Hickling Broad Dossier
(1980 -2024)
Part of the review of lake restoration practices and their performance in the Broads National Park, 1980-2013

Produced by the Broads Authority

September 2025

The study was a Broads Authority and Natural England initiative, undertaken by Geoff Phillips, Helen Bennion, Martin Perrow, Carl Sayer, Bryan Spears and Nigel Willby, funded by a Broads Authority and Natural England with additional contributions from Environment Agency, Essex & Suffolk Water and Norfolk Wildlife Trust.

Update Note (2025)
In 2025, selected sections of this report were updated by Geoff Phillips, incorporating additional data spanning the period 2014–2025. These updates apply to:

- **Section 5**: Water Quality
- **Section 7**: Macrophytes
- **Section 10**: Aquatic Birds
- **Section 11**: Summary of Interactions

While other sections remain unchanged, a revised overview of the major ecological interactions at Hickling Broad has also been included (Section 12). This update was made with support of the **Norfolk Wildlife Trust** as landowners

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Contents

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.

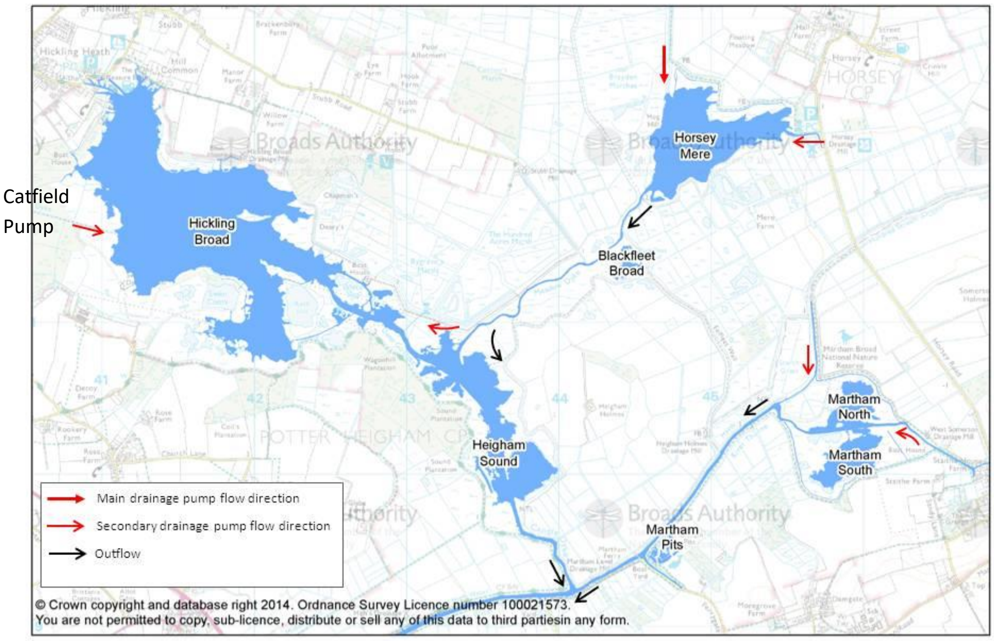

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Figure 1 Geographical context of Hickling Broad

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.

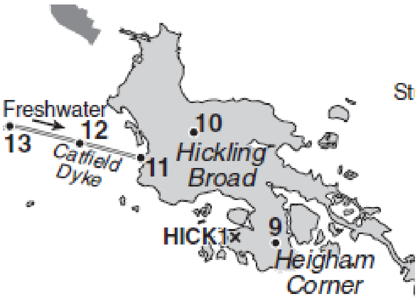


Figure 2 Location of the HICK1 sediment core.

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).

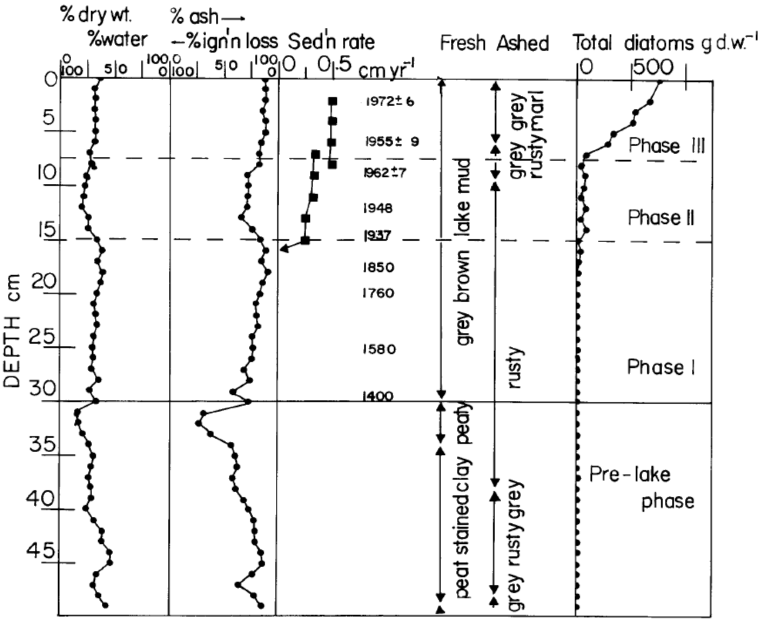


Figure 3 Physical characteristics, sedimentation rates and diatom content of the core taken by Moss (1978). Dates are deduced from ^{210}Pb datings.

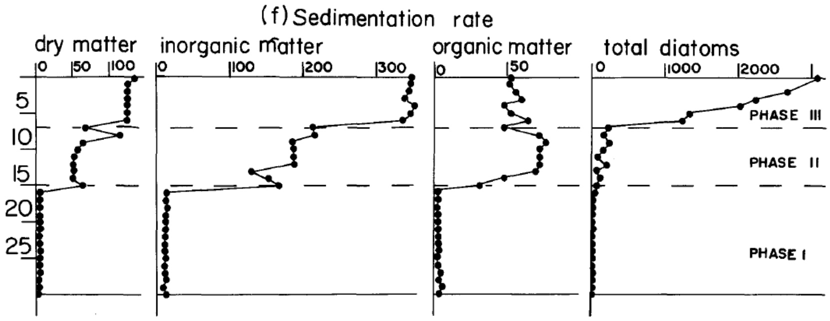


Figure 4 Sedimentation rates of dry, inorganic and organic matter and of total diatoms. Figures are relative and were derived by multiplication of the sedimentation rate (mm yr-1) at the time by, respectively, the percentage ash in dried sediment, the percentage loss on ignition at 500°C from dried sediment, 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).

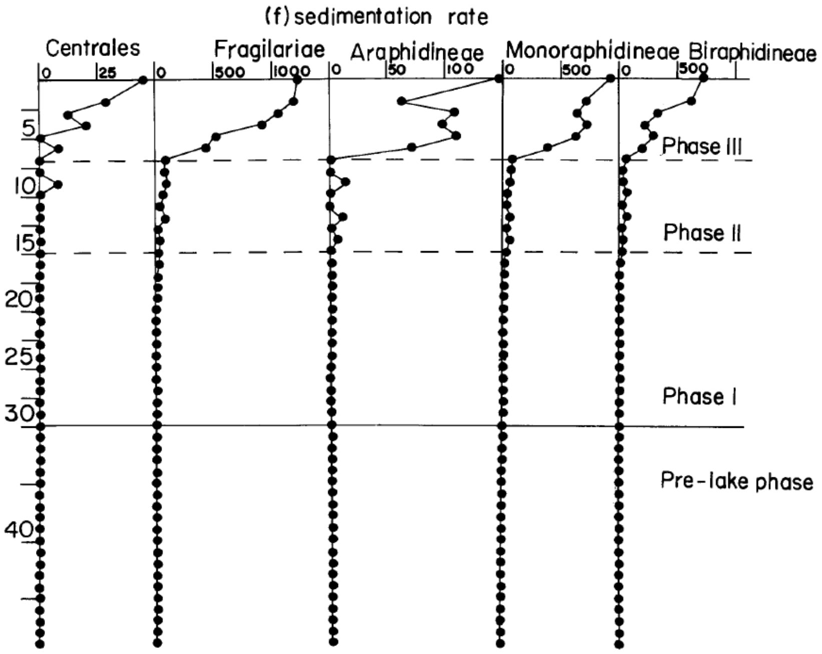


Figure 5 Sedimentation rates (relative values) of major groups of diatoms in the core. Data for the pre-lake phase assume a similar sedimentation rate to that during phase 1 in the absence of definite dates for these layers (from Moss, 1978).

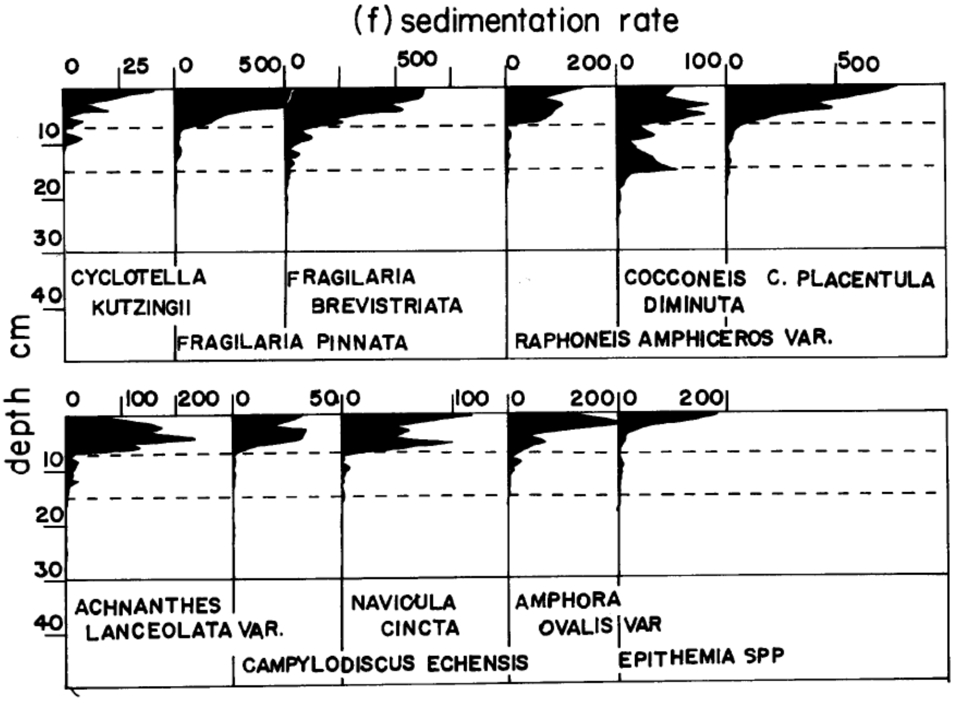


Figure 6 Sedimentation rates (relative values) for Cyclotella kutzingiana Thr. and the most common diatoms in the core – Fragilaria pinnata Ehr., F. brevistriata Grun., Raphoneis amphiceros var rhomboides Cl., Cocconeis diminuta Pant., C. placentula (Ehr.), Achnanthes lanceolata var elliptica Cl., Campylodiscus echensis Her., Navicula cincta Cl., Amphora ovalis var. pediculus Kutz. and Epithemia spp. (from Moss, 1978).

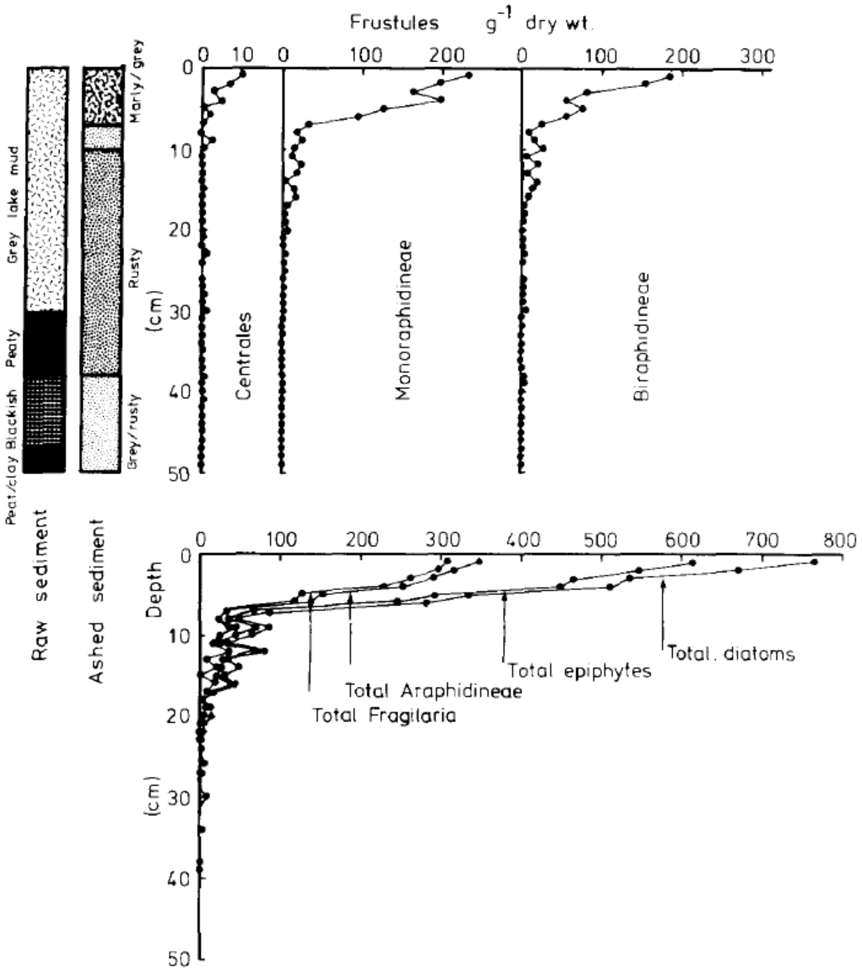
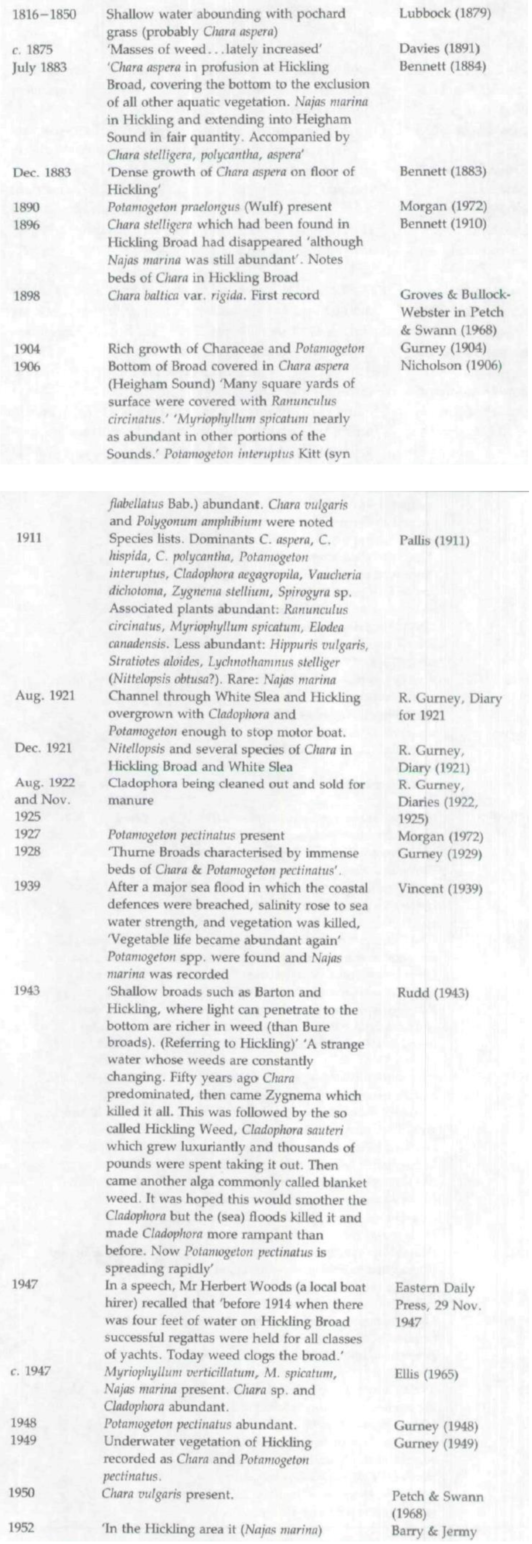


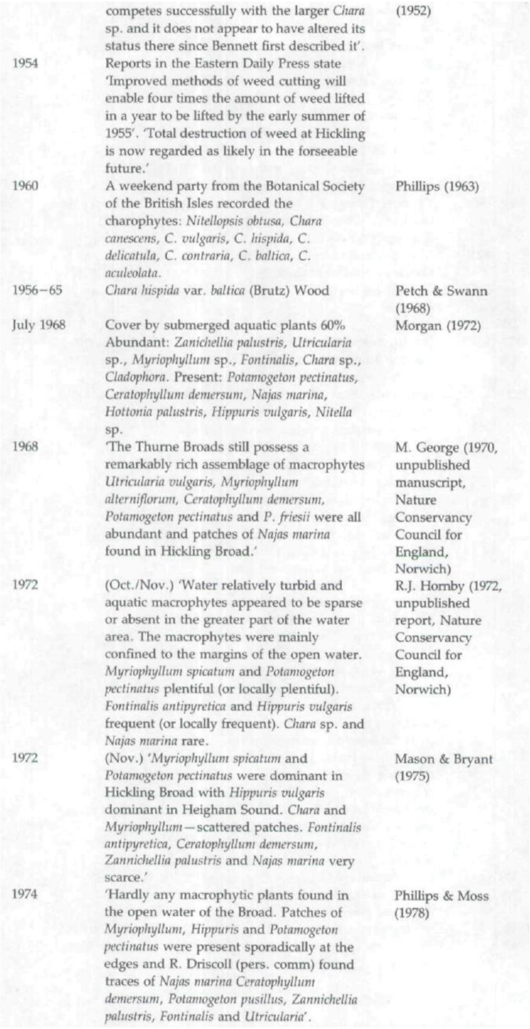
Figure 7 Changes in diatom numbers in a sediment core collected from Hickling Broad in the late 1970s. Numbers are given per unit dry weight of sediment (taken from Phillips et al. 1978).

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).



1816–1850	Shallow water abounding with pochard grass (probably *Chara aspera*)	Lubbock (1879)
c. 1875	'Masses of weed…lately increased'	Davies (1891)
July 1883	'*Chara aspera* in profusion at Hickling Broad, covering the bottom to the exclusion of all other aquatic vegetation. *Najas marina* in Hickling and extending into Heigham Sound in fair quantity. Accompanied by *Chara stelligera, polycantha, aspera*'	Bennett (1884)
Dec. 1883	'Dense growth of *Chara aspera* on floor of Hickling'	Bennett (1883)
1890	*Potamogeton praelongus* (Wulf) present	Morgan (1972)
1896	*Chara stelligera* which had been found in Hickling Broad had disappeared 'although *Najas marina* was still abundant'. Notes beds of *Chara* in Hickling Broad	Bennett (1910)
1898	*Chara baltica* var. *rigida*. First record	Groves & Bullock-Webster in Petch & Swann (1968)
1904	Rich growth of Characeae and *Potamogeton*	Gurney (1904)
1906	Bottom of Broad covered in *Chara aspera* (Heigham Sound) 'Many square yards of surface were covered with *Ranunculus circinatus*.' '*Myriophyllum spicatum* nearly as abundant in other portions of the Sounds.' *Potamogeton interuptus* Kitt (syn *flabellatus* Bab.) abundant. *Chara vulgaris* and *Polygonum amphibium* were noted	Nicholson (1906)
1911	Species lists. Dominants *C. aspera, C. hispida, C. polycantha, Potamogeton interuptus, Cladophora aegagropila, Vaucheria dichotoma, Zygnema stellium, Spirogyra* sp. Associated plants abundant: *Ranunculus circinatus, Myriophyllum spicatum, Elodea canadensis*. Less abundant: *Hippuris vulgaris, Stratiotes aloides, Lychnothamnus stelliger* (*Nittelopsis obtusa*?). Rare: *Najas marina*	Pallis (1911)
Aug. 1921	Channel through White Slea and Hickling overgrown with *Cladophora* and *Potamogeton* enough to stop motor boat.	R. Gurney, Diary for 1921
Dec. 1921	*Nitellopsis* and several species of *Chara* in Hickling Broad and White Slea	R. Gurney, Diary (1921)
Aug. 1922 and Nov. 1925	Cladophora being cleaned out and sold for manure	R. Gurney, Diaries (1922, 1925)
1927	*Potamogeton pectinatus* present	Morgan (1972)
1928	'Thurne Broads characterised by immense beds of *Chara* & *Potamogeton pectinatus*'.	Gurney (1929)
1939	After a major sea flood in which the coastal defences were breached, salinity rose to sea water strength, and vegetation was killed, 'Vegetable life became abundant again' *Potamogeton* spp. were found and *Najas marina* was recorded	Vincent (1939)
1943	'Shallow broads such as Barton and Hickling, where light can penetrate to the bottom are richer in weed (than Bure broads). (Referring to Hickling)' 'A strange water whose weeds are constantly changing. Fifty years ago *Chara* predominated, then came Zygnema which killed it all. This was followed by the so called Hickling Weed, *Cladophora sauteri* which grew luxuriantly and thousands of pounds were spent taking it out. Then came another alga commonly called blanket weed. It was hoped this would smother the *Cladophora* but the (sea) floods killed it and made *Cladophora* more rampant than before. Now *Potamogeton pectinatus* is spreading rapidly'	Rudd (1943)
1947	In a speech, Mr Herbert Woods (a local boat hirer) recalled that 'before 1914 when there was four feet of water on Hickling Broad successful regattas were held for all classes of yachts. Today weed clogs the broad.'	Eastern Daily Press, 29 Nov. 1947
c. 1947	*Myriophyllum verticillatum, M. spicatum, Najas marina* present. *Chara* sp. and *Cladophora* abundant.	Ellis (1965)
1948	*Potamogeton pectinatus* abundant.	Gurney (1948)
1949	Underwater vegetation of Hickling recorded as *Chara* and *Potamogeton pectinatus*.	Gurney (1949)
1950	*Chara vulgaris* present.	Petch & Swann (1968)
1952	'In the Hickling area it (*Najas marina*)	Barry & Jermy



	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 forseeable future.'	
1960	A weekend party from the Botanical Society of the British Isles recorded the charophytes: *Nitellopsis obtusa, Chara canescens, C. vulgaris, C. hispida, C. delicatula, C. contraria, C. baltica, C. aculeolata.*	Phillips (1963)
1956–65	*Chara hispida* var. *baltica* (Brutz) Wood	Petch & Swann (1968)
July 1968	Cover by submerged aquatic plants 60% Abundant: *Zanichellia 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)

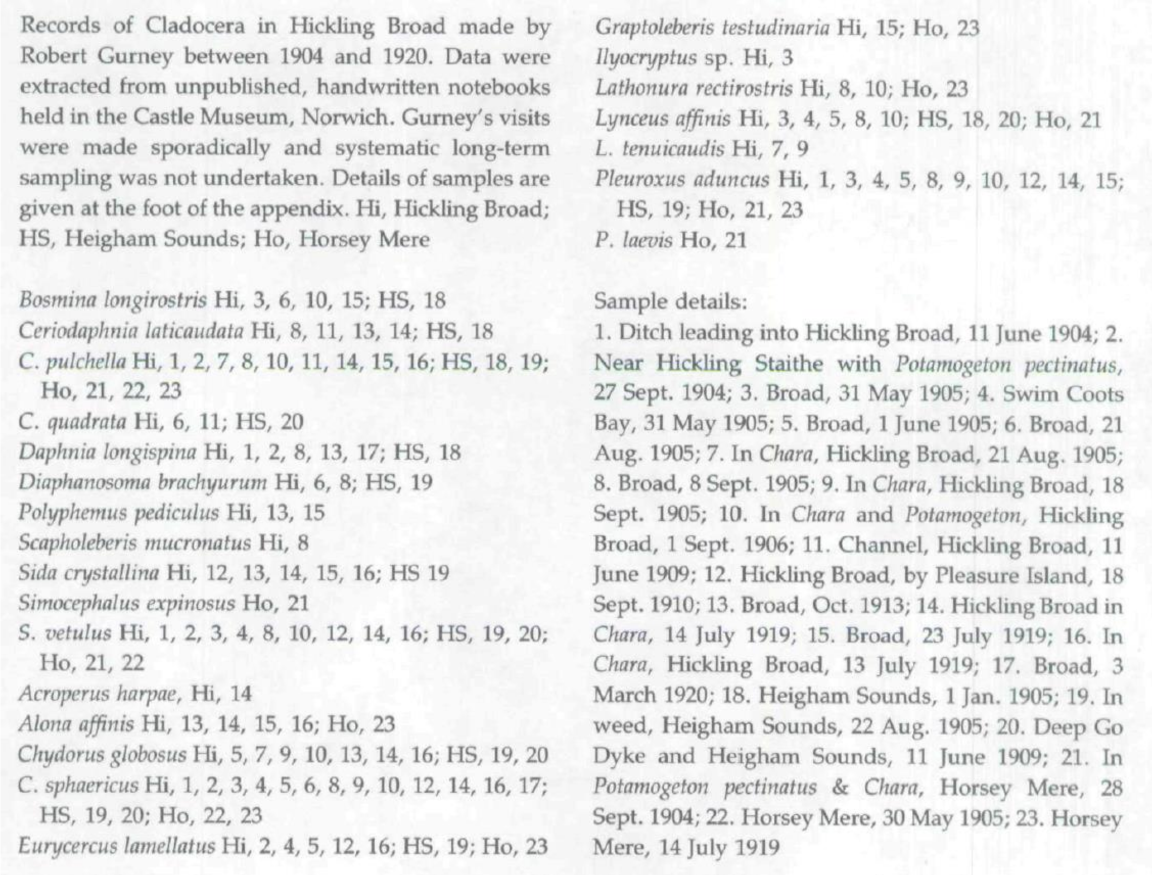
There are numerous historic records for submerged and floating-leaved plants in Hickling Broad. Table 1, extracted from Bales *et al.* (1993), details anecdotal evidence of aquatic vegetation in Hickling Broad and Heigham Sound from the 1800s to 1970s. Most information is not quantitative and surveys 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 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.

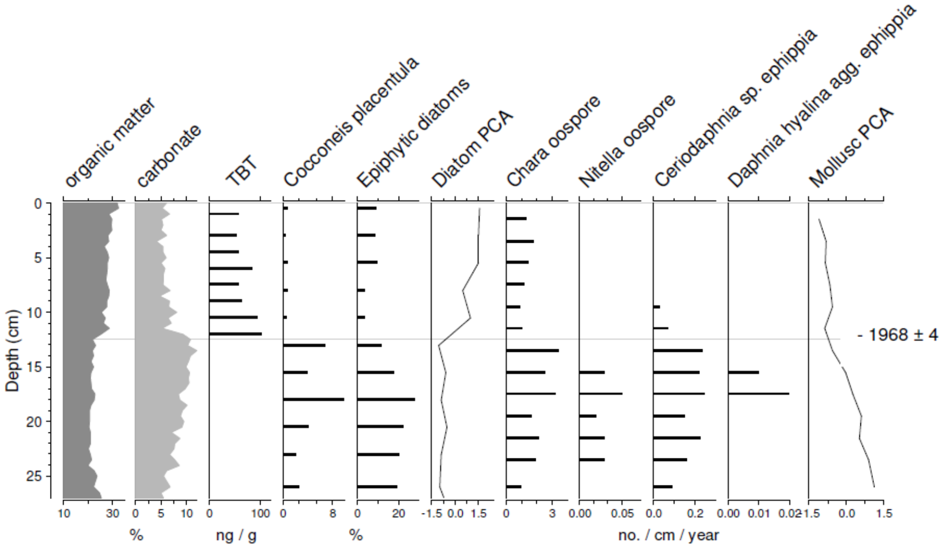


Figure 8 Summary of HICK1 palaeolimnological results (taken from Hoare, 2007).

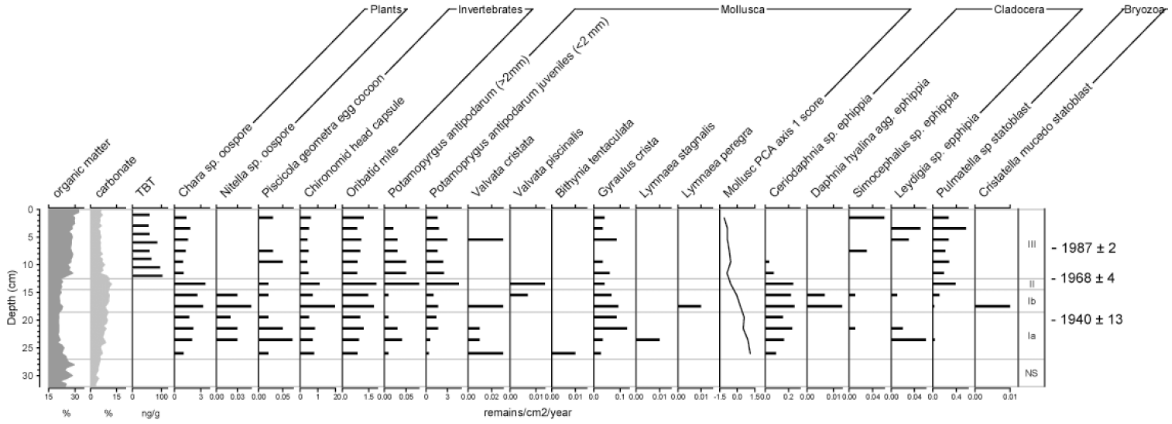


Figure 9 Macrofossil remains from core HICK1 (taken from Hoare, 2007).

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.

