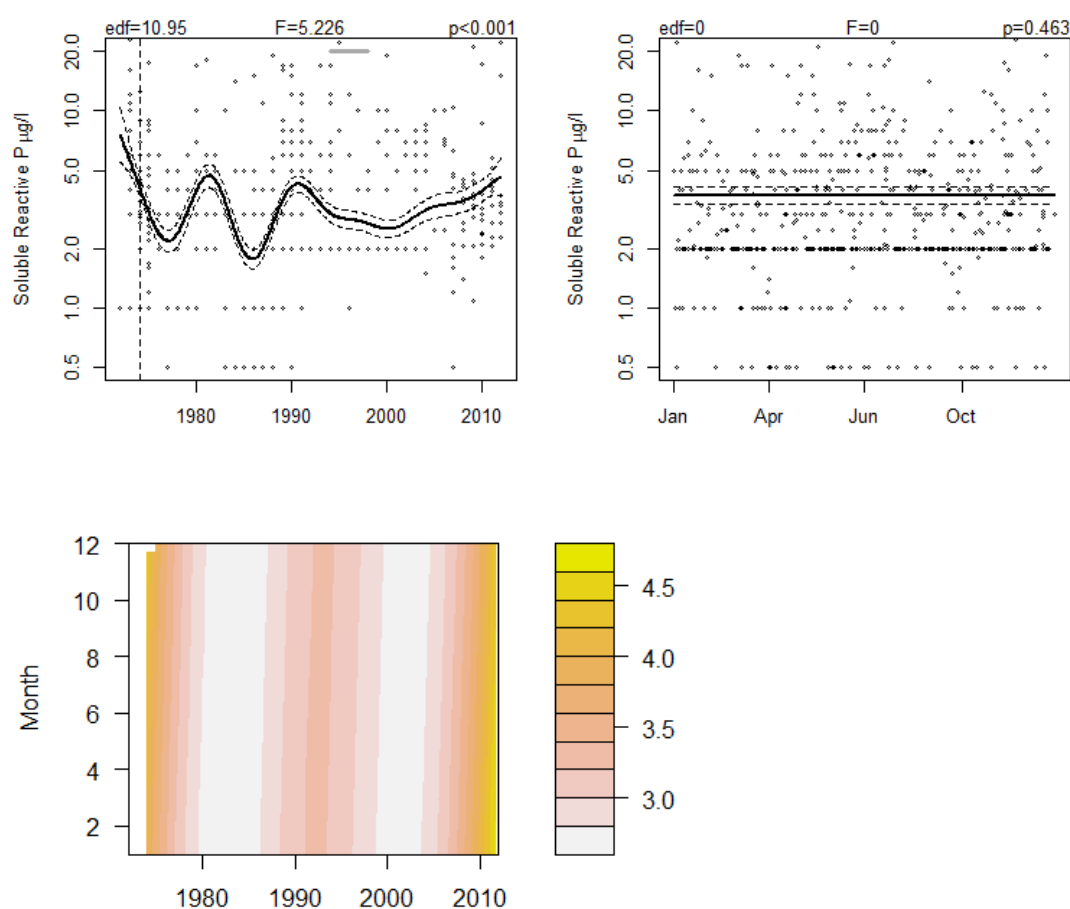
Figure 16 Changes in SRP for Hickling Broad: a) Long term trend (Annual); b) Seasonal trend; c) contour plot showing changes in seasonality with time. (Trends extracted using GAM model, contour plot using GAM model)

5.3   Total Oxidised Nitrogen (TON)

In common with many broads there has been a highly significant decline in TON concentration in Hickling Broad since 1982 (Mann Kendall tau= -0.554 p<0.001) (Figure 17). There has been a similar decline in winter TON (Mann Kendall tau= -0.379 p=0.002) (Figure 18) and in comparison to other broads over the last decade Hickling has very low levels of TON with a median winter mean concentration of 0.36 mg l^{-1}. Changes in winter TON are clearly linked to rainfall, with the highest TON concentrations occurring in years when river discharge is high (Figure 18), although the lowest TON concentrations have occurred since 2001 when rainfall and thus river discharge have been high. This may reflect warmer conditions and greater rates of denitrification.

The concentrations of TON in Catfield Pump are substantially higher and, although these have decreased, the changes are less marked and there has been no significant decline since 1983, suggesting that the lower TON in the Broad is not a direct result of changes in the small pumped catchment of Hickling Broad.
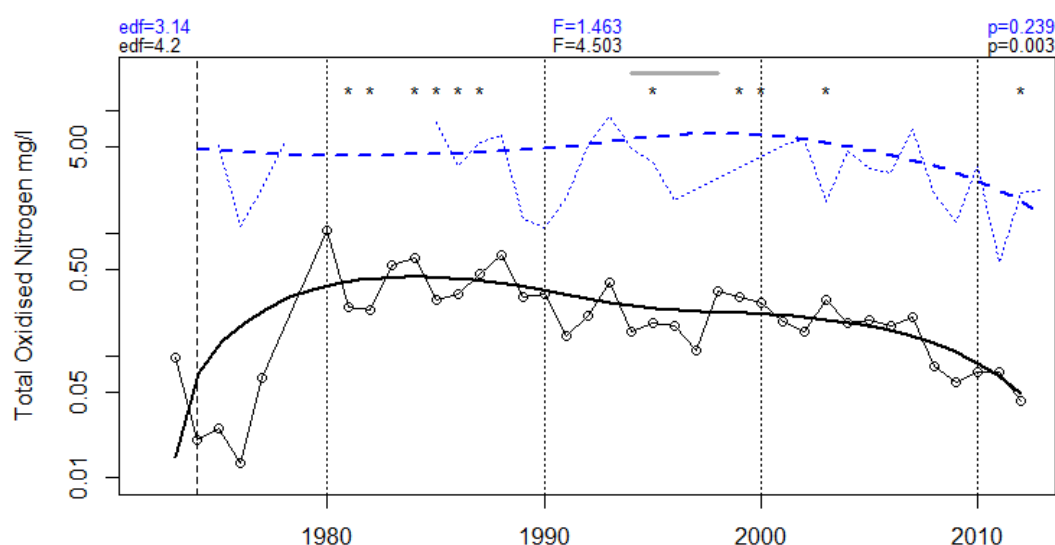
Figure 17 Trend in annual mean TON for Hickling Broad (solid black line), compared with trend in Catfield Pump, showing GAM smoothers, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting.
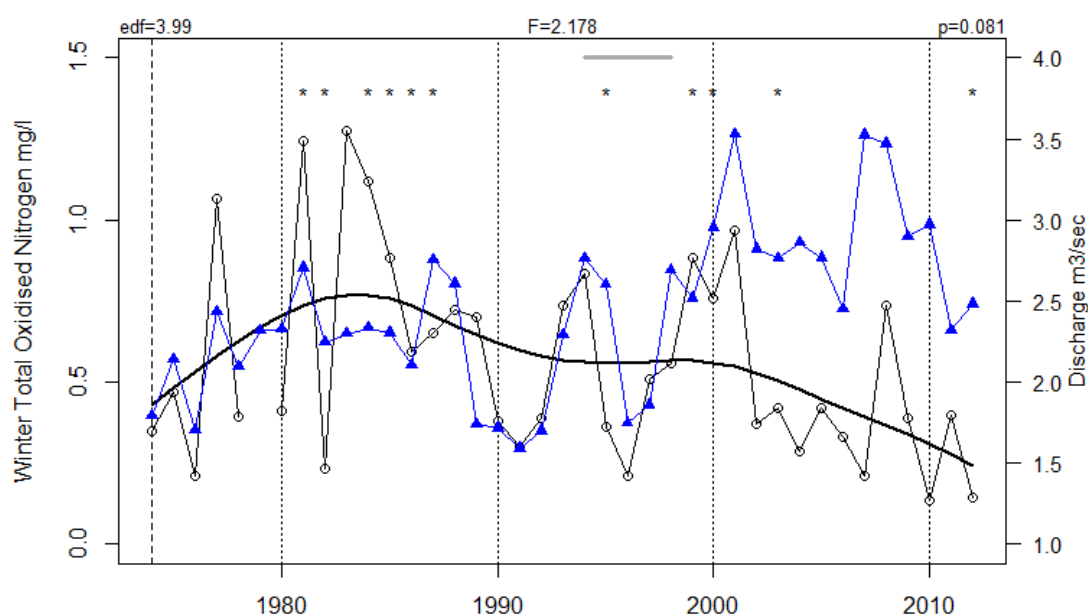
Figure 18 Trend in winter (Nov-Feb) mean TON for Hickling Broad (solid black line) compared to trend in river discharge, showing GAM smoother for TON, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting.

The seasonal pattern of TON in Hickling is similar to other Broads with maximum values in February, which rapidly decline to very low values by May. In comparison to other broads the period of low TON is longer with concentrations only increasing again from November (Figure 19b).

As SRP in Hickling has remained unchanged the ratio of summer TON/SRP has declined as TON has reduced (Figure 20). The reduction from 1989 is associated with a period of increased temperature and as suggested above may reflect an increase in denitrification.
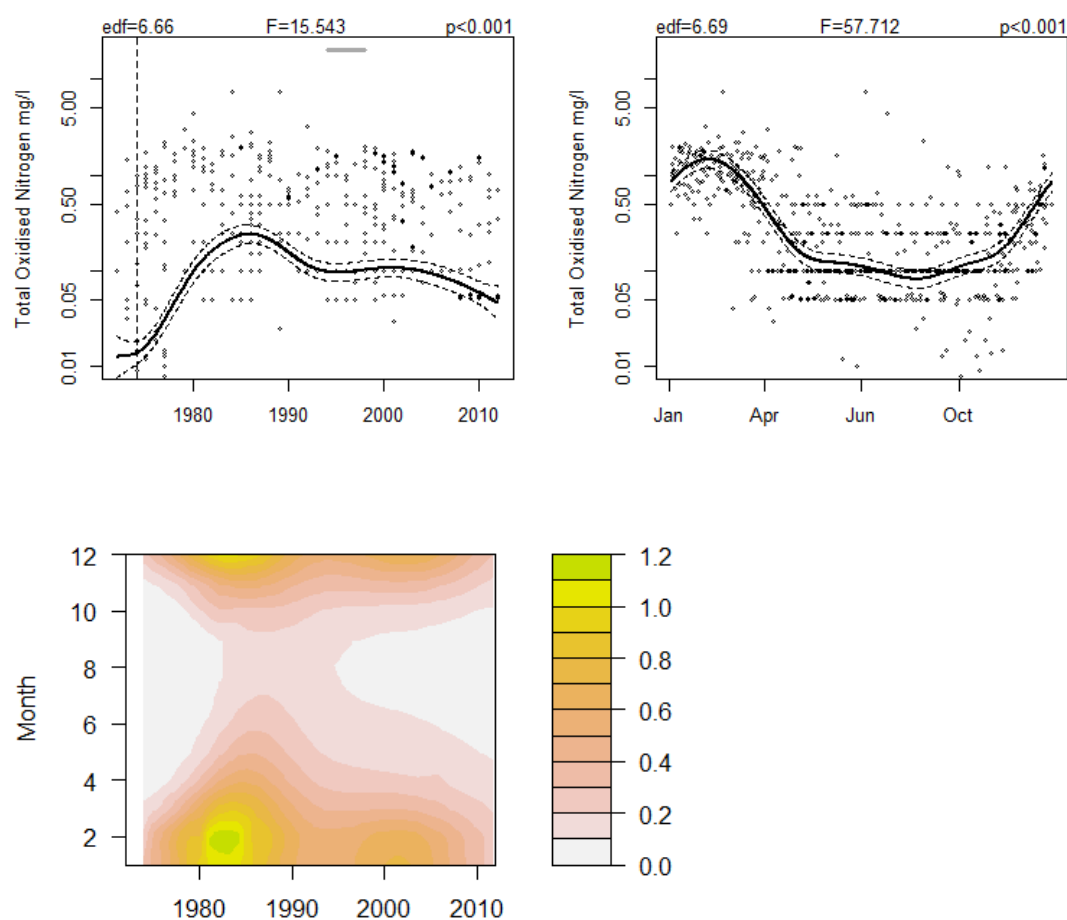
Figure 19 Changes in TON for Hickling Broad: a) Long term trend (Annual); b) Seasonal trend. (Trends extracted using Mixed GAM model); c) contour plot showing changes in seasonality with time. (Trends extracted using Mixed GAM model, contour plot using GAM model)
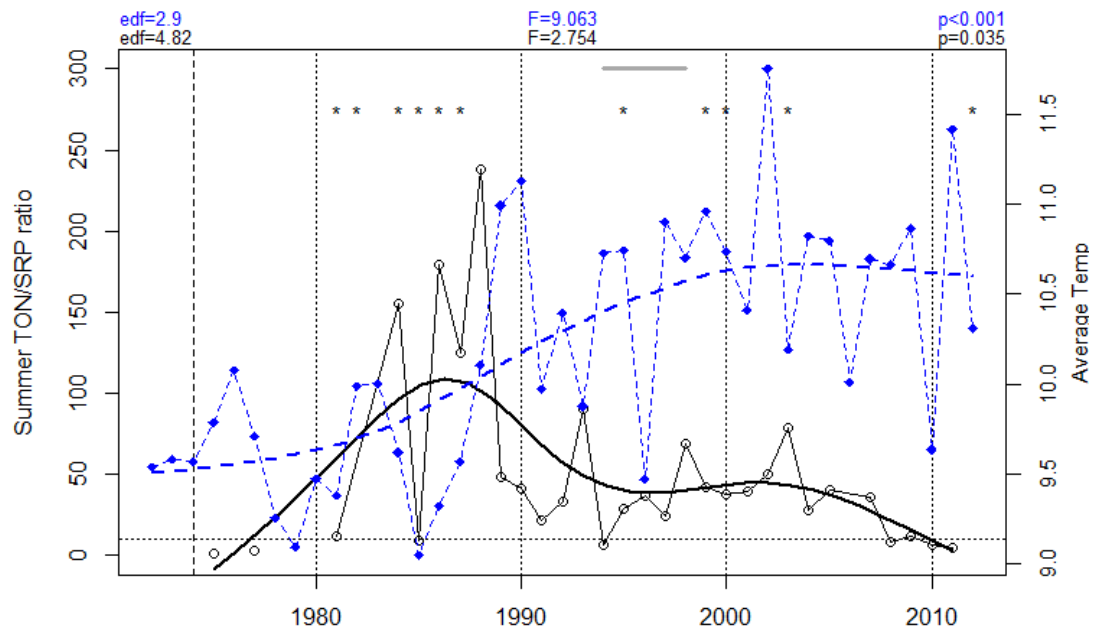
Figure 20 Trend in ratio of summer mean TON/SRP in Hickling Broad (black line), compared to change in average air temperature (blue dotted line). Horizontal line marks Redfield ratio, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting, lines shows GAM models.
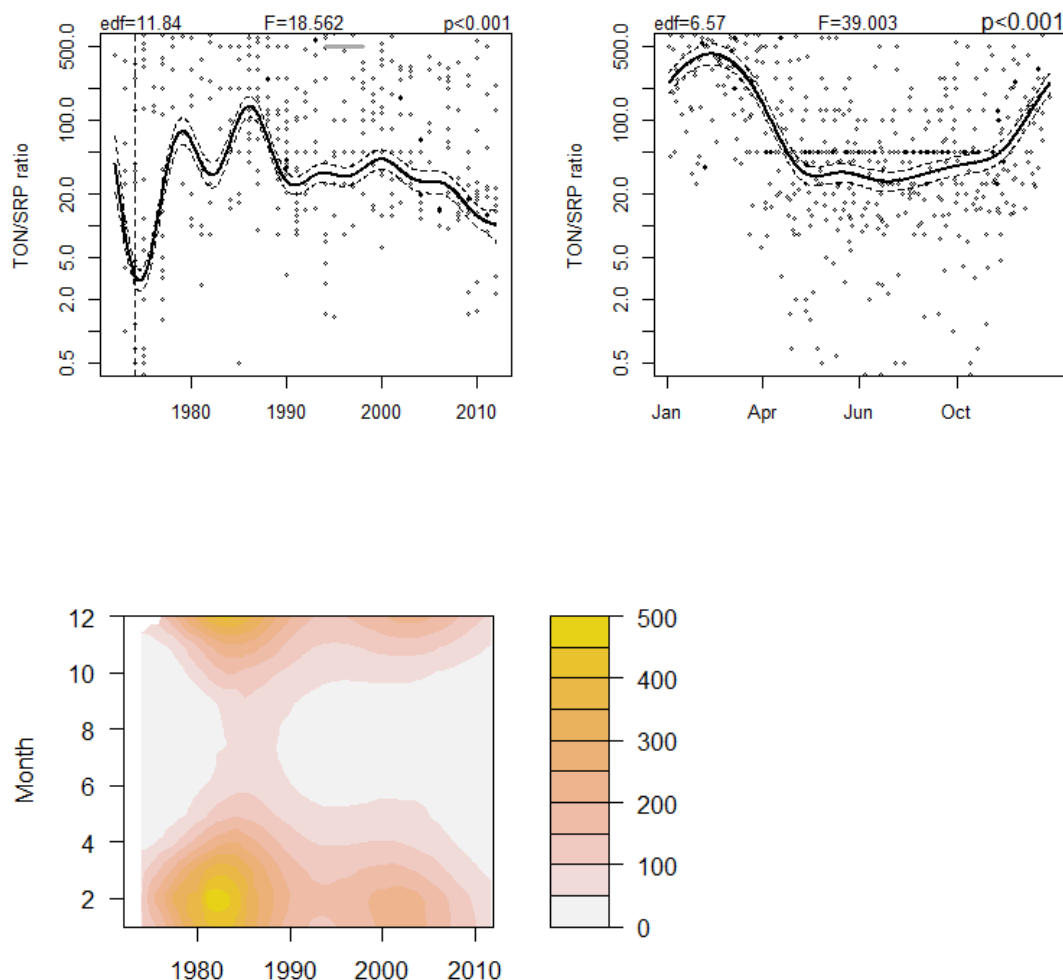
Figure 21 Changes in SRP/TON ratios for Hickling Broad: a) Long term trend (Annual); b) Seasonal trend. (Trends extracted using GAM model); c) contour plot showing changes in seasonality with time. (Trends extracted using Mixed GAM model, contour plot using GAM model)

5.4   Chloride

In common with the other Thurne Broads, chloride levels in Hickling are high giving rise to brackish conditions. Chloride concentrations in Hickling have shown little long-term change over the last 30 years averaging 1500-1600 mg/L, lower than in Horsey Mere and higher than the Martham Broads (Figure 22). While the hydrology of Hickling implies that it must always have been brackish (at least since monastic times), present salinities are two-three times higher than the average salinity recorded from 1892 to 1933 (Holmes et al., 2010) and are significantly higher than the Favourable Condition threshold of 600 mgl[-1]. The major input of brackish water to Hickling is via pumped drainage from the Brograve levels to the north via Horsey Mere.   This large pumped input flushes water from Horsey Mere along Meadow Dyke into Heigham Sound. Tidal action then mixes this with water from further downsteam, which is then then pushed back into Hickling on the rising tide. Hickling

also receives a small volume of freshwater from Catfield pump (Figure 22), which provides some dilution and accounts for the slightly lower chloride concentration in Hickling in comparison to Horsey Mere. The relative influence of this brackish land drainage water increases in dry years, as in 1991-93 and 1997, when freshwater inputs from Catfield dyke are lower and tidal water pushes higher up the system.