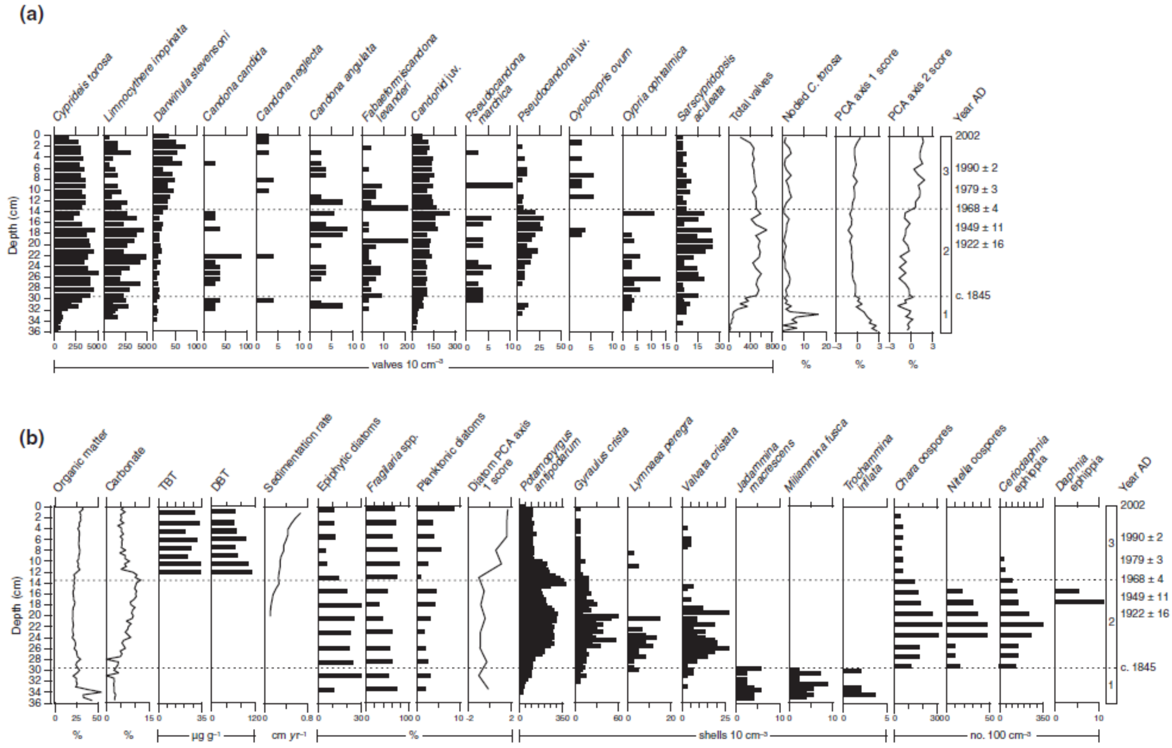


Figure 10 Palaeolimnological data from core HICK1; (a) ostracod faunal assemblages; (b) physical sedimentology, toxic pollutant and other palaeoecological data. Fragilaria spp. includes several species within the Staurosira-Staurositella-Pseudostaurosira complex. TBT and DBT are tributyltin and dibutyltin, 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 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
Winter 2015/16*	Dredging of Hickling marked channel (3,400 m3) with sediment to Duck Broad constructed reedbed
Winter 2016/17*	Dredging of marked channel and restoration of reedbed at Churchill's Bay and Studio Bay (5,100 m3)
Summer 2017*	Experimental cut of stoneworts in 10 m x 10 m plots on the western side of the Broad, with prior and subsequent monitoring for biomass, height and species, with monitoring repeated in 2018 and 2019.
Winter 2017/18*	Dredging (mud pumping) of marked channel towards North Bay with sediment to arable land (4,900 m3)
Winter 2018/19*	CANAPE project Year 1 – construction of 1 ha constructed reed swamp lagoon using geotextile tubes filled with dredged sediment from the marked channel and the approach to Catfield Dyke (5,800 m3)
Winter 2019/20*	CANAPE project Year 2 – filling the Chara Bay lagoon with dredged sediment from the marked channel and Catfield Dyke (9,800 m3)
Winter 2020/21*	CANAPE project Year 3 – filling the Chara Bay lagoon with dredged sediment from the marked channel and Catfield Dyke (11,900 m3)
Winter 2021/22*	Additional filling of the Chara Bay and Churchill's Bay lagoons with dredged sediment from River Thurne and Heigham Sound

5 Water Quality

Data prior to 1980 taken from UEA studies, other data from Environment Agency, updated from EA WQ Archive https://environment.data.gov.uk/water-quality/view/download 20 May 2025.

5.1 Total Phosphorus

In the previous edition of this dossier, it was reported that total phosphorus in Hickling Broad had reduced significantly (Mann-Kendall $\tau = -0.486$, p 0) since the late 1970s, with a median concentration for the last decade of 69 µg P L^{-1}— similar to several other broads in the Bure catchment, but higher than those in the Thurne system. At that time the broad was classified as having Moderate phosphorus status under the Water Framework Directive (WFD). Since then, phosphorus concentrations have declined further, and in 2022 the site was classified as having Good status (Cycle 3 WFD, with total phosphorus falling below the 60 µg P L^{-1} threshold. TP levels are now comparable to other broads in the Thurne system (Figure 11).

However, this improvement has not been continuous: there was a marked increase in TP concentrations in the early 2000s, followed by a decline after 2012. This rise was not observed in other broads in the Thurne system (Figure 11), and as it coincided with a decline in submerged vegetation in the broad (see Section 7), it is very likely that these events are related.

The link with plant growth is further reinforced when the seasonal pattern of TP in Hickling Broad is considered. Like many very shallow lakes, maximum values typically occur in July–August (Figure 12a), driven by the release of soluble phosphorus from sediment or sediment resuspension due to wind. However, this summer peak was much less pronounced during periods of substantial macrophyte growth from 1995 to 2005, and again after 2012.

The overall decline of phosphorus is likely to be a result of the disappearance of a very large black-headed gull roost (Bales et al. 1993). However TP concentration in Catfield pump, which discharges into Hickling Broad, has also declined. While this could indicate a reduced phosphorus load, the fluctuations in TP concentration at Catfield Pump are negatively correlated with rainfall ($r2 = 0.36$ $p<0.001$), suggesting that the reduction is due to increased dilution rather than decreased input.

The short term increase in TP in Hickling (2005 – 2010) mirrors a rise in both total iron (Figure 24) and suspended solids (Figure 25), indicating that some of the observed phosphorus increase may be associated with elevated levels of suspended sediment. This, in turn, may reflect enhanced wind-induced sediment resuspension following the loss of macrophyte cover.

The relationship between changes in plant growth, TP, and other parameters will be explored in more detail in Section 11. Current evidence suggests that, while the long-term decline in TP likely supported the re-establishment of submerged vegetation, more recent changes in plant cover are more likely to have driven the observed shifts in TP concentrations, rather than the reverse.

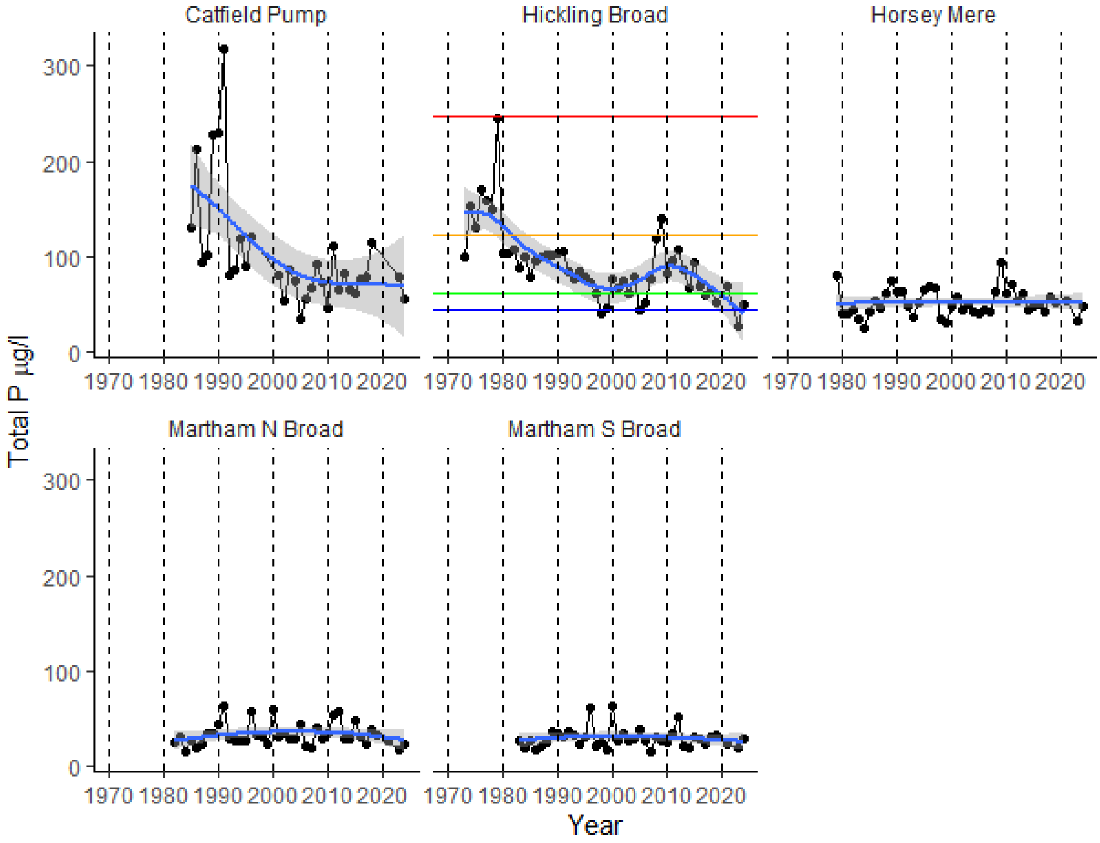


Figure 11. Trend in annual mean TP for a) Catfield Pump, b Hickling Broad, c) Horsey Mere, d) Martham N Broad, e) Martham S Broad. Line showing GAM smooth highlighting long-term trend. Horizontal lines mark WFD boundary values for Hickling Broad (HG = 44,GM = 62, MP = 123,PB = 247 µg/L)

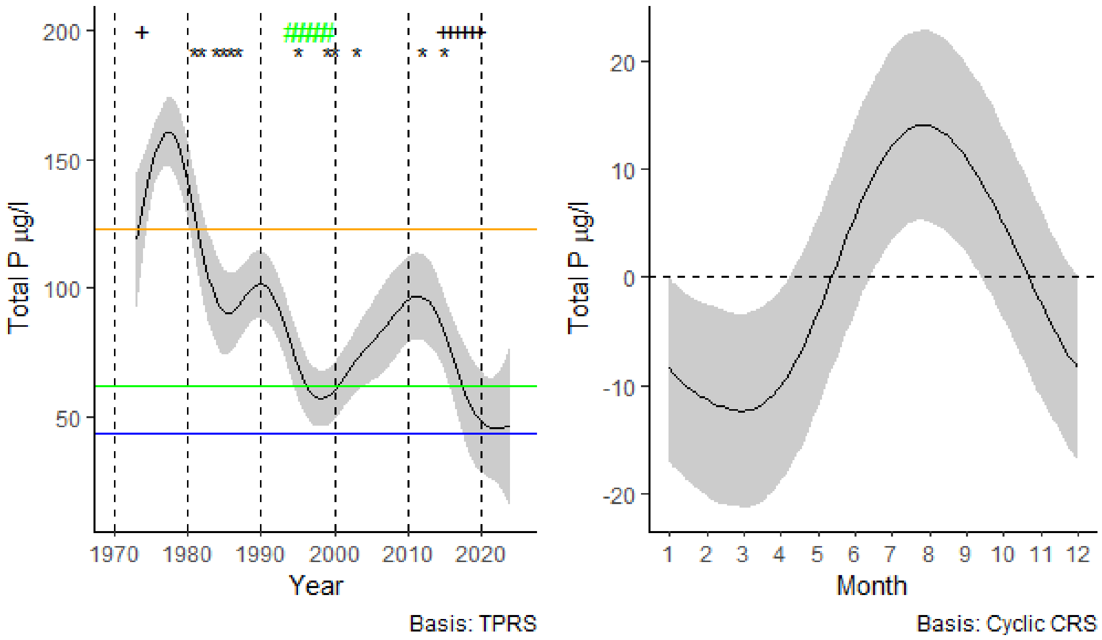


*Figure 12. GAM smooths showing a) predicted long-term and b) seasonal change of total phosphorus in Hickling Broad. (shading shows the confidence limits). * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging. Horizontal lines mark WFD boundary values for Hickling Broad (HG = 44,GM = 62, MP = 123 µg/L)*

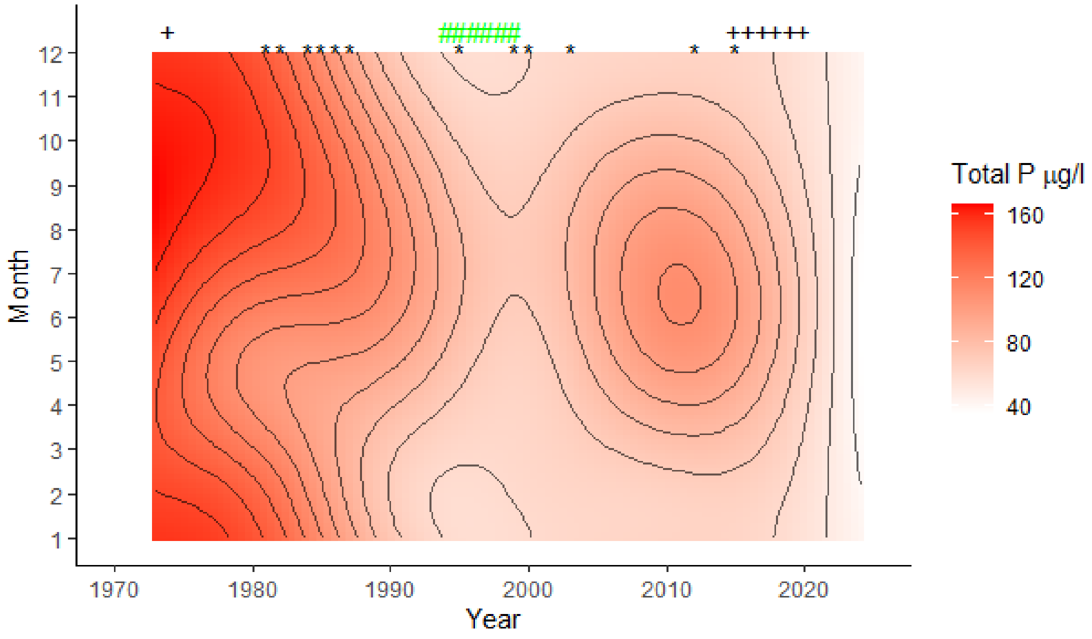


*Figure 12c. Surface plots showing predicted change in TP concentration in Hickling Broad using a GAM model incorporating an interaction between year and month as 2D contour. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

5.2 Soluble Phosphorus

Soluble Reactive Phosphorus (SRP) has always been extremely low in Hickling Broad (Figure 15). There has been a decline in concentration since the 1970s but with no significant trend since 2000 (Mann-Kendall τ = -0.0671937 *p* = 0.336). Concentrations are now similar to other broads in the River Thurne system.

There is no seasonal variation of SRP (Figure 14b & 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.

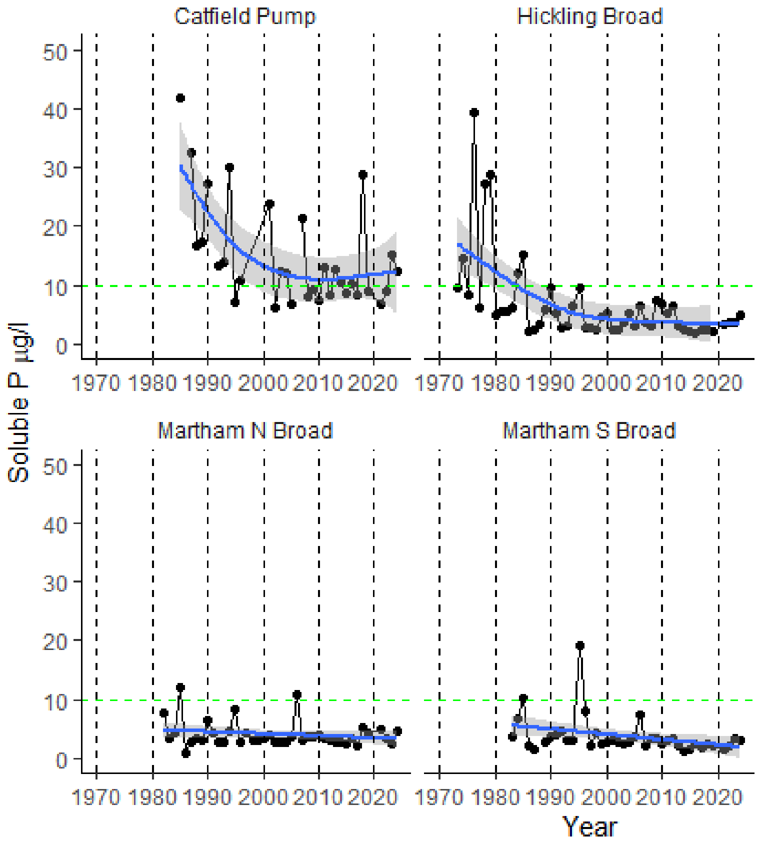


Figure 13. Trend in annual mean soluble phosphorus for a) Catfield Pump, b Hickling Broad, c) Horsey Mere, d) Martham N Broad, e) Martham S Broad. Line showing GAM smooth highlighting long-term trend. Horizontal green line marks 10 ug/L

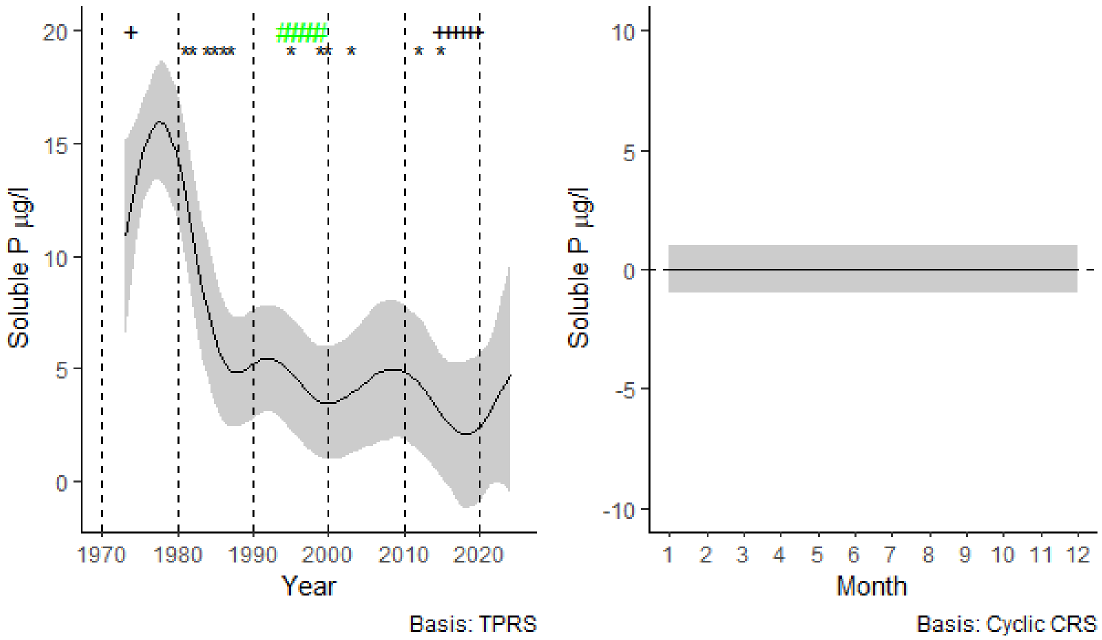


*Figure 14. GAM smooths showing a) predicted long-term and b) seasonal change of soluble phosphorus in Hickling Broad. (shading shows the confidence limits). * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

5.3 Total oxidised nitrogen (nitrate + nitrite)

In common with many broads, including the other Thurne broads, there has been a highly significant decline in TON (Nitrate + Nitrite nitrogen) concentration in Hickling Broad since 1982 (Mann-Kendall τ = -0.45 *p* = <0.001) (Figure 15). Concentration in Catfield Pump are substantially higher but have similarly declined (Mann-Kendall τ = -0.31 *p* = 0.009).

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 16b & c).

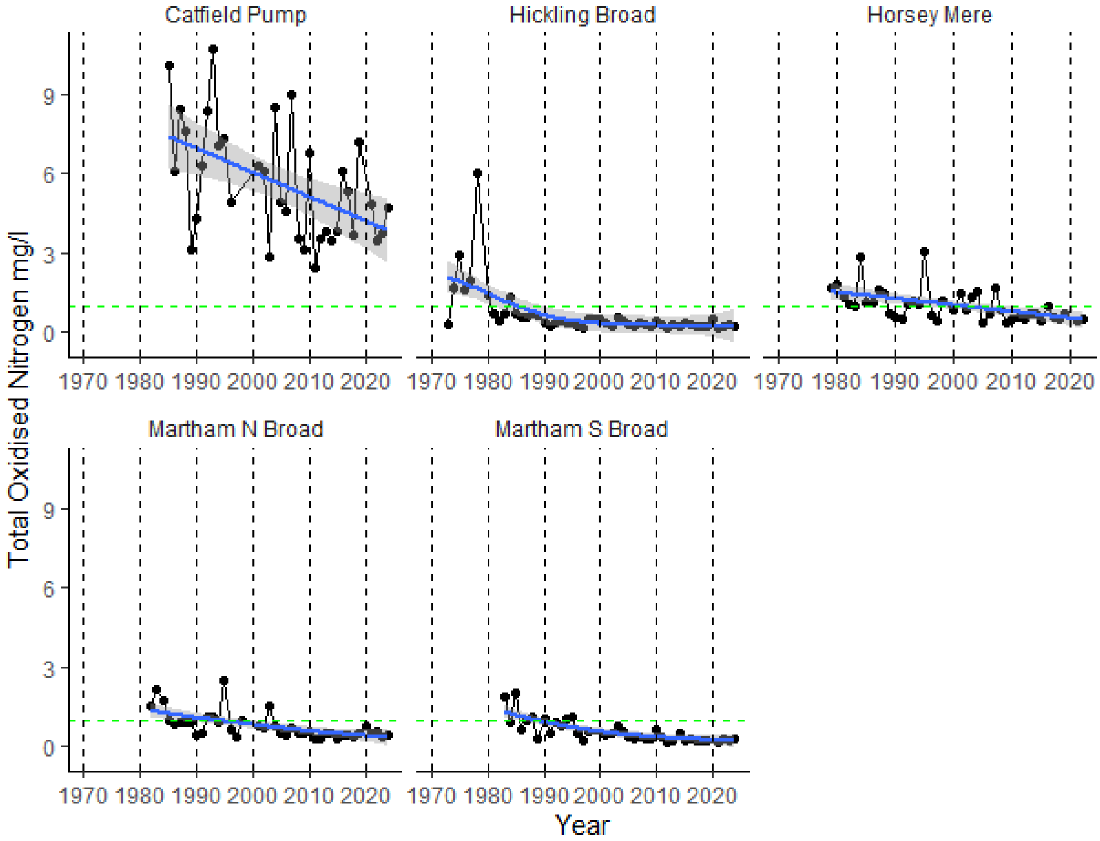


Figure 15. Trend in annual mean total oxidised nitrogen (TON) for a) Catfield Pump, b Hickling Broad, c) Horsey Mere, d) Martham N Broad, e) Martham S Broad. Line showing GAM smooth highlighting long-term trend. Horizontal green line marks 1.0 mg/L

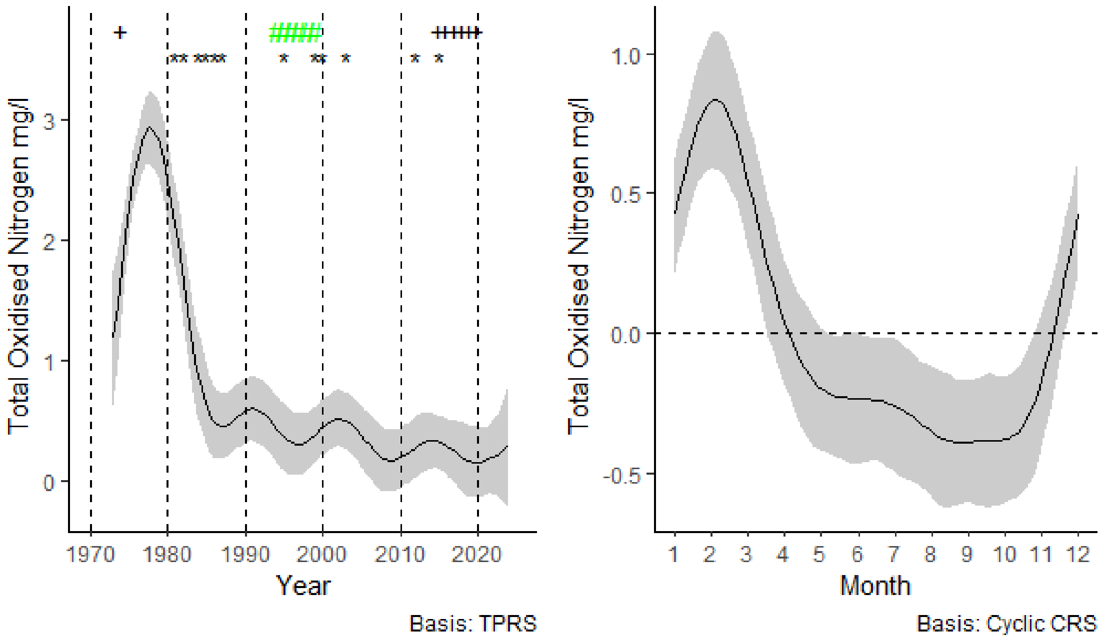


*Figure 16. GAM smooths showing a) predicted long-term and b) seasonal change of total oxidised nitrogen in Hickling Broad. (shading shows the confidence limits). * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

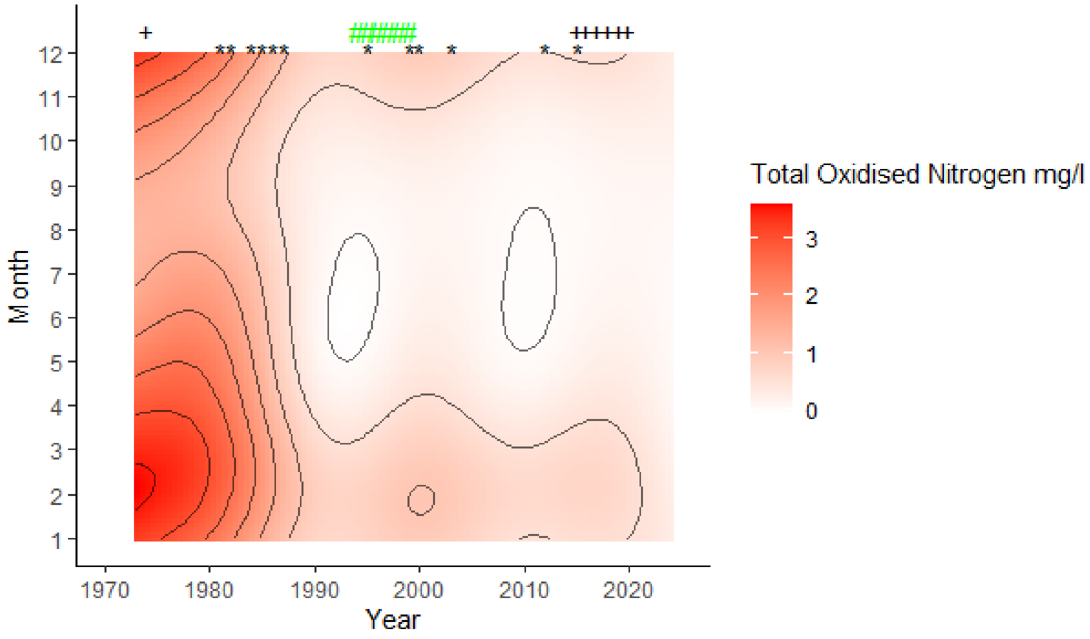


*Figure 16c. Surface plots showing predicted change in total oxidised nitrogen concentration using a GAM model incorporating an interaction between year and month as 2D contour. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

5.4 Ammonium nitrogen

Ammonium nitrogen concentration in Hickling have declined significantly since 1982 (Mann-Kendall τ = -0.43 *p* = <0.001) (Figure 17), with similar trends in the other Thurne Broads, particularly Horsey Mere. In contrast to phosphorus and TON there was no significant decline in Catfield Pump (Mann-Kendall τ = -0.18 *p* = 0.125). Concentrations are high in winter, but low between April and September (Figure 18b & c), the long-term decline being caused by a reduction in the winter concentrations. These results clearly suggest the source of ammonium nitrogen for Hickling Broad being almost certainly Horsey Mere via tidal mixing and the result of the very substantial decline in Horsey Mere (Mann-Kendall τ = -0.48 *p* = <0.001).

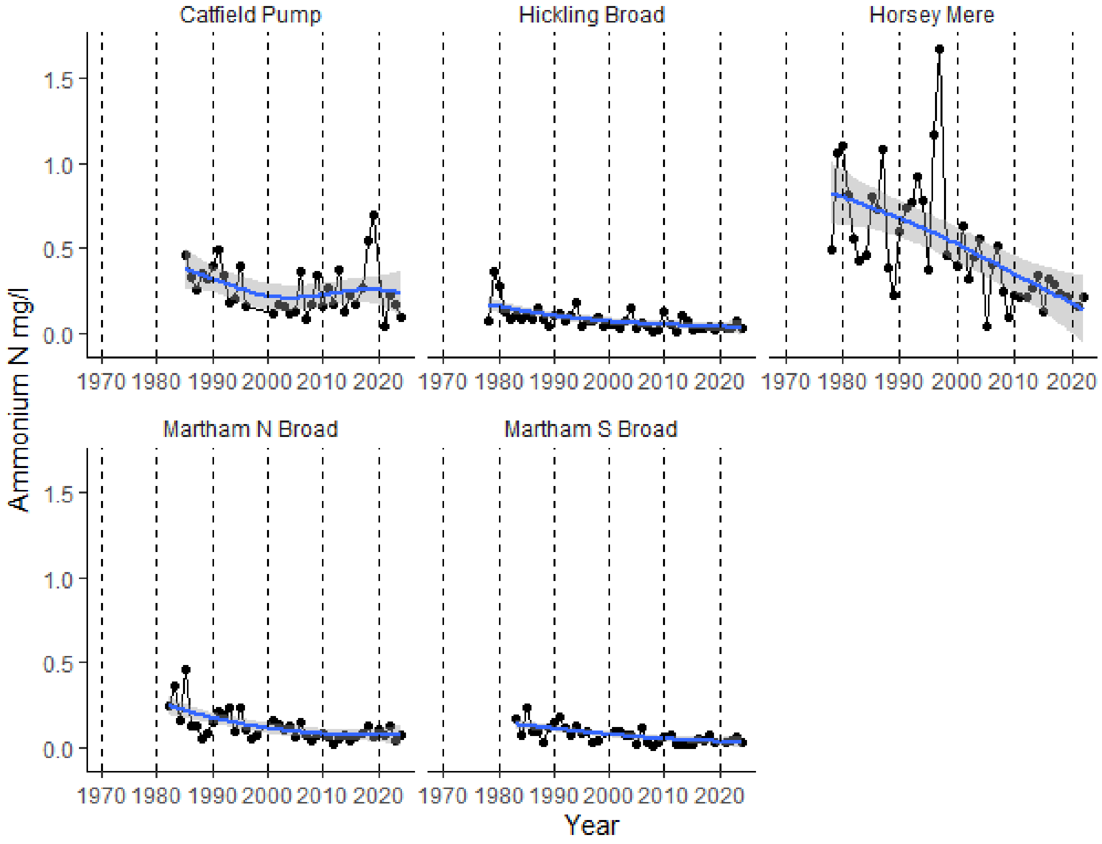


Figure 17. Trend in annual mean ammonium nitrogen for a) Catfield Pump, b Hickling Broad, c) Horsey Mere, d) Martham N Broad, e) Martham S Broad. Line showing GAM smooth highlighting long-term trend.

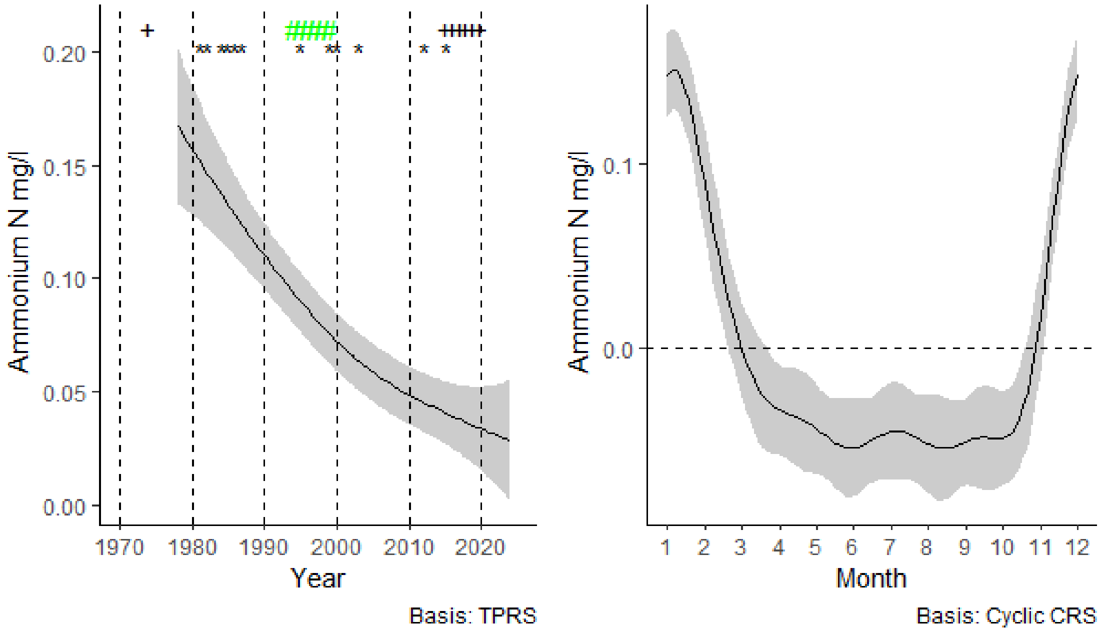


*Figure 18. GAM smooths showing a) predicted long-term and b) seasonal change of ammonium nitrogen in Hickling Broad. (shading shows the confidence limits). * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

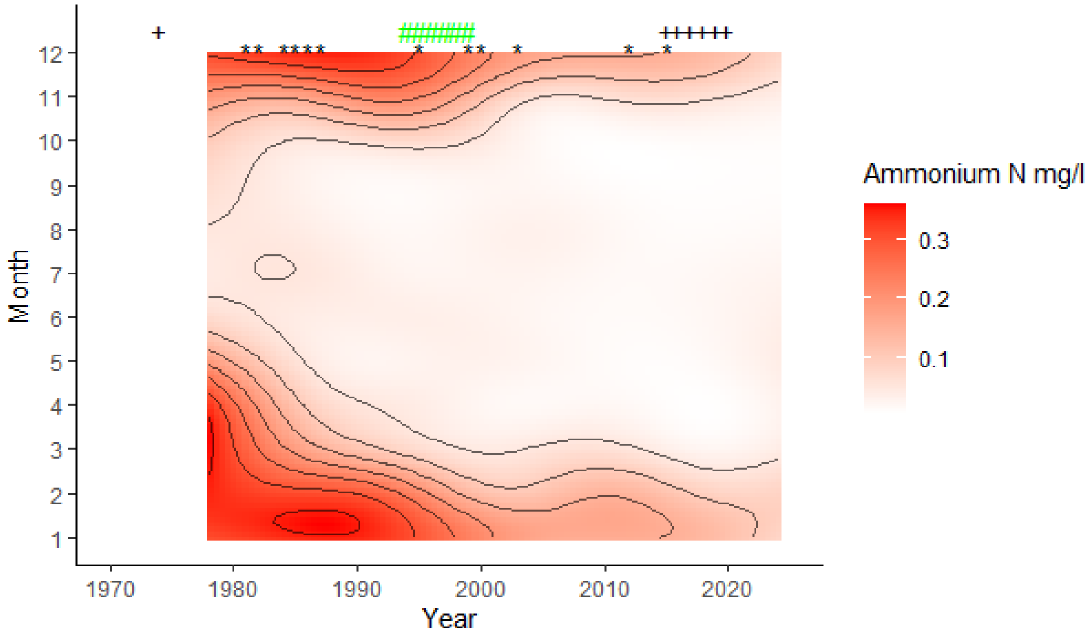


*Figure 18c. Surface plots showing predicted change in ammonium nitrogen concentration using a GAM model incorporating an interaction between year and month as 2D contour. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

5.5 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 (Mann-Kendall $\tau = -0.02$ $p = 0.884$) averaging 1500-1600 mg/L, lower than in Horsey Mere and higher than the Martham Broads (Figure 19). 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 downstream, which is then then pushed back into Hickling on the rising tide. Hickling also receives a small volume of freshwater from Catfield pump 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 20b & c). 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.

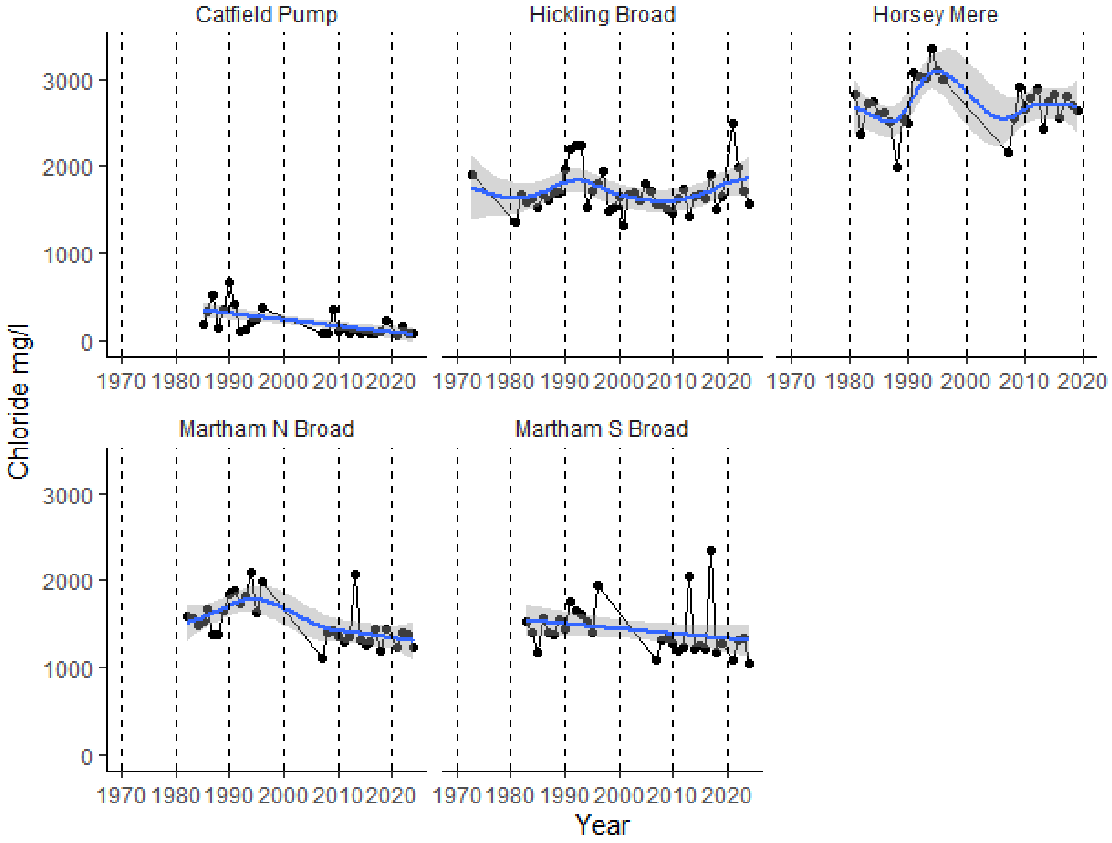


Figure 19. Trend in annual mean chloride concentration for a) Catfield Pump, b Hickling Broad, c) Horsey Mere, d) Martham N Broad, e) Martham S Broad. Line showing GAM smooth highlighting long-term trend.

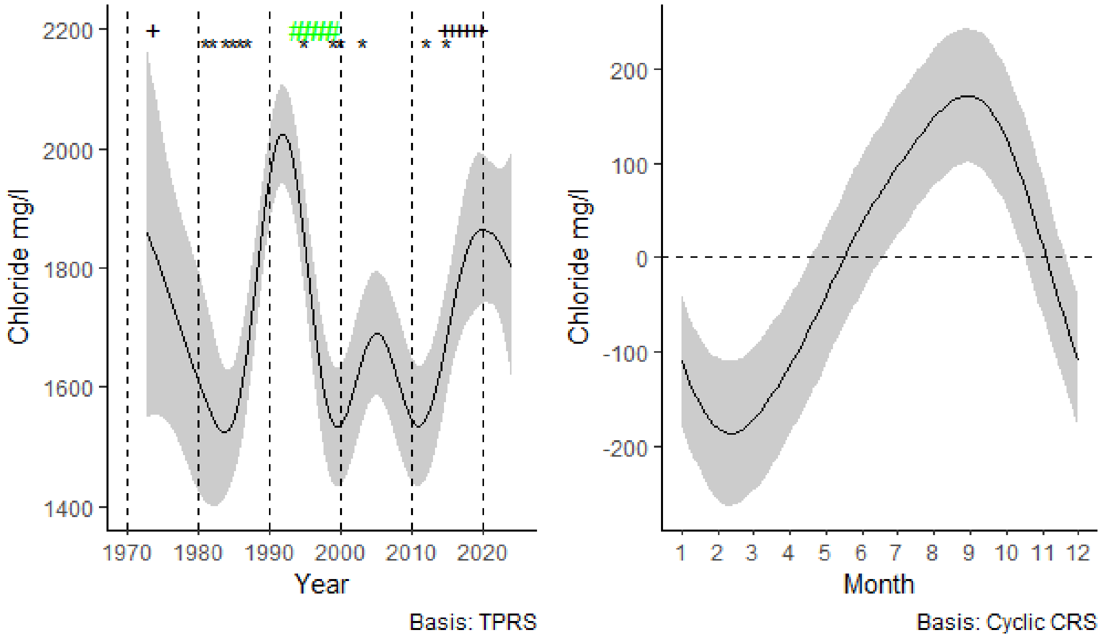


*Figure 20a. GAM smooths showing a) predicted long-term and b) seasonal change of chloride in Hickling Broad. (shading shows the confidence limits. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

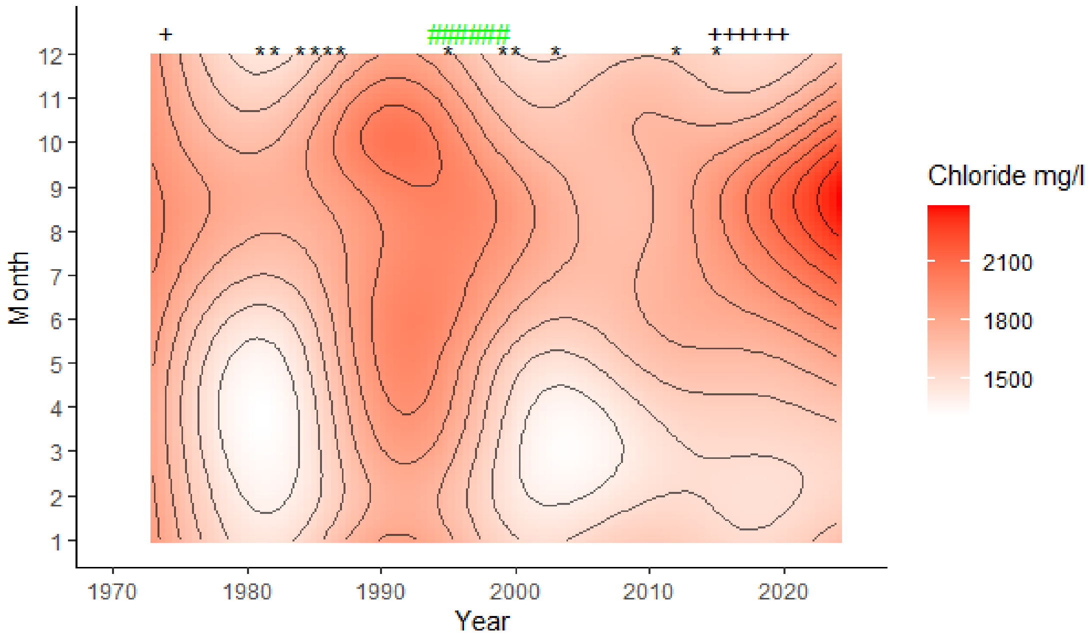


*Figure 20c. Surface plots showing predicted change in chloride concentration using a GAM model incorporating an interaction between year and month as 2D contour. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

5.6 Conductivity

Conductivity records as expected follow the trends described above for chloride, there are fewer long-term records available but are included here for information.

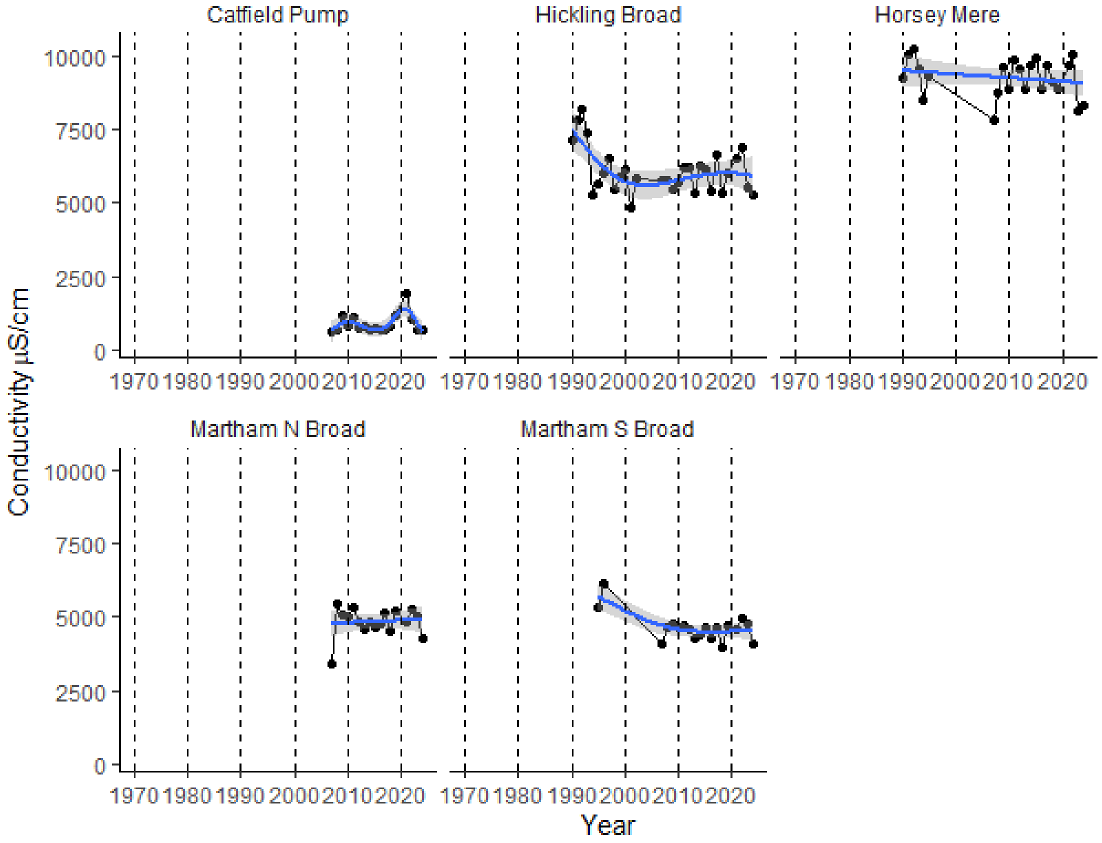


Figure 21. Trend in annual mean conductivity for a) Catfield Pump, b Hickling Broad, c) Horsey Mere, d) Martham N Broad, e) Martham S Broad. Line showing GAM smooth highlighting long-term trend.

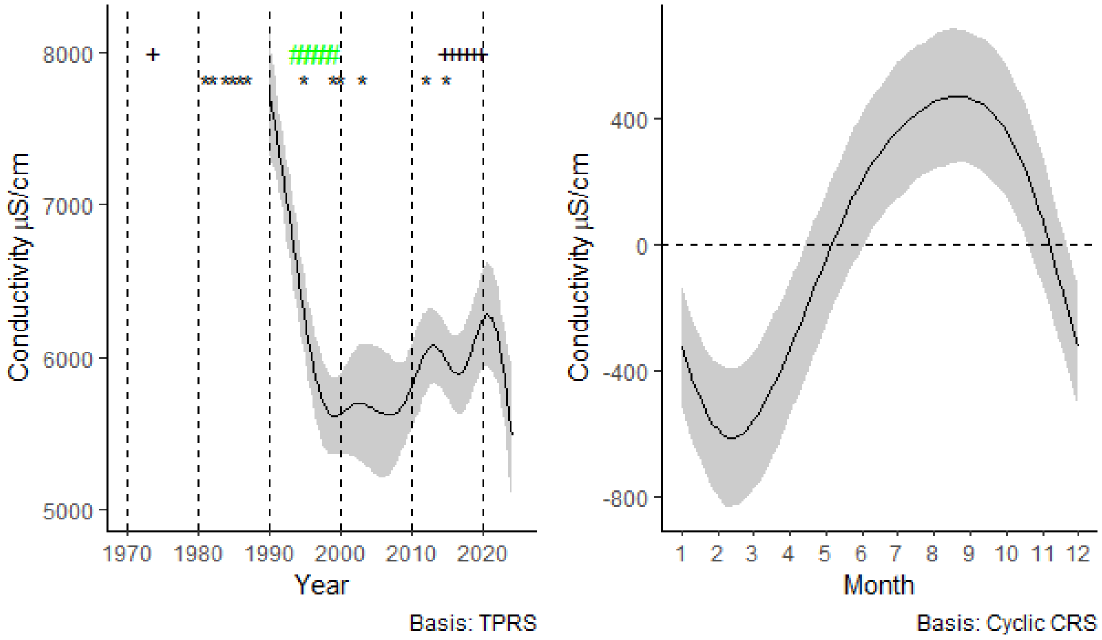


*Figure 22. GAM smooths showing a) predicted long-term and b) seasonal change of conductivity in Hickling Broad. (shading shows the confidence limits. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

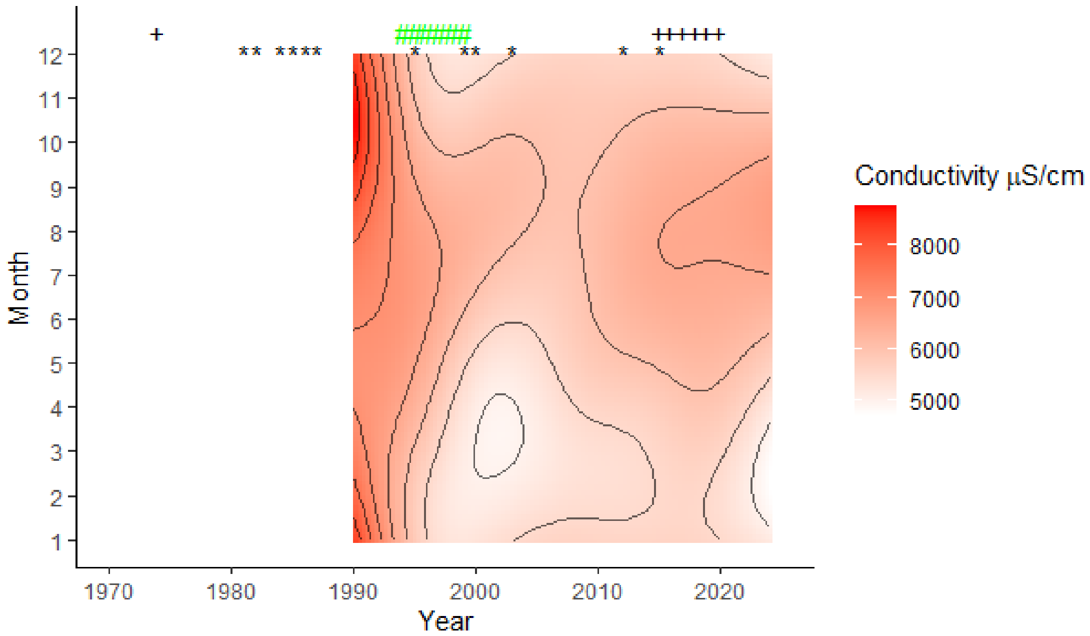


*Figure 22c. Surface plots showing predicted change in conductivity using a GAM model incorporating an interaction between year and month as 2D contour. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

5.7 Total Iron

Given the high concentration of iron hydroxide (ochre) discharged from the Brograve land drainage pump into Horsey Mere and subsequently into Hickling it is worth considering changes in total iron concentration. Iron concentrations in Hickling are substantially lower than in Horsey Mere, averaging 296 µgl-1 compared to 1400 µg l-1 (Figure 23). There has been no overall significant trend in Iron concentration in Hickling (Mann-Kendall τ = -0.03 *p* = 0.790), despite a significant increase in Horsey Mere (Mann-Kendall τ = 0.27 *p* = 0.017). 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. Despite the absence of a long-term trend, iron concentrations in Hickling have shown notable variation, with lower values recorded in the early 1990s, early 2000s, and again in 2022 (Figure 24a). The 1990s low coincided with a period of dry weather, whereas the later declines occurred during periods of extensive macrophyte growth — a relationship explored further in Section 11.

The seasonal pattern of total iron typically shows peak concentrations in winter and early spring (December–February) (Figure 24b). However, this seasonal signal has changed over time: a summer decrease was evident in the early 1990s, followed by an increase during 2005–2015 (Figure 24c). These temporal shifts partially align with periods of macrophyte growth, suggesting a potential connection between aquatic vegetation and total iron levels.

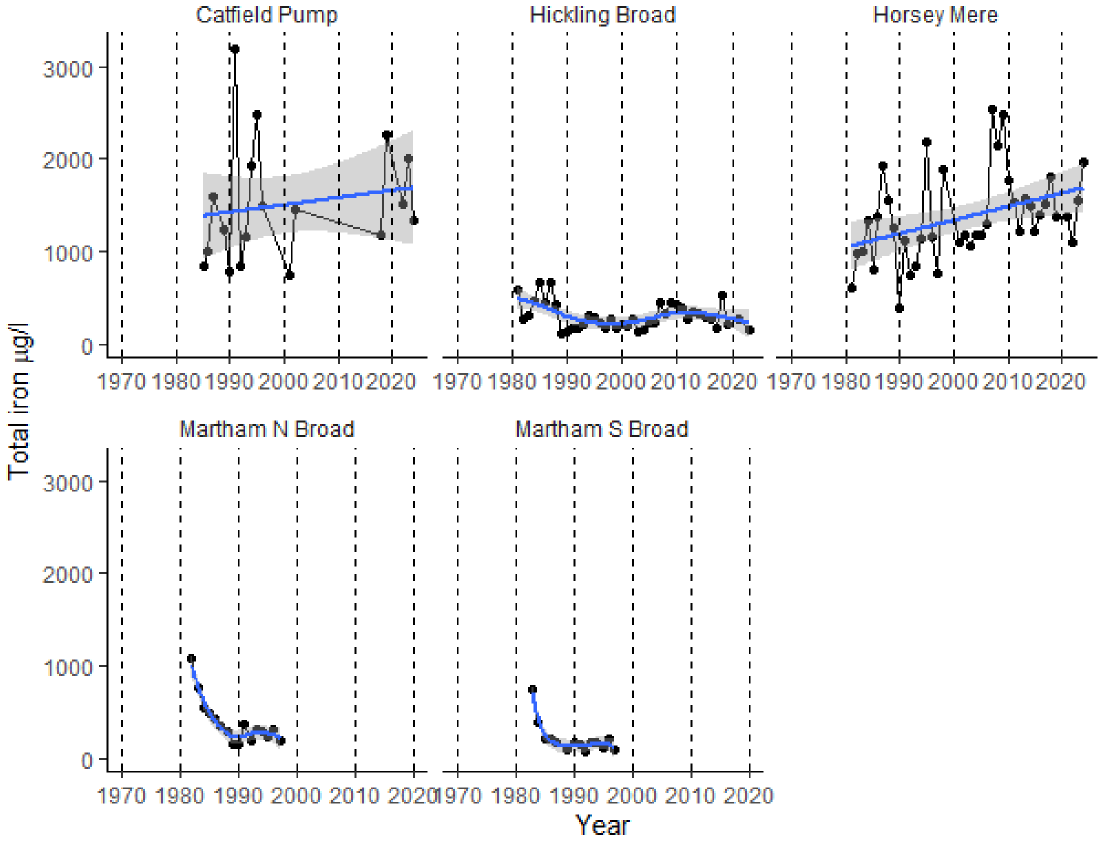


Figure 23. Trend in annual mean total iron concentration for a) Catfield Pump, b Hickling Broad, c) Horsey Mere, d) Martham N Broad, e) Martham S Broad. Line showing GAM smooth highlighting long-term trend.