The seasonal pattern of chloride in Hickling shows a minimum in spring when the Catfield pumped input is highest and a maximum in August and September (Figure 23b). In dry years the period of elevated chloride occurs as early as April and extends to August and may also be influenced by saline water pushing up the Thurne system from the lower estuary of the River Bure during tidal surges (Figure 23c).
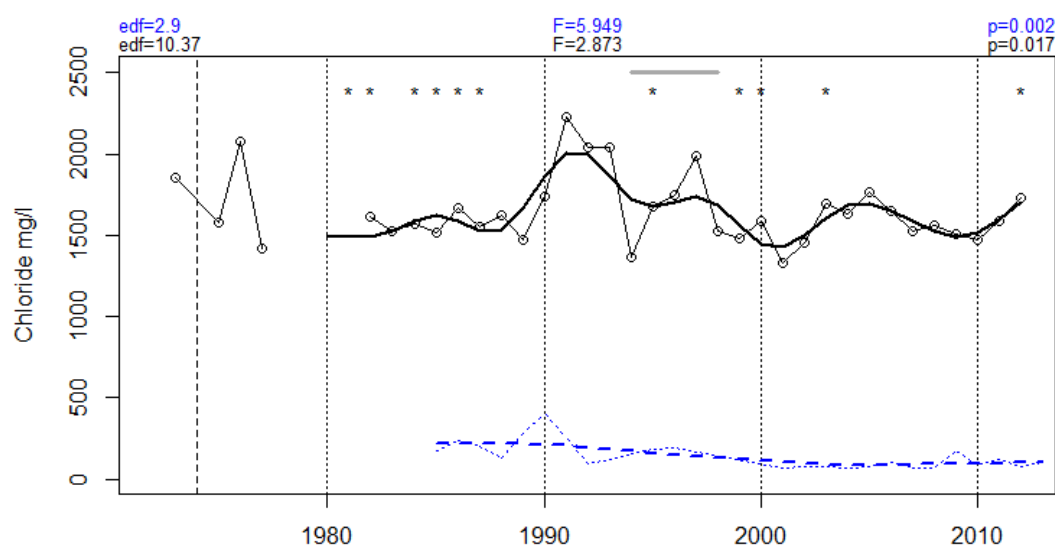
Figure 22 Trend in annual mean chloride for Hickling Broad (solid black line), compared with trend in Catfield Pump, showing GAM smoothers, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting.

Figure 23 Changes in chloride for Hickling Broad: a) Long term trend (Annual); b) Seasonal trend (Julian Days); c) contour plot showing changes in seasonality with time. (Trends extracted using mixed GAM model, contour plot using GAM model)
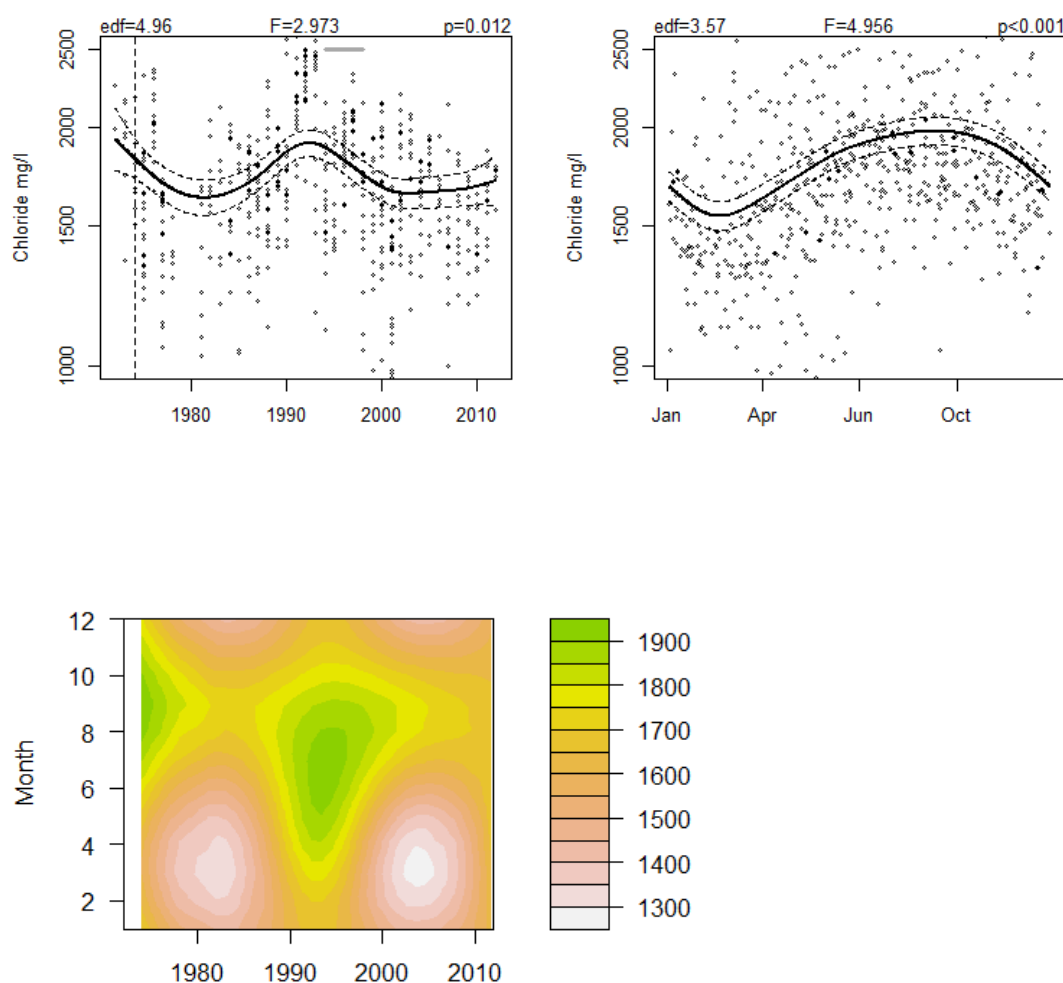
5.5 Total Iron

Given the high concentration of iron hydroxide (ochre) discharged from the Brograve land drainage pump into Horsey Mere and thus into Hickling it is worth considering changes in total iron concentration. Iron concentrations in Hickling are substantially lower than in Horsey Mere, averaging 247µg l^{-1} in comparison to 1025 µg l^{-1} (Figure 24). This is much lower than the difference in chloride and probably reflects that much of the iron in Horsey sediments out before reaching Hickling Broad. Total iron levels in Hickling are lowest in dry years and in general were higher in the mid-1980s than they were from 1995 to 2006, although they have risen again in the last 6 years. These changes may reflect changes in mixing of water from Horsey, but they are also likely to be influenced by the amount of ochre in Horsey, as the recent increase in Hickling occurs at the same time as a much more significant change in Horsey. In

Hickling Broad there is very little evidence of seasonality if considering the whole 30-year data set. However, from 1990 – 2000 total iron was much lower during the summer, the period when macrophyte growth in Hickling increased substantially.

Figure 24 Trend in annual mean total iron for Hickling Broad (solid black line), compared with trend in Catfield Pump, showing GAM smoothers, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting.
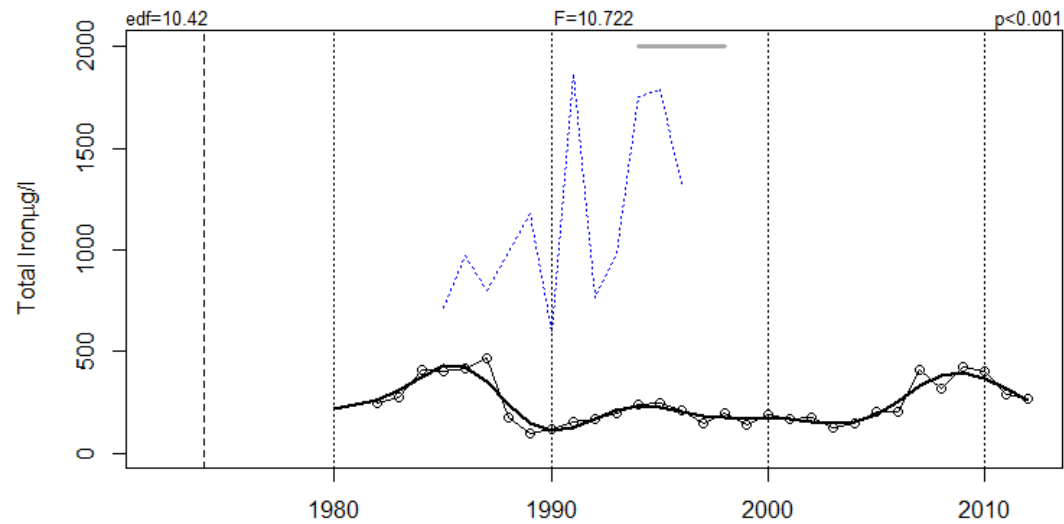
Figure 25 Changes in total iron for Hickling Broad: a) Long term trend (Annual); b) Seasonal trend; c) contour plot showing changes in seasonality with time. (Trends extracted using mixed GAM model, contour plot using GAM model)
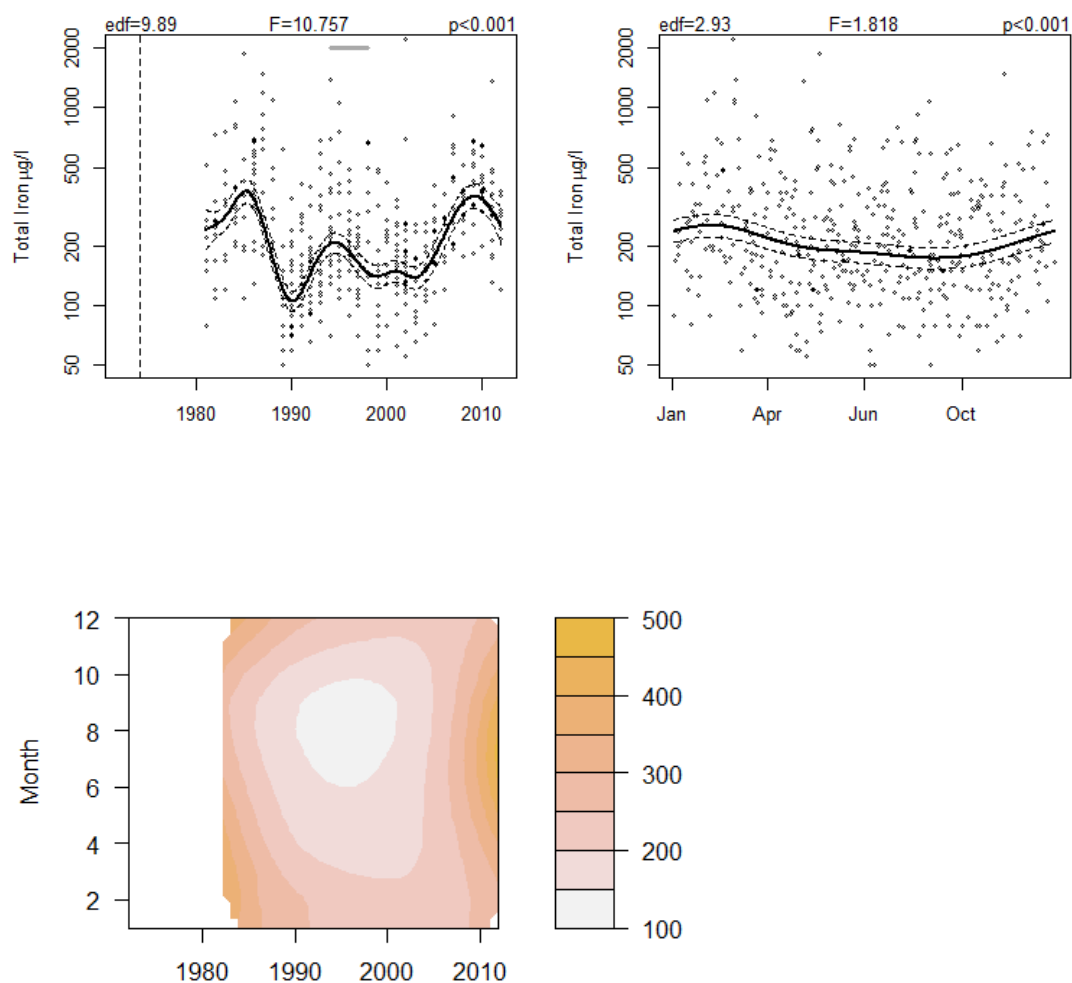
5.6 Suspended Solids

Total suspended solids in Hickling are currently similar to the values recorded prior to 1994 (Figure 26). They were substantially lower from 1997 to 2004, with extremely low values in 1998,1999, 2003,2004, the periods of maximum *Chara* growth. This may suggest that high plant cover effectively reduces sediment re-suspension by wind action in Hickling. However, it is also difficult to rule out an external driver of suspended solids, which then controls the light environment for macrophytes.

The seasonal pattern shows a minimum value in winter and a maximum in early summer (June-July) (Figure 27b). This contrasts with the seasonal pattern of chlorophyll (Figure 29b), suggesting that phytoplankton is not the main component of suspended solids.

Figure 26 Trend in annual mean suspended solids for Hickling Broad (solid black line), compared with trend in Catfield Pump, showing GAM smoothers, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting.
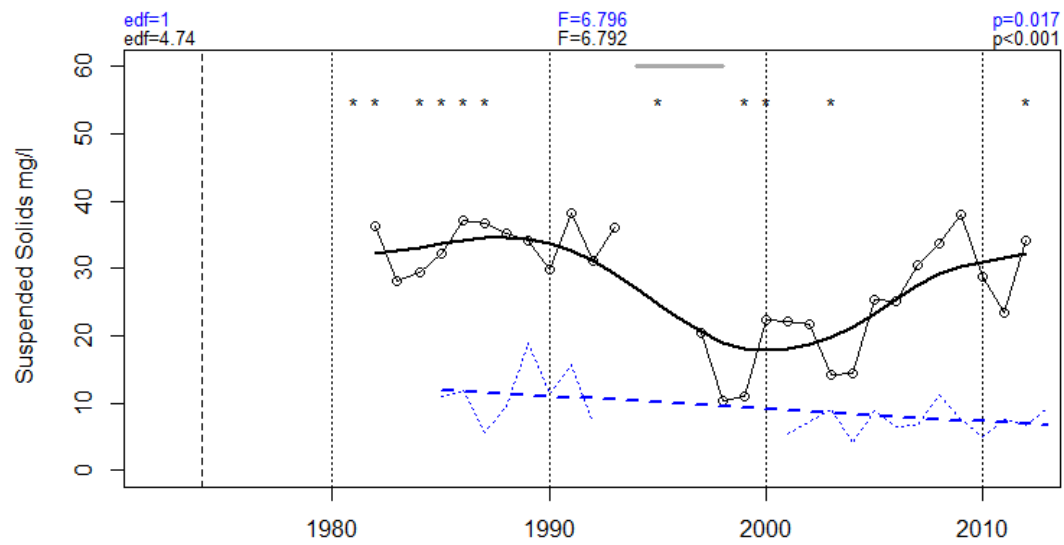
Figure 27 Changes in suspended solids for Hickling Broad: a) Long term trend (Annual); b) Seasonal trend; c) contour plot showing changes in seasonality with time. (Trends extracted using mixed GAM model, contour plot using GAM model

5.7 Chlorophyll *a*

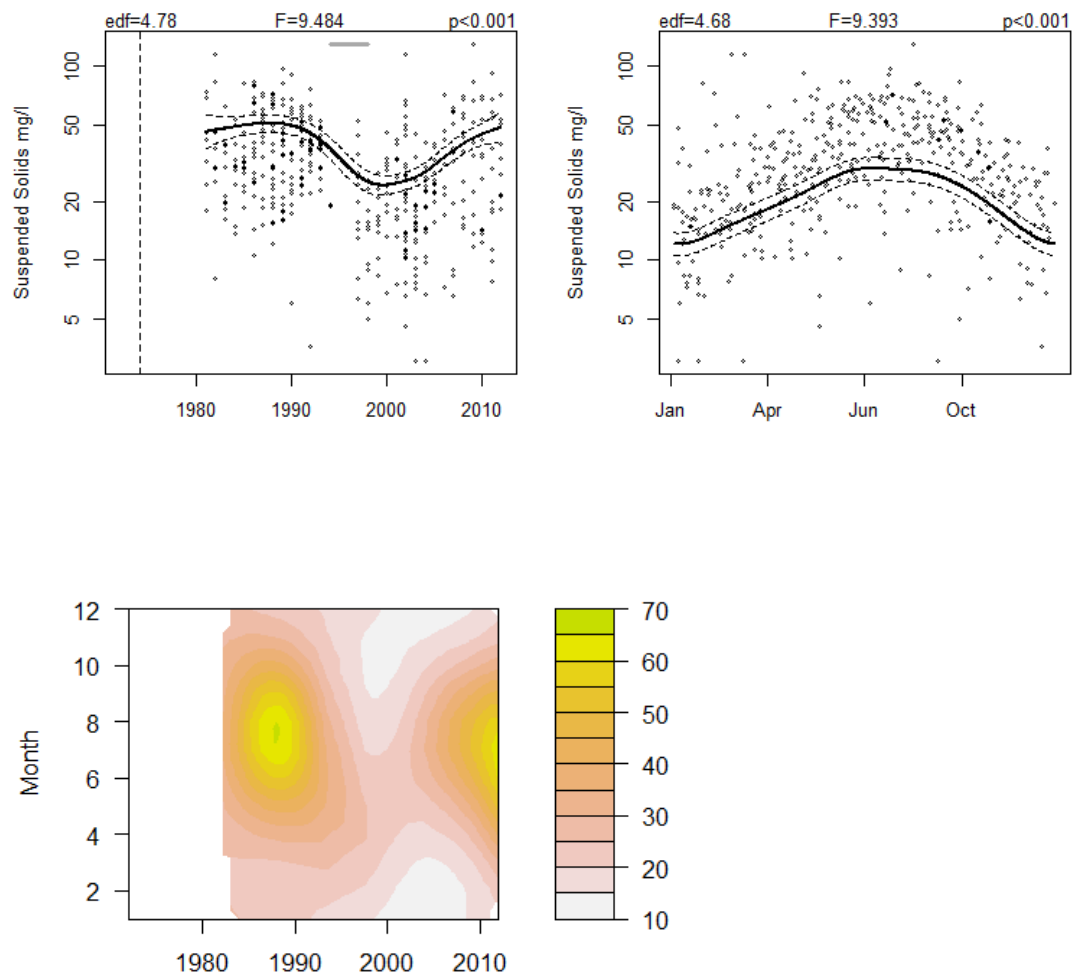The chlorophyll concentration in Hickling Broad has reduced significantly since the mid-1970s (Mann Kendall tau = -0.603, p<0.001) reaching a minimum value in 2008 (Figure 28). Since 2000 there has been no significant trend in chlorophyll (Mann Kendall tau = -0.091, p=0.732) and chlorophyll remains relatively high with a median concentration over the last decade of 42 µgl^{-1}. Hickling is currently classed as Moderate status for chlorophyll *a* but Poor status overall for phytoplankton. There was a pronounced drop in chlorophyll in 1998 and 1999 during the period of maximum abundance of *Chara*, and again in 2010, the former presumably due to monopolisation of phosphorus by macrophytes.
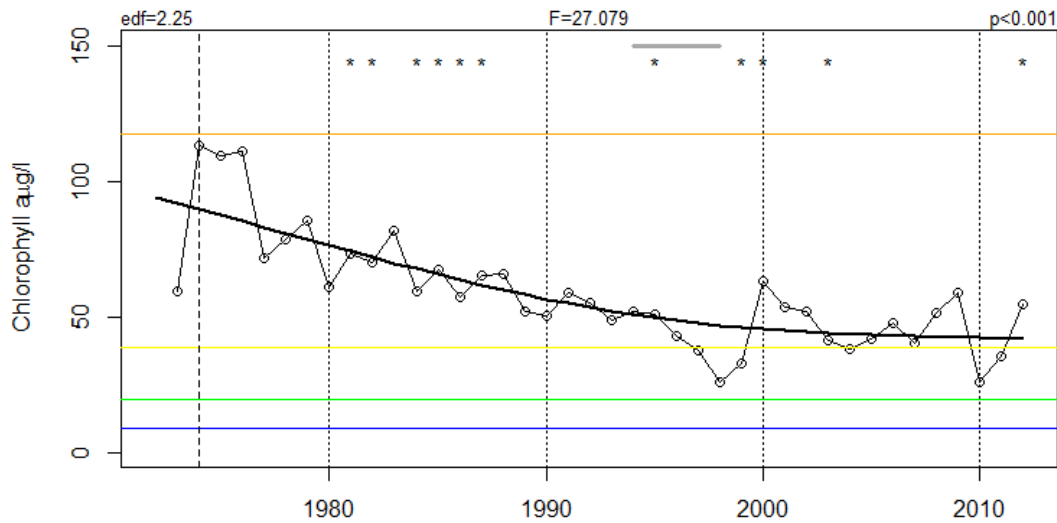
Figure 28 Trend in annual mean chlorophyll a for Hickling Broad (solid black line), showing GAM smoothers, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting.