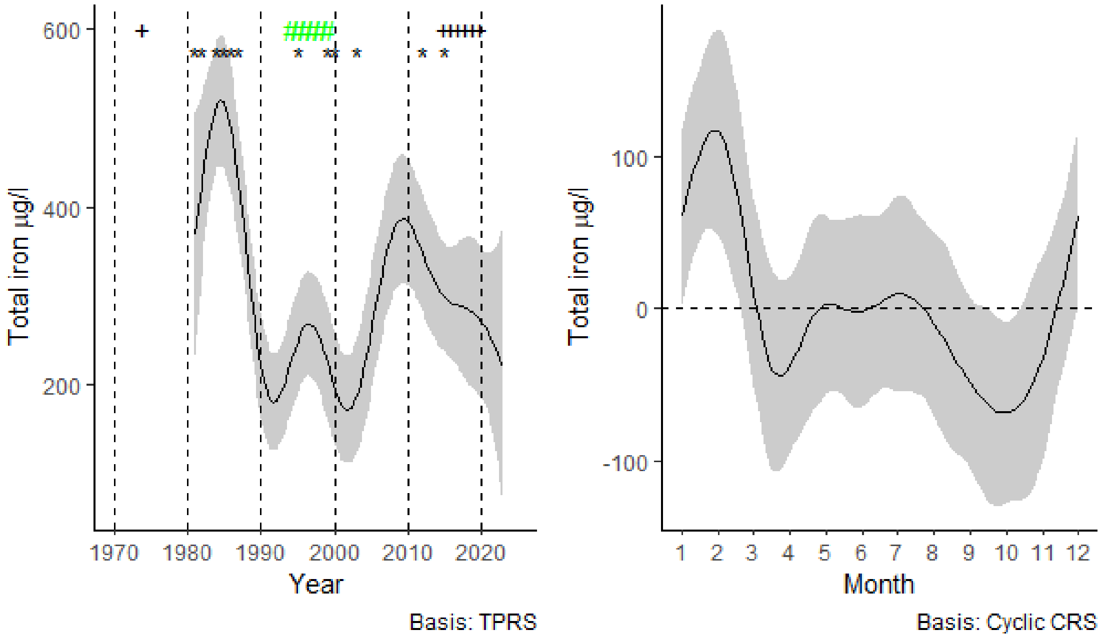


*Figure 24. GAM smooths showing a) predicted long-term and b) seasonal change of total iron in Hickling Broad. (shading shows the confidence limits. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

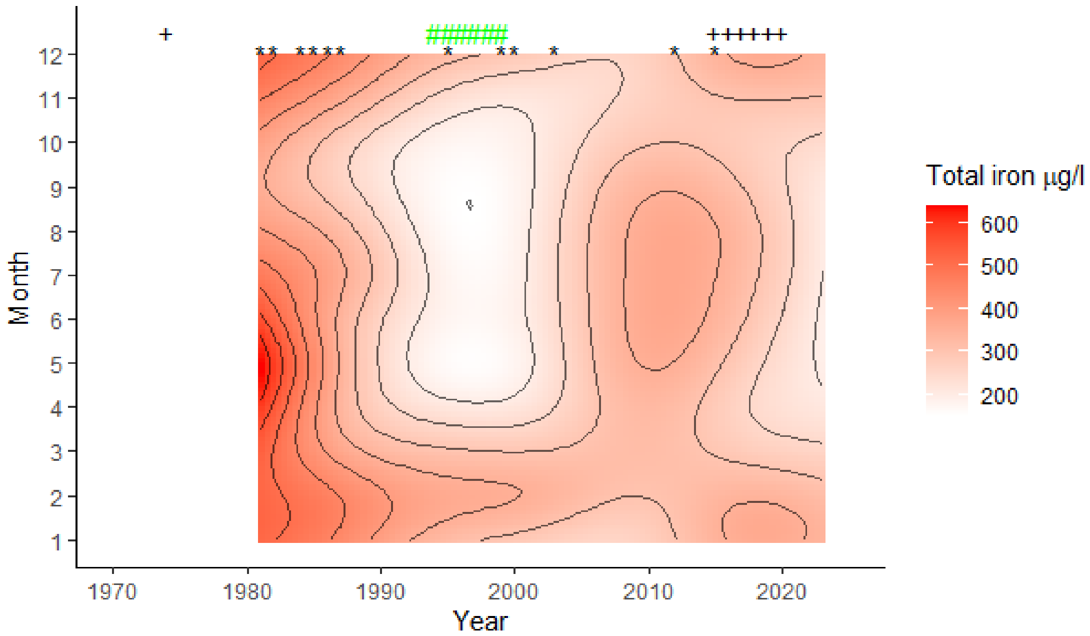


*Figure 24c. Surface plots showing predicted change in total iron concentration using a GAM model incorporating an interaction between year and month as 2D contour. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

5.8 Suspended solids

Total suspended solids in Hickling show no overall trend (Mann-Kendall $\tau = -0.16$ $p = 0.237$), but like total iron, they have exhibited significant long-term variation. Values were generally low between 1997 and 2004, with particularly low concentrations in 1998, 1999, 2003, 2004, and 2024 — all periods associated with maximum Chara growth (Figures 25 & 26a). Suspended solids concentrations in Hickling can be higher than in the other Thurne Broads, possibly reflecting the relatively shallow water depth and the influence of wind-induced sediment resuspension.

The seasonal pattern shows minimum values in winter and a maximum during summer (May–September) (Figure 27b & c). This contrasts with the seasonal pattern of chlorophyll (Figure 28c), suggesting that phytoplankton is not the main component of suspended solids. The seasonal pattern, along with long-term changes that closely mirror plant growth, further supports the conclusion that high macrophyte cover effectively reduces sediment resuspension by wind action in Hickling.

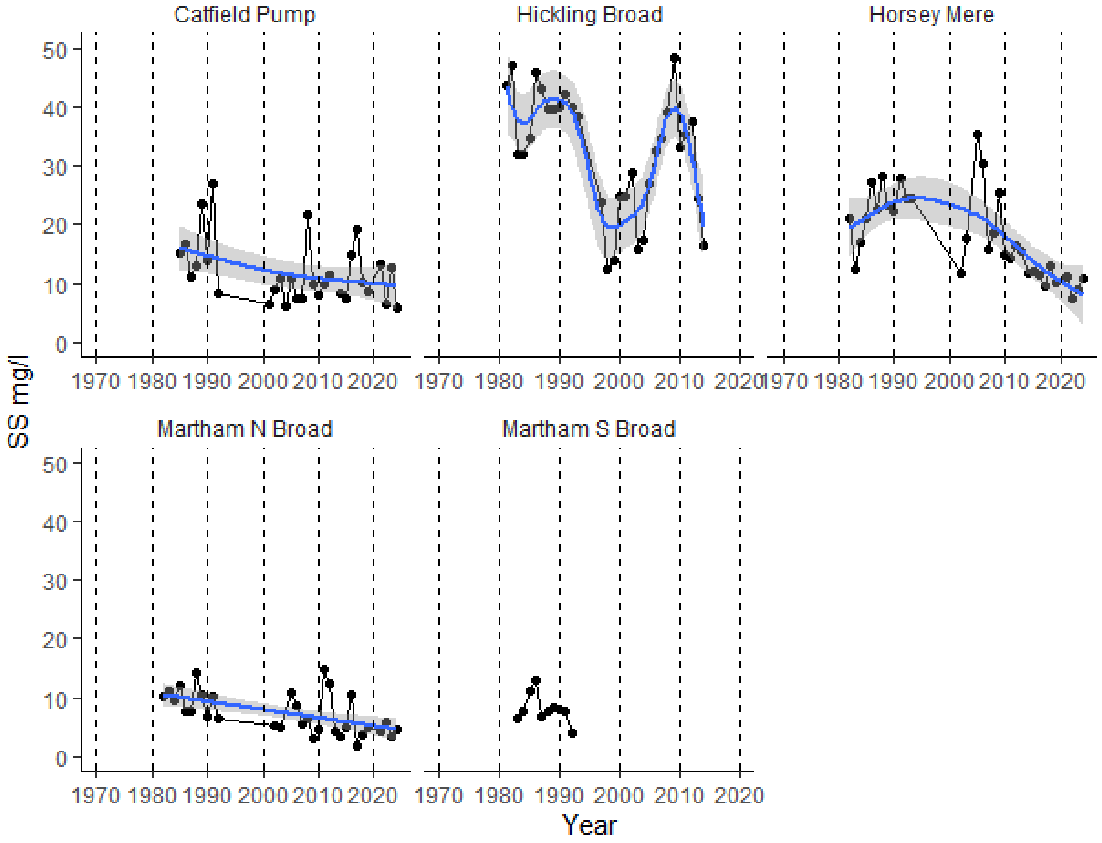


Figure 25. Trend in annual mean suspended solids concentration for a) Catfield Pump, b Hickling Broad, c) Horsey Mere, d) Martham N Broad, e) Martham S Broad. Line showing GAM smooth highlighting long-term trend.

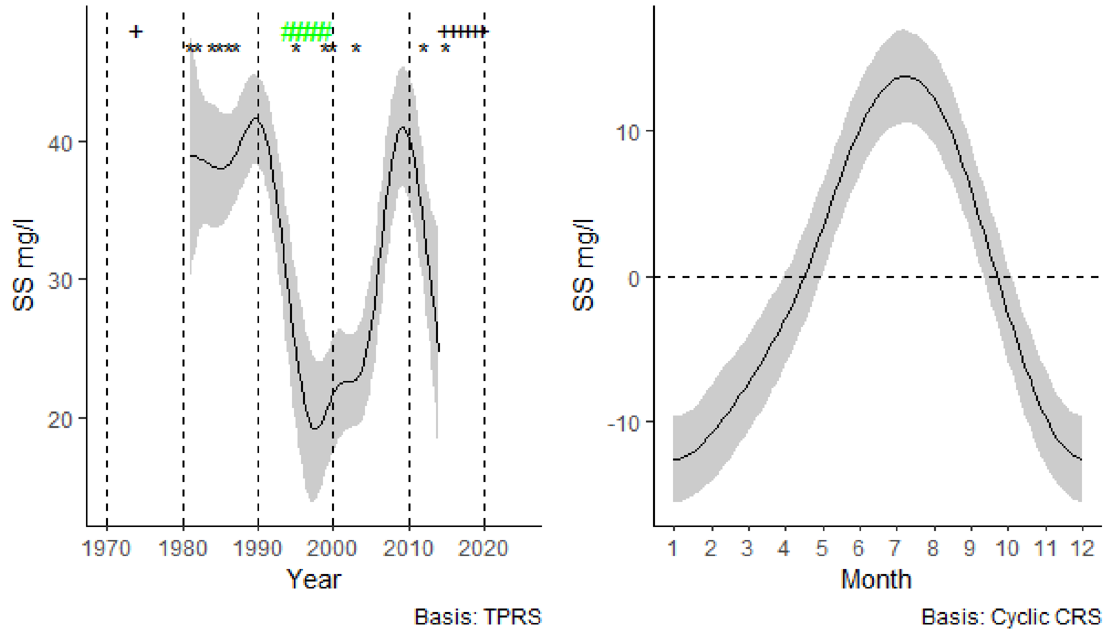


*Figure 26. GAM smooths showing a) predicted long-term and b) seasonal change of suspended solids in Hickling Broad. (shading shows the confidence limits. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

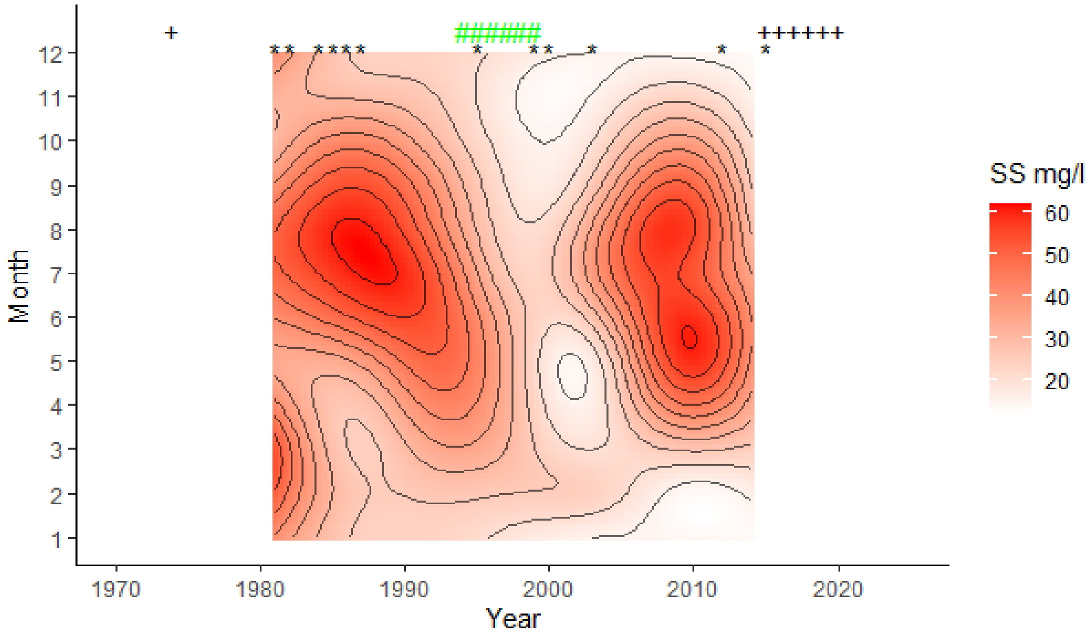


*Figure 26c. Surface plots showing predicted change in suspended solids concentration using a GAM model incorporating an interaction between year and month as 2D contour. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

5.9 Chlorophyll a

The chlorophyll concentration in Hickling Broad has reduced significantly since the mid-1970s (Mann-Kendall $\tau = -0.54$ $p = <0.001$) reaching a minimum annual mean value of 23 µg/L in 2023 (Figure 27), which is very close to the good/moderate boundary of 20 µg/L. However, the phytoplankton of Hickling is currently classified as having Poor status (2022 classification), a deterioration from the Moderate status reported in 2019.

The decline in chlorophyll has not followed a steady trajectory, with a noticeable flattening of the long-term trend (Figure 28a) driven by a marked increase in concentrations during 1999 and 2000 (Figure 27). This rise followed a period of much lower chlorophyll levels from 1996 to 1998, coinciding with the first significant resurgence of Chara in the Broad. Even lower chlorophyll values have been recorded more recently during the most recent period of Chara dominance

The seasonal cycle of chlorophyll a in Hickling differs from that of most other broads, exhibiting a single peak in February–March and relatively low concentrations in summer (Figures 28b & 28c), particularly in years dominated by Chara (1996–2000, 2019–2024). This may be partly due to the summer phytoplankton community being dominated by small cyanobacteria—such as Microcystis aeruginosa (EA records, 2010–2012) and *Aphanothece* spp. (1988–1989, Bales et al. 1993). However, competition for available phosphorus between phytoplankton and Chara may also be a contributing factor.

Since 1969, Hickling has experienced persistent fish kill events associated with toxic blooms of the flagellated alga Prymnesium parvum, which have predominantly occurred in winter and spring. There is some evidence that the frequency and severity of these blooms have declined since the 1980s, a period when fish kills were more frequent. No confirmed fish kills have been recorded at Hickling since the spring events of 2012 and 2015, although P. parvum was still detected in routine counts carried out until 2020 and subsequently to 2022 in qPCR analysis when routine monitoring stopped and is assumed to still be present in Hickling Broad.

The specific environmental drivers of Prymnesium bloom development in Hickling—and more broadly—remain uncertain, and toxic effects may occur even in the absence of visible blooms. Recent research by Wagstaff et al. (2021) has provided new insight into the toxicity mechanism, identifying the role of *Prymnesium* toxins and the importance of a lytic virus, PpDNAV. While the study did not demonstrate causality, it presented strong correlative evidence that PpDNAV contributes significantly to the toxicity of Prymnesium blooms and associated fish kills, likely by triggering toxin release through viral lysis of algal cells.

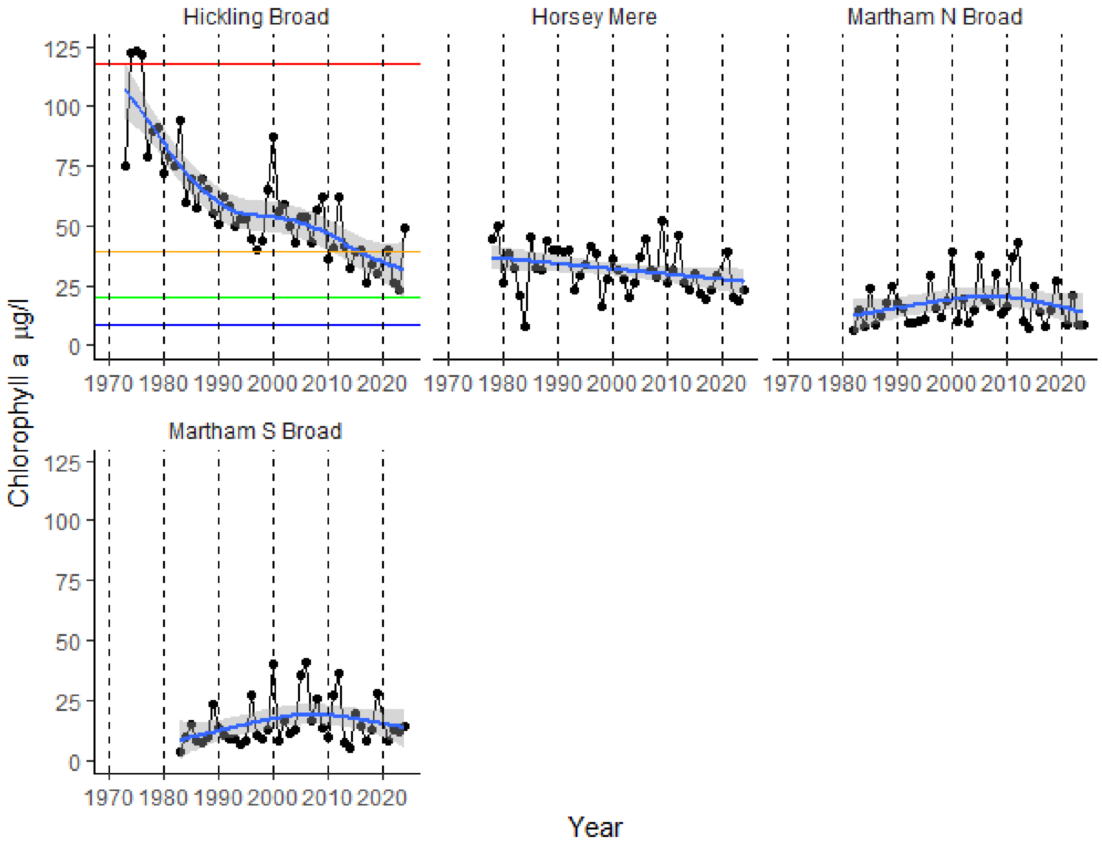


Figure 27. Trend in annual mean chlorophyll a concentration for a) Hickling Broad, b) Horsey Mere, c) Martham N Broad, d) Martham S Broad. Line showing GAM smooth highlighting long-term trend. Horizontal lines mark WFD boundary values for Hickling Broad (HG = 9,GM = 20, MP = 39,PB = 118 µg/L)

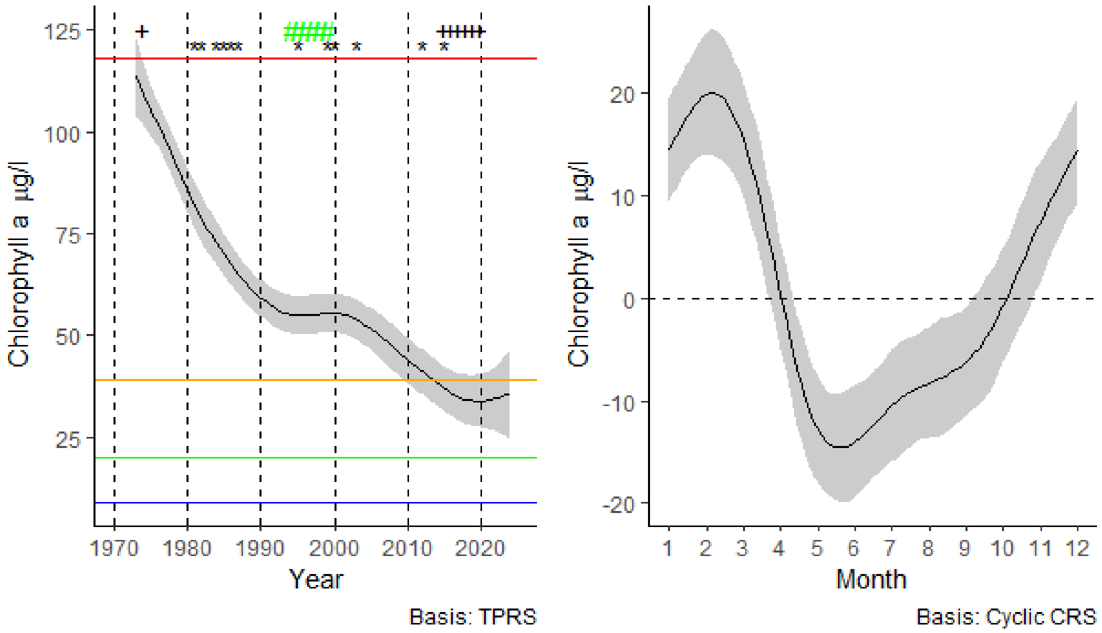


*Figure 28. GAM smooths showing a) predicted long-term and b) seasonal change (relative to long-term mean) of chlorophyll a in Hickling Broad. (shading shows the confidence limits. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

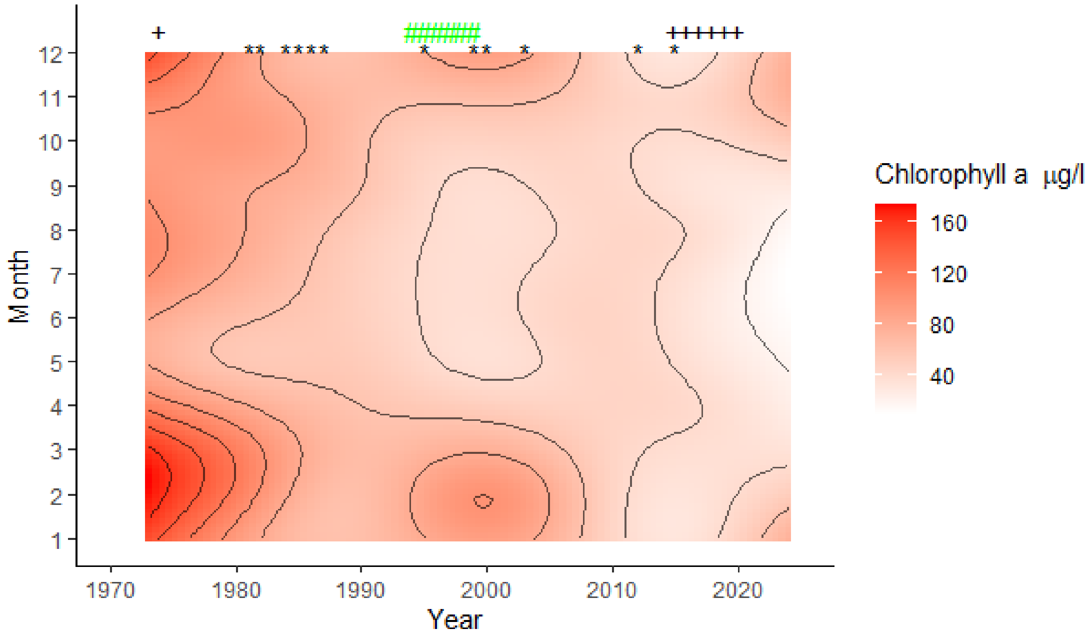


*Figure 28c: Surface plots showing predicted change in chlorophyll a concentration using a GAM model incorporating an interaction between year and month as 2D contour. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

5.10 Chlorophyll a / Total Phosphorus ratio

In common with the other Thurne broads, Hickling Broad exhibits a high chlorophyll-to-total phosphorus (Chl:TP) ratio, with a median value of 0.63 (Figure 29). The elevated salinity in Hickling limits the development of large populations of grazing cladocerans, resulting in a generally high phytoplankton yield per unit of phosphorus.

There has been a slight long-term decreasing trend in the Chl:TP ratio (Mann-Kendall τ = -0.2 p = 0.045), although this has been punctuated by distinct peaks during periods of Chara proliferation. These peaks reflect reductions in total phosphorus concentrations associated with macrophyte development, which had relatively little effect on the annual biomass of phytoplankton.

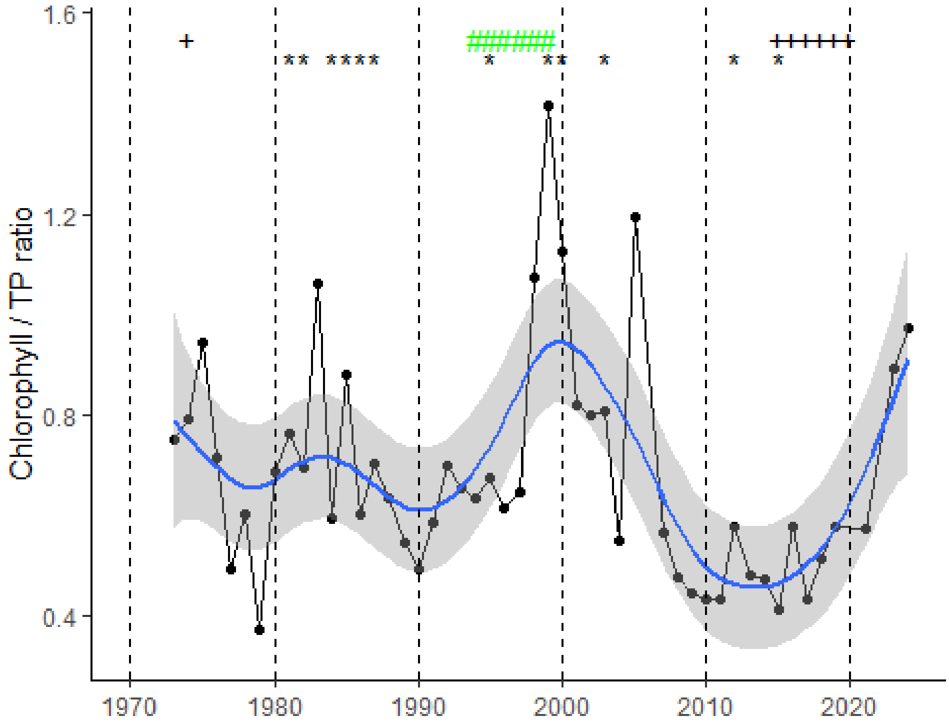


*Figure 29. Trend in annual mean chlorophyll a / TP ratio for Hickling Broad (solid black line) showing GAM smoothers, * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging. (2006 data omitted as no samples March-June).*

5.11 Transparency (Secchi disc depth).

From 1978 to 2024, there was a slight but statistically insignificant overall increase in water transparency in Hickling Broad (Mann-Kendall τ = 0.16 p = 0.118) (Figure 30). Within this period, transparency exhibited notable fluctuations, with distinct phases of both increased and decreased clarity. Periods of higher transparency largely coincided with the growth phases of Chara (Figure 31a).

Transparency typically peaked during winter and reached its lowest levels in July (Figure 31b). However, this characteristic summer decline was noticeably reduced during years when Chara was abundant (Figure 31c), suggesting either a stabilising influence of macrophyte cover on water clarity or that high water clarity is a prerequisite for Chara growth. See Section 11 for further discussion.

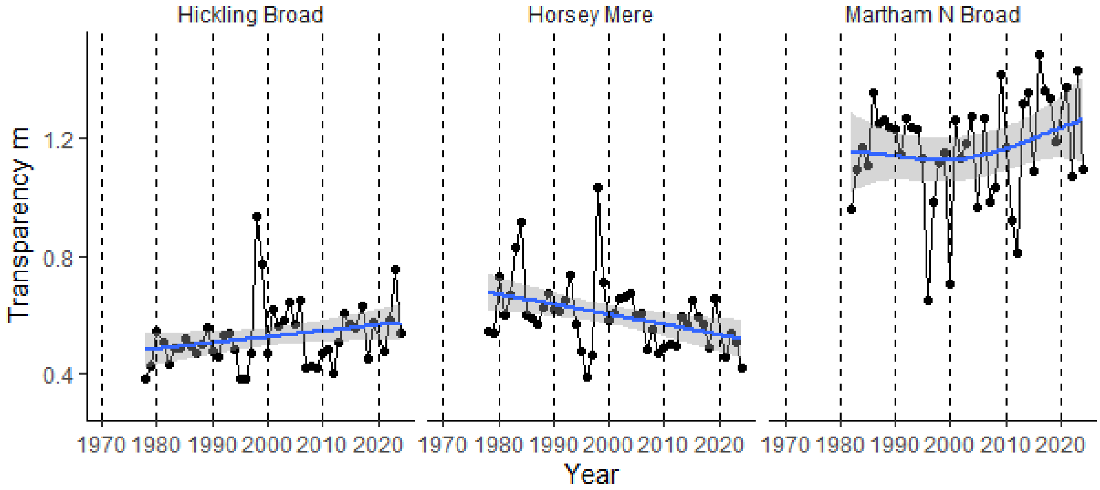


Figure 30. Trend in annual mean transparency (Secchi depth) for a) Hickling Broad, b) Horsey Mere, c) Martham N Broad. Line showing GAM smooth highlighting long-term trend.