The seasonal cycle of chlorophyll *a* for Hickling is unlike most other broads, having a single peak in March with a relatively low summer concentration (Figure 29b & c). This may be partly caused by the dominance of the summer phytoplankton by small cyanobacteria (*Microcystis aeruginosa* (EA records 2010-2012) or *Aphanothece* spp 1988-1989 (Bales et al. 1993).

Since 1969 Hickling has experienced persistent problems of fish kills associated with toxic blooms of the flagellate alga *Prymnesium parvum*. These have mostly occurred in winter and spring. There is some evidence that the incidence and severity of these blooms has reduced since the 1980s in which there was a run of fish kills. The precise drivers for bloom development in *Prymnesium* in Hickling and more generally are unknown and toxic effects may occur in the absence of blooms. However, it is generally accepted that N deficiency, probably coupled with increasing salinity and lower temperatures are the commonest scenario for the release of the toxin prymnesin. Hagström (2013) has reviewed the occurrence of *Prymnesium* blooms in Hickling Broad and identifies a number of associated conditions.

Figure 29 Changes in chlorophyll *a* for Hickling Broad: a) Long term trend (Annual); b) Seasonal trend; c) contour plot showing changes in seasonality with time. (Trends extracted using Mixed GAM model, contour plot using GAM model)
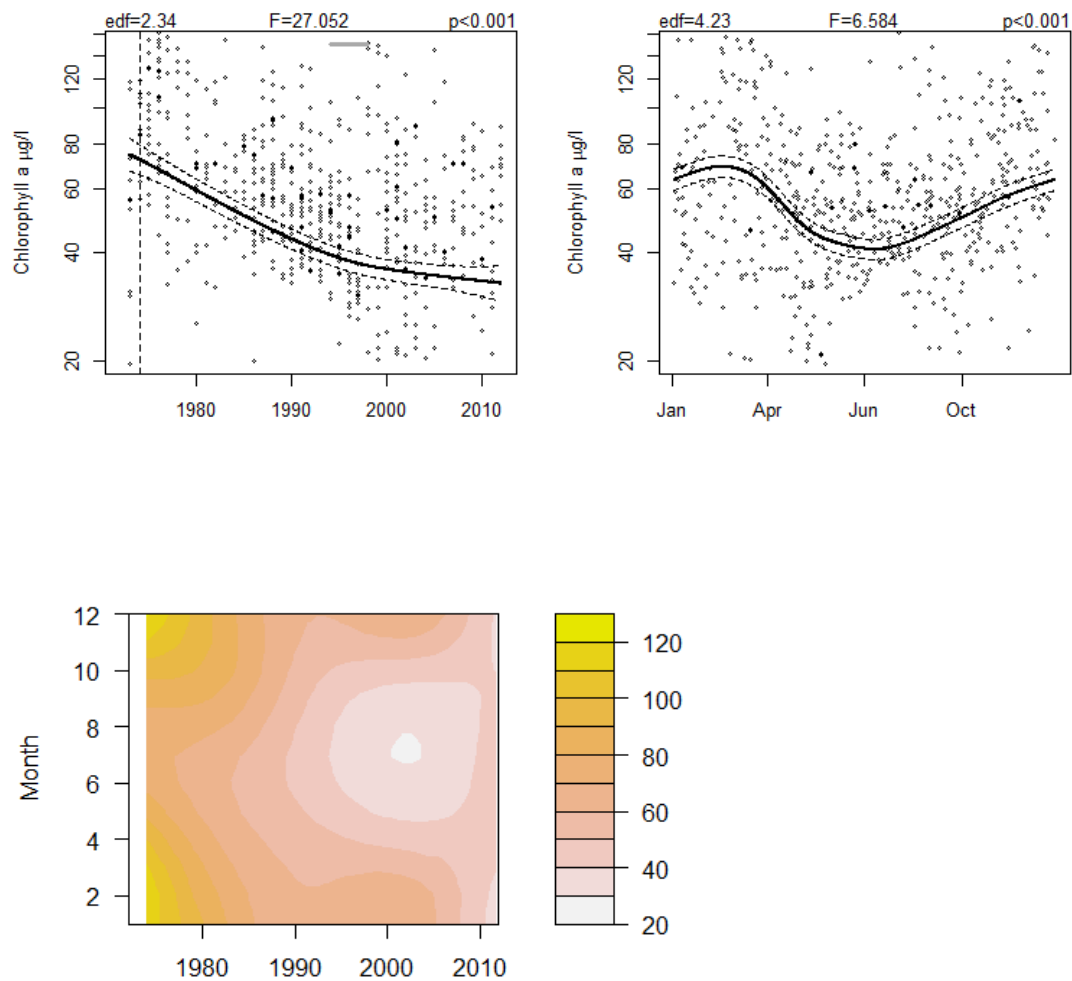
5.8 Chlorophyll *a*:Total Phosphorus ratio

In common with the other Thurne broads, the chlorophyll/total phosphorus ratio for Hickling Broad is high with a median value of 0.55 over the last decade (Figure 30). The salinity of Hickling precludes the development of larger grazing cladoceran populations and there is thus generally a high yield of chlorophyll per unit of phosphorus in the broad.

Since 2000 there has been a significant decline in the ratio (Mann Kendall tau= -0.636 p=0.004). The reason for this is unknown but it appears that the increase in total phosphorus during this period has not resulted in an increase in chlorophyll. Possible factors include a less favourable light climate associated with suspended solids or iron, reduced bioavailability of P or increasing N limitation.
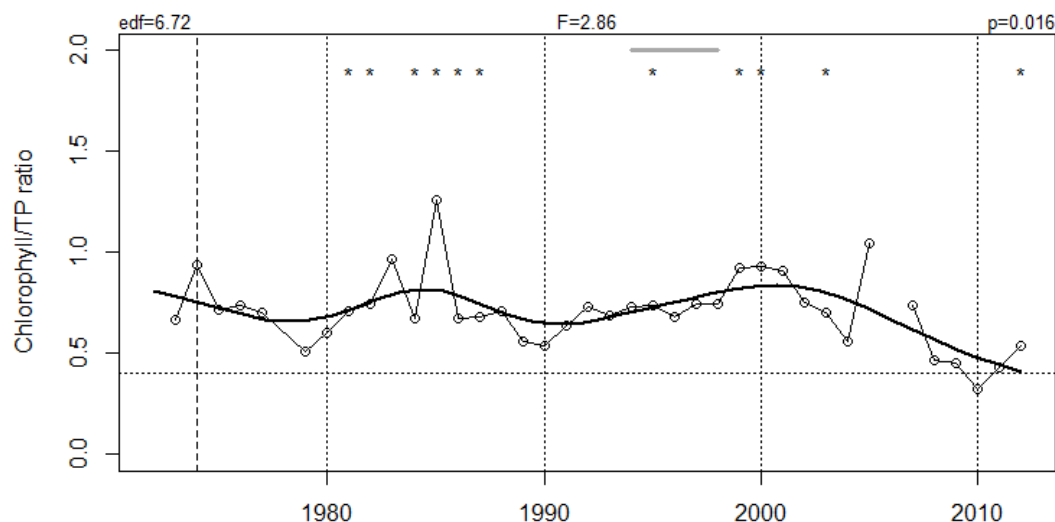
Figure 30 Trend in annual mean chlorophyll a/TP ratio for Hickling Broad (solid black line) showing GAM smoothers, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting (2006 data omitted as no samples March-June).

5.9 Secchi Depth

From 1978 to 2006 there was a slight increasing trend of water transparency in Hickling Broad (Mann Kendall tau= 0.286 p = 0.021), although more recent data suggests this has subsequently declined again (Figure 31). There was a marked short-term improvement of transparency in 1998 and 1999 during the period of *Chara* dominance.

There is a relatively poor relationship between secchi depth and chlorophyll (Figure 32). When chlorophyll is high, transparency is decreased, but the relationship has a plateau at values of chlorophyll above 50µgl^{-1} suggesting other sources of turbidity. This is confirmed by the relationship of Secchi depth with suspended solids (Figure 33), which shows a significant linear relationship. As discussed in 5.6 the explanation for the higher suspended solids relative to chlorophyll in Hickling is unclear. Bales et al. (1993) show a significant relationship between light extinction coefficients for Hickling and counts of small cyanobacteria. The summer dominance of small cyanobacteria in the plankton of Hickling may therefore be an explanation. However, the light climate over the last six years in Hickling is very poor in comparison to other broads, averaging less than 0.5m (Figure 26).
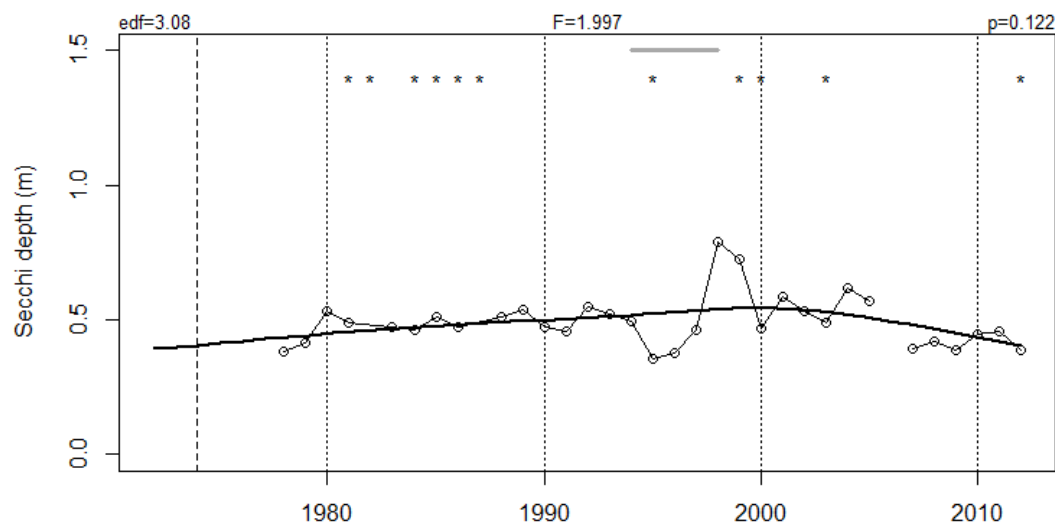
Figure 31 Trend in annual mean Secchi depth for Hickling Broad (solid black line) showing GAM smoother, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting (2006 data omitted as no samples March-June)
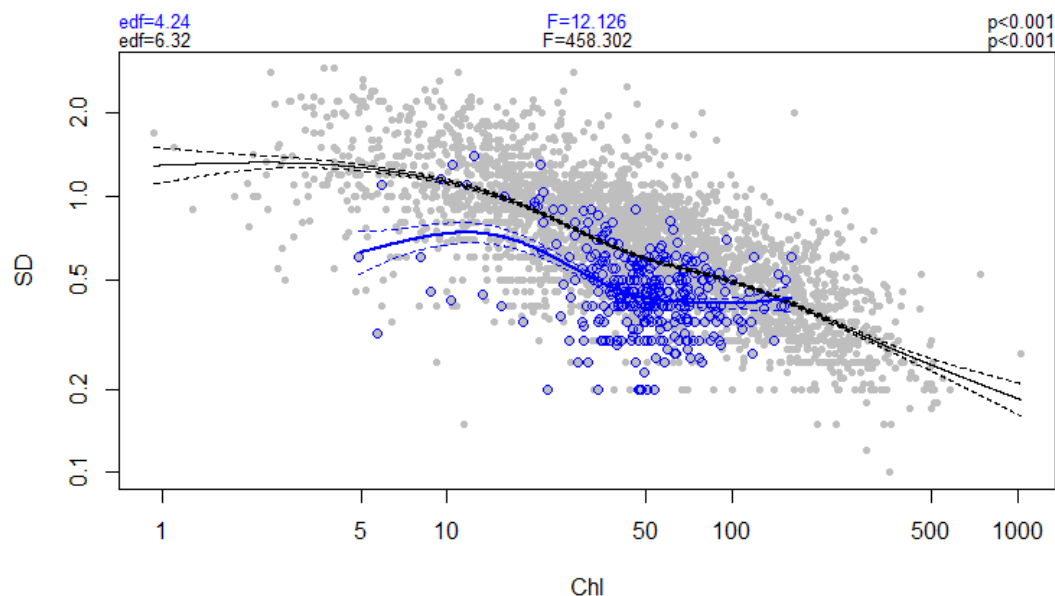

Figure 32 Relationship between water transparency (Secchi depth) and chlorophyll a in Hickling Broad. Lines show GAM smoothers for all broads

Figure 33 Relationship between water transparency (Secchi depth) and suspended solids in Hickling Broad. Lines show GAM smoothers for all broads (black) and for Hickling (blue). (Data shown are only for samples where secchi depth is < water depth).

6   Sediment chemistry

6.1   Review of available data on sediment P and P release

Available grey and peer reviewed literature was reviewed and data on pore water P concentrations used in this report were originally reported by (Jackson and Phillips 1990). Data on pore water chemistry were available from cores collected in 1990 (Jackson 1991) and are summarised in this report. Methods of sample collection and analysis are reported in detail by (Jackson 1991); Jackson and Phillips (1990). We review the available data below in the context of evidence of internal P loading.

Data on sediment TP concentrations from triplicate sediment cores collected 21st August 2013 from 4 sample points in Hickling Broad were provided by the Broads Authority (Andrea Kelly, 8th October 2013). TP concentrations across the upper 15 cm were assessed using homogenised slurries sampled from three sediment depth ranges (1 cm to 3 cm; 7 cm to 11 cm; and 13 cm to 15 cm).

6.2   Review of historical data (i.e. pre 2012)

Soluble reactive P concentrations in pore waters of the upper 12 cm of sediment cores collected from Hickling Broad in 1990 are reported (Table 4). The values indicate low pore water SRP concentrations in relation to other broads surveyed

during 1990 (SRP range for five other broads: 0.110 mg L^{-1} to 0.906 mg L^{-1}; (Jackson 1991) and indicate a low likelihood of significant diffusive release of SRP from interstitial waters to the water column at the time of sampling. In addition, Fe^{2+} concentrations were moderate in pore waters at 0.527 mg L^{-1} during the sample period (range: 0.00 mg L^{-1} to 2.32 mg L^{-1}; Jackson (1991)) and indicate both reducing conditions in the sediment and a plentiful supply of Fe with which P release can be controlled under oxidising conditions. This is in general agreement with the results of the P release incubations in which P uptake by sediments was reported. However, the single high Fe^{2+} value reported (Table 2) coincided with relatively high S^{2-} concentrations (range of other lakes: 0.00 mg L^{-1} – 1.39 mg L^{-1}). High concentrations of S^{2-} may indicate competition with P for Fe binding sites by S, a process which occurs under reducing sediment conditions required for the production of H$_2$S. Figures presented by Jackson (Figure 34) indicate that surface sediment interstitial waters tended to be > 200 mV and that conditions became reducing (i.e. < 0 mV) at depths of below 1 cm. Collectively, these results suggest that there is sufficient chemical barrier to P release from Hickling Broad sediments, and that the potential for competition between S and P species for Fe binding sites may potentially restrict SRP removal within Fe-P complexes under oxidising conditions. The reduction of Fe-P sediment complexes is expected to occur at redox potentials of less than about 50 mV. It should be noted that given data were only available from one sample point the interpretation of results above should be treated with caution, and only represents conditions on the 1st August 1990.

Table 4. Average concentrations of pore water soluble reactive phosphorus (SRP), Fe^{2+} and S^{2-} in the upper 12 cm of sediment in Hickling Broad in 1990. SRP release rates are estimated from sediment core incubations (n=3) sampled in 1990. (Jackson 1991)

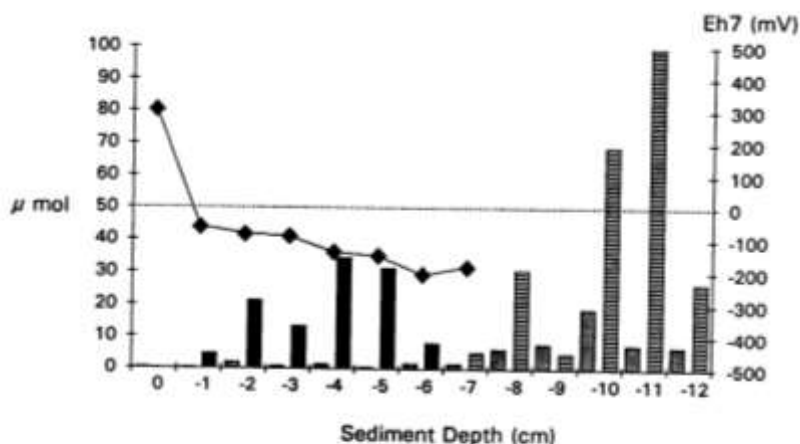| Date	SRP (mg L^{-1})	Fe^{2+} (mg L^{-1})	S^{2-} (mg L^{-1})	SRP Release rate (mg m^2 d^{-1})
1/8/1990	0.110	0.527	0.764	-0.8

Figure 4.2.3.1 Hickling Broad 1/8/90 - Variation in interstitial soluble phosphorus (▤), ferrous iron (▮) and soluble sulphide (≡) concentration and redox potential (-♦-) with depth.

Figure 34 Copy of figure from (Jackson 1991)

6.3 Current baseline sediment phosphorus

Variation in sediment TP concentrations with sediment depth in 2013 up to 15 cm is shown (Figure 35). The average TP concentration across the upper 15 cm of sediment was 0.36 mg TP g^{-1} dw in 2013. In the context of recommendations made by (Sas 1989), the reported sediment TP concentrations indicate low potential internal P loading in Hickling Broad.

The relationship between sediment TP concentration and sediment depth is shown (Figure 35) and indicates a gradual decrease in sediment TP concentration with increasing sediment depth up to about 15 cm. This relationship is commonly reported in lakes suffering from elevated catchment P loading as described by Carey and Rydin (2011). The gradient of the slope in the relationship between $\log_e$ sediment TP and sediment depth can be used to estimate the potential for sediment P release following reduction of catchment P loading where strongly negative slopes indicate the strongest P release potential. The slope for Hickling Broad across the upper 15 cm of sediment in 2013 was – 0.034. Similar relationships calculated using data from 96 lakes ranging in TP concentrations from 3 to 1162 µg TP L^{-1} indicate that TP accumulation in the upper 15 cm of sediment in Hickling Broad is similar to that of other moderately eutrophic lakes.

As a general principle, since marine and brackish waters are naturally rich in sulphate there is the potential for sulphide to reach high concentrations in reducing sediments. Due to increased sequestering of iron by sulphide in such systems the efficiency of co-precipitation of P by iron oxides is reduced (Blomqvist et al. 2004). The sediments of Hickling Broad may therefore display characteristically poor retention of P. This may be reflected in lower potential for internal loading of P from sediment during anoxia. Rising salinity will exacerbate the problem of non-retention of P in the sediment, although this may be partially offset by an increased iron supply delivered by pumped drainage via Horsey Mere.
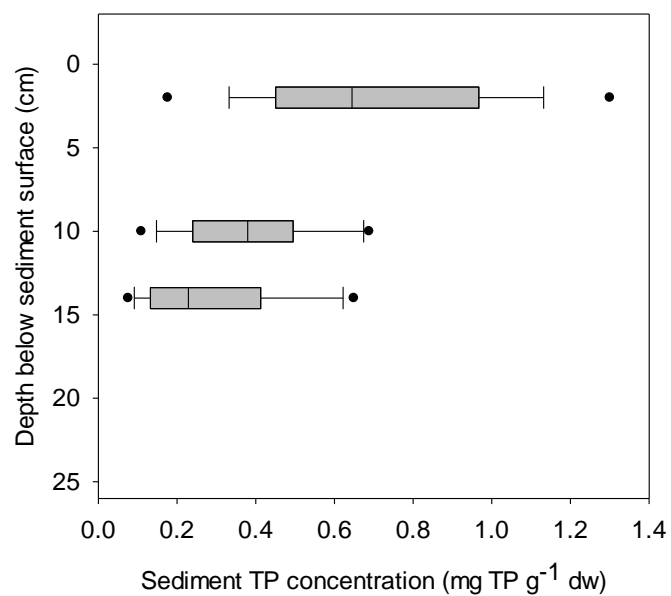
Figure 35 Variation in sediment TP concentration with sediment depth in Hickling Broad in 2013. Sediment samples were collected from 1 cm to 3 cm; 7 cm to 11 cm and 13 to 15 cm sediment depth ranges.

7 Macrophytes

7.1 Cover: macrophytes and algae

Hickling is unique among the broads in showing a pronounced and more or less steady increase in plant cover from 1986 to a peak in 1999 followed an equally rapid decline, interrupted only by strong performance in 2003. In the period 2007-2009 total plant cover in Hickling was lower than at any time in the previous 25 years (Figure 36). 2011 onwards has seen a partial recovery but cover in the period 2011-13 was almost identical in each year rather than increasing.
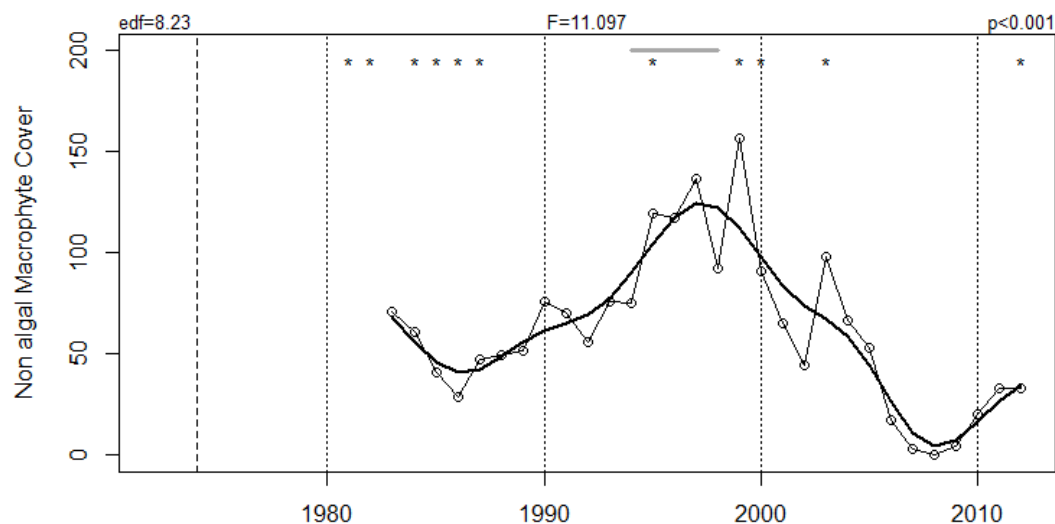
Figure 36 Trend in cover of non-algal macrophytes for Hickling Broad (solid black line), showing GAM smoother, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting.

Filamentous algal cover is perennially low in Hickling and is unusually low for a brackish lake with elevated P concentrations (Figure 37). One possibility is that this reflects periphyton grazing by the mysid shrimp *Neomysis integer* which is common in Hickling but absent from the freshwater broads, and is a known algal grazer (Irvine et al. 1993). Barker et al. (2008) report the results of a mesocosm experiment with Hickling sediment and water in which salinity was manipulated and P loading increased relative to the present levels. *Neomysis* was absent from the mesocosms

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and in this case filamentous algal growth was much higher than that observed in the field in Hickling. Observations by Gurney in the early 1900s also suggest that filamentous algal growths in Hickling were considerably higher in the past when salinity was lower and *Neomysis* was absent. The extent of filamentous algal growth is similarly low in all the other Thurne Broads and is very much lower than the amounts found in the non-brackish broads generally, even when their P concentrations are lower. Filamentous algal growths in other broads are typically benthic in the early part of the season, later moving to the surface under their own buoyancy and the support of a canopy of macrophytes. A combination of turbidity and low sediment nutrient release may therefore be additional contemporary constraints on the growth of filamentous algae in Hickling.
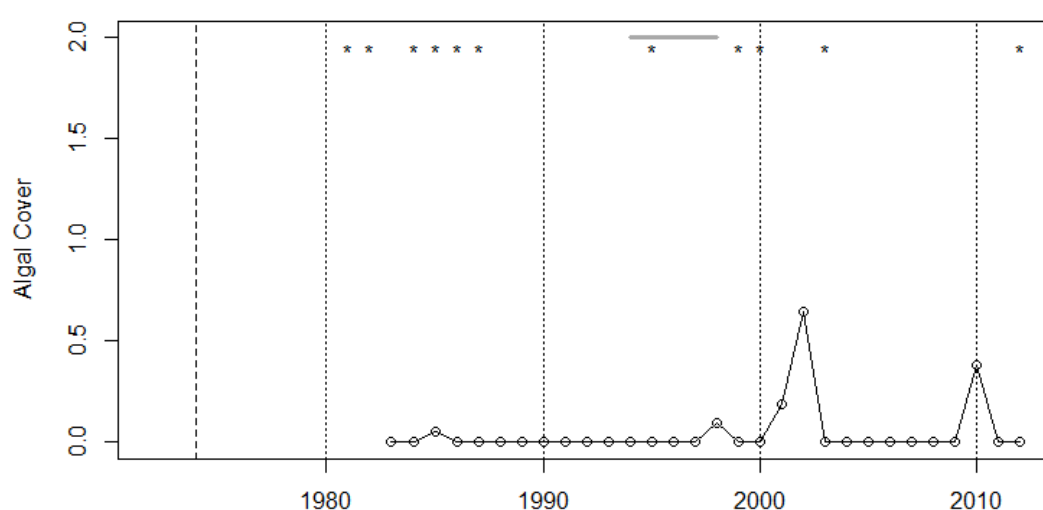
Figure 37 Trend in algal macrophyte cover for Hickling Broad, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting.

7.2 Species richness

Plant species richness was relatively stable (~10 – 13 species) from 1987 to 2001, after which it declined sharply mirroring the overall decline in cover (Figure 38). The recovery since 2008 to ~10 species is in line with the partial recovery in cover since this time. Richness in Hickling is generally less volatile than cover, which suggests that small populations of the majority of species are able to persist through periods of adversity. In the same way the significant increase in cover from 1987 to 1999 appears to have been dependent on increasing populations of already established species rather than new colonists.

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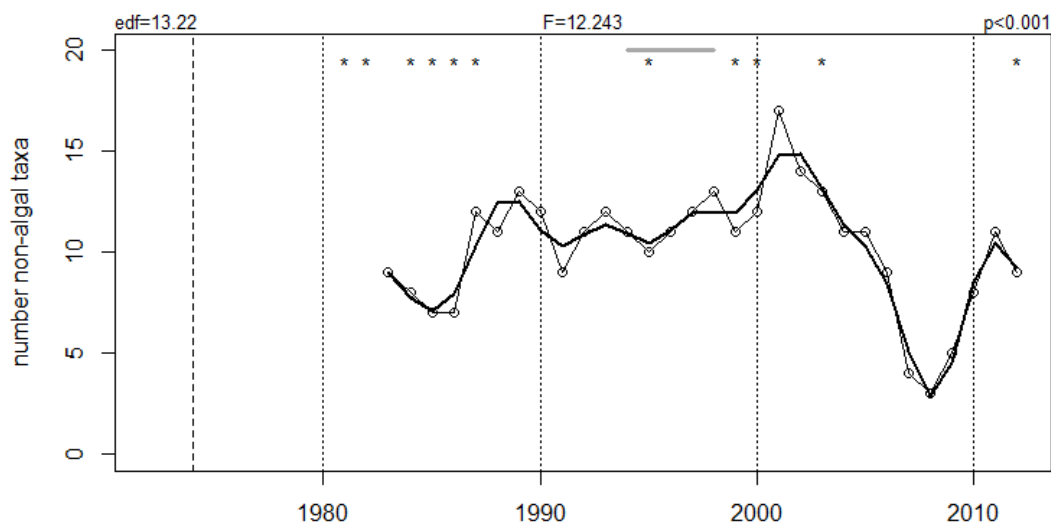
Figure 38 Trend in macrophyte species richness for Hickling Broad (solid black line), showing GAM smoother, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting.

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7.3   Compositional change

The compositional data (Figure 39) suggest a series of competitive interactions and facilitation amongst the major species with external drivers in the form of growing conditions and herbivory by birds. The most striking feature of the recent vegetation history in Hickling is the 10-year build up and collapse of *Chara* beds which reached a peak coverage of >80% in 1999. The exact drivers for the initial build-up of *Chara* beds are unclear but appear to involve some or all of a combination of declining TP and chlorophyll, sunny years, especially in 1995 (the sunniest of the past 30 years), a period of low grazing by water birds possibly favouring the expansion of a supporting crop of *Myriophyllum spicatum*, and a series of drier than average years, especially in 1996 and 1997 which were presumably associated with lower pumping rates on the Brograve and Horsey levels. There is no evidence from this data or commissioned reports that localised cutting of *Chara* from 1994-99 to maintain sailing had an overall negative effect on the *Chara*, although it presumably constrained the net cover below its potential maximum and may have contributed to a decline in *Myriophyllum* cover over this period. By 2000 the extent of *Chara* was reduced by about 25% with a proportionally even larger decline in *Myriophyllum*. As with the initial build-up of *Chara* the causes of decline in the aquatic vegetation in Hickling from 1999-2005 are likely to have multiple causes. One factor was almost certainly grazing by coot whose numbers reached a peak of 5000 birds in October 1999. Other likely contributing factors include an unusually wet year in 2001 and the effect of anoxia, caused by

accumulation of plant detritus, on sediment chemistry. There was a partial recovery of *Chara* in 2003, which was again an unusually sunny year (only exceeded in the last 30 years by 1995 and 1976), but *Chara* has been absent from Hickling since 2006. In 2013 there was limited growth of *Chara* in Hickling equivalent to a coverage of 5%. This coincided with the best summer growing conditions since 2003 and may have been aided by the increasing cover of *Myriophyllum*. Whether the 2013 growth is sufficient to kick-start a re-expansion of *Chara* should be evident in 2014.

Najas marina, and to a lesser extent the fine-leaved pondweeds *P. crispus* and *P. pusillus* appear to benefit from the window of clear water conditions associated with the *Chara* boom. Although *Najas* has persisted at low cover in Hickling since the 1980s (and indeed for much longer in terms of historical data) as an annual it is likely to depend on high water clarity and to be restricted to sheltered margins at other times.

Potamogeton pectinatus shows a somewhat different pattern to the other species. There is a possible competitive interaction with *Myriophyllum* and it seems likely that the latter would also be favoured over *P pectinatus* by declining P concentrations. The seeds, tubers and foliage of *P pectinatus* are also grazed vigorously by water birds and one factor contributing to the continued decline of *P pectinatus* may therefore be the large increase in wintering Teal and Greylag Goose at Hickling since 2000 (see section 10).

The most recent spatial distribution of plants in Hickling (2013) is considered in Figure 42. There is a strong indication of physical control by wind exposure on the distribution of plants, with greatest coverage at the upwind eastern end of the broad and in sheltered bays and marginal areas. Given this distribution wind strength and direction might be expected to be an important additional influence on inter-annual fluctuations in overall plant abundance.
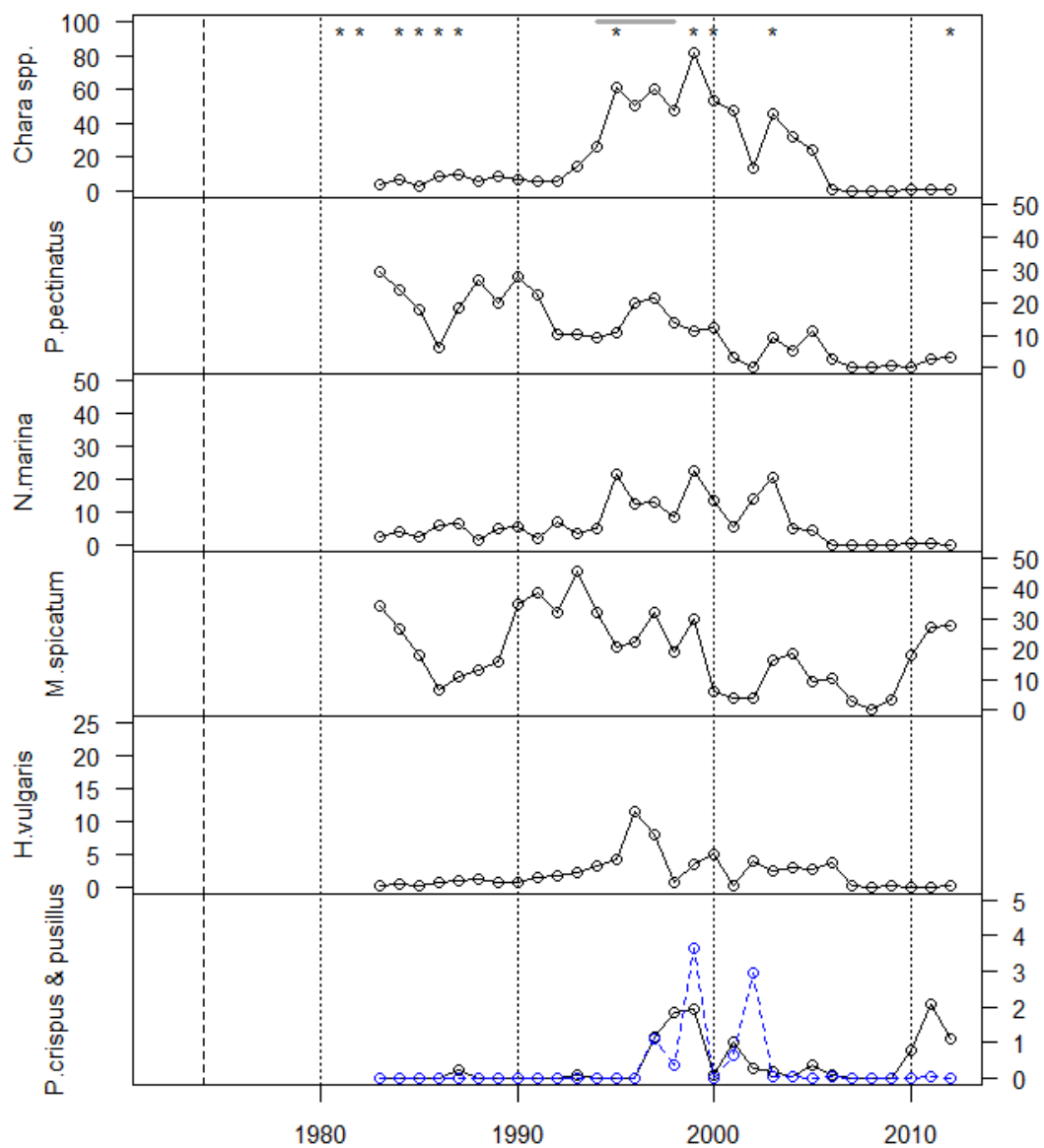
Figure 39 Trend in macrophyte species in Hickling Broad. Vertical lines mark key events, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting.
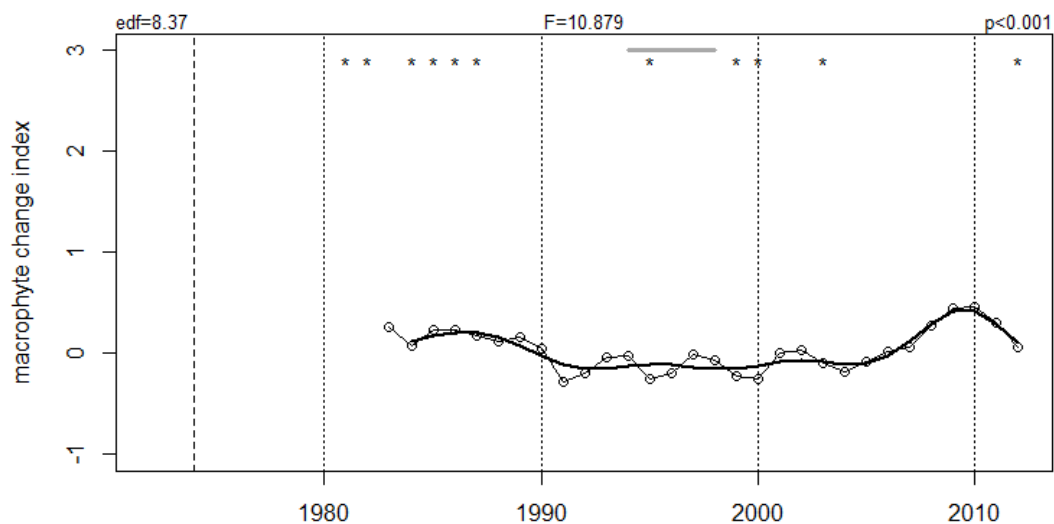
Figure 40 Trend in macrophyte Change Index for Hickling Broad, stars mark fish kills due to *Prymnesium*, grey bar marks period of macrophyte cutting.
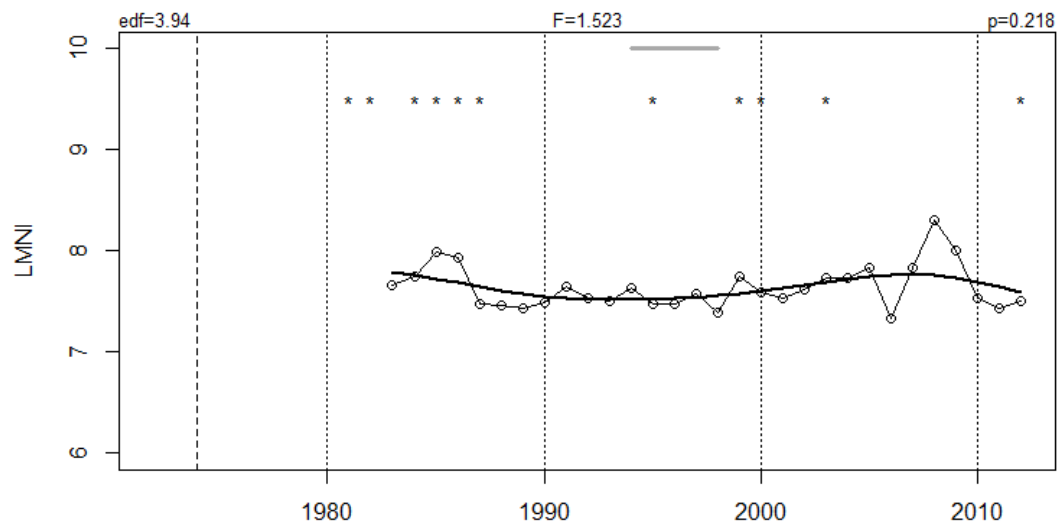
Figure 41 Trend in lake macrophyte nutrient index (LMNI) for Hickling Broad (solid black line), showing GAM smoother. Vertical lines mark key events, grey bars mark period of sediment removal.
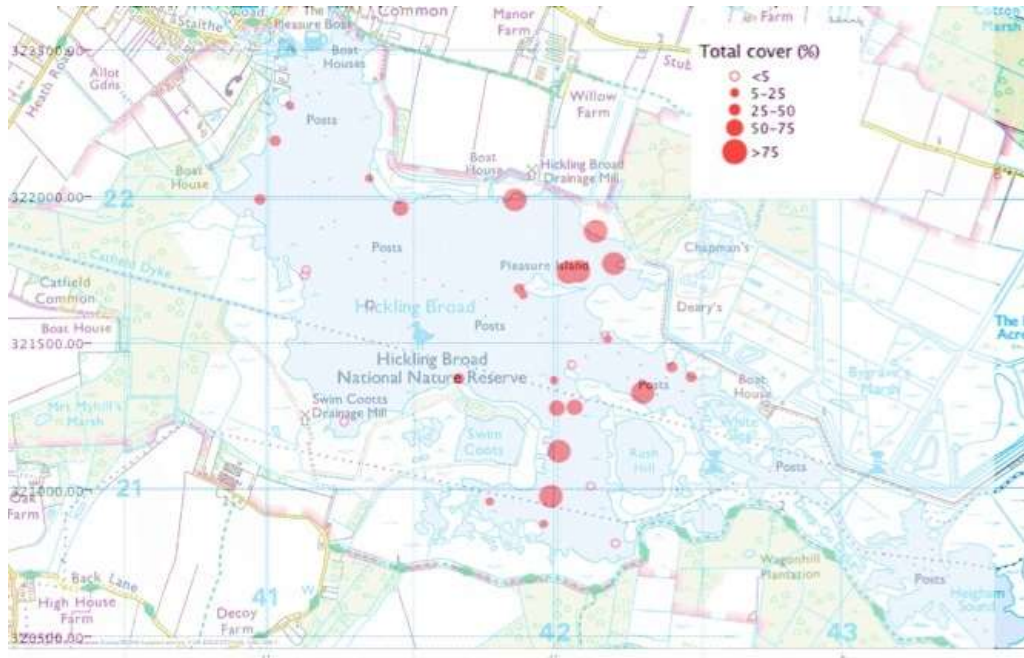
Figure 42 Spatial distribution of plants in Hickling Broad in 2013. Note the concentration of plants in sheltered bays and margins.