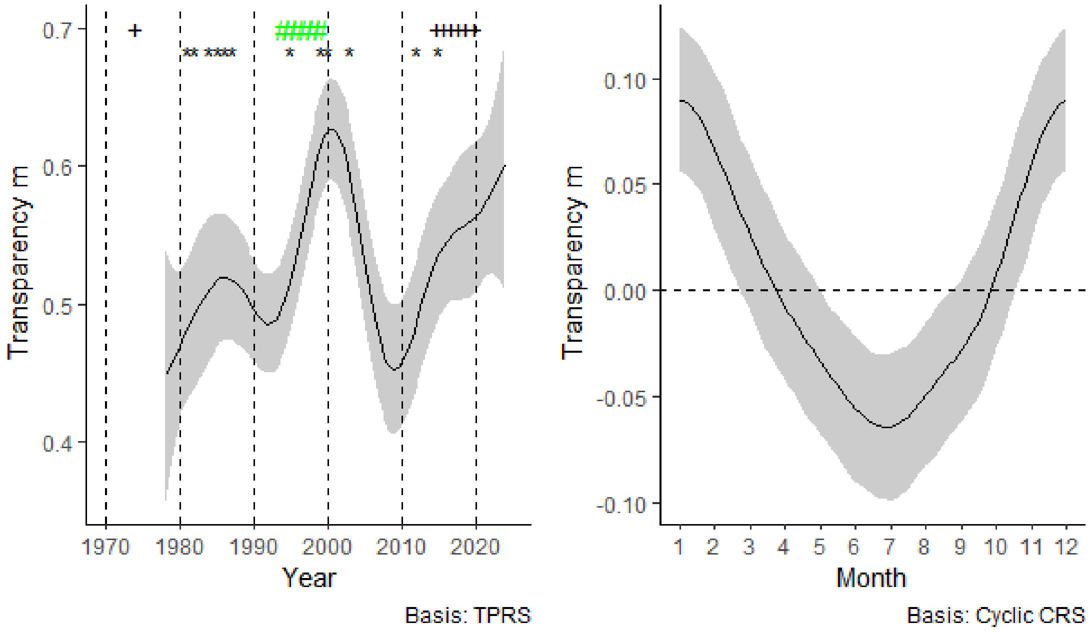


*Figure 31. GAM smooths showing a) predicted long-term and b) seasonal change of Secchi disc transparency in Hickling Broad. (shading shows the confidence limits. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

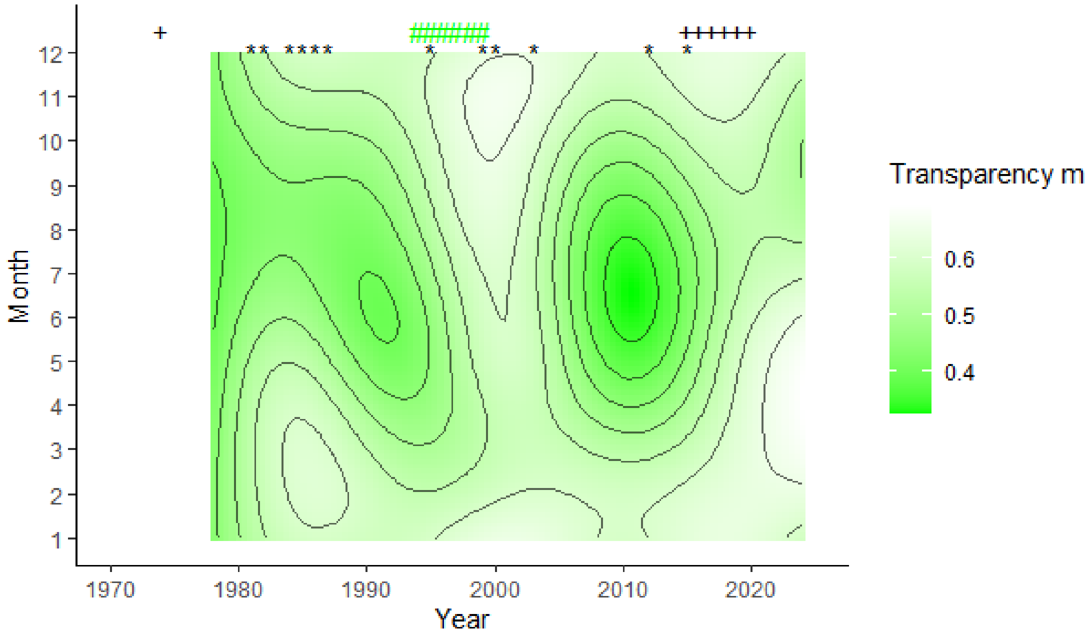


*Figure 31c: Surface plots showing predicted change in transparency (Secchi depth) using a GAM model incorporating an interaction between year and month as 2D contour. * mark fish kills due to Prymnesium, green # mark period of macrophyte cutting, + mark winter dredging.*

There is a relatively weak relationship between Secchi depth and chlorophyll concentration in Hickling Broad (Figure 32). While high chlorophyll levels are generally associated with reduced water transparency, the relationship plateaus at concentrations above 50 µg/L, suggesting that additional sources of turbidity are influencing light penetration. Furthermore, data points from Hickling Broad consistently fall below the fitted relationship derived from other lakes in the Broads area, further indicating that non-phytoplankton sources contribute to turbidity.

This is supported by the relationship between Secchi depth and suspended solids (Figure 33), which shows a significant linear correlation, closely matching that observed in other Broads lakes. As discussed in Section 5.8, the cause of elevated suspended solids relative to chlorophyll in Hickling remains uncertain. One possible explanation is wind-driven resuspension of sediment. However, Bales et al. (1993) reported a significant relationship between light extinction coefficients in Hickling and the abundance of small cyanobacteria, suggesting that summer dominance of such taxa in the phytoplankton community may also play a role.

Regardless of the cause, the light climate in the absence of Chara is notably poor compared to other Broads, with summer Secchi depths often averaging below 0.5 m (Figure 31c).

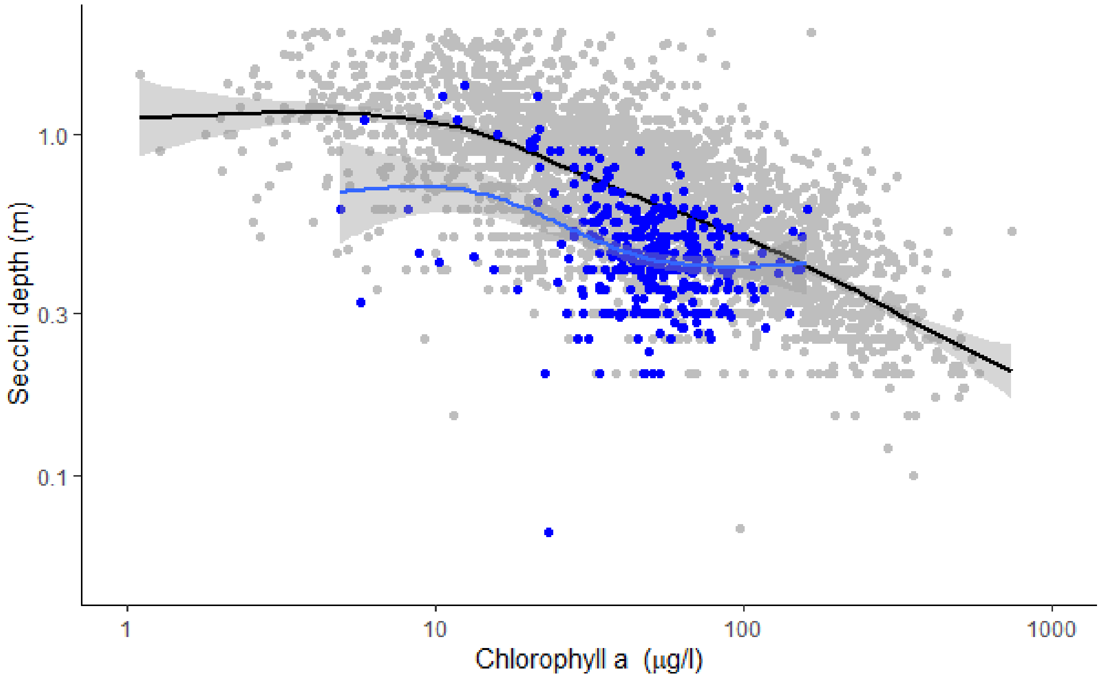


Fig 32. Relationship between water transparency (Secchi depth) and chlorophyll a in Hickling Broad (blue dots) in comparison to all broads (black dots). Lines show GAM smoothers for all broads (black) and for Hickling (blue). (Data shown are only for samples where Secchi depth is < water depth).

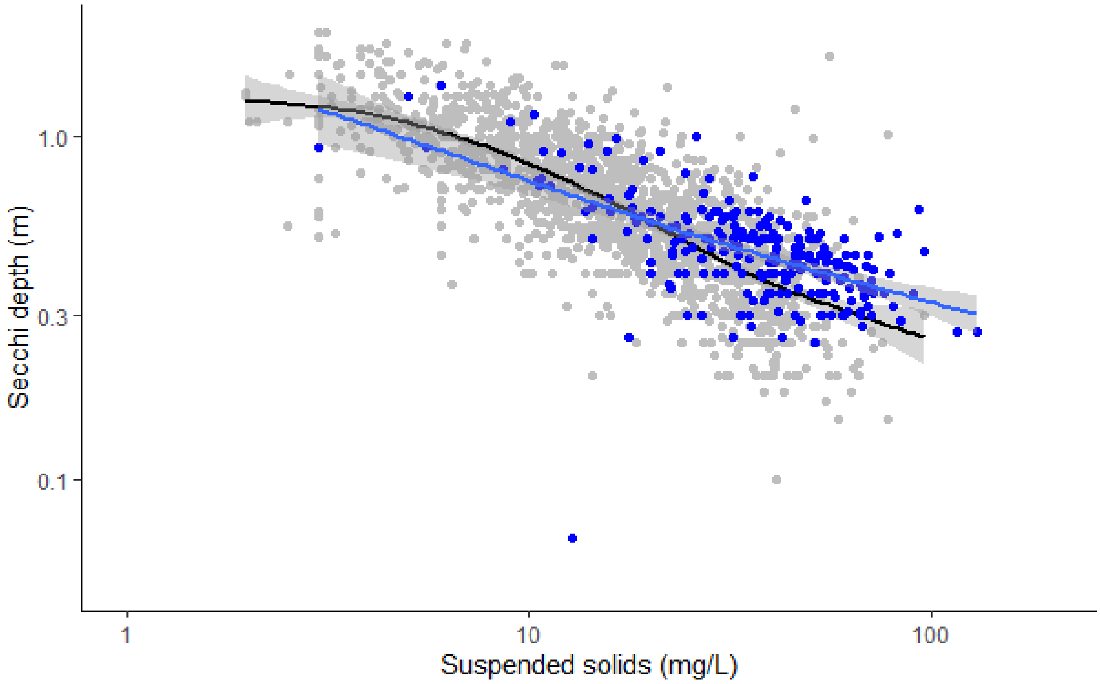


Figure 33. Relationship between water transparency (Secchi depth) and suspended solids in Hickling Broad (blue dots) in comparison to all broads (black dots). Lines show GAM smoothers for all broads (black) and for Hickling (blue). (Data shown are only for samples where Secchi depth is < water depth).

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 3). 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_2S. Figures presented by Jackson (Figure 11) 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 3. 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

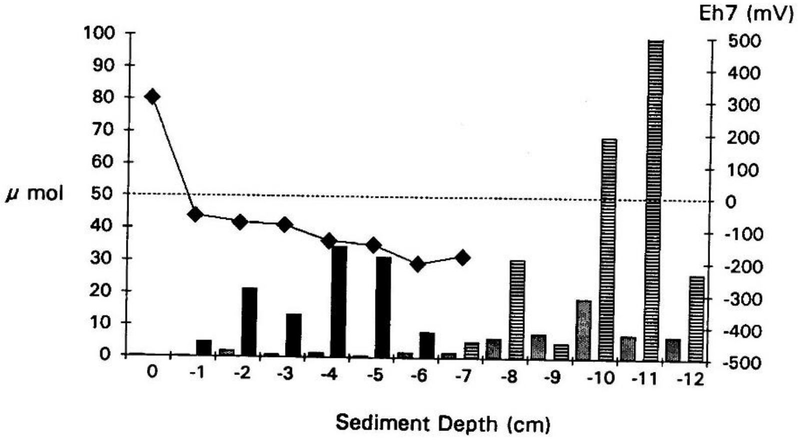


Figure 4.2.3.1 Hickling Broad 1/8/90 - Variation in interstitial soluble phosphorus (▩), ferrous iron (▮) and soluble sulphide (≡) concentration and redox potential (-♦-) with depth.

Figure 11 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 12). 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 12) 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.

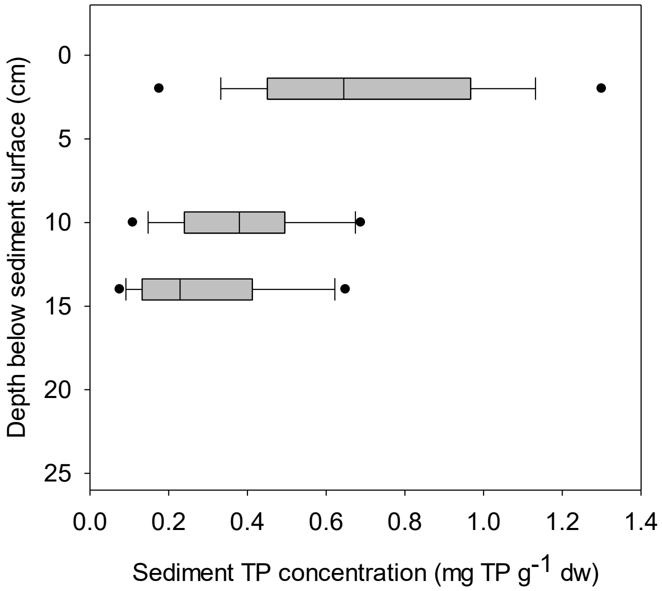


Figure 12 Variation in sediment TP concentration with sediment depth in Hickling Broad in 2013. Sediment samples were collected from 1 cm to 3 cm; 7 cm to 11 cm and 13 to 15 cm sediment depth ranges.

7 Macrophytes

Two survey methods have been used to assess aquatic plant abundance in Hickling Broad. Initially, surveys employed a rake sampling technique along fixed transects, with abundance scores derived from the vegetation retrieved along these lines. In 2014, the methodology shifted to a point-based approach, in which plant abundance was assessed at 40 fixed sampling locations across the broad (see Broads Authority, 2024, for details). Both methods produce an index of plant cover, but to enable comparison and integration of results across the two approaches, the data were normalised.

Normalisation was based on the observation that maximum total plant abundance recorded in 1999 (using the transect method) was comparable to that observed in 2024 (using the point-based method). To standardise the datasets, values from each method were scaled to a 0–1 range by dividing each year’s total abundance by the maximum recorded within its respective survey type.

It should be noted that Hydroacoustic surveys are undertaken annually, particularly at Hickling Broad, to assess aquatic plant cover, height, and volume using sonar technology. These surveys complement point-based botanical sampling by providing detailed spatial data on plant distribution and abundance. While the results are not included in this report, they are published separately in the Broads Authority’s annual water plant reports. Hydroacoustic monitoring supports navigation management and long-term ecological assessment by enabling efficient, broad-scale measurement of vegetation structure. It also enhances understanding of habitat change and the impacts of restoration, and may be incorporated in future reporting for a more comprehensive analysis.

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 recovery achieving a cover as high as the 1999 peak by 2022, a level which has remained until the latest 2024 survey.

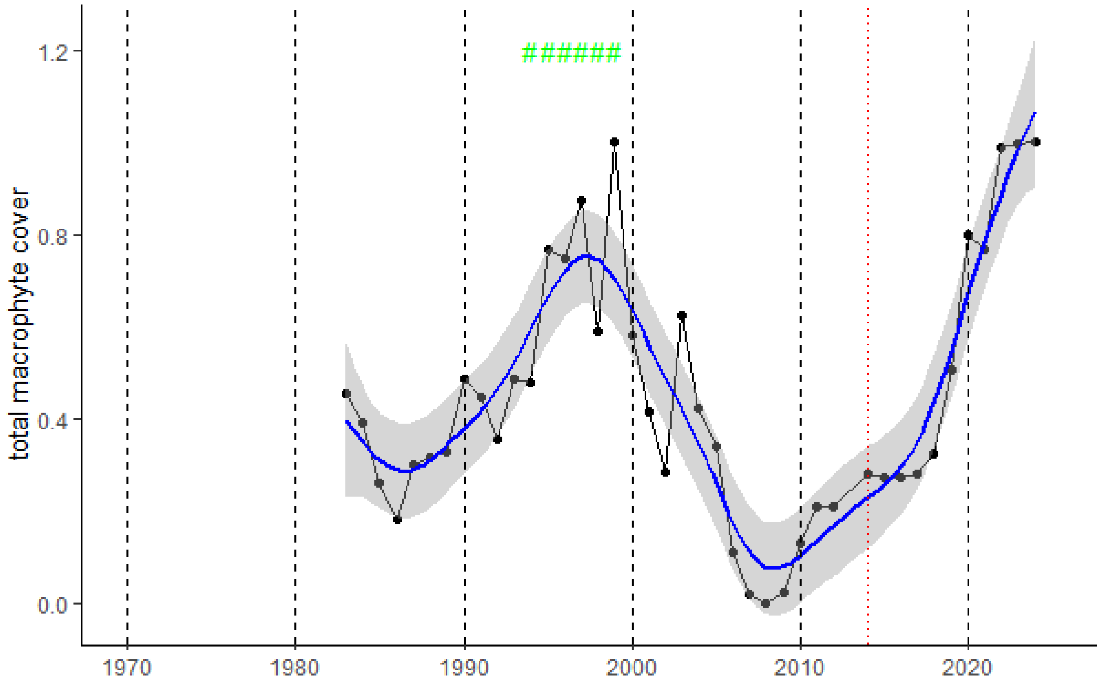


Figure 36. Trend in cover of all macrophyte taxa for Hickling Broad 1983-2024. GAM smooth shows trend, green # mark periods of weed cutting, red dotted line marks change to point survey method

Filamentous algal cover is perennially low in Hickling and is unusually low for a brackish lake with elevated P concentrations, although it shows a slight increase in the last two years (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 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.

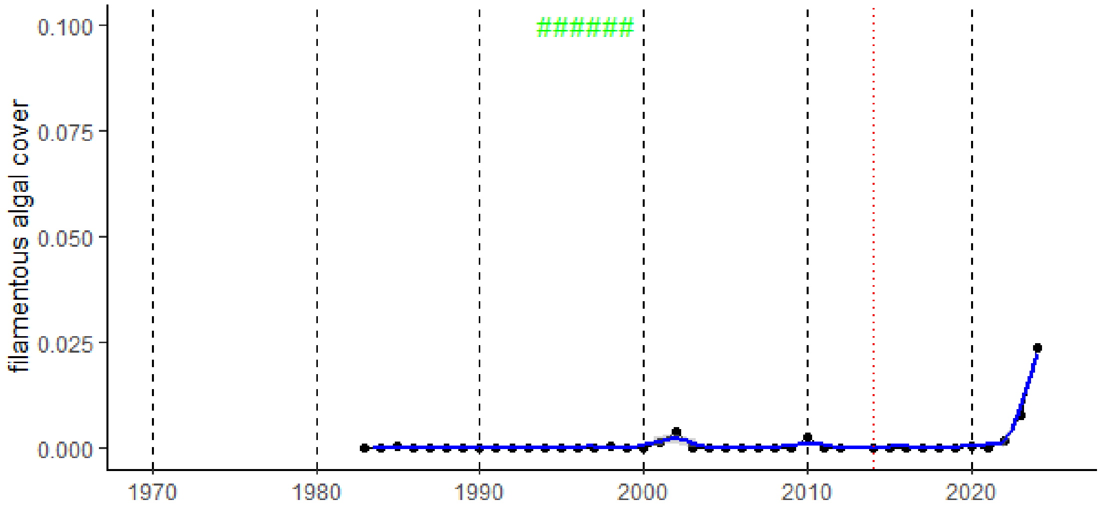


Figure 37. Trend in algal macrophyte cover for Hickling Broad 1983-2024. GAM smooth shows trend, green # mark periods of weed cutting, red dotted line marks change to point survey method

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 ~15 species is in line with the 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. The change in survey method in 2014 revealed a steady increase in the diversity of these macrophytes to ~8 species (Figure 39).

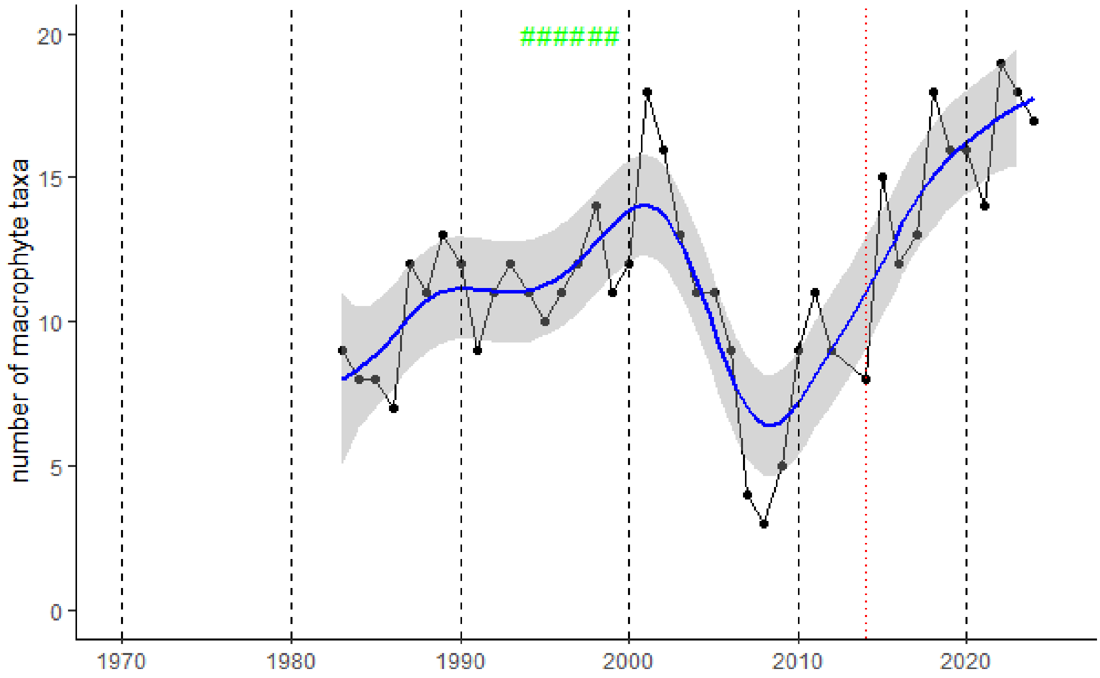


Figure 38. Trend in number of macrophyte taxa (treating all Chara species as single taxon) for Hickling Broad 1983-2024. GAM smooth shows trend, green # mark periods of weed cutting, red dotted line marks change to point survey method

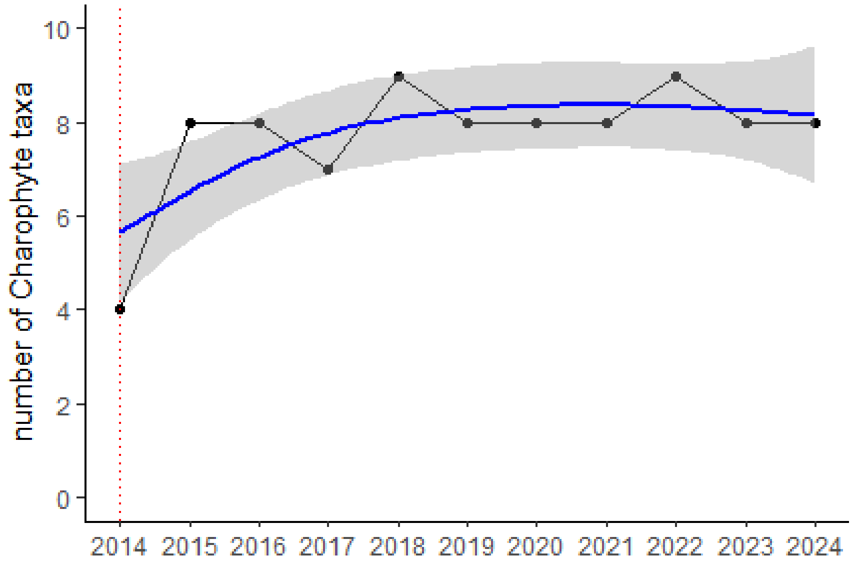


Figure 39. Trend in number of Charophyte taxa for Hickling Broad 2014-2024. GAM smooth shows trend, red dotted line marks change to point survey method

7.3 Compositional change

The compositional data (Figure 40) 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 in 1999 and subsequently by 2022. 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 was 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*. This 2013 growth was apparently sufficient to kick-start a re-expansion of Chara which began in 2014. Water level records (available for Hickling since 1997) reveal that this new expansion of Chara occurred when water levels were unusually low in Hickling Broad (See section 11) which would have contributed to a good light climate.

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 Mute Swan, Teal and Greylag Goose at Hickling since 2000 (see section 10).

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 (Figure 41). 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.

Chara intermedia forms by far the majority of the Charophyte cover, but most of the other species recorded have also increased in abundance since 2014 (Figure 42).

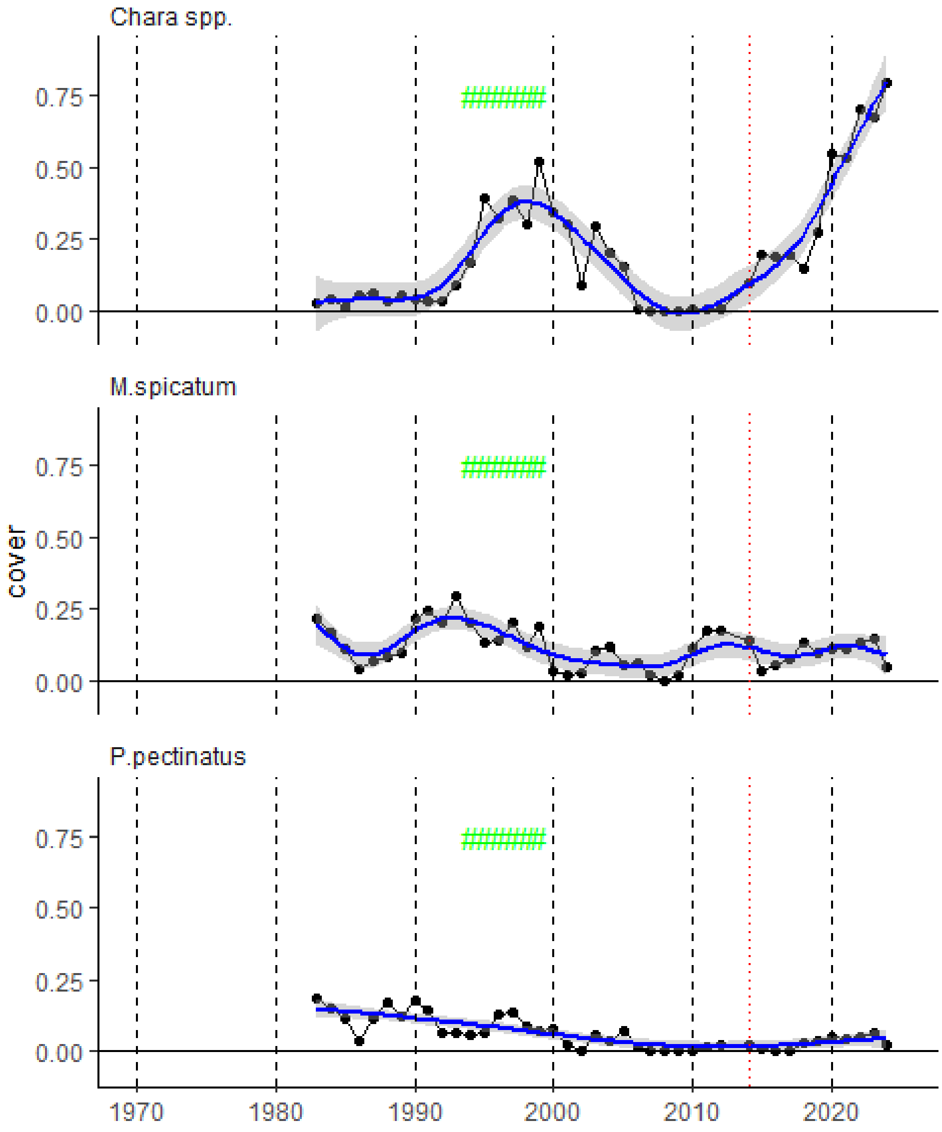


Figure 40. Trend in key macrophyte taxa in Hickling Broad 1983-2024.

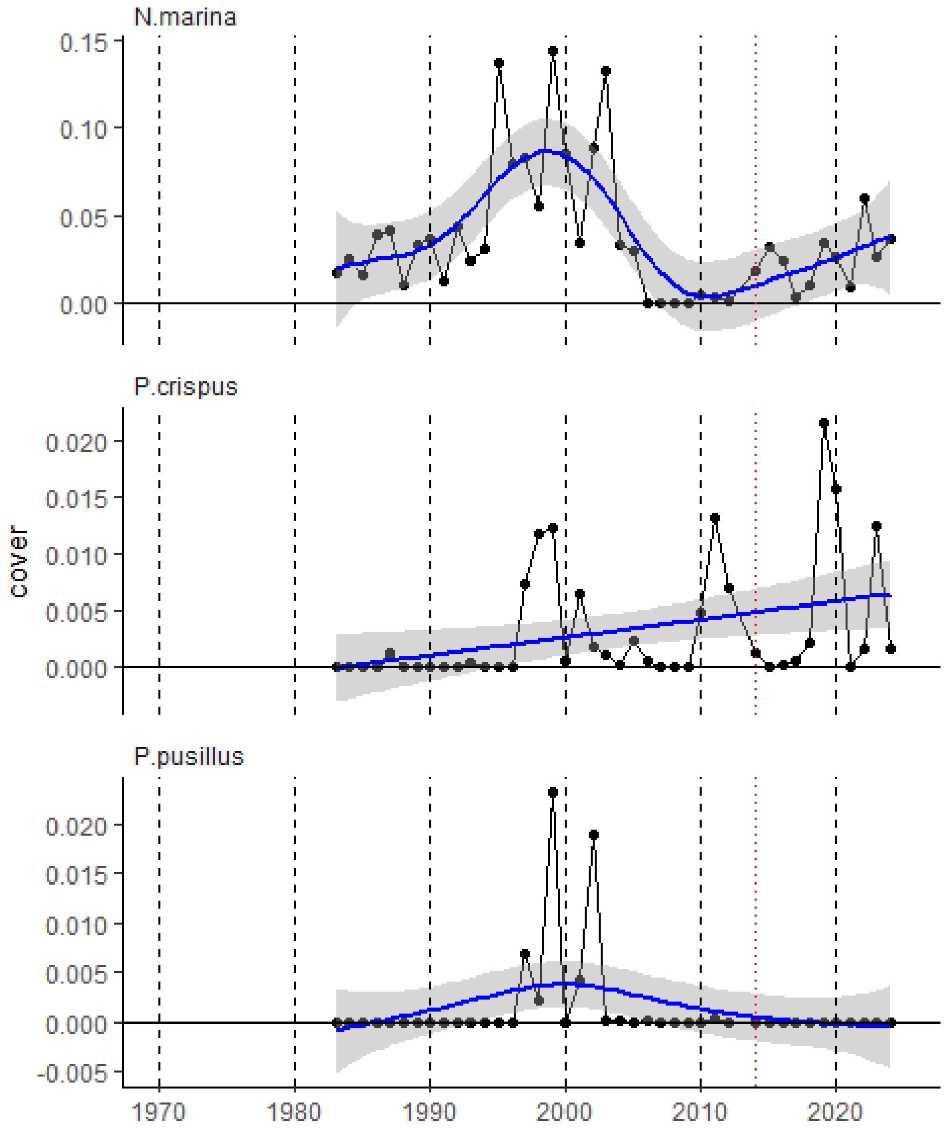


Figure 41. Trend in other macrophyte taxa in Hickling Broad 1983-2024 (note variable y axis scale).