8 Zooplankton

Mesocosm studies designed to simulate the effects of different salinity scenarios in Hickling indicate that the zooplankton community is essentially a product of the combined effect of salinity and fish density (Barker et al. 2008). Ambient salinities are too high for the survival of large bodied cladoceran grazers and the zooplankton is largely dominated by the small cyclopoid copepod *Eurytemora*.

9 Fish

9.1 Abundance and composition

Four surveys of Hickling Broad were undertaken by the EA using seine nets (1986,1989,1993 and 1998). In total, twelve species were captured, European eel, Common bream, Gudgeon, Roach, Rudd, Pike, Smelt, Three-spined stickleback, Ten-spined stickleback, Sea Bass, Perch, Common goby and Flounder. Of these only Roach, Pike, Perch and Flounder were present in all four surveys.

In general, overall fish density within the Hickling Broad increased from 0.23 ind. m^{-2} in 1986 to 0.33 ind. m^{-2} in 1998. Whilst the densities are relatively low, only the limnetic zone is sampled by seine net, with the littoral margin, which can support large numbers of fish, particularly fry, not sampled. The earlier two surveys were possibly influenced by a high frequency of *Prymnesium*-induced fish kills during the

1980s with the higher fish densities in the later surveys being consistent with recolonisation.

Roach were the dominant fish numerically, contributing between 67% (1986) to 93% (1993) of individuals, with Common bream contributing 11% and 13% in 1986 and 1989 respectively. As the Common bream were likely to be large adult fish the potential contribution to the overall biomass would likely be higher. The two species remain the popular catches for anglers on account of the number of Roach and the large size of the bream (Wilson 2013).

Although Common bream appeared to decline over the period over which seine net surveys were conducted, in electric fishing surveys of the connected Catfield Dyke (undertaken annually by the EA between 2004 and 2012, 2007 excepted) the species maintained a presence, and in general was the most abundant species after Roach. Pike were also more prominent within the fish community through this method of sampling, as was Rudd, present in each of the surveys, and more in keeping with the increasing presence in anglers' catches (Wilson 2013). The estimated densities from the electric fishing surveys of Catfield Dyke have been consistently low at <0.1 ind. m^{-2} since 2006.

10 Water birds

Being the largest of the broads, and given its proximity to grazing marshes and the coast Hickling holds large numbers of water birds. Since about 2004 Hickling has held nationally important numbers of wintering Teal, Greylag and Gadwall.

Few species show statistically significant long term trends in terms of seasonal maxima and there are several unfortunate gaps in the record at critical times (Figure 43 - 46). The most striking increases are in wintering Teal and Gadwall and in numbers of Greylag in both seasons, the latter having tended to replace Canada Geese. The increase in these three species is consistent with overall national trends, feral Greylag in particular having shown a dramatic increase across GB since the early 1980s. Wintering Teal numbers at Hickling in recent years have averaged 3000 with a peak of 4550 in 2005/6. These numbers have remained high against a national decline in Teal over the past 8 years. Wintering Wigeon at Hickling have shown a modest but significant decrease that is contrary to the national trend over the same period. In terms of summer numbers both Mallard and Shoveller have shown a marked and significant long-term decline, though wintering Mallard numbers have held up against a national long-term decline. The decline in summer Mallard is surprising as the British breeding population has increased significantly over the last few decades. There have also been weaker and marginally significant decreases for summer Pochard and Tufted Duck. In the case of Pochard the observed decline is consistent with a pronounced national decline that has been ongoing since the mid-1990s. The causes for declines in summer bird numbers at Hickling are likely to be varied - some merely echo national or international trends with larger scale

underlying causes, but locally important factors may include conversion of some grazing marshland to arable, competitive interactions amongst species including
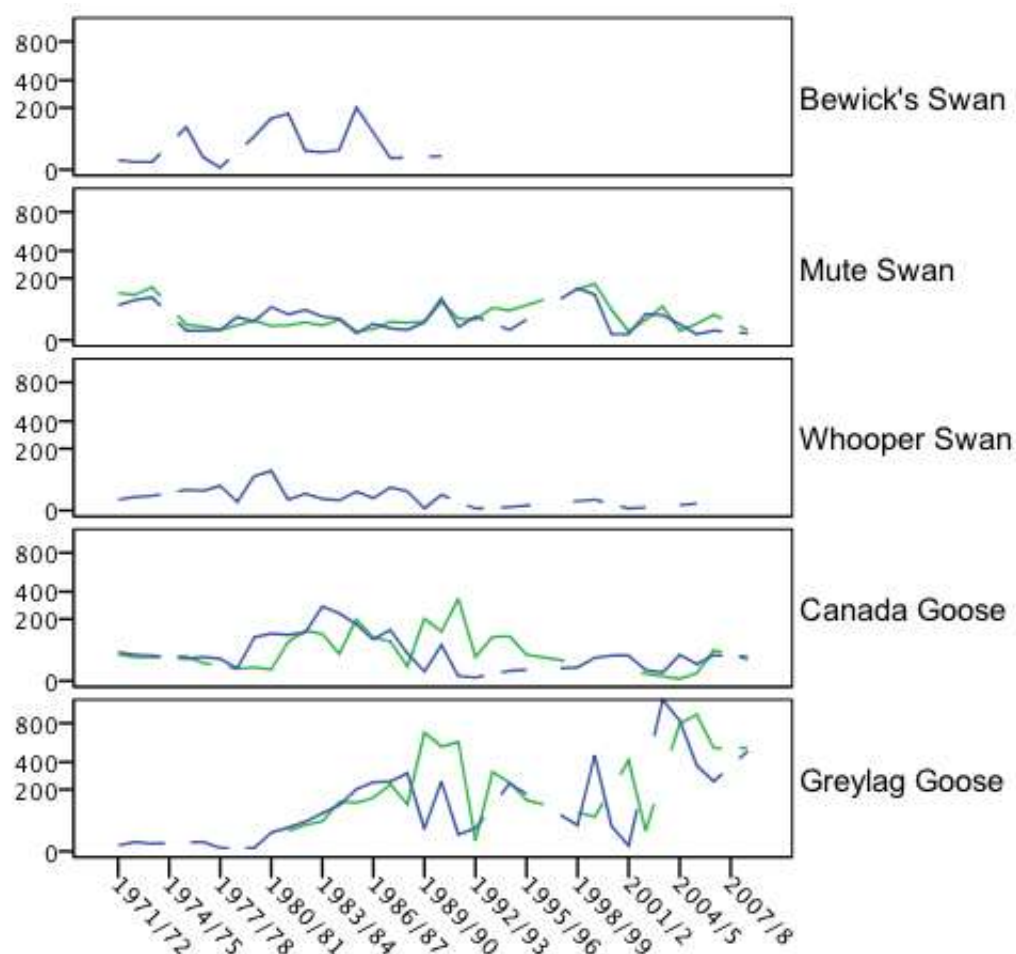
Figure 43 Changes in abundance (as maximum seasonal counts) of swans and geese in Hickling Broad since 1971 based on BTO WeBS counts. Winter (blue) = Oct-March; Summer (green) = April-Sept.

possible displacement by feral geese, disturbance by powered boats and sailing dinghies and loss of marginal reedswamp.

There are a number of inter-annual and short-term fluctuations in bird numbers that can be clearly linked to the abundance of aquatic vegetation. The most prominent of these is for coot that clearly tracks the increase and collapse in *Chara* abundance, reaching a peak of 5000 birds in October 1999, including a secondary peak in 2003 when there was partial recovery of *Chara*. Birds are both drivers of and responders to changes in aquatic vegetation, as demonstrated by the pattern in numbers of the major diving duck, Tufted Duck and Pochard, which largely mirror those of coot, and presumably reflect the temporarily increased availability of plant-associated invertebrate prey during periods of high vegetation abundance. Significant crashes in

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diving duck populations in Lough Neagh have recently been linked to major declines in the invertebrate food base (Tomankova et al. 2014). As omnivores coot in Hickling are likely to have benefited from both direct grazing of plants and feeding on plant-associated invertebrates. Numbers of other species show weaker trends that are
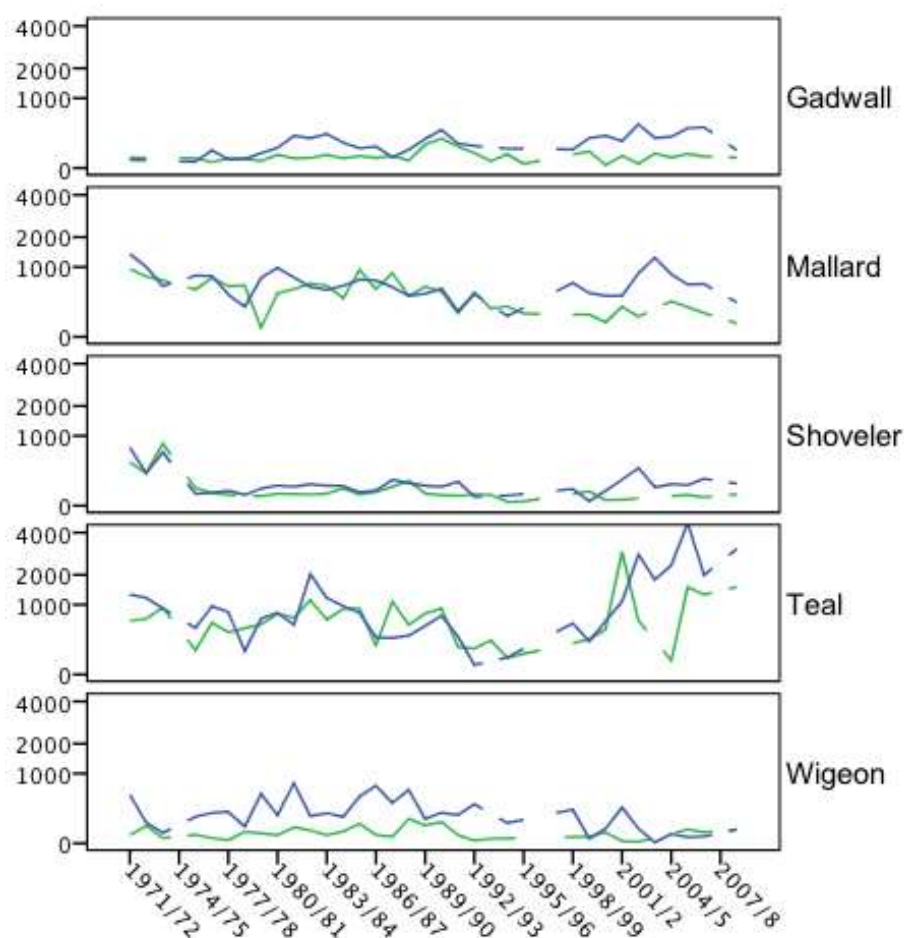

Figure 44 Changes in abundance (as maximum seasonal counts) of dabbling duck in Hickling Broad since 1971 based on BTO WeBS counts. Winter (blue) = Oct-March; Summer (green) = April-Sept.

potentially associated with plant abundance. Summering Mute Swan increased year on year until 1999 after which they decreased strongly mirroring the crash in *Chara*. Only wintering Teal and resident Greylag have shown a strongly increasing trend over the last decade and it is unclear if this may be implicated in limiting the recovery of aquatic vegetation in Hickling.

The major omission to this account concerns gulls, which for some reason do not figure in the WeBS data for Hickling. Black-headed gull numbers in the early 1970s peaked at a staggering 250,000 birds and contributed to substantial guanotrophication but declined rapidly following the closure and capping of the Martham landfill site (Moss and Leah 1982).
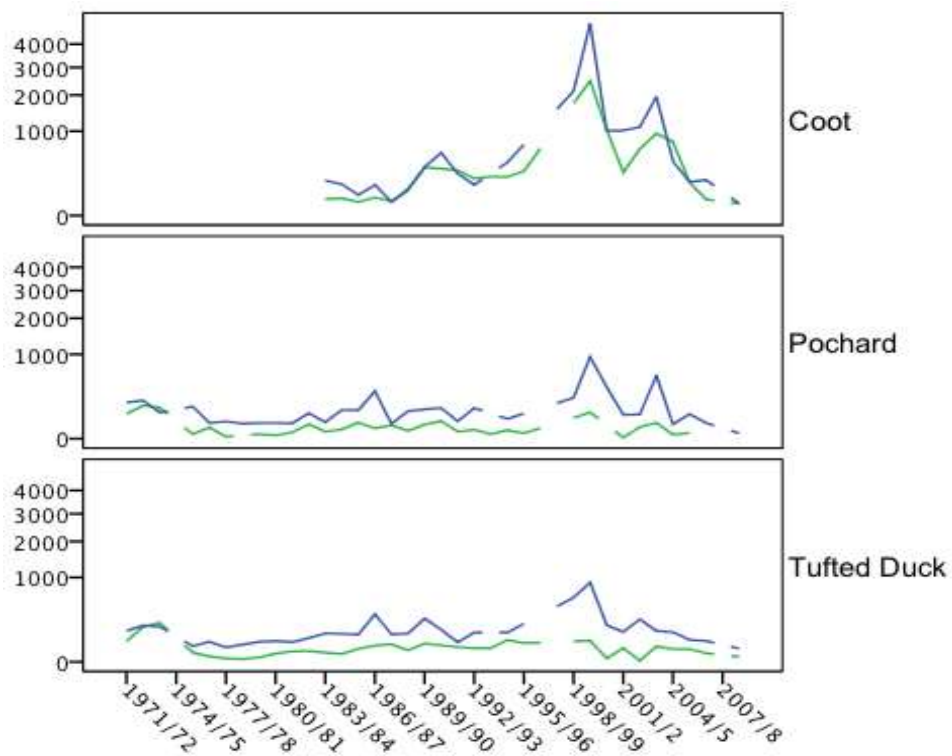
Figure 45 Changes in abundance (as maximum seasonal counts) of coot and major diving duck in Hickling Broad since 1971 based on BTO WeBS counts. Winter (blue) = Oct-March; Summer (green) = April-Sept.
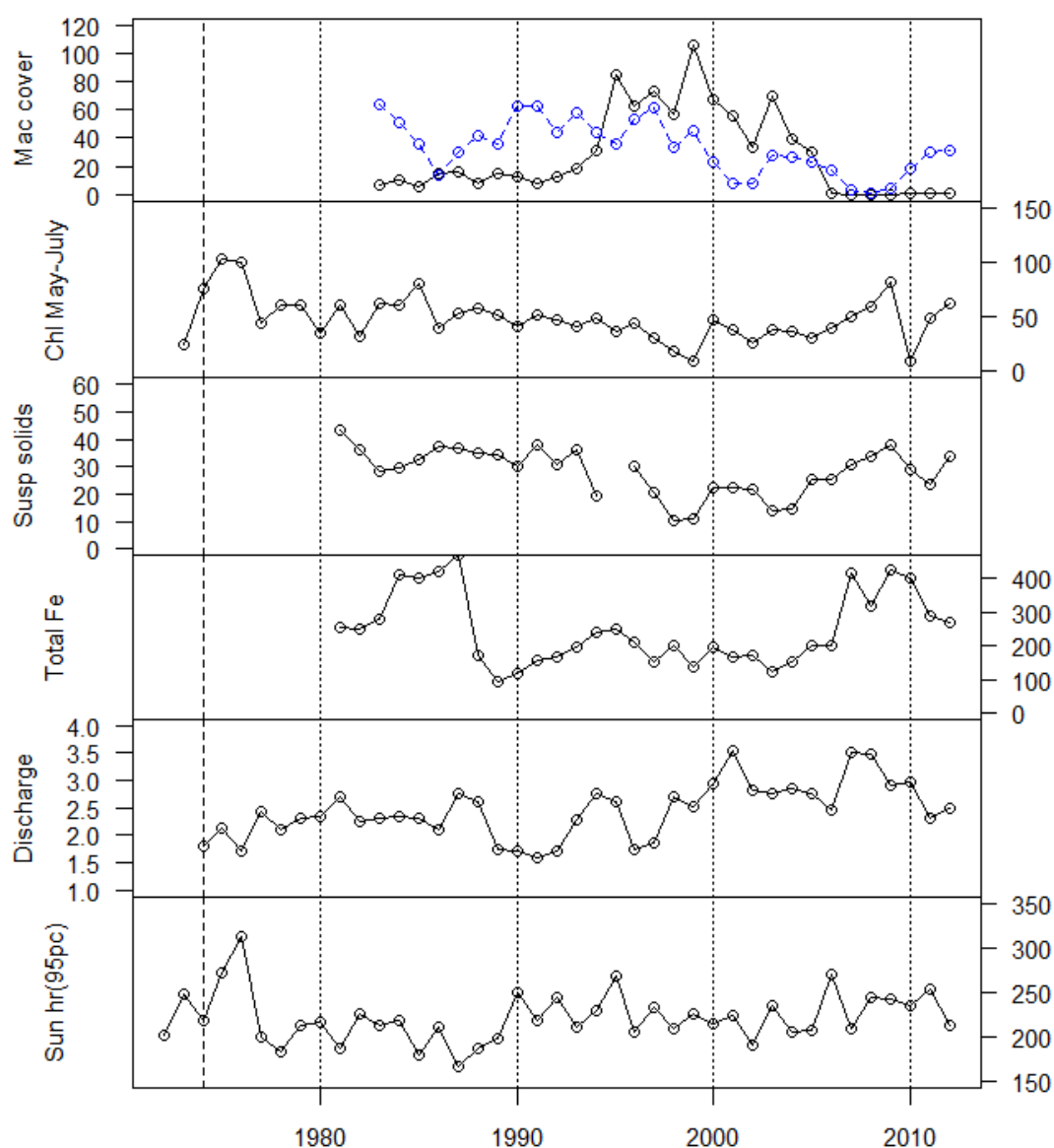
11 Summary of interactions

Figure 46 Comparison of annual time series in Hickling Broad for a) cover of Charophytes + *Najas* **(solid line) &** *M. spicatum + P. pectinatus + H. vulgaris* **(blue dotted line) b) mean May-July chlorophyll a (μg.l^{-1}), c) annual mean suspended solids (mgl^{-1}), d) annual mean total iron (μg l^{-1}), e) annual mean river discharge (m$_3$.s^{-1}) and f) 95th percentile of hours of sun. Vertical lines mark key events, stars mark fish kills due to** *Prymnesium***, grey bar marks period of macrophyte cutting.**
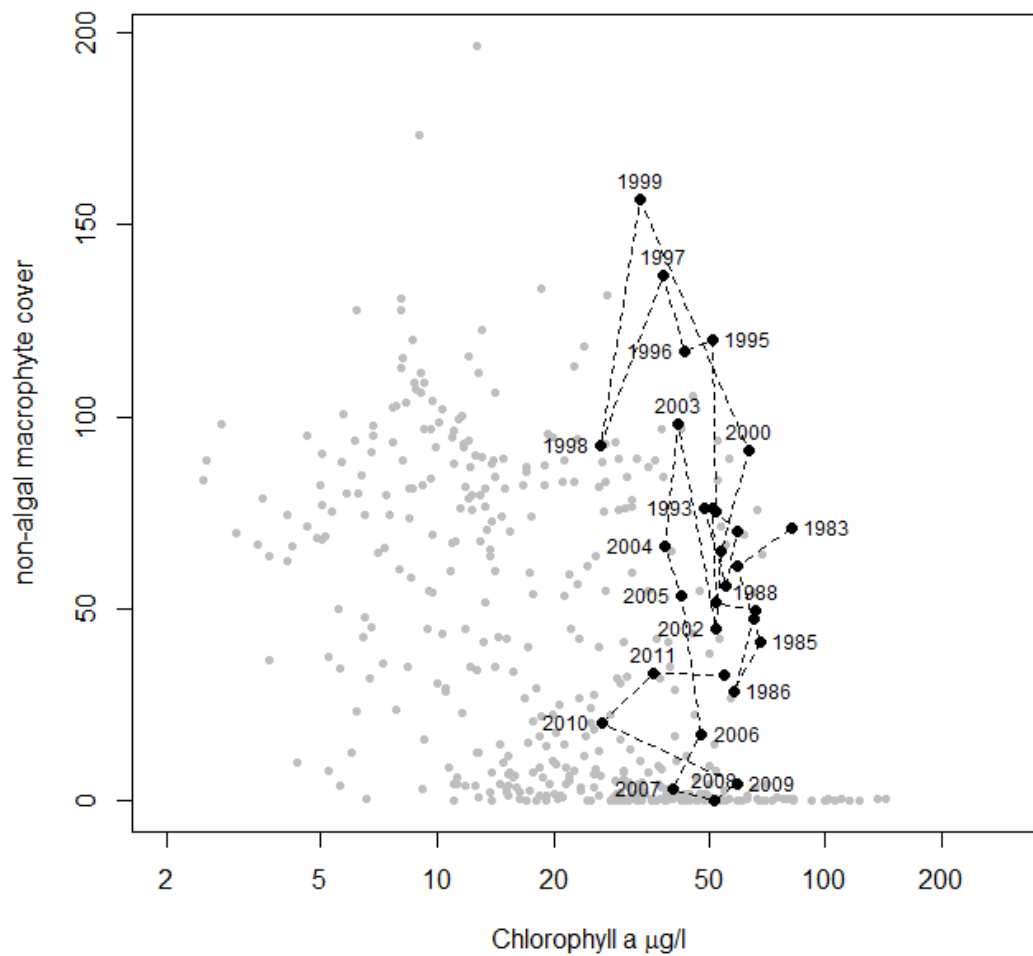
11.1 Macrophytes and Chlorophyll

In Hickling Broad there is a very wide range of response of macrophyte cover over a narrow range of chlorophyll (Figure 47). There is a similar response in comparison to total phosphorus (Figure 48), Secchi depth transparency (Figure 50) and a significant

linear relationship (adjR^2 = 0.379 p=0.004) with total suspended solids (Figure 49). What is clear from all of these relationships is that the cover of macrophytes in Hickling is higher for a given level of chlorophyll/suspended solids or water transparency than in other broads. Even in the case of TP the macrophyte yield over the range 80-100ug/L is much higher than that observed in other broads at similar TP. This may in part be a function of the relatively shallow depth of much of Hickling Broad, or a result of the salinity (e.g. through reduced cell size of phytoplankton, increased epiphytic algal grazing by *Neomysis*).

The dominant macrophytes in Hickling Broad are species which produce long floating mats of foliage. For example, *Myriophyllum spicatum* plants produce the majority of their leaves in the upper 40 cm of the water column by May (Phillips and Moss 1978) thus making them less sensitive to water transparency. *Potamogeton pectinatus* has a similar strongly apical growth form and the cover of these two species has no relationship with either chlorophyll or suspended solids (Figure 53a & Figure 54a). In contrast species with leaves distributed throughout the water column such as *Chara* and *Najas marina* have (non-linear) relationships with chlorophyll and suspended solids (Figure 53b & Figure 54b).

Thus while the dominance of canopy forming species in Hickling may partly explain the higher abundance of macrophytes in this turbid and still plankton-dominated lake other factors such as salinity and its shallow depth may also have a role. This is a very important feature of the interactions in Hickling Broad and further work is required to identify the explanation.

Figure 47 Relationship between non-algal macrophye cover and mean growing season (April-September) chlorophyll a, showing trajectory of change in Hickling Broad. Grey points mark values for all other broads
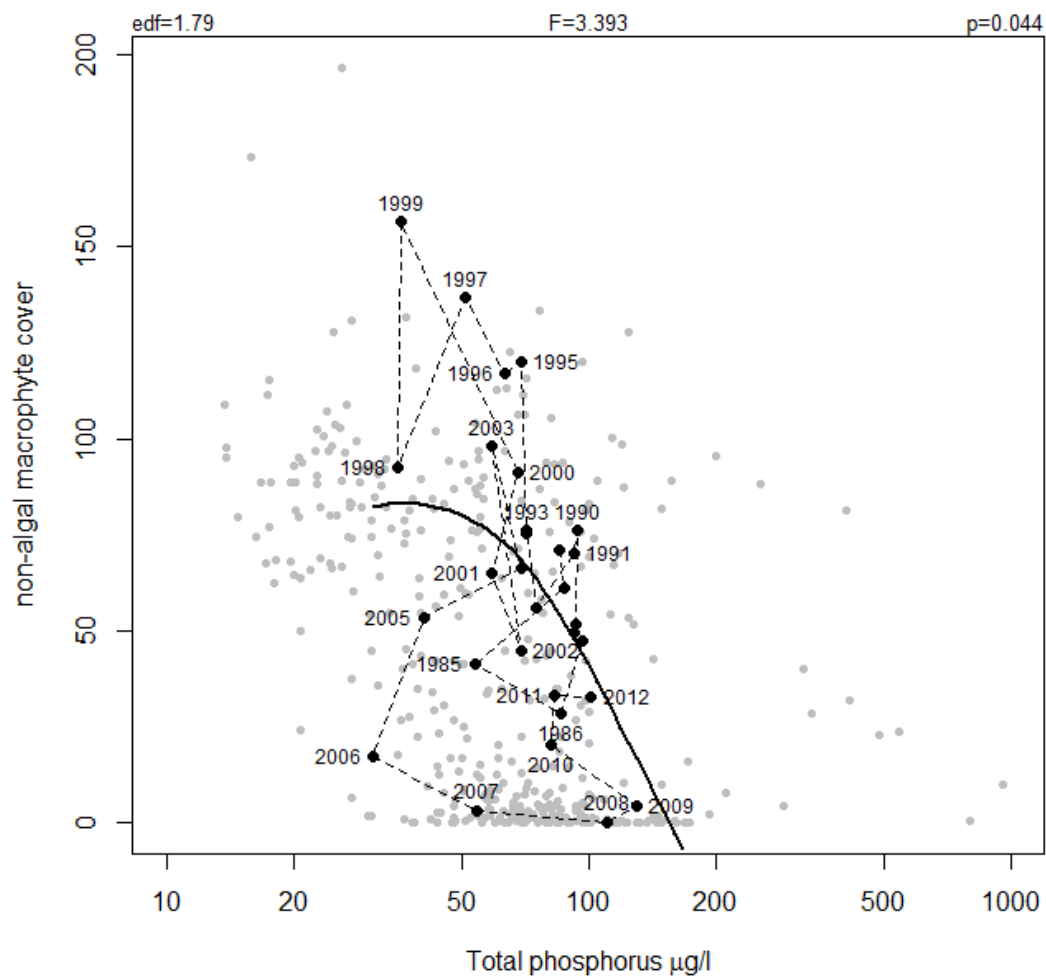
Figure 48 Relationship between non-algal macrophye cover and total phosphorus, showing trajectory of change in Hickling Broad. Grey points mark values for all other broads
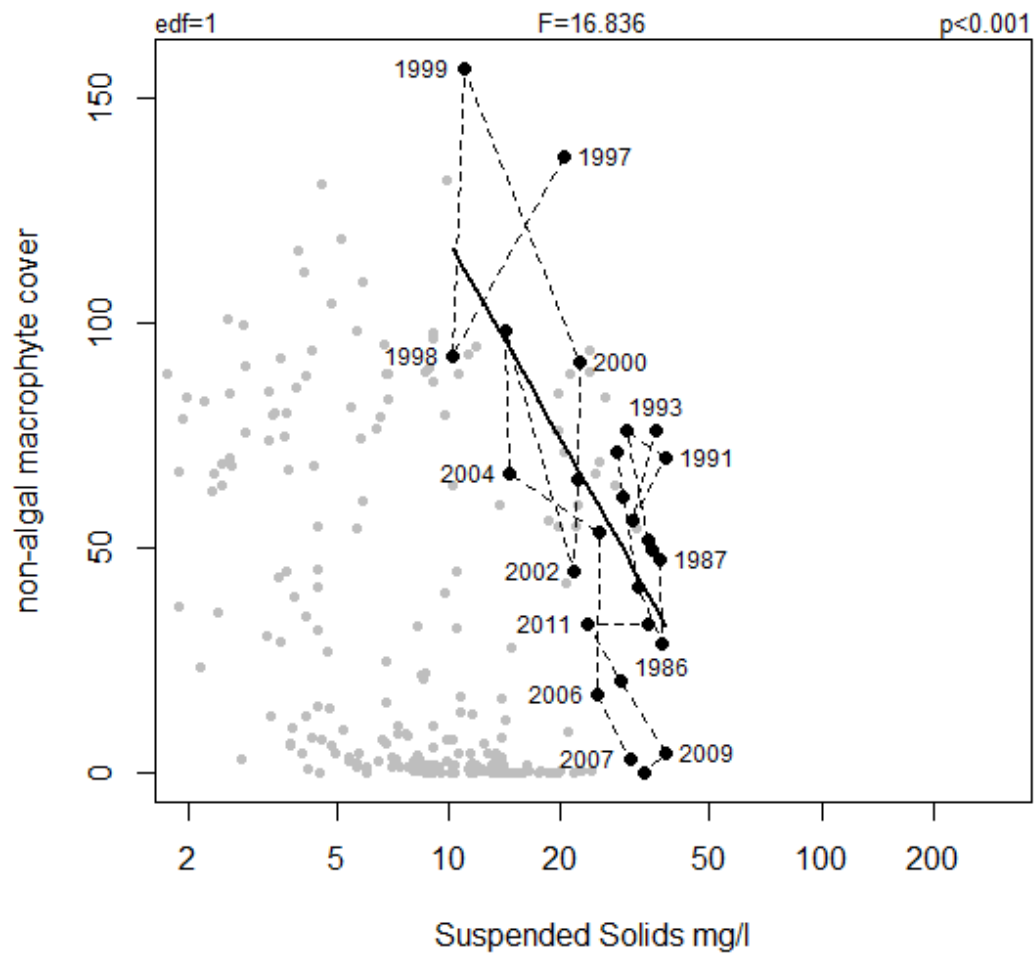
Figure 49 Relationship between non-algal macrophye cover and total suspended solids, showing trajectory of change in Hickling Broad. Grey points mark values for all other broads

Figure 50 Relationship between non-algal macrophye cover and Secchi transparency, showing trajectory of change in Hickling Broad. Grey points mark values for all other broads

Figure 51 Relationship between cover of a) Canopy forming macrophytes (*M.spicatum* + *P. pectinatu*s) and b) Non-canopy forming macrophytes (*Chara* spp + *N. marina*) with mean annual chlorophyll a for all broads in comparison to Hickling Broad (solid red) and other Thurne Broads (open circle), lines show upper 90th quantiles fitted by quantile regression

Figure 52 Relationship between cover of a) Canopy forming macrophytes (*M.spicatum* + *P. pectinatu*s) and b) Non-canopy forming macrophytes (*Chara* spp + *N. marina*) with mean annual suspended solids for all broads in comparison to Hickling Broad (solid red) and other Thurne Broads (open circle) , lines show upper 90th quantiles fitted by quantile regression.

11.2 Chlorophyll *a* v TP

The trajectory of change of mean annual chlorophyll *a* and total phosphorus is shown in Figure 53 in comparison with other European lakes (Phillips et al. 2008). It

illustrates the high yield of chlorophyll in Hickling and that the reduction in chlorophyll is a direct result of a reduction in phosphorus. However, the post-2009 rise in TP is not accompanied by an uplift in chlorophyll.
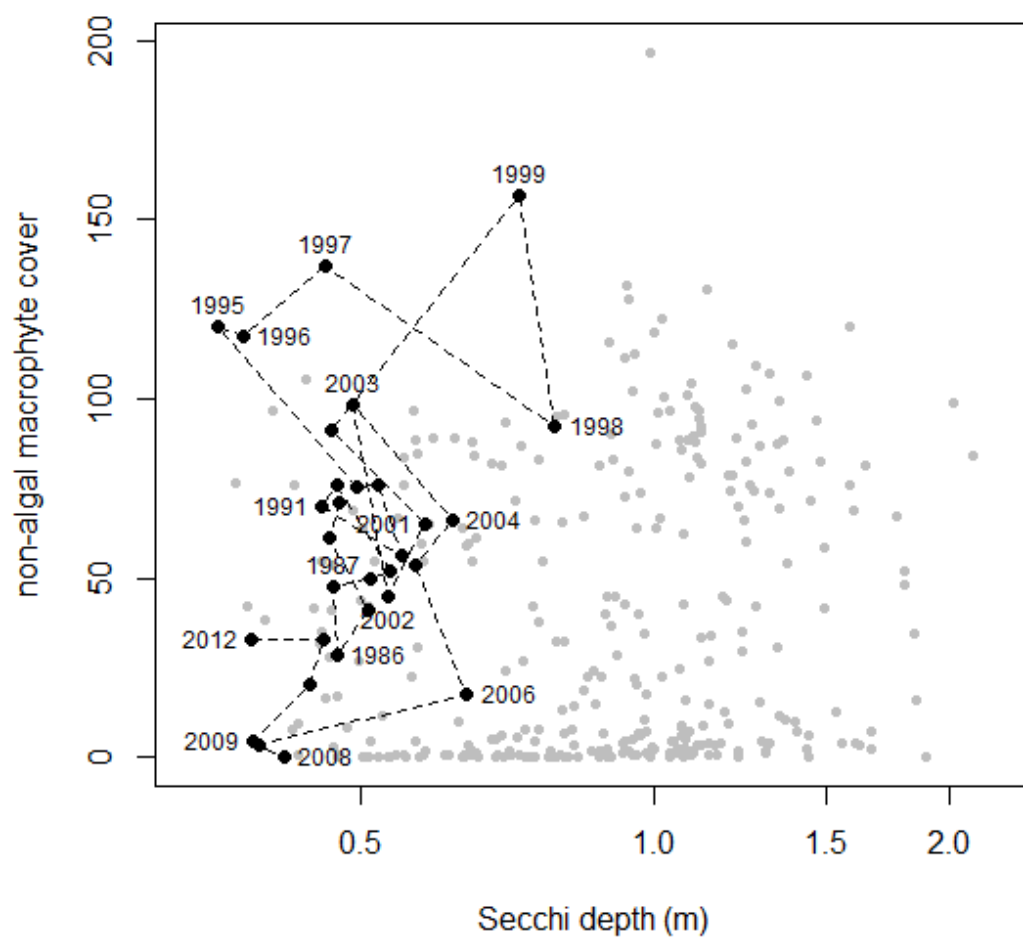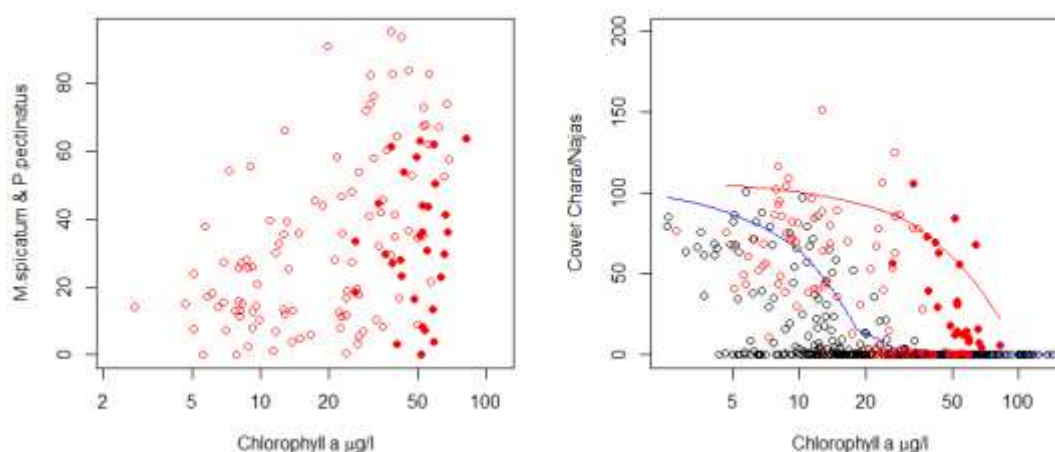