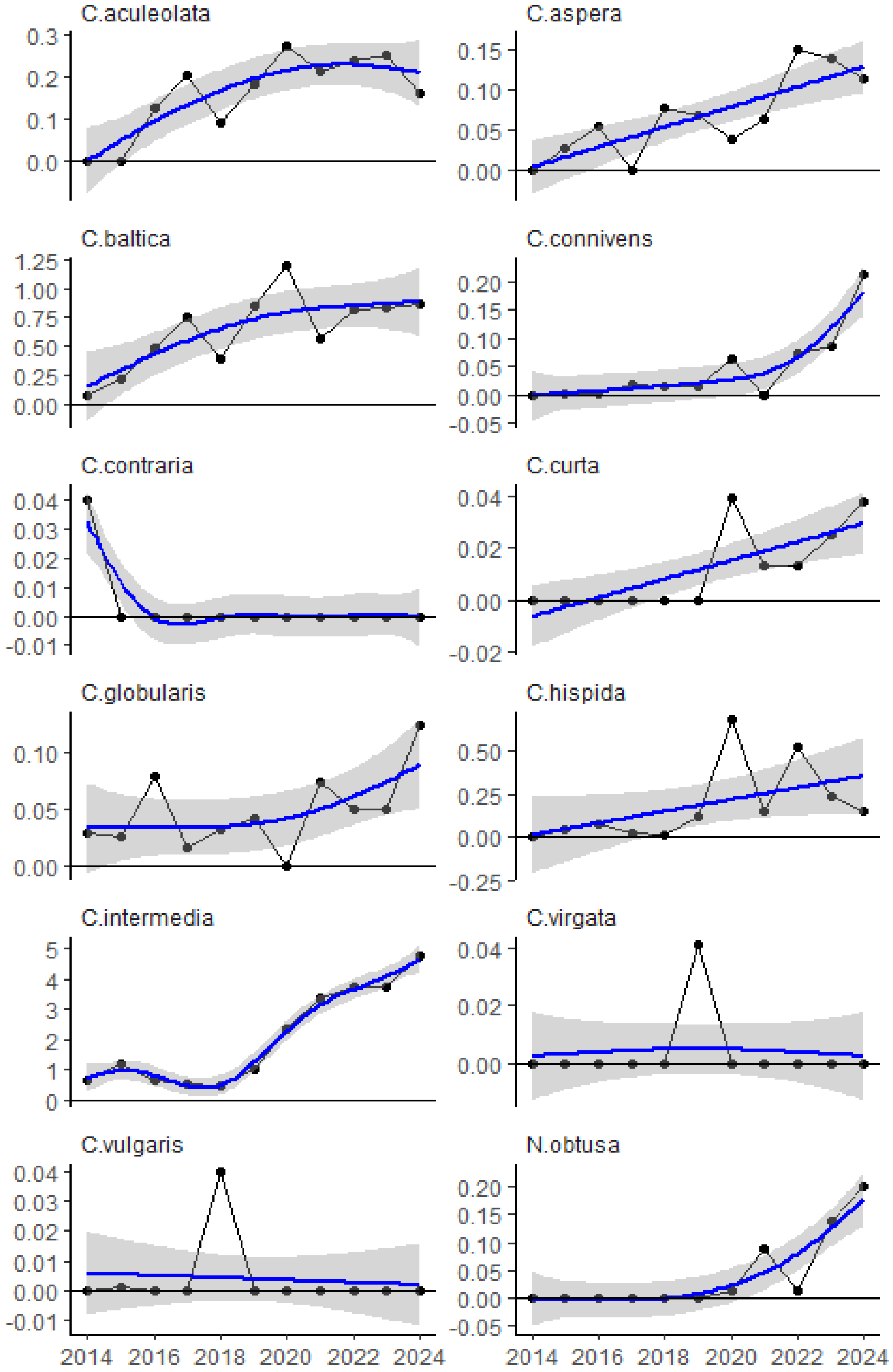


Figure 42. Trend in Charophyte species in Hickling Broad 2014-2024 (note variable y scale).

7.4 Spatial distribution

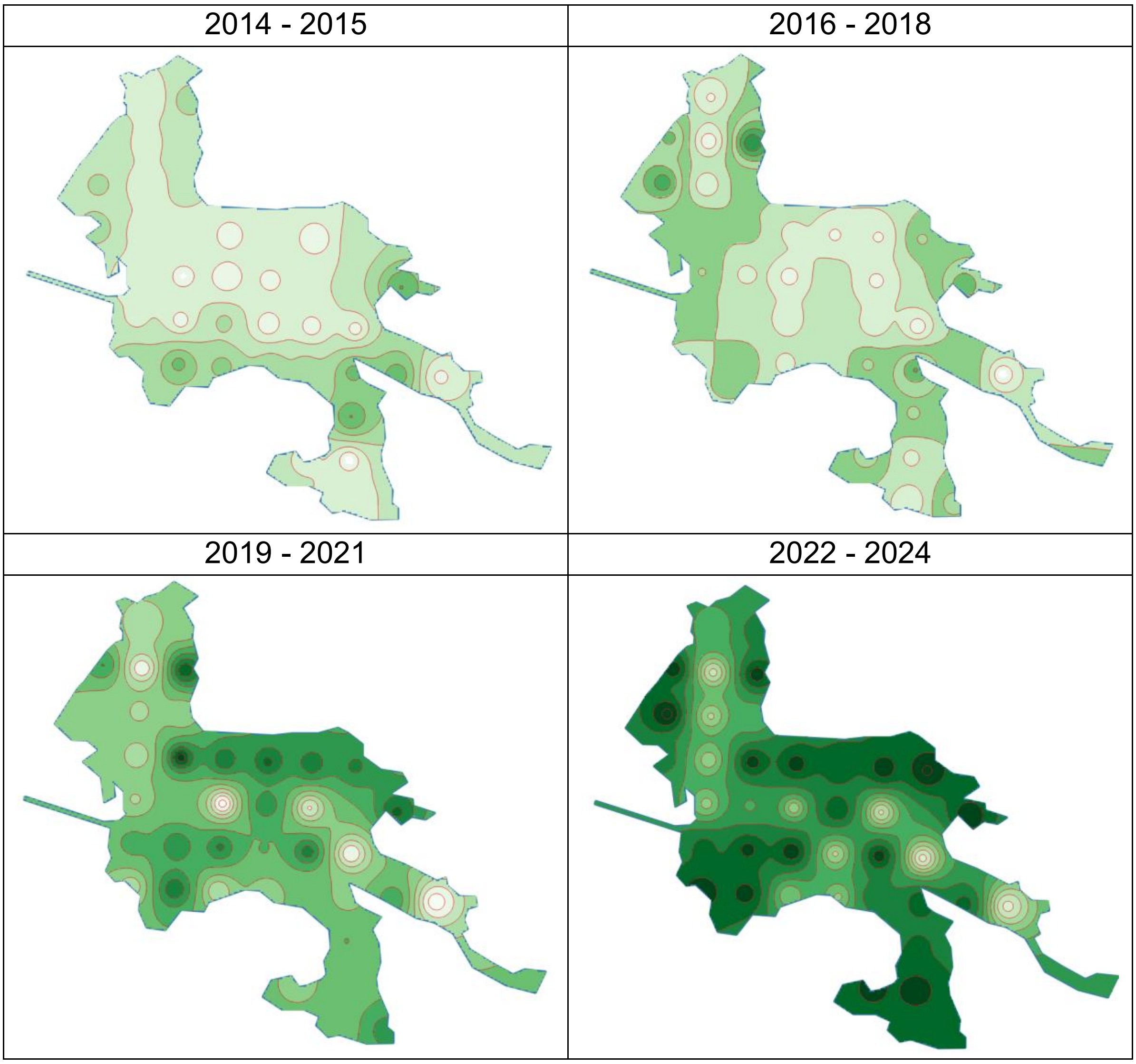
The spatial distribution of plants in Hickling Broad is shown in Figure 43. At the beginning of the recovery period (2014–2015), there is clear evidence of physical control by wind exposure, and potentially a requirement for adequate light, with the highest plant coverage at the upwind eastern end of the broad, as well as in sheltered bays and marginal areas. Over the following three years (2016–2019), plant abundance increased in these areas and expanded into the northwestern end of the broad, gradually spreading toward the more central regions. Between 2019 and 2021, coverage further extended into the central area, particularly along the shallower northern edge. By 2022–2024, plant abundance was high across most of the broad, except in the central dredged and cut channel, where coverage remained limited.

The changes in the spatial distribution of three key plant species in Hickling Broad are shown in Figure 44. The most abundant species, *Chara intermedia*, first appeared in the shallower and more sheltered eastern bays during 2014–2015, and then expanded to the northwestern edges between 2016 and 2019. It subsequently developed and reached peak abundance in the central areas—initially along the northern edge (2019–2021), and later spreading to include the southern and southwestern bays.

Although *Myriophyllum spicatum* was also dominant in the sheltered bays, it had a slightly wider distribution than *Chara* during 2014–2015. In contrast to *Chara*, *Myriophyllum* became most abundant in the central area between 2016 and 2018, before declining there and increasing in abundance in the western end of the broad and the northwestern bay during 2019–2021. Its distribution has since remained broadly similar, and although it is now found throughout the broad, it remains most abundant in the western and northwestern areas.

The distribution of *Najas marina*, a much less abundant species, has remained relatively unchanged from 2014 to 2024. It is primarily found along the sheltered northwestern edge and in the western side of the southeastern bay. This stable and limited distribution highlights its preference for sheltered conditions, likely reflecting its stiff and relatively brittle growth form.

The changing distributions of the two most abundant submerged aquatic species, *Chara intermedia* and *Myriophyllum spicatum*, support the earlier observation that *Myriophyllum* may provide the shelter necessary for *Chara* to establish. However, once *Chara* becomes established, it appears to outcompete *Myriophyllum*, resulting in the current pattern where the two species dominate different areas of the broad—*Myriophyllum* in the northwestern region, and *Chara* across most of the remaining open water areas.



2014 - 2015

2016 - 2018

2019 - 2021

2022 - 2024

Figure 43. Distribution of submerged macrophyte vegetation in Hickling Broad 2014-2014. Colours represent mean total abundance score for years shown split into 10 classes (<3 treated as zero). (Contours mark abundance intervals of 10)

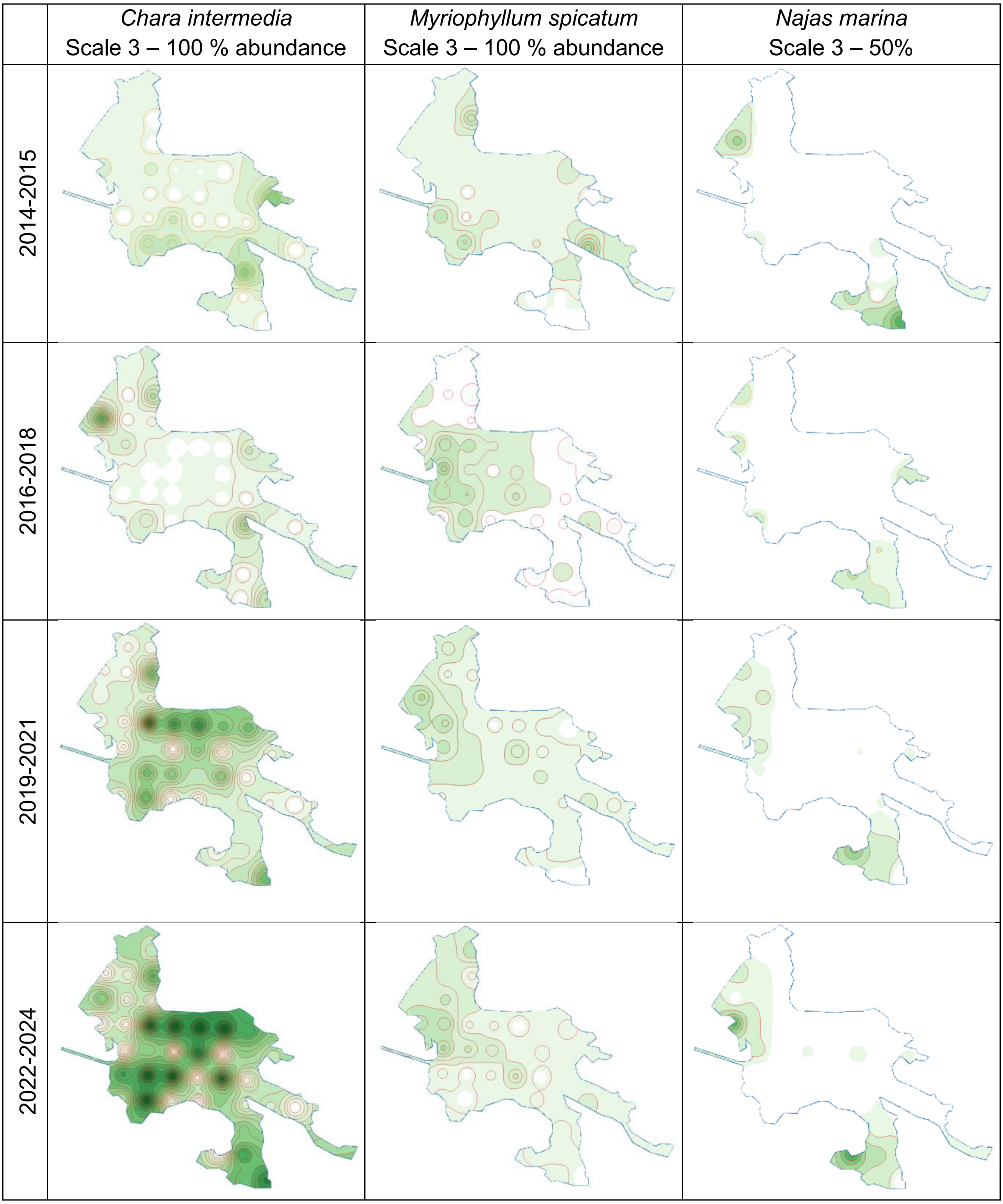


Figure 44. Distribution of abundance scores for a) Chara intermedia, b) Myriophyllum spicatum, c) Najas marina in Hickling Broad 2014 – 2024. Colours represent mean total abundance score for years shown split into 10 classes (3-100), note scale for Najas marina is 3-50. (Contours mark abundance intervals of 5)

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 situated close to both grazing marshes and the coast, Hickling supports large numbers of waterbirds (Figures 46–48). In the previous edition of this dossier (2016), we reported that since 2004, Hickling had supported nationally important numbers of wintering Teal, Greylag Goose, and Gadwall. Numbers of these species increased after 2000, consistent with national trends—particularly for feral Greylag, which has shown a dramatic rise across Great Britain since the early 1980s. Gadwall numbers have continued to grow, in line with the strong national upward trend. However, more recently, Teal and Greylag numbers at Hickling have declined. While the Teal decline mirrors the national pattern, Greylag numbers have continued to increase nationally, suggesting site-specific factors may be influencing their local decline. Wigeon have also shown a significant decline at Hickling since 2012, again matching national trends. Mallard numbers have declined markedly over the long term, consistent with a broader national decline in winter counts.

There are clear inter-annual and short-term fluctuations in bird numbers that can be closely linked to changes in aquatic vegetation. The most notable example is Coot, whose numbers closely tracked the rise and fall of Chara abundance—peaking at over 5,000 individuals in October 1999, with a secondary peak in 2003 during a partial recovery of Chara.

Waterbirds are both drivers and responders to changes in aquatic vegetation. This is evident in the parallel trends seen in diving ducks such as Tufted Duck and Pochard, whose numbers largely mirror those of Coot. These patterns likely reflect increased availability of plant-associated invertebrate prey during periods of high vegetation cover. Similar dynamics have been observed elsewhere; for example, dramatic crashes in diving duck populations at Lough Neagh have been linked to major declines in the invertebrate food base (Tománková et al. 2014). As omnivores, Coot in Hickling likely benefited from both direct grazing on vegetation and feeding on associated invertebrates.

However, despite their strong association with submerged vegetation, Coot numbers did not increase following the recovery of Chara after 2011. This is surprising given the substantial resurgence in aquatic vegetation during that period, a pattern that was reflected, albeit to a lesser extent, in the population responses of Pochard and Tufted Duck. In contrast, Mute Swan numbers at Hickling have increased during the recent phase of macrophyte expansion and appear to have replaced Coot as the primary grazer of aquatic vegetation

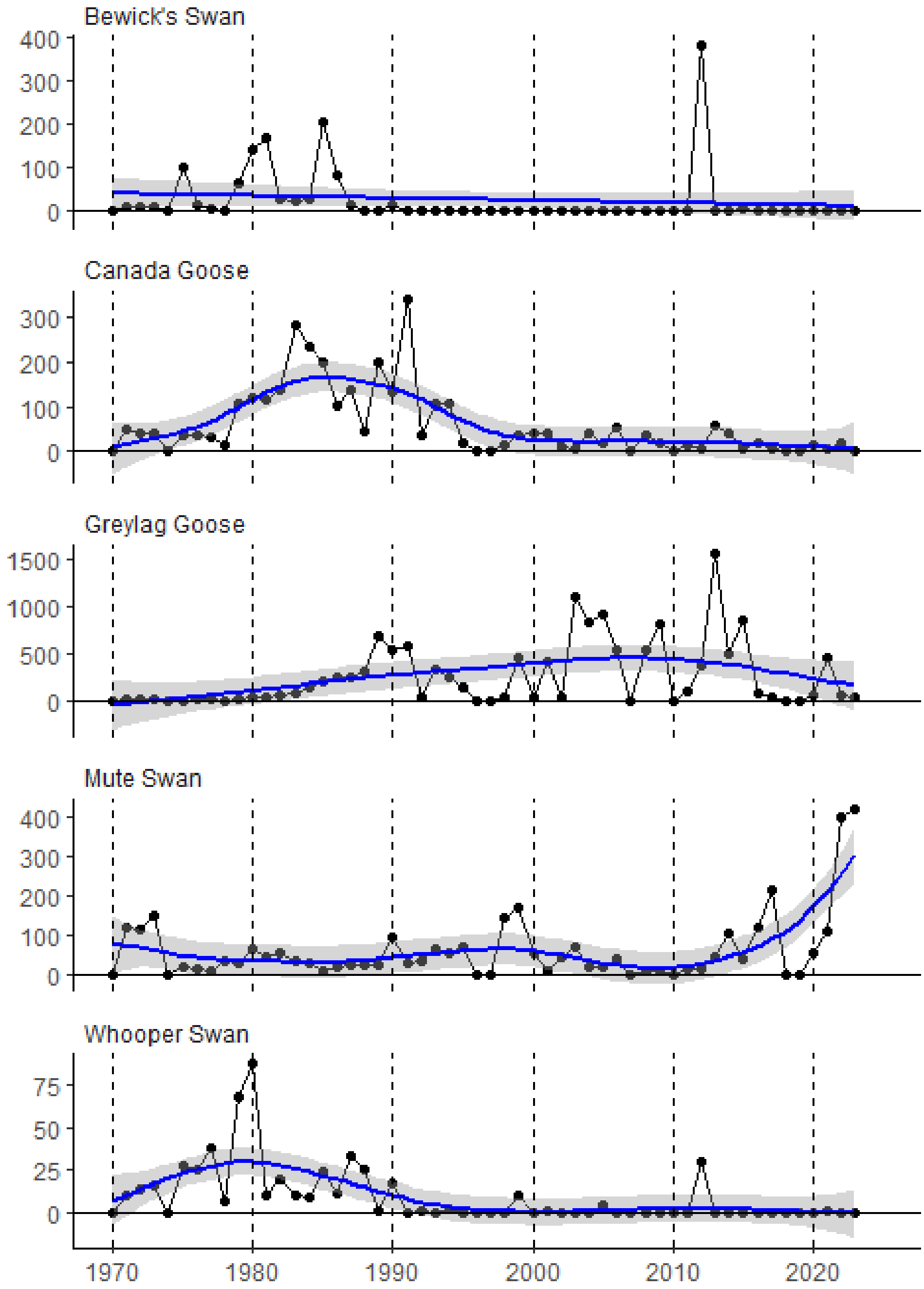


Figure 46. Changes in abundance (as maximum seasonal counts) of swans and geese in Hickling Broad since 1971 based on BTO WeBS counts.

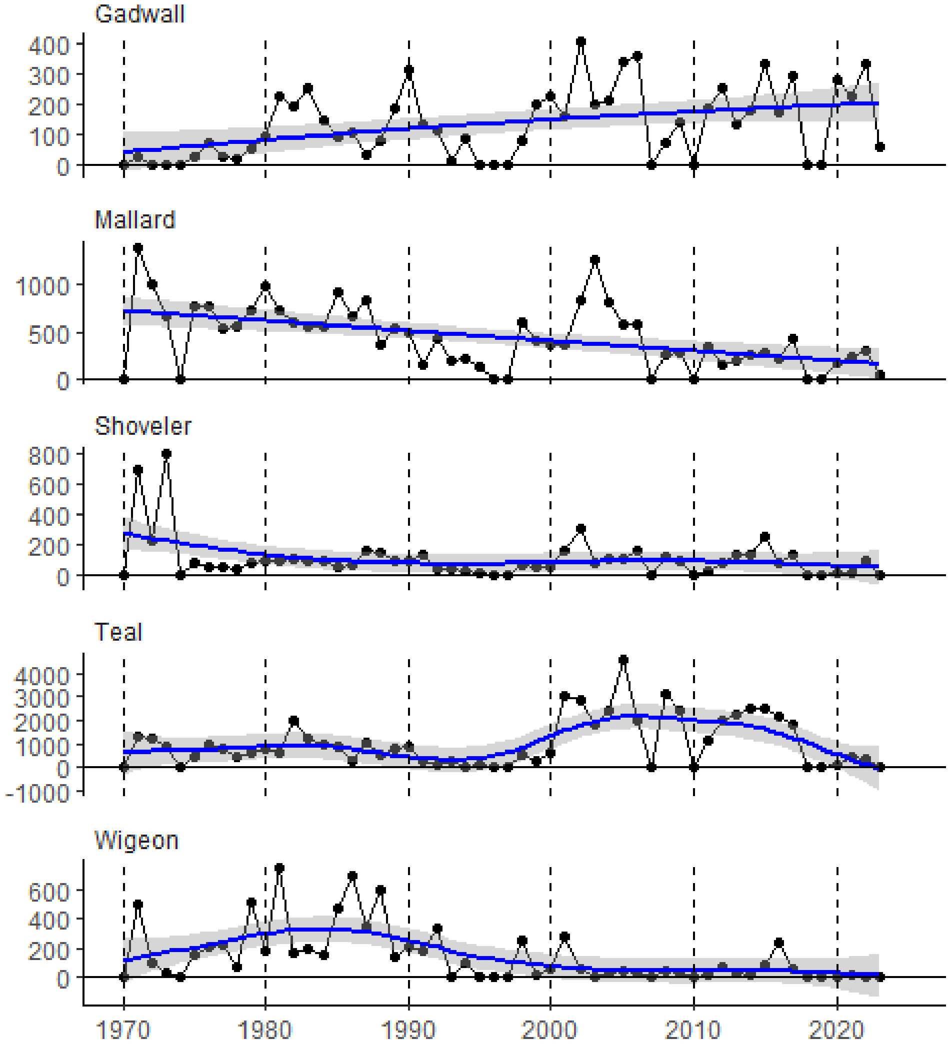


Figure 47. Changes in abundance (as maximum seasonal counts) of dabbling duck in Hickling Broad since 1971 based on BTO WeBS counts.

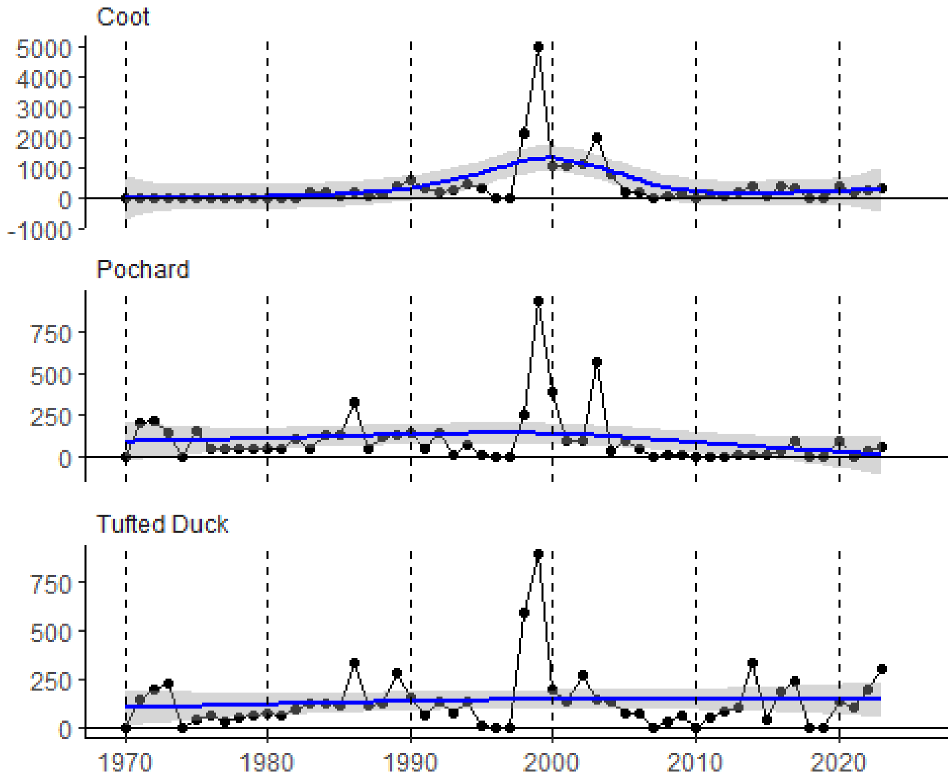


Figure 48. Changes in abundance (as maximum seasonal counts) of coot and major diving duck in Hickling Broad since 1971 based on BTO WeBS counts.

11 Summary of Interactions

This section outlines the key ecological interactions occurring in Hickling Broad, drawing on the long-term monitoring data presented in previous sections covering c50 year monitoring period. It focuses on changes in the submerged plant community, with particular attention to *Chara intermedia*, which has exhibited pronounced long-term fluctuations. These are characterised by an approximate 10-year cycle of build-up and collapse, with peak abundances observed in 1999 and again in 2022 with high plant abundance remaining though to 2024.

Given the proximity of Hickling Broad to Horsey Mere, and their shared location within the tidally influenced Upper Thurne system (see section 1 for details), it is informative to compare ecological changes in both lakes when addressing this question. It should be remembered that Horsey is significantly influenced by saline and ochrous water from land drainage pumps, however this comparison provides a broader context for understanding site-specific dynamics.

The following sections then focus more specifically on Hickling Broad, offering a detailed examination of its ecological changes over time and culminating in the development of a conceptual model. This model is intended to aid understanding of the factors influencing *Chara* and other macrophyte dynamics within the Broad.

11.1 Relationship between chlorophyll and total phosphorus

Hickling Broad is shown in Figure 49, alongside comparative data from other Thurne Broads and the European reference line for very shallow lakes (Phillips et al., 2008). Chlorophyll-a concentrations in Hickling and Horsey Mere are broadly similar, although chlorophyll yield—that is, the chlorophyll-to-phosphorus ratio—is slightly lower in Horsey. The figure highlights the elevated chlorophyll yield in Hickling, supporting the interpretation that recent reductions in chlorophyll-a are most likely a direct response to declining phosphorus concentrations.

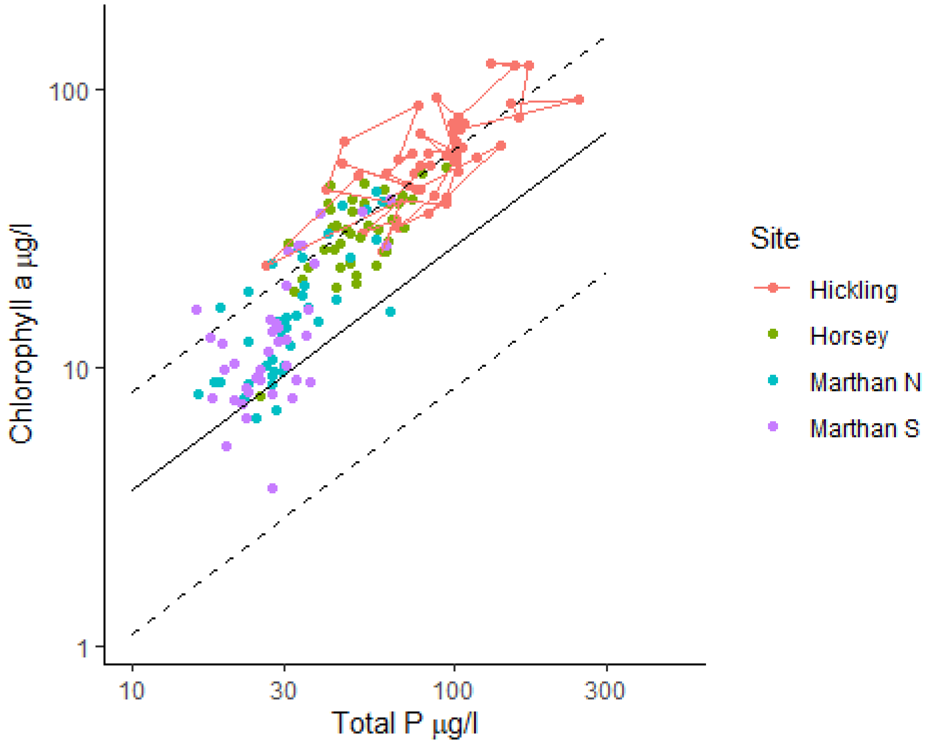


Figure 49. Relationship between annual mean chlorophyll a and total phosphorus for Thurne Broads, showing trajectory of change in Hickling Broad. Diagonal dotted lines show relationship for European lakes (Phillips et al. 2008).

11.2 Comparisons between Hickling Broad and Horsey Mere

In this section, we compare long-term (yearly) and seasonal (monthly) trends for Hickling using *difference smooths*. To do this, Generalized Additive Models (GAMs) are fitted to the data within the same model framework, and the difference between the resulting smooth terms is plotted. These difference smooths are not intended to display the trend patterns themselves, but rather to highlight deviations between them. When the long-term and seasonal trends are identical, the difference smooth will appear as a flat line at zero.

When the difference is positive, it indicates that the first site (e.g., Hickling) has a higher value than the comparison site; when the difference is negative, the first site has a lower value. This approach emphasizes relative variation in trends rather than absolute values.

11.2.1 Total Phosphorus

Total phosphorus concentrations in Hickling Broad have shown a marked decline over time, while levels in Horsey Mere have remained relatively stable (see Section 5.1). In both lakes, phosphorus concentrations are consistently highest during the summer months, reflecting typical seasonal patterns. Comparisons of the long-term (yearly) and seasonal (monthly) smooths fitted using GAMs for each lake (Figure 50) highlight both shared patterns and key differences. Notably:

- Before 1990, Hickling Broad exhibited substantially higher total phosphorus levels than Horsey Mere.
- During the 1990s, when *Chara* became dominant for the first time in Hickling, phosphorus concentrations in Hickling dropped well below those in Horsey.
- Following the loss of *Chara*, total phosphorus levels in both lakes became similar.
- After *Chara* re-established post-2015, Hickling again showed lower total phosphorus compared to Horsey Mere.

Despite these interannual differences, the seasonal pattern of total phosphorus — with peaks in summer — remained consistent in both lakes throughout the study period.

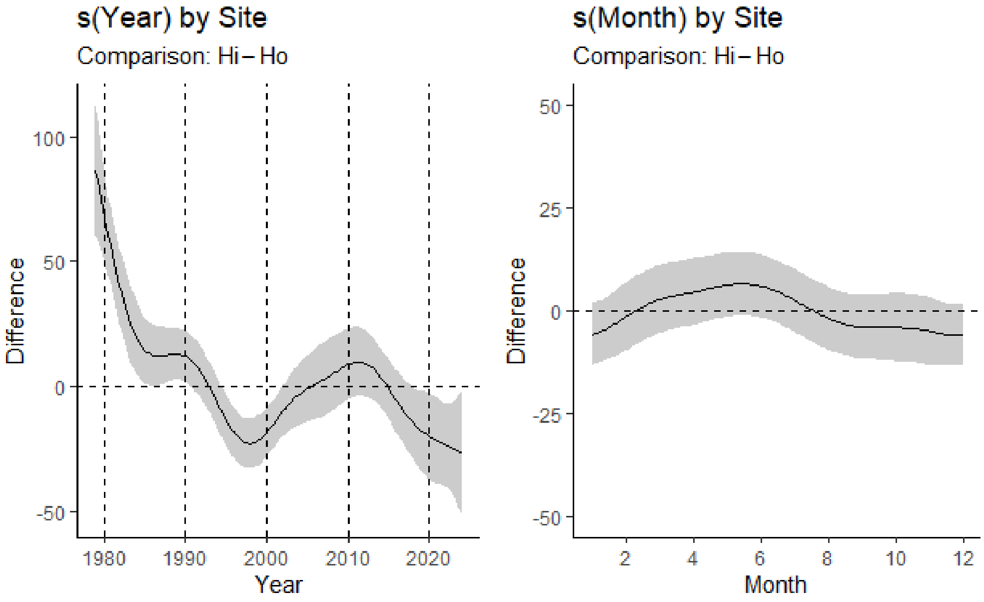


Figure 50. Estimated differences in smooth terms (±95% confidence interval) between Hickling Broad and Horsey Mere (Hi – Ho). a) Long-term smooth of the total phosphorus over calendar year; b) Seasonal smooth over month.
Positive values indicate higher values at Hickling Broad relative to Horsey Mere.

In many shallow lakes, total phosphorus concentrations are closely linked to phytoplankton biomass, particularly during the summer months. This relationship is often driven by the release of phosphorus from sediments, which can exceed the immediate uptake capacity of phytoplankton. However, while chlorophyll-a and total phosphorus are clearly associated, regression analyses indicate a more complex interaction in both Hickling Broad and Horsey Mere.

A multiple regression model for Hickling Broad, using standardised predictors—suspended solids, chlorophyll-a, and soluble reactive phosphorus—was highly significant ($R^2 = 0.51$, $p < 0.001$) (Table 5a). Of the three predictors, only suspended solids had a statistically significant effect ($p < 0.001$), highlighting sediment resuspension as the dominant driver of total phosphorus in this lake. In contrast, chlorophyll-a was not a significant predictor, suggesting that phytoplankton contributes relatively little to total phosphorus compared to particulate material from sediments.

In Horsey Mere, a similar model yielded a lower but still significant fit ($R^2 = 0.366$, $p < 0.001$) (Table 5b). In this case, both suspended solids and chlorophyll-a had significant effects, though the standardised coefficient for chlorophyll-a was much smaller. This indicates that, while sediment resuspension, and potentially ochre additions from the Brograve drainage pump, remains the primary source of phosphorus, phytoplankton also contributes to the particulate phosphorus pool in Horsey Mere.

In both lakes, soluble reactive phosphorus was not a significant predictor, implying that any phosphorus released from the sediments is likely rapidly removed from the water column—either through uptake by macrophytes and algae, or by adsorption to iron-rich sediments that characterise these systems.

Table 5. Model predicting total phosphorus in a) Hickling Broad and b) Horsey Mere

This table summarizes the results of a standardised multiple regression where:
- TP = Total Phosphorus (response variable)
- SS = Suspended Solids (predictor)
- Chl = Chlorophyll-a (predictor)
- PO4 = Soluble Reactive Phosphorus (predictor)

Model formula: TP ~ SS_std + Chl_std + PO4_std

a) Hickling Broad Regression Coefficients

Predictor	Estimate	Std. Error	t-value	p-value
(Intercept)	0.087	0.001	59.915	<0.000***
Suspended Solids (SS_std)	0.028	0.001	19.268	<0.000***
Chlorophyll-a (Chl_std)	0.003	0.001	1.759	0.079'
Phosphate (PO4_std)	-0.002	0.001	-1.077	0.282

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

Model Fit Statistics
Residual standard error: 0.02869 on 387 degrees of freedom
Multiple R-squared: 0.5047
Adjusted R-squared: 0.5008
F-statistic: 131.4 on 3 and 387 DF, p-value: < 0.001

b) Horsey Mere Regression Coefficients

Predictor	Estimate	Std. Error	t-value	p-value
(Intercept)	0.052	0.001	41.783	<0.001***
Suspended Solids (SS_std)	0.017	0.001	11.489	<0.001***
Chlorophyll-a (Chl_std)	0.004	0.001	2.635	0.009**
Phosphate (PO4_std)	0.001	0.001	0.761	0.447

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

Model Fit Statistics
Residual standard error: 0.02517 on 404 degrees of freedom
Multiple R-squared: 0.3707
Adjusted R-squared: 0.366
F-statistic: 79.3 on 3 and 404 DF, p-value: < 0.001

11.2.2 Comparison of chlorophyll a

Like total phosphorus, chlorophyll-a concentrations have declined substantially in Hickling Broad over the long term. In Horsey Mere, concentrations have also decreased, but the change has been less pronounced (see Section 5.9). Figure 51a compares the long-term and seasonal smooths fitted to chlorophyll-a concentrations in the two lakes.

The interannual trends in chlorophyll-a reveal more pronounced differences than those seen for total phosphorus. In 1980, chlorophyll-a levels were substantially higher in Hickling than in Horsey, but this difference declined sharply over the 1980s. By 1990, both lakes exhibited similar concentrations. This parity persisted during the first period of *Chara* dominance in Hickling. Around 2000, Hickling briefly showed slightly higher chlorophyll-a levels, but from the mid-2000s onward, concentrations in Hickling declined again relative to Horsey. By 2015, when the second phase of *Chara* recovery began, chlorophyll-a concentrations in Hickling were significantly lower than in Horsey Mere.

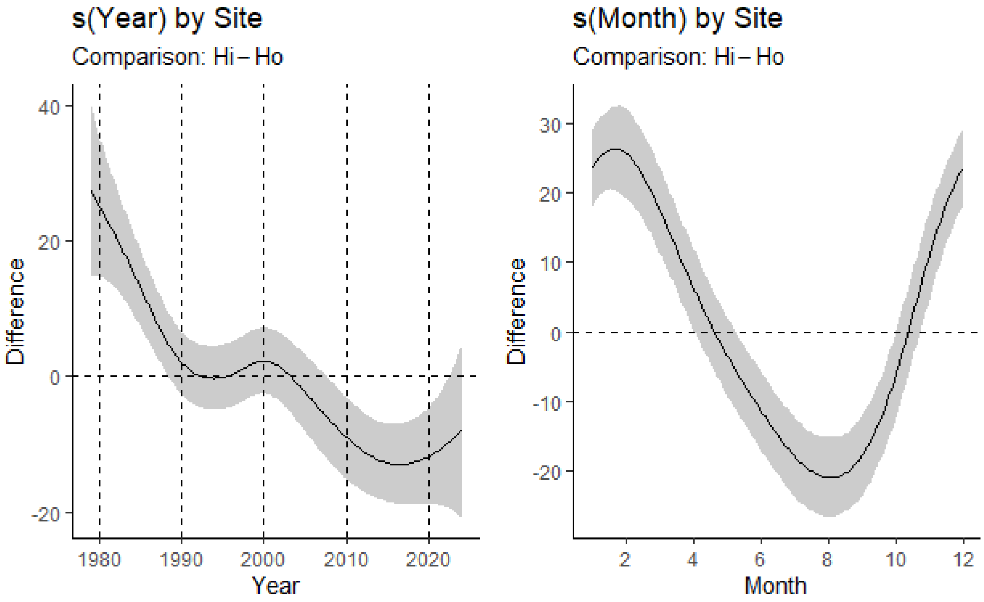


Figure 51. Estimated differences in smooth terms (±95% confidence interval) between Hickling Broad and Horsey Mere (Hi – Ho). a) Long-term smooth of chlorophyll a over calendar year; b) Seasonal smooth over month.
Positive values indicate higher values at Hickling Broad relative to Horsey Mere.

In contrast to total phosphorus, there is a marked difference in the seasonal patterns of chlorophyll-a between the two lakes (Figure 51b). In Hickling Broad, peak phytoplankton biomass occurs early in the year (February), whereas in Horsey Mere, the peak is observed in September, with only a modest spring increase in March/April. This divergence likely reflects differences in hydrology: Horsey receives substantial land drainage inflows during winter and spring, which flush the lake and may delay phytoplankton growth. In contrast, Hickling experiences lower water exchange, driven mainly by tidal influence, allowing earlier blooms. Additionally, the dominance of *Chara* in Hickling likely contributes to suppressed summer phytoplankton biomass through shading and nutrient competition.

11.2.3 Comparison of transparency (Secchi disc depth)

The long-term trends in water transparency diverge notably between the two lakes: Hickling Broad has experienced a significant increase in clarity, while Horsey Mere has shown a significant decline (see Section 5.11). Given the ecological importance of underwater light availability, both annual and seasonal trends in transparency are shown in Figure 52, where smooths are plotted as deviations from each lake's overall mean.

While the contrasting overall trends are clearly visible, both lakes exhibit some aspects of a shared temporal pattern—a peak in transparency around 2020 and a trough around 2010. However, the seasonal dynamics differ markedly: in Hickling Broad, the highest water clarity occurs during the winter months, despite the presence of a winter phytoplankton bloom. In contrast, Horsey Mere displays an early summer peak (May) in transparency, a consequence of the spring and subsequent summer phytoplankton populations.

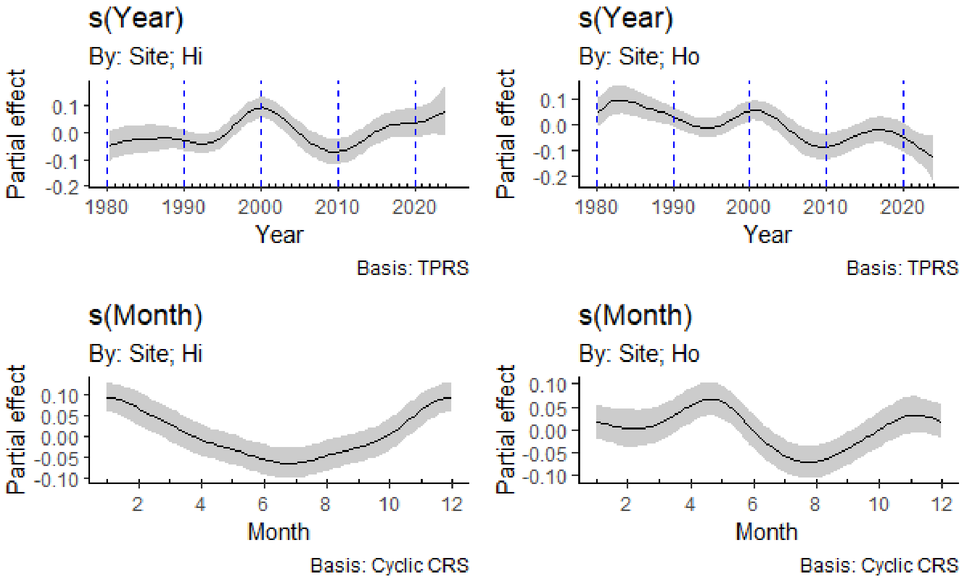


Figure 52. Partial effect smooths showing long-term (Year) and seasonal (Month) trends for Secchi disc transparency in Hickling Broad and Horsey Mere.

Figure 53a compares the long-term difference in transparency between the lakes, showing a shift from lower clarity in Hickling during the 1980s, to similar conditions between 2000 and 2010, and finally to clearer water in Hickling post-2020. Figure 50b highlights the differences in seasonal transparency, with Horsey exhibiting a distinct early summer peak that is absent in Hickling.

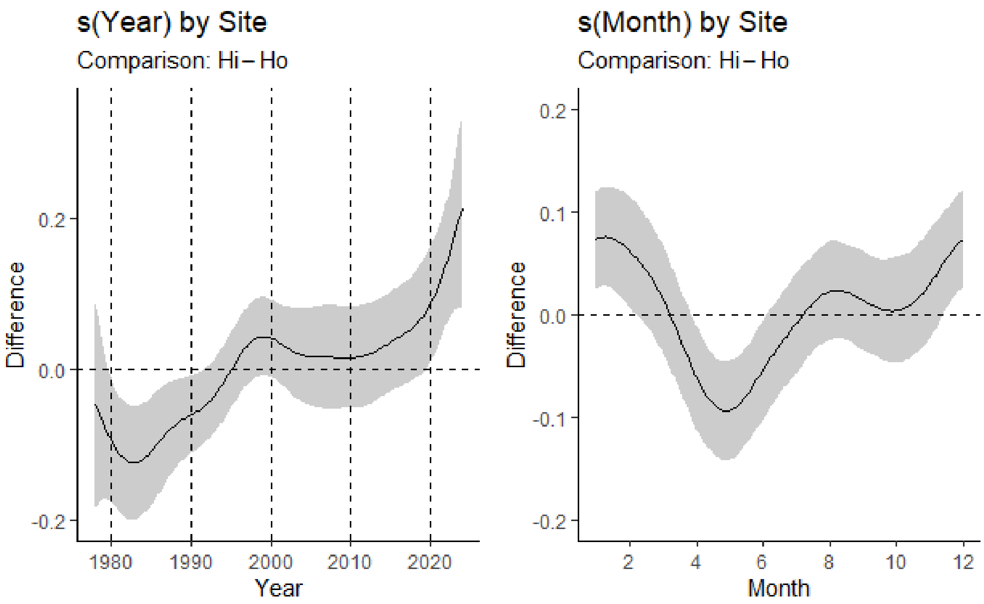


Figure 53. Estimated differences in smooth terms (±95% confidence interval) between Hickling Broad and Horsey Mere (Hi – Ho). a) Long-term smooth of Secchi disc transparency over calendar year; b) Seasonal smooth over month.
Positive values indicate higher values at Hickling Broad relative to Horsey Mere.

These differences likely reflect variation in the dominant drivers of water clarity in the two systems. Results from multiple regression models (Table 6), using standardised predictors of chlorophyll-a and suspended sediment, provide further insight. Both models were statistically significant ($p < 0.001$), though the model fit was stronger for Hickling ($R^2 = 0.39$) than for Horsey ($R^2 = 0.11$).

In Hickling Broad, both predictors were highly significant ($p < 0.001$), with standardised coefficients of –0.57 for suspended sediment and –0.21 for chlorophyll-a. This indicates that resuspended sediment is the primary factor limiting transparency, with phytoplankton playing a secondary role. In Horsey Mere, chlorophyll-a was again a significant predictor (–0.22, $p < 0.001$), but suspended sediment had a smaller and less significant effect (–0.17, $p = 0.004$). These results suggest that, unlike Hickling, Horsey's water clarity is more strongly influenced by phytoplankton than by sediment resuspension.

Table 6. Model predicting transparency in a) Hickling Broad and b) Horsey Mere

This table summarizes the results of a standardised multiple regression where:

- SD = Secchi Depth (response variable)
- Chl = Chlorophyll-a (predictor)
- SS = Suspended Solids (predictor)

Model formula: SD ~ Chl_std + SS_std

a) Hickling Broad Regression Coefficients

Predictor	Estimate	Std. Error	t-value	p-value
(Intercept)	0.000	0.041	0.000	1
Chlorophyll-a	-0.210	0.041	-5.110	<0.001***
Suspended Solids	-0.565	0.041	-13.769	<0.001***

Signif. codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Model Fit Statistics

Residual standard error: 0.7801 on 366 degrees of freedom
Multiple R-squared: 0.3948
Adjusted R-squared: 0.3915
F-statistic: 119.4 on 2 and 366 DF, p-value: < 0.001

b) Horsey Mere Regression Coefficients

Predictor	Estimate	Std. Error	t-value	p-value
(Intercept)	0.000	0.048	0.000	1.000
Chlorophyll-a	-0.219	0.058	-3.776	<0.001***
Suspended Solids	-0.167	0.058	-2.890	0.004**

Signif. codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Model Fit Statistics

Residual standard error: 0.9423 on 382 degrees of freedom
Multiple R-squared: 0.1166
Adjusted R-squared: 0.112
F-statistic: 25.21 on 2 and 382DF, p-value: < 0.001

11.2.4 Hickling and Horsey, a summary of their differences.

Thus, in summary, Hickling Broad emerges as a lake where the primary driver of eutrophication—phosphorus—has declined significantly, whereas in Horsey Mere, levels have remained largely unchanged. This difference underscores the localised ecological impact of the disappearance of the gull roost at Hickling (Bales et al., 1993).

The hydrological differences between the two lakes (see Section 1) help explain their contrasting seasonal phytoplankton dynamics: Horsey shows the typical pattern for very shallow lakes, with spring and summer chlorophyll peaks, whereas Hickling exhibits mostly a winter or very early spring peak.

Following the decline in total phosphorus, both lakes now typically show undetectable levels of soluble reactive phosphorus. In Hickling, the winter high concentrations that once supported the winter/spring phytoplankton bloom have largely disappeared—with the notable exception of 2000, a point addressed in the following section. As a result of this decline, chlorophyll-a concentrations in Hickling have converged with those in Horsey, reflecting a shift toward more nutrient-limited conditions.

These changes have led to improved water clarity in Hickling, although the data show that resuspended sediment remains the more significant constraint on transparency in this lake, compared to phytoplankton, probably due to it being both larger and shallower and thus more subject to wind disturbance.

Being shallower than Horsey, submerged plants in Hickling may have been more responsive to these changes. The improved light environment and reduced nutrient load likely enabled the conditions necessary for the first large-scale colonisation of *Chara intermedia* in the early 1990s. In contrast, Horsey Mere did not experience *Chara* dominance during this period, although an increase in *Myriophyllum spicatum* growth was observed.

The period of *Chara* dominance in Hickling persisted until 2000. The causes of its subsequent decline—and its later recovery—are not obviously explained by differences between these two lakes and are explored in more detail in the next section.

11.3 The influence of meteorological conditions

To provide broader environmental context, long-term meteorological data from the UK Met Office station at Lowestoft were examined, alongside records of river discharge and water level, all of which exert a background influence on the ecological dynamics within the Hickling system (Figure 54).

As expected, the long-term meteorological data reveal an overall increase in temperature of approximately 2 °C, accompanied by increases in both rainfall and hours of sunshine—trends consistent with broader climate change patterns observed across the UK. There is some indication of a step change in rainfall and sunshine hours during the early 1990s, and a further step increase in river discharge around the end of 1999. Aside from these discontinuities, fitted GAM smooths suggest relatively steady trends over the full monitoring period.

Water level records begin in the 1990s, with the earliest data available from the River Bure at Acle Bridge, several years before consistent records were established for Hickling Broad itself. Despite this, the data show clear long-term patterns that closely

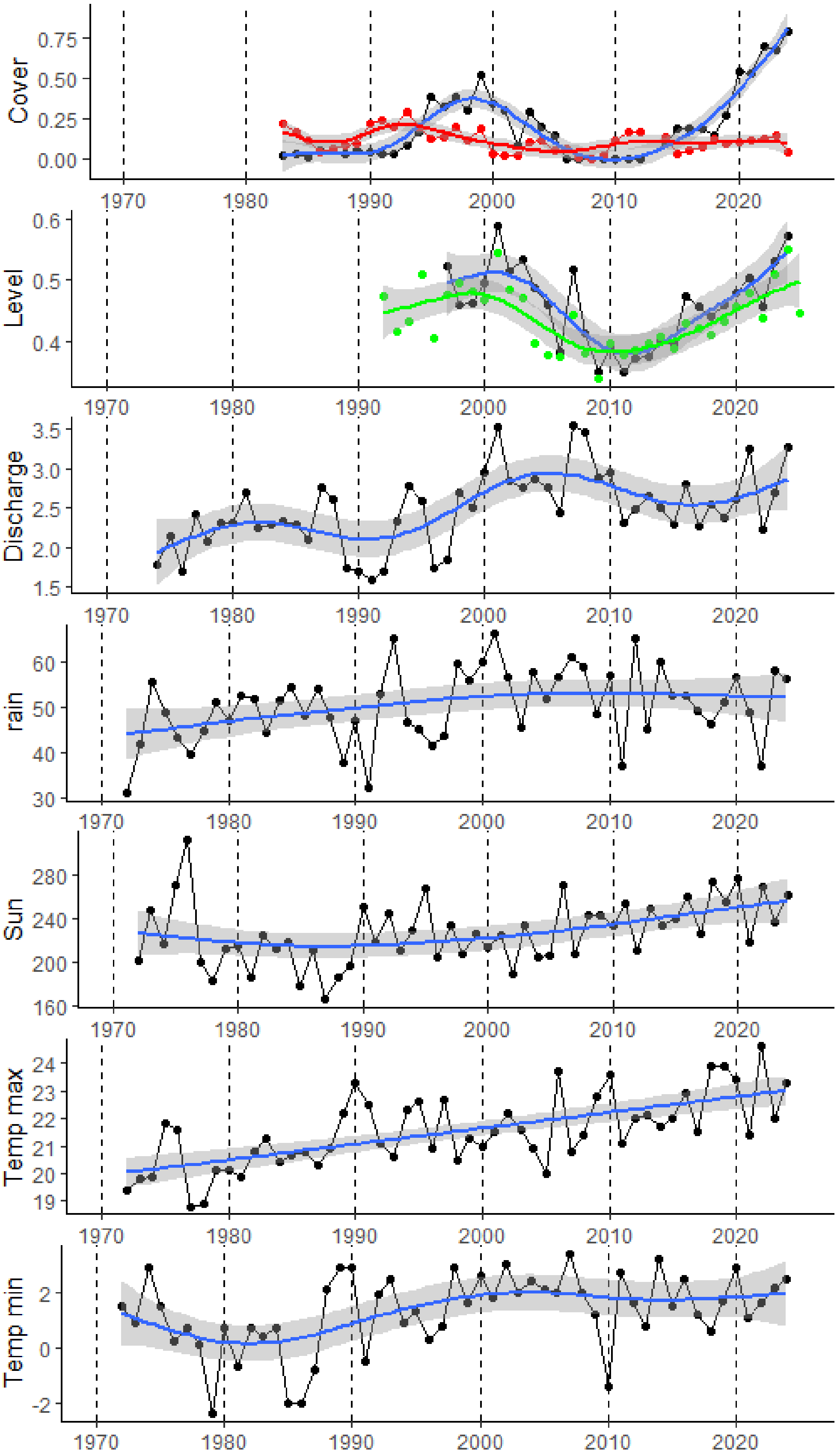


Figure 54. Time series for annual mean values for a) macrophyte cover (blue Chara, red M.spicatum), b) water levels (blue Hickling, green R.Bure at Acle Bridge), c) river discharge (R.Bure Horstead Mill), d) rainfall, e) hours of sunshine (95th percentile),f) maximum air temperature, g) minimum air temperature.

mirror fluctuations in *Chara* abundance—specifically, elevated water levels during the early 2000s, a subsequent decline, and a renewed rise from around 2010 onward. In contrast, no consistent long-term similarities are observed between trends in rainfall or river discharge and those in water level. This suggests that water levels in Hickling Broad are primarily influenced by tidal forcing from the North Sea, transmitted via the interconnected tidal rivers Bure and Thurne. The striking similarity between long-term water level patterns and fluctuations in *Chara* abundance is considered further in the following sections.

11.4 The influence of water quality

Annual average values for key water quality indicators are shown in Figure 55. As outlined previously and in Section 5, these data show a long-term decline in total phosphorus and chlorophyll-a (a proxy for phytoplankton biomass), alongside an increase in water transparency, measured by Secchi disc depth. This pattern is indicative of a reduction in eutrophication, which, in a very shallow lake such as Hickling, would typically be expected to support increased growth of submerged vegetation. However, both increases and declines in plant abundance must be considered to explain the pronounced fluctuations in charophyte dominance observed over time.

Notably, during the period of low *Chara* abundance between the two major peaks (circa 2005–2010), there was an increase in total phosphorus (Figure 55b), a slowing in the decline of chlorophyll-a (Figure 55c), a reduction in water transparency (Figure 55d), and a marked rise in suspended solids (Figure 55e). In other words, when *Chara* was largely absent, multiple indicators of eutrophication increased. While this correlative evidence does not establish causation, the pattern is consistent with the interpretation that deteriorating water quality may have contributed to the collapse of *Chara* populations. This occurred despite the presence of unusually low water levels during the same period—conditions that would typically favour submerged macrophyte growth by enhancing light availability.

There is no clear external pressure that accounts for the observed rise in phosphorus during this period, which might otherwise have triggered a cascade of increased phytoplankton, reduced water clarity, and subsequent vegetation loss. A more plausible explanation is that the loss of *Chara* itself contributed to elevated phosphorus levels, either through nutrient release from decaying biomass or reduced nutrient uptake. Previous analysis has shown that, in Hickling Broad, suspended sediment—not phytoplankton—is the dominant contributor to total phosphorus. The absence of dense plant cover may have allowed increased sediment resuspension, particularly under low water levels, thereby reinforcing a negative feedback loop that impaired water clarity and inhibited *Chara* recovery.

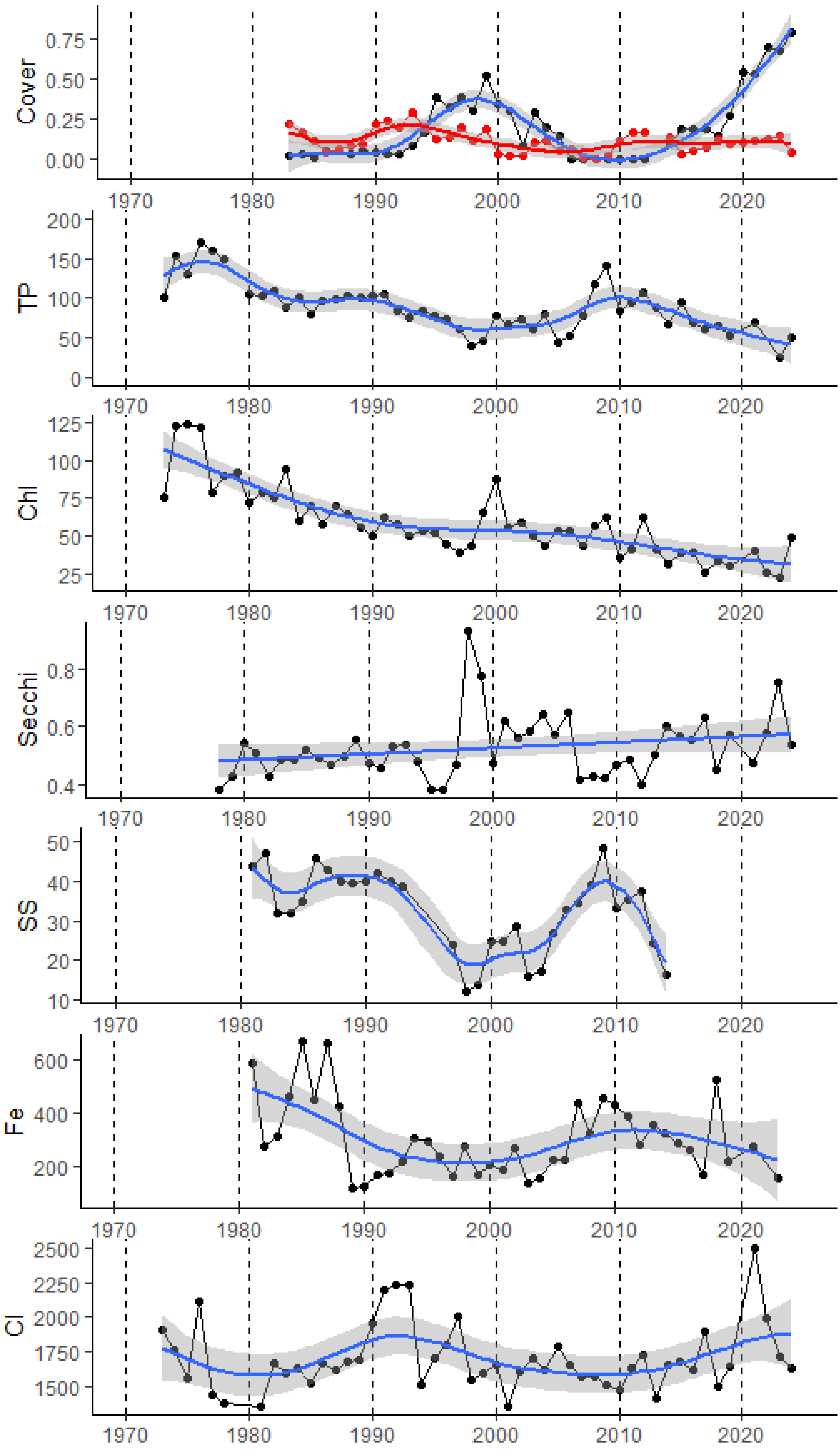


Figure 55. Time series for annual mean values for a) macrophyte cover (blue Chara, red M.spicatum), b) total P, d) chlorophyll a, e) Secchi depth, f) suspended solids, g) total iron, h) chloride.

Using annual mean values as shown in Figure 55 masks potential seasonal patterns. To better visualize seasonal effects, Figure 56 presents two-dimensional shaded contour plots derived from a GAM model that captures both year-to-year variation and seasonal changes within each year. This clearly shows a marked concordance of low water level, low Secchi transparency, high suspended solids and total phosphorus during the growing season (April - September). In contrast, although summer chlorophyll a is clearly lower during the summer the correspondence with water clarity and growth of Chara is less marked.

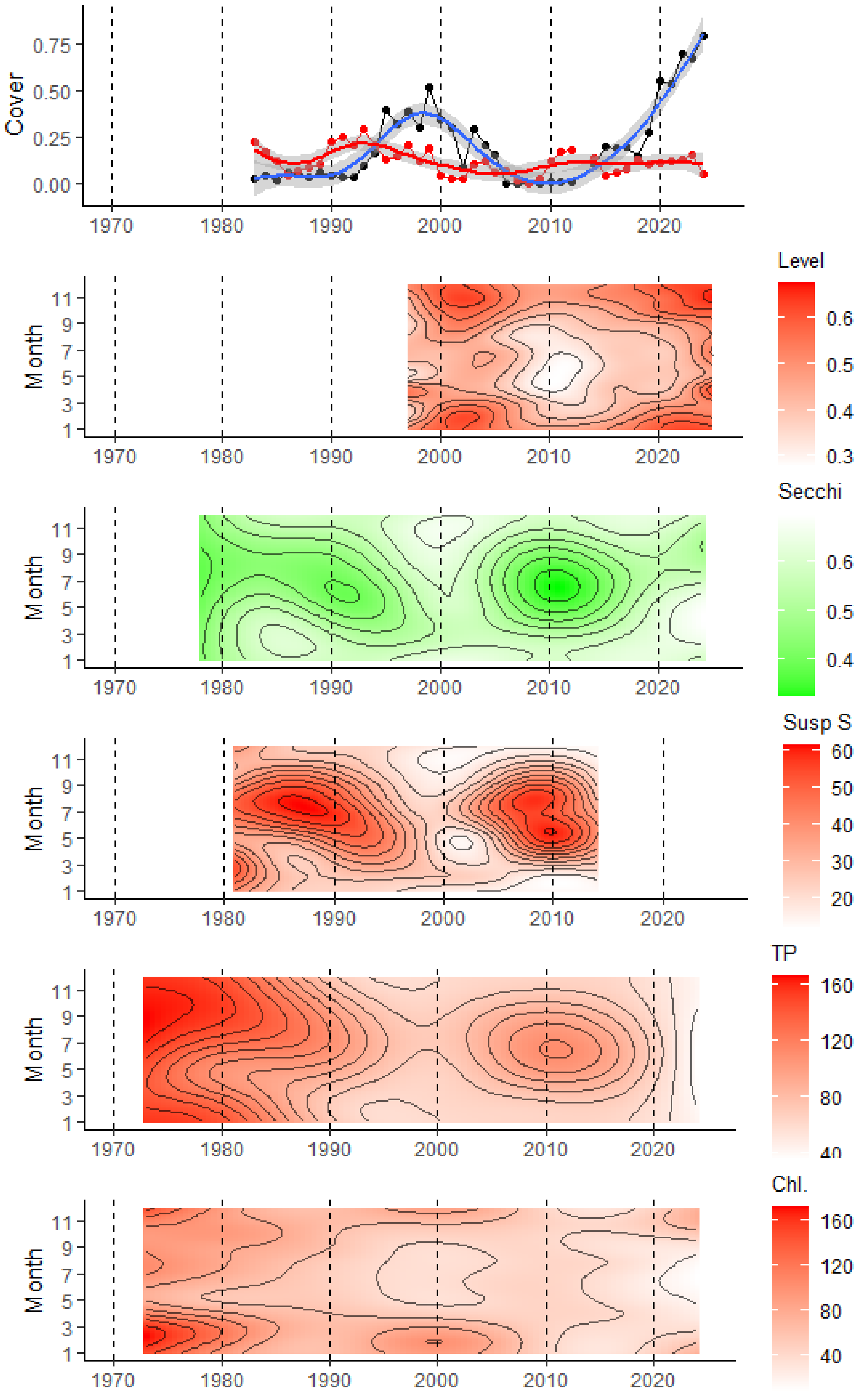


Figure 56. Relationships between a) macrophyte cover (blue Chara, red M.spicatum), b) water level, c) Secchi depth, d) suspended solids, e) total phosphorus, f) chlorophyll a. (d - f presented as a surface generated from 2d GAM smooths).

11.5 Influence of water birds

Figure 57 compares the population dynamics of Coot and Mute Swan—the two dominant submerged plant-grazing waterbirds at Hickling Broad—with trends in macrophyte (Chara) and phytoplankton (chlorophyll-a) abundance. Both bird species show population peaks that are closely aligned with periods of high *Chara* cover. In the first peak (1999), Coot were the more abundant grazer, whereas in the second peak (2020), Mute Swan numbers predominated.

These waterbirds are known to be significant grazers of *Chara* and can substantially reduce its overwintering biomass, potentially limiting the capacity for spring regrowth and acting as negative feedback on macrophyte dominance. Grazing activity is also likely to contribute to internal phosphorus loading, either through sediment disturbance or nutrient excretion.

It is notable that marked peaks in chlorophyll-a concentration were observed in the winter/spring of 1999 and again in 2024, aligning with the periods of elevated bird abundance. These peaks likely indicate increased phytoplankton biomass, which would have reduced water transparency. Combined with elevated water levels during these periods—further reducing light penetration—this reduction in clarity may have limited light availability to the benthic zone, suppressing *Chara* growth and potentially initiating its subsequent decline.

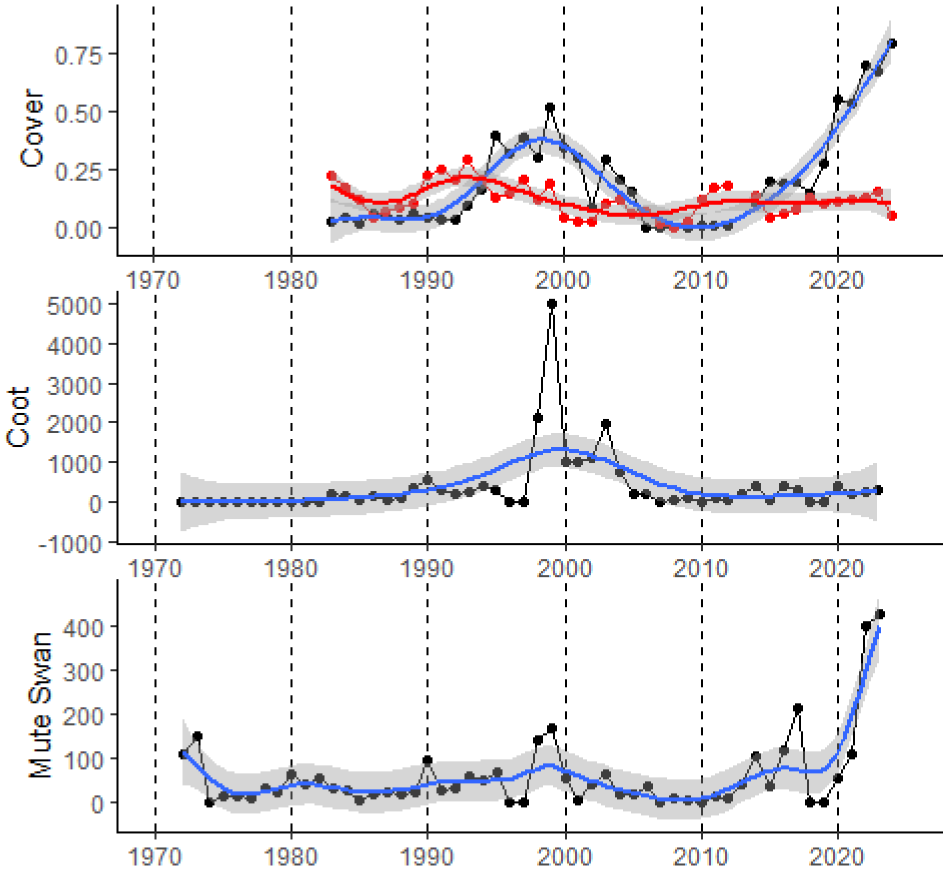


Figure 57. Time series for annual mean values for a) macrophyte cover (blue Chara, red M.spicatum), b) Coot number and c) Mute Swan number.

11.6 Modelling interactions

Modelling the interactions that may account for changes in *Chara* abundance presents a considerable challenge. An initial set of exploratory models has been developed, comprising simple time series analyses of annual data and comparisons of generalized additive model (GAM) smooths. Both approaches are based on annual summary datasets and are constrained by incomplete data coverage—notably, the water level time series begins only in 1997, suspended sediment data end in 2015, and gaps are present during the COVID-19 period. Despite these limitations, the models provide preliminary evidence that helps to clarify the key environmental drivers of *Chara* dynamics. These findings may offer useful insights for anticipating future ecological trajectories in Hickling Broad.

Time series models in this analysis employed cross-correlation functions (CCFs) to explore potential cause–effect relationships between variables. Prior to applying these models, the data were differenced to remove long-term trends and achieve stationarity, a prerequisite for valid cross-correlation analysis.

The GAM smoothers used were based on simple models, and no attempt was made to assess or account for serial autocorrelation in the residuals. As a result, the smooths may be over-fitted (i.e., overly flexible or "wiggly"). However, given that the outputs are intended for visual comparison rather than formal statistical inference, this level of smoothing is considered acceptable for exploratory purposes. That said, future analyses would benefit from more rigorous model structures that explicitly account for temporal autocorrelation.

11.6.1 Time Series models

The results of the time series models are shown in table 7. Two distinct sets of relationships emerged, reflecting contrasting feedback mechanisms influencing *Chara* population stability: one representing positive feedback loops that promote *Chara* persistence (i.e., system stabilisation), and the other representing negative feedback loops associated with system destabilisation, particularly due to grazing impacts.

As anticipated from the earlier sections, *Chara* abundance was significantly cross-correlated with both total phosphorus and transparency, with negative lags of –2 and –3 years, respectively[1].

These results suggest that *Chara* exerts a delayed influence on its environment: reducing total phosphorus concentrations after approximately two years and improving water transparency after about three years. Such delayed effects are

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[1] In interpreting these results, it is important to note that the lag *k* reported by *ccf(x, y)* estimates the correlation between *x[t+k]* and *y[t],* meaning that negative lags indicate that changes in *x* precede changes in the y. So in this example (negative lag) changes in Chara precede changes in the other variable

indicative of a causal influence of *Chara*, likely through nutrient sequestration and sediment stabilisation. Other significant relationships occurred at zero lag, where directionality cannot be inferred. Nevertheless, the overall pattern confirms the key point that suspended sediment is the primary driver of transparency and total phosphorus and that light availability is a key driver of Chara abundance.

Coot and Mute Swan[2], the two most important waterbirds grazing on submerged vegetation, showed a positive correlation with *Chara* abundance at lag 0, suggesting that these grazers are attracted to areas with abundant *Chara*, likely due to habitat preference or foraging opportunity. However, both species were also significantly negatively correlated with *Chara* abundance at a lag of -3 years, indicating that increased grazing pressure may lead to reduced *Chara* cover after a delay of approximately three years.

In addition, bird abundance was positively associated with water transparency at lag 0 but negatively associated at a lag of –1 year. This pattern suggests that birds are more likely to occupy clear-water habitats, yet their foraging activity may contribute to a subsequent decline in clarity, possibly through sediment disturbance or plant removal.

Taken together, these relationships indicate that grazing by Coot and Mute Swan represents a significant negative feedback loop—one that may help explain the longer-term decline of *Chara intermedia* in the system.

Given the visual similarity between Chara abundance and water level trends (Figures 54 and 56), the potential effect of water level on *Chara* was examined using cross-correlation analysis. This analysis produced the strongest observed correlation (+0.556), but at a lag of +4 years—indicating that *Chara* abundance was positively correlated with water level four years later.

Despite the strength of this correlation, there is no plausible ecological mechanism by which *Chara* could influence water level with such a delay. As such, the relationship is likely spurious. It may reflect an artefact of the CCF method, particularly when applied to differenced time series, which emphasises short-term fluctuations rather than long-term trends. This result underscores the importance of cautious interpretation when inferring causality from statistical associations in ecological time series.

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[2] Based on differences in body mass (~10–12 kg vs. ~0.8 kg) and daily vegetation intake (~3–4.5 kg vs. <300 g), a Mute Swan exerts approximately 10–15× the grazing pressure of a Eurasian Coot. A factor of 8–10× is commonly used in modelling to conservatively reflect this disparity (Lauridsen et al., 1993; Wood et al., 2012).

Table 7 Summary of significant cross-correlation (acf) values and time lag at which they occur

Positive Feedback Loop – Chara-Stabilization (Clear State)

Cause → Effect	acf	Lag Years	Interpretation
Chara Cover → Total Phosphorus	–0.344	-2	Chara reduces TP after 2 years.
Chara Cover → Transparency	+0.347	-3	Chara improves transparency after 3 years.
Hours of Sun → Chara Cover	+0.334	0	High light availability promotes Chara growth.
Suspended Solids → Chara cover	-0.492	0	Low suspended solids increase Chara.
Suspended Solids → Transparency	–0.472	0	Increased particles immediately reduce clarity.
Suspended Solids → Total Phosphorus	+0.501	0	Sediment resuspension elevates phosphorus levels.

Negative Feedback Loop – Grazing Impact (Turbid State)

Cause → Effect	acf	Lag Years	Interpretation
Coot + Swan → Chara Cover	+0.447	0	Chara presence attracts grazers (habitat preference).
Coot + Swan → Chara Cover	–0.351	-3	increased grazing reduces plant abundance approximately 3 years later.
Coot + Swan → Transparency	+0.503	0	Grazers occur at higher water clarity.
Coot + Swan → Transparency	-0.347	-1	Grazers reduce transparency approximately 1 year later.

Mediating effect – Water level

Cause → Effect	CCF	Lag Years	Interpretation
Water Level → Chara Cover	+0.556	+4	Water level in 4 years time reflects Chara abundance now, a spurious correlation.

11.6.2 Comparison of Smooths

To better assess overall trends, Figures 58–61 present annual partial effect smooths and the corresponding difference smooths, illustrating the relationships between *Chara* abundance and key environmental variables. These smooths are directly comparable to those in Figure 55, but here they are fitted over the same temporal window and constrained to exhibit similar levels of smoothing (i.e., wiggliness), facilitating more meaningful comparisons across variables. Note that the partial effect scales do not correspond to the original units of each variable. This is because the smooths are constrained to sum to zero rather than reflect absolute values. Their purpose is not to represent magnitude, but to highlight the shape and timing of trends.

The patterns shown reinforce previously observed relationships. In particular, *Chara* abundance closely mirrors trends in transparency and grazing bird numbers (Figures 58a and 58b), with their difference smooths (Figures 59a and 59b) remaining relatively flat. Notably, in the *Chara–grazer* comparison, peaks in the difference curve around 1995 and 2000 suggest that increases in *Chara* may have preceded subsequent increases in grazing pressure. A similar, though less pronounced, lag is apparent in the *Chara–transparency* difference, possibly indicating a delayed response of water clarity to vegetation change. These relationships illustrate the key positive and negative feedback loops identified earlier.

In contrast, *Chara* and chlorophyll-a show little synchrony (Figure 58c), consistent with previous findings that transparency in Hickling is primarily influenced by suspended solids, not phytoplankton. Nevertheless, a series of distinct peaks in chlorophyll-a are evident, notably around 2000, 2008–2010, and again near 2024. The first coincides with a peak in grazing bird numbers, while the 2008–2010 rise occurred during a period of low water levels and elevated suspended solids. These events reflect how *Chara* decline can indirectly lead to increases in phytoplankton, with a subsequent recovery of *Chara* after 2010 again associated with declining chlorophyll-a.

The correlation between water level and *Chara* is already established and is clearly visible in Figure 60a. However, the declining trend in *Chara* starting around 1999 occurred while water levels were still rising, and recovery occurred when water levels were low. This is illustrated by the marked dip in the *Chara–water level* difference smooth (Figure 61a). Thus, while water level is positively correlated with *Chara* over the long term, its role during transitional phases may be more complex and potentially confounding. The remaining smooths for *Chara* vs suspended solids (Figure 60b) and total phosphorus (Figure 60c) highlight their negative relationships, and the strong similarity between the suspended solids and phosphorus trends. This is further supported by the similar shapes of the difference smooths (Figures 61b and 61c) indicating that these variables likely co-vary and may operate as a linked driver of water clarity and nutrient dynamics.

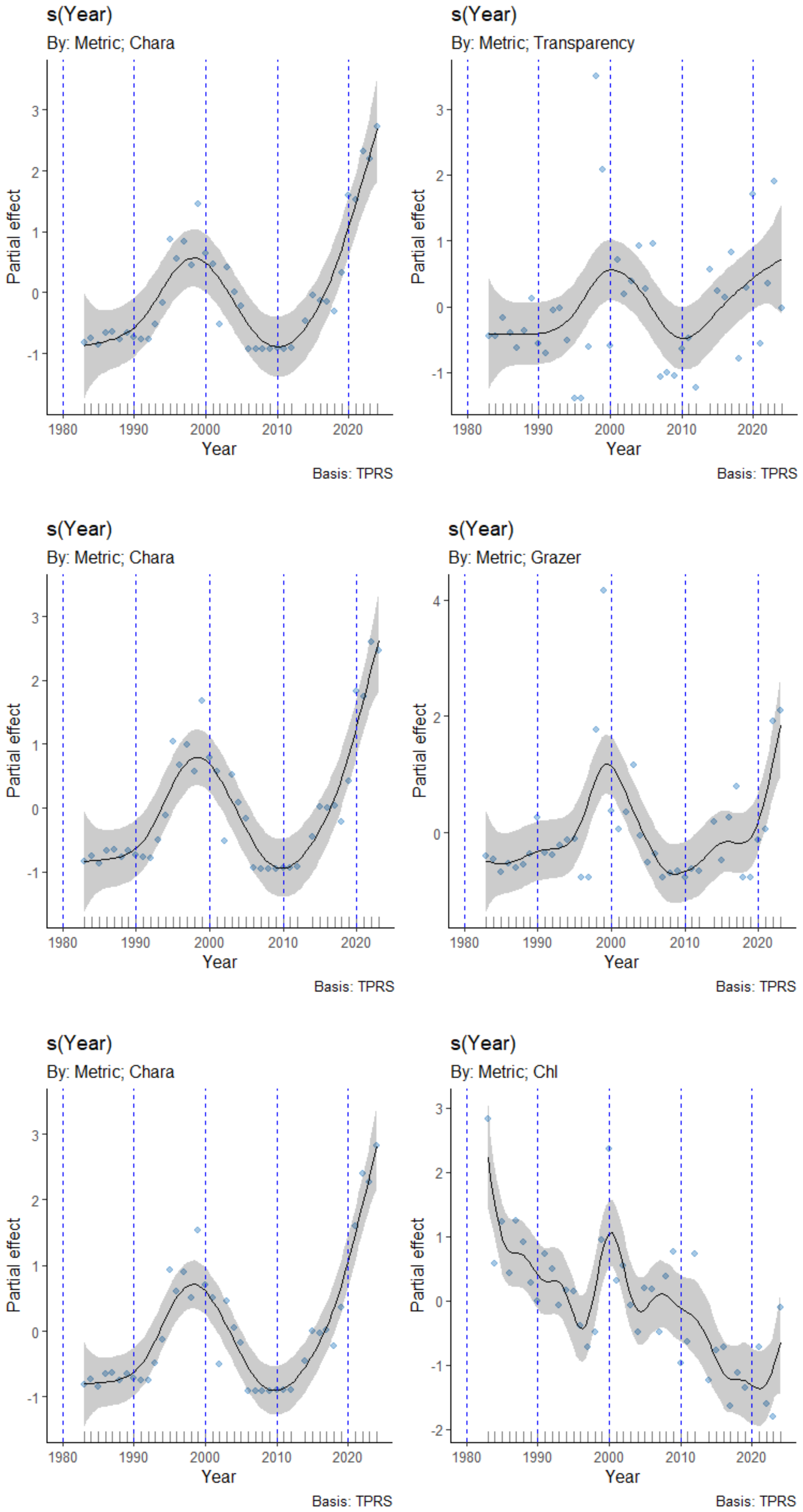

Figure 58. Partial effect trends (s of Year) of Chara abundance vs a) transparency, b) bird grazer abundance, and c) chlorophyll-a concentration in Hickling Broad.

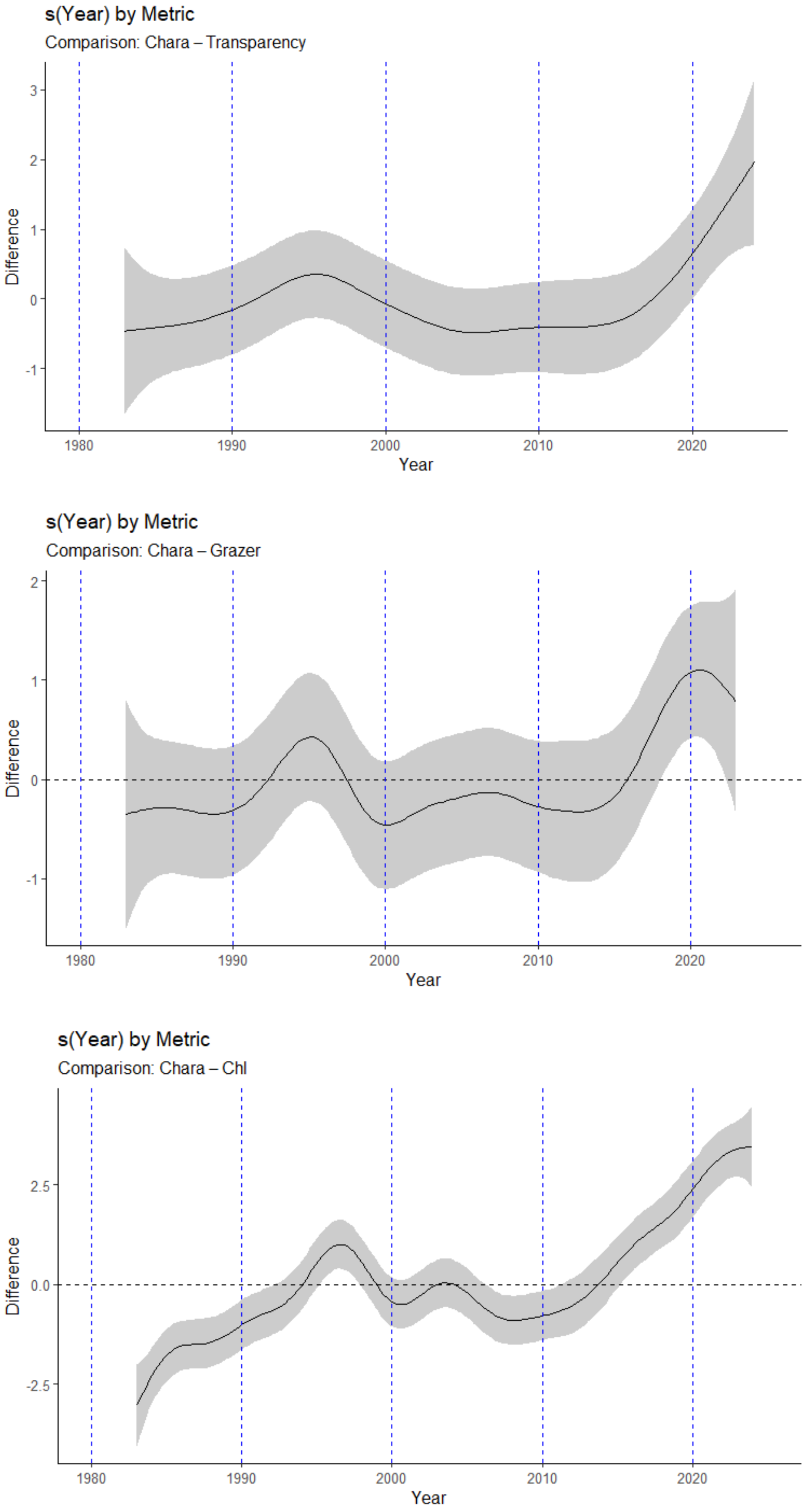

Figure 59. Comparative partial effect smooths showing the difference between the trends of Chara abundance and a) water transparency (Secchi depth), b) bird grazer abundance, and c) chlorophyll-a concentration in Hickling Broad.

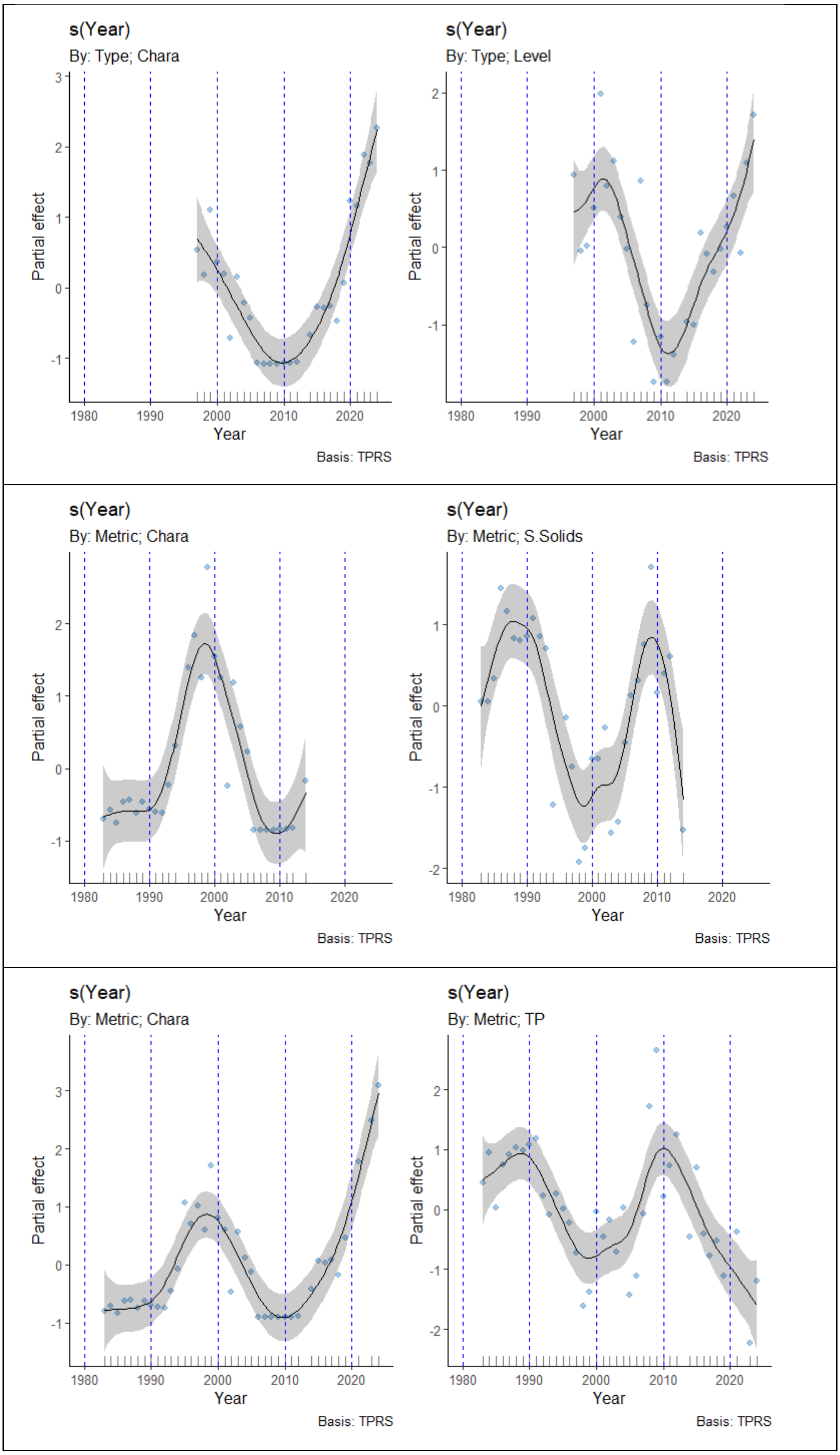

Figure 60. Partial effect trends (s of Year) of Chara abundance in relation to a) water level), b) suspended solids concentration, and c) total phosphorus concentration in Hickling