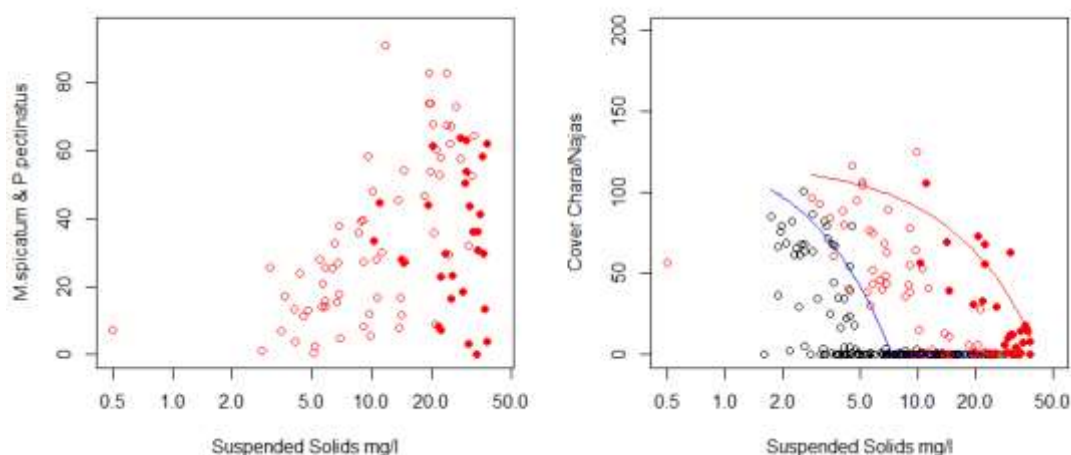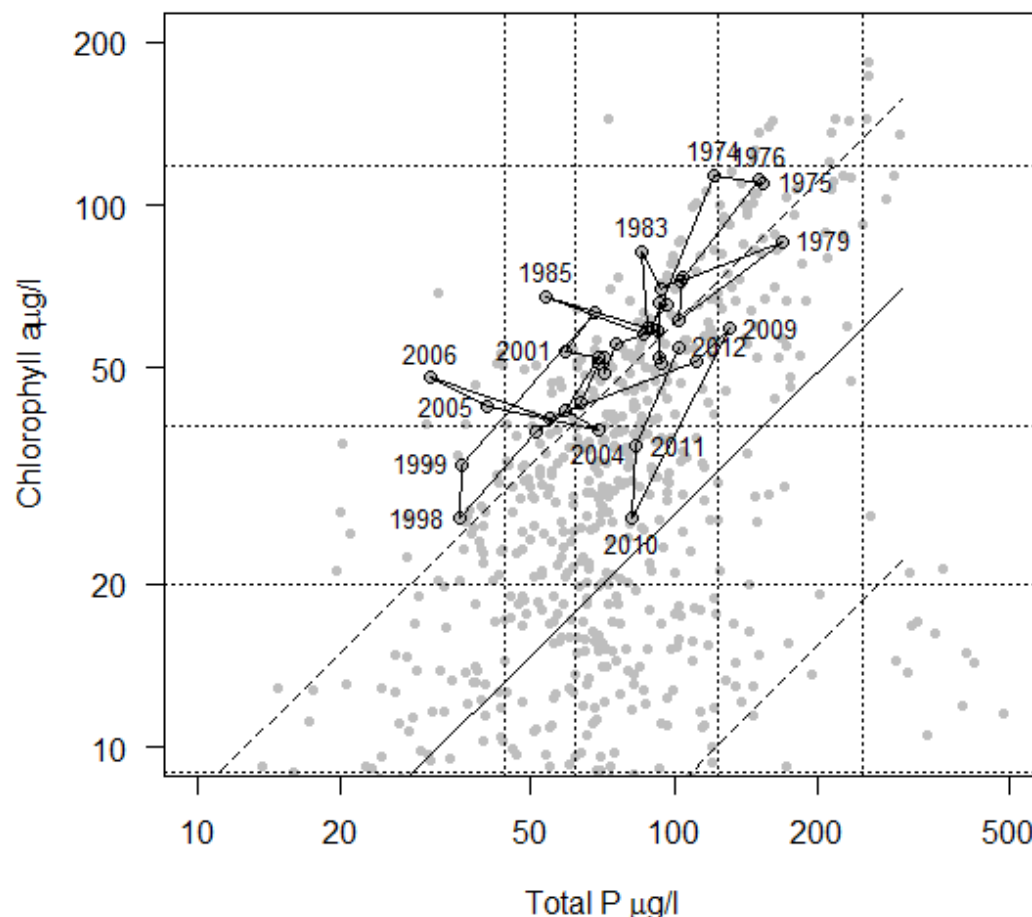**Figure 53 Relationship between annual mean chlorophyll a and total phosphorus, showing trajectory of change in Hickling Broad Grey points mark values for all other broads, horizontal and vertical dotted lines show WFD boundary values, diagonal dotted lines show relationship for European lakes (Phillips et al. 2008).**

11.3 Chlorophyll v Suspended Solids

There is a clear relationship between chlorophyll and suspended solids in the broads taken as a whole. However, in Hickling Broad there are higher levels of suspended solids in comparison to chlorophyll, illustrating the differences in this broad. As discussed above this is either a feature of the relatively shallow water and large exposed area of the lake allowing more wind re-suspension of sediment, or because the summer phytoplankton is dominated by very small cyanobacteria which contain relatively smaller amounts of chlorophyll per unit of biomass.
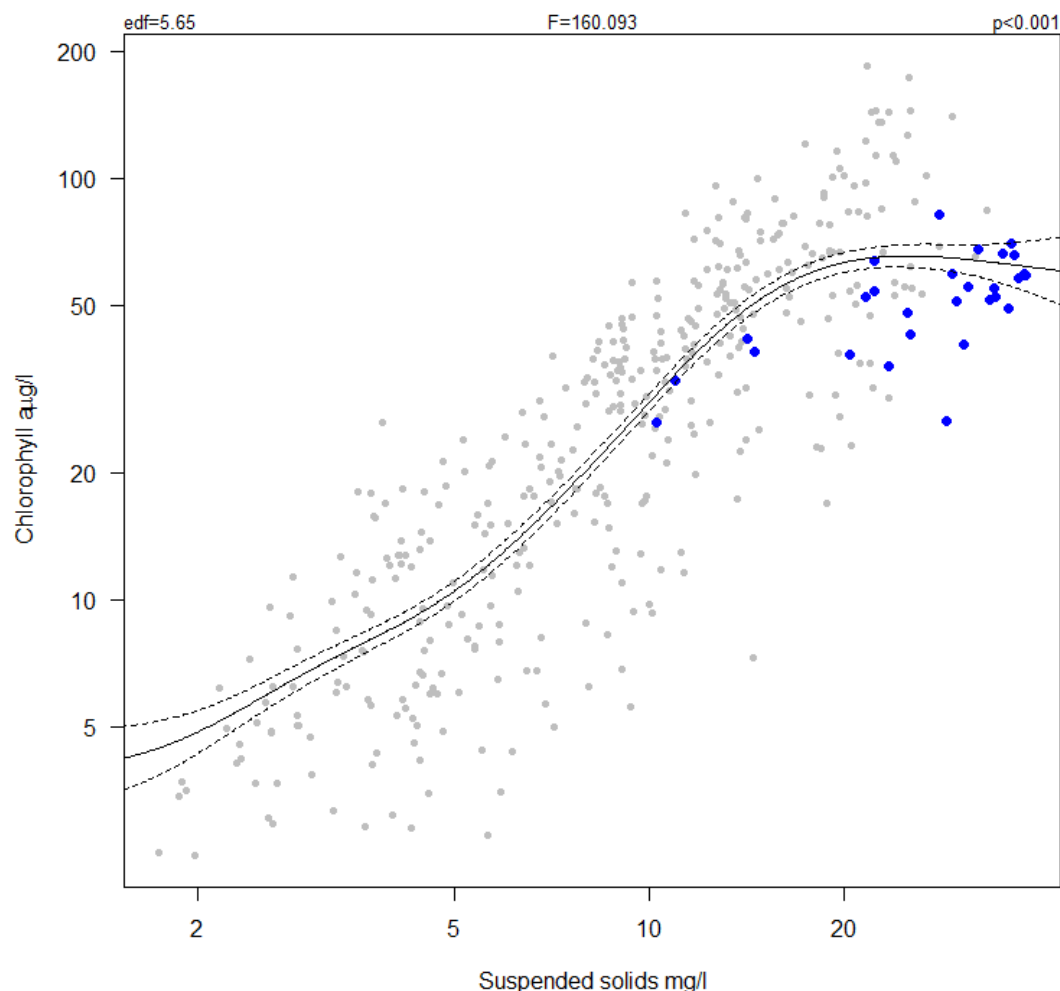
Figure 54 Relationship between annual mean chlorophyll a and suspended solids in Hickling Broad (blue points), grey points mark values for all other broads.

11.4 Evidence of overall system response to environmental factors and management

There have been few significant management actions in Hickling Broad directed at lake restoration and the main question that arises for this lake is what has caused the changes in macrophyte abundance. Given the proximity of the lake to Horsey Mere and the complex tidally mixed system of the Upper Thurne broads it is potentially helpful to compare changes in Hickling with those in Horsey Mere when considering this question.

In Hickling, in contrast to Horsey Mere, there has clearly been a reduction in phosphorus and chlorophyll a since the 1970s (Figure 55a & b) and this is probably largely the result of the decline in the Black headed gull roost, although improvements in phosphorus in the lower River Bure as a result of effluent P removal, may also make a contribution via tidal mixing. However, after allowing for this trend there is a clear relationship between the inter-annual changes in these two lakes with a cross correlation function of the differenced time series being highly significant (R=0.674 p<0.001) at a lag of 1 indicating the strong connectivity of these lakes. The recent rise in phosphorus in Hickling is also seen in Horsey and may thus be caused by changes in the pumped inputs direct to Horsey rather than to Hickling. Chlorophyll shows a similar pattern (Figure 55b) and again has a highly significant cross correlation (R=0.710 p<0.001) at a lag of 1. The pattern of total iron in Hickling is also very closely related to that in Horsey (cross correlation differenced time series R=0.737 p<0.001, lag 0), although the concentration in Hickling is substantially lower (Figure 55c note scale difference) confirming that the source of iron in Hickling is the pumped inputs to Horsey Mere.

Macrophyte cover in Hickling and Horsey also show similar trends, although a cross correlation shows no evidence of positive correlation. Prior to 1993 macrophyte cover in Hickling is dominated by *M. spicatum* and *P. pectinatus* and these are the species that perennially dominate in Horsey Mere where their total cover is slightly higher (Figure 56a) – possibly reflecting lower wind exposure or limited grazing by birds in Horsey compared to Hickling. Both lakes show a dip in cover in the mid-1980s, a period that was markedly colder (Figure 58c) and when total iron peaked in both lakes (Figure 55c). Following this cold period temperature have increased markedly and there was a period of much lower rainfall, indicated by extremely low river discharge (Figure 58b). At this time these macrophytes (*M. spicatum* and *P. pectinatus*) increase again returning to cover values similar to those seen prior to the mid-80s dip (Figure 56a). At the same time in Hickling Broad, but not in Horsey, *Chara* also begins to increase, reaching a massive peak by 1999 before decreasing and virtually disappearing by 2006 (Figure 56b). After 2000 *M. spicatum* and *P. pectinatus* also decline; this starts earlier in Hickling than in Horsey, but the change in overall cover of macrophytes is similar in both lakes (Figure 57a). The similarity in the change in overall macrophyte cover in Hickling and Horsey is striking, in effect cover in Hickling is identical to Horsey, except that from 1995-1999 Hickling has a very substantial biomass of *Chara* (Figure 57a).

During this period *Chara* occupied the bulk of the water column and during this time suspended solids in Hickling were clearly reduced (Figure 56c). Before and after this period suspended solids in Hickling are higher than they are in Horsey. An element of this is probably phytoplankton, as chlorophyll levels are also higher in Hickling (Figure 55a), but the clear dip in suspended solids during the period of *Chara* dominance probably reflects a lack of sediment re-suspension which is likely to be relatively high in this large very shallow and wind-exposed lake.

The cause of these changes in macrophyte cover in Hickling and Horsey are difficult to ascertain. The two species which dominated these lakes, *M. spicatum* and *P. pectinatus*, are relatively insensitive to water clarity as they rapidly form floating

canopies with the bulk of the photosynthetic biomass sitting on or just beneath the water surface. They are thus able to dominate in these lakes despite relatively high phytoplankton biomass (or turbidity from other sources). They grow from turions and tubers and it is suggested that the particularly cold winter of 1985 may have reduced their growth. The increase in *Chara* is associated with a period of marked warming and sunnier summers, occurring at a time when water clarity was improving as phosphorus and chlorophyll in Hickling declined. The presence of well-established stands of *M. spicatum* and *P. pectinatus* may also have been instrumental in creating quiescent conditions close to the bed in some areas that facilitated the initial growth of *Chara*. Comparing the differenced time series of average hours of sun with time series of both *Chara* and the sum of *M. spicatum* + *P. pectinatus*, shows a significant correlation at lag 0 for *Chara* (R= 0.559 p<0.001), but not for the canopy forming species. The explosive growth of *Chara* thus seems to be associated with a combination of conditions, improved water clarity and warmer sunnier summers. What is less clear is why *Chara* did not also increase in Horsey Mere, where chlorophyll and suspended sediment levels are lower and transparency (Figure 59) is higher than in Hickling, although there is a small but not insignificant increase in *Chara* coverage in Horsey during 1999. Possible explanations may be the higher chloride and total iron levels in Horsey (Figure 58a & Figure 55c), as well, presumably, as sulphide in sediments, which may collectively have supressed *Chara* growth. Recent evidence for effects of iron on aquatic plants comes from studies of the potential use of iron as an additive to promote reductions in internal loading through P binding in the sediment (Immers et al., 2013). This suggests differential effects on *Chara* species (e.g. *C. globularis* more sensitive than *C. virgata*) and greater impacts on the strongly rooted *Potamogeton pectinatus* compared with the weakly and facultatively rooted *Elodea nuttallii*. Van der Welle et al. (2007) also discuss possible impacts of sulphide on rooted macrophytes which is potentially relevant to the *Chara* dieback.

Similarly, it is not clear what caused the decline of all the macrophytes in both of these lakes from 2000. Temperature remains high, so cold winters are unlikely to be an explanation. Chlorophyll remains constant, although there is a decline in transparency in Hickling. Similarly suspended solids, total iron and total phosphorus all increase, but only from 2005 which is after the decline of the macrophytes. It thus seems unlikely that these changes directly caused the change in macrophytes and may even be a symptom of the macrophyte loss.

Bird grazing may offer a possible explanation. Bird numbers increased markedly during and after the peak of *Chara* and were generally high from 1998 to 2008 with the composition shifting from coot to teal dominance over this period. Unfortunately gaps in bird count records in the mid-1990s make statistical analysis difficult but comparing GAM smoothers suggests that bird grazing during the winter increased substantially and extends into the period of macrophyte decline (Figure 57c). Bird numbers were also particularly low in the early 1990s when the growth of *Chara* started.

It is possible that other environmental factors for which we do not have direct measurements are somehow implicated in the macrophyte decline. The last decade

or so has, on average, been mild and wet which together suggest the likelihood of stronger winds. This is likely to increase hydraulic stress on plants and promote sediment re-suspension. Barker et al. (2008) also investigated the role of salinity as a possible driver of plant collapse in Hickling through mesocosm studies, concluding that at ambient or higher salinity P release from sediment was increased and overall system sensitivity to other factors was heightened, although salinity per se was not considered a problem. Salinity has remained almost stable in Hickling for at least the last 30 years and is only 5-10% higher than in the Martham Broads which have supported a highly diverse and aquatic flora throughout the period of BA monitoring, albeit at significantly lower TP concentrations.

In summary it is difficult to provide a convincing explanation of the changes in macrophyte abundance in Hickling Broad. In general, the macrophytes that have dominated the lake are relatively insensitive to chlorophyll as they have the bulk of their biomass in the upper water column or on the surface and thus competition with phytoplankton for light may be relatively unimportant. The increase in *Chara* seems likely to be a coincidence of warmer sunnier summers and its decline at least partly a result of bird grazing. The *Chara* may have displaced the less light-sensitive macrophyte species but the overall decline in all macrophytes in both Horsey and Hickling cannot be satisfactorily explained.
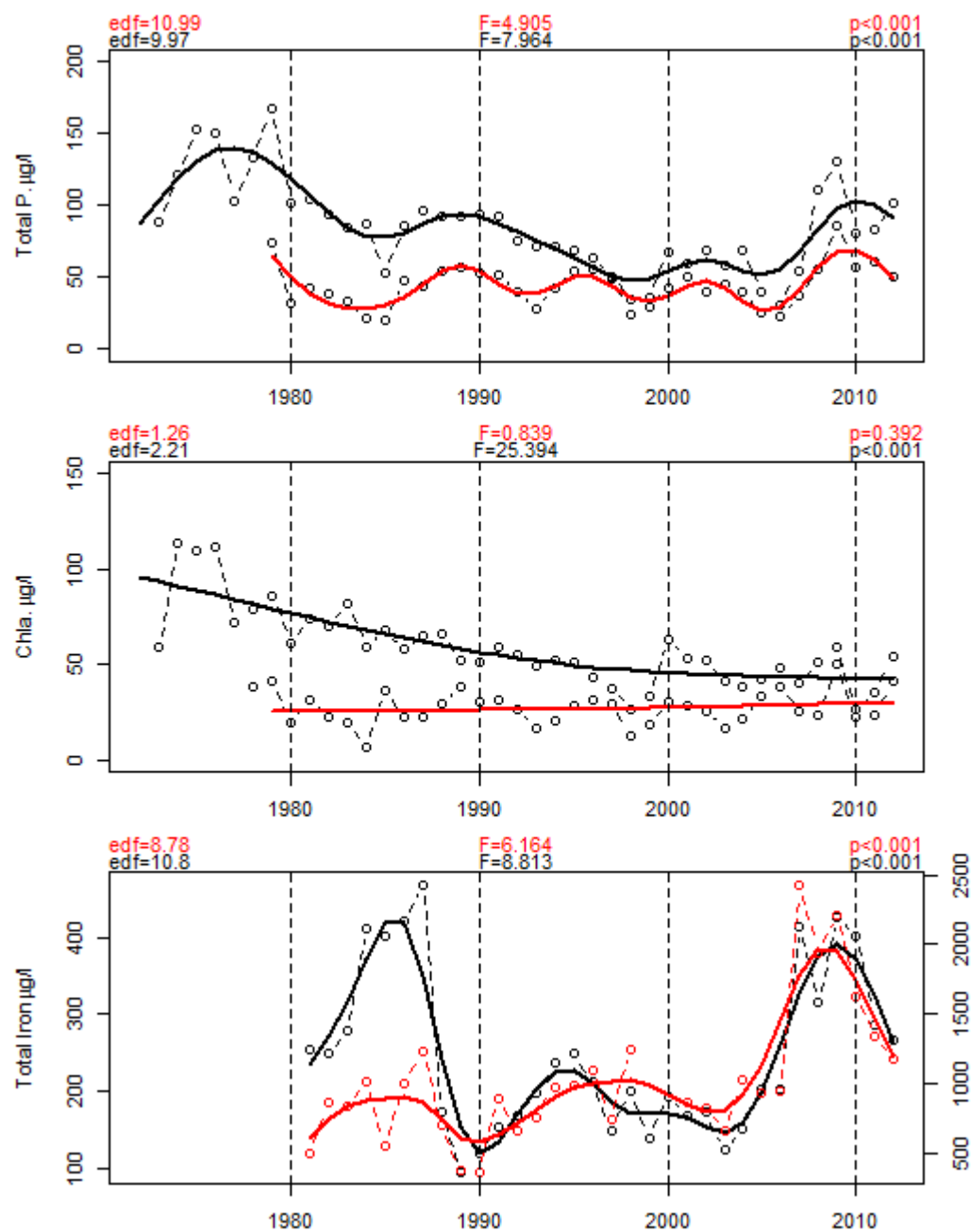
Figure 55 Time series of annual means for Hickling Broad (black) and Horsey Mere (red), a) total phosphorus, b) chlorophyll a, c) total iron (Horsey scale on right axis)
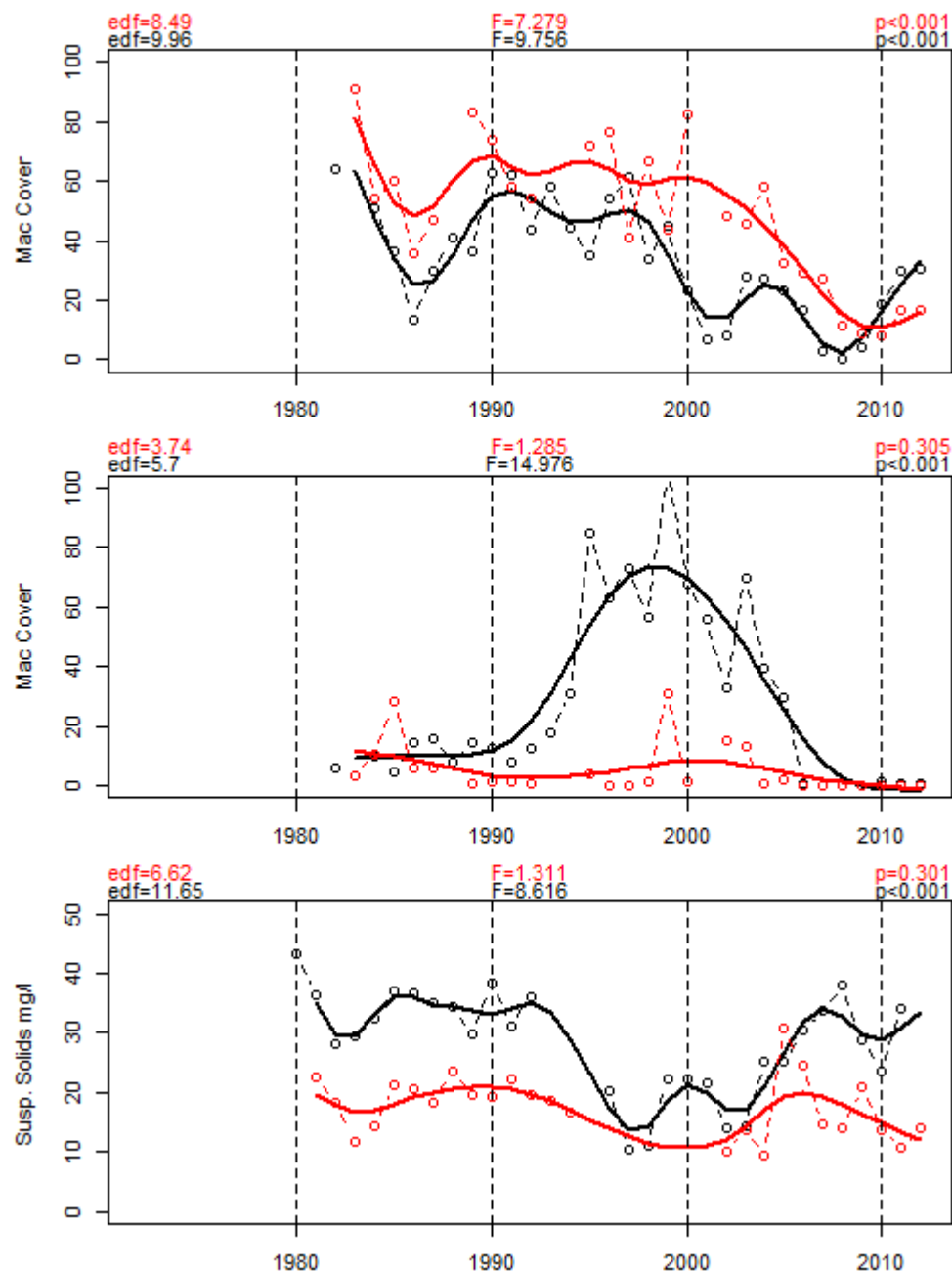
Figure 56 Time series of annual means for Hickling Broad (black) and Horsey Mere (red), a) cover of canopy forming macrophytes (*P. pectinatus + M. spicatum*), b) cover of *Chara spp + N. marina*, c) total suspended solids
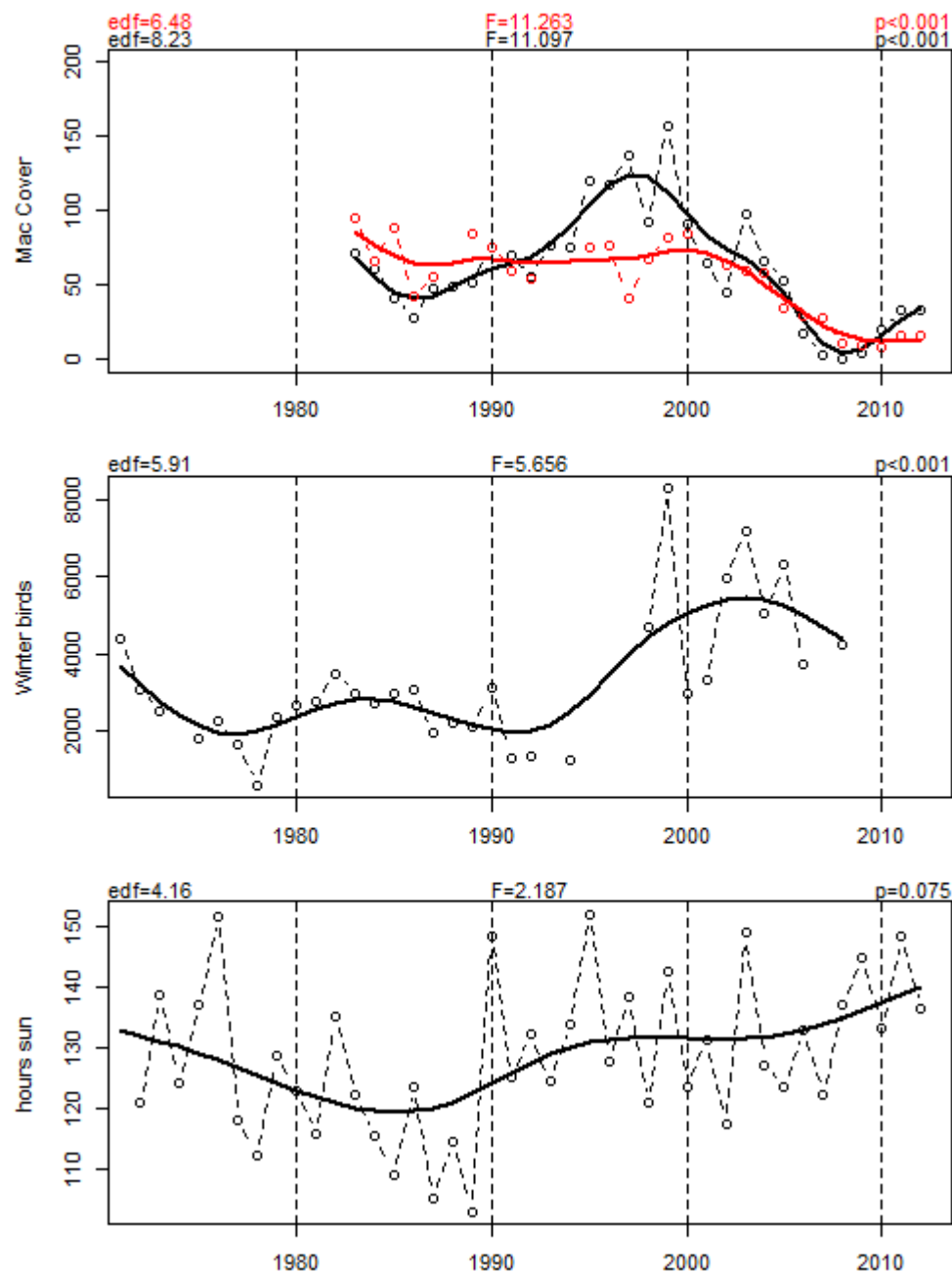
Figure 57 Time series of a) annual mean macrophyte cover in Hickling (black) and Horsey Mere (Red) compared with b) number of birds on Hickling during winter and c) annual mean hours of sunshine (Lowestoft)
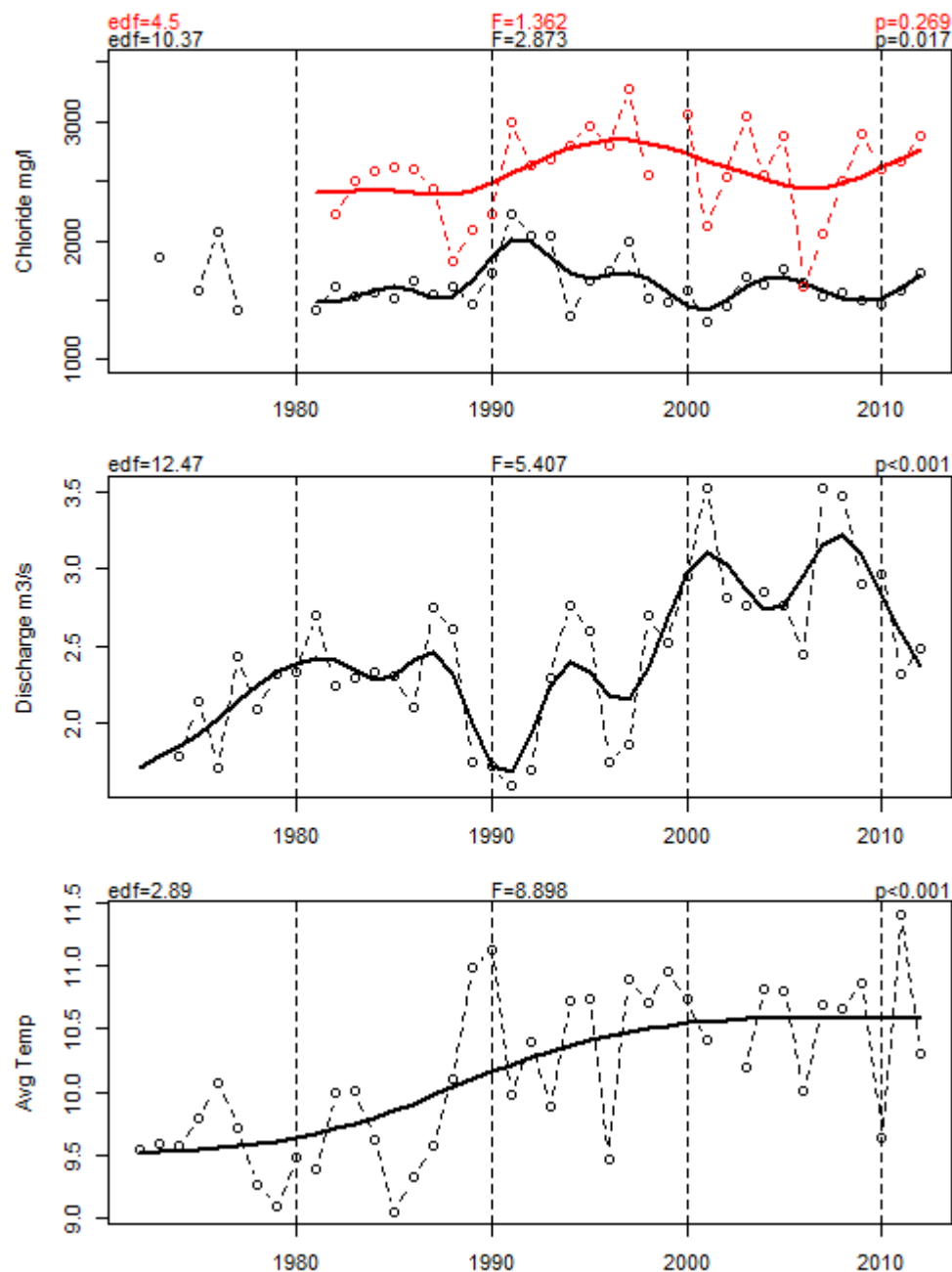
Figure 58 Time series of annual mean for Hickling Broad (black) and Horsey Mere (red), a) chloride and b) river discharge (R Bure Horstead Mill), c) average air temperature (Lowestoft)
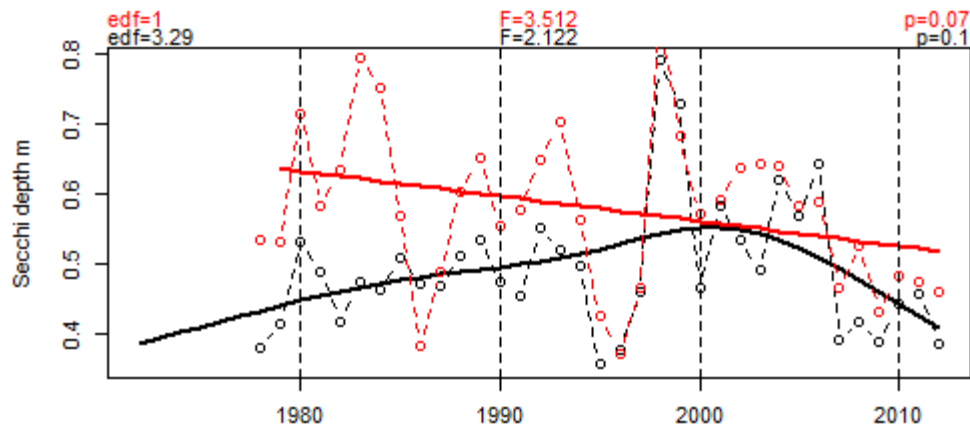
Figure 59 Time series of annual mean transparency as Secchi depth in Hickling (black) and Horsey (red)

11.5 Future management options

In considering management options for Hickling several points need to be borne in mind:

- Hickling cannot be viewed in isolation and is highly responsive to management and agricultural usage within the catchment of Horsey Mere.
- External factors which cannot be controlled, such as weather conditions and bird numbers, are likely to influence the effectiveness of any management activities.
- Macrophytes respond to but also promote changes in environmental variables so underlying mechanisms and causality can prove hard to establish.
- Although our analysis points to several factors or combinations of factors that have probably contributed to the decline of vegetation in Hickling this cannot be explained with any certainty and consequently confidence in the effectiveness of any particular form of management is low.

Three main groups of options can be identified, none of which should be considered exclusive.

- Extensification of agriculture on the Horsey/Brograve levels through conversion of existing arable land to pasture. This would need to be accompanied by a resumption of shallow drainage and acceptance of higher ditch water levels. Direct benefits are likely to both Horsey and Hickling through reduced iron, phosphorus and salinity inputs. Larger scale implications for agricultural subsidy.
- Source control, possibly accompanied by increased freshwater input from the Catfield catchment. This would reduce P inputs and increase flushing and dilution, although there have been perennial problems with abstraction on Catfield fen so there is there is little realistic prospect of sourcing additional freshwater inputs via this route.

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- Sediment removal. Purely from a nutrient-reduction perspective sediment removal is unlikely to be especially effective as Hickling sediment appears likely to retain little P under current levels of salinity. Bed stabilisation, propagule bank exposure and increased water depth may all benefit macrophytes directly, or through improvements in the submerged light climate. It is perhaps the opportunities for bank reclamation and creation of hydraulic refugia and habitat complexity that offer the most compelling ecological arguments for sediment removal in Hickling. Plants in Hickling are currently confined to sheltered bays and margins implying a strong hydraulic control on their distribution. Vegetation would be only one of various potential beneficiaries of imaginative deployment of dredged material if a truly visionary scheme in line with the Dutch Marker Wadden were to be adopted. It is strongly recommended that trials of sediment removal to different designs and standards are undertaken on an experimental basis before a broad-wide programme is embarked upon. This should include comparison of shallow sediment removal over an area of several hectares, versus pit or trench dredging as a means of refocusing adjacent sediment without direct physical disturbance. Table 5 examines various well-rehearsed arguments in relation to sediment removal and their specific relevance to Hickling Broad.

Table 5. Review of potential support for a sediment removal programme and its relevance to Hickling Broad.

Function	Support for dredging	Comment
Reduction of internal loading	Low	Non-retentive sediment due to competitive binding of iron by sulphide. Therefore internal loading is naturally limited.
Increased water depth	Low	Hickling is shallow and turbid (unless dominated by plants). Deepening it is unlikely to improve the submerged light climate unless there is an accompanying equivalent reduction in turbidity. Current dominant species are rhizomatous and independent of light regime but could be reduced unless dredging avoids existing beds.
Bed stabilisation	Moderate	Wind-induced re-suspension of sediment is likely to be a common source of turbidity. Increasing depth by removing fine sediment should increase clarity. Hickling sediment is, however, already comparatively cohesive and unlikely to limit macrophyte establishment.
Propagule bank exposure	Moderate	Hickling has historically been dominated by charophytes which form a large persistent oospore bank and would be likely to benefit from removal of soft sediment
Bank reclamation	High	Dredged material offers the opportunity to reclaim and restore sections of eroded bank, especially in areas of reed dieback and goose grazing. Potential benefits to macrophytes through increased shoreline complexity and reduced wave reflection from steep eroded banks.
Contaminant removal	Unknown	Sediment removal offers the opportunity to reduce the concentration of potentially phytotoxic metals (iron, copper) and sulphide.
Creation of hydraulic refugia	High	Hickling is large and wind-exposed. Macrophytes are currently confined to sheltered bays. Imaginative used of dredged material to create bunds or islands could significantly increase hydraulic refugia.

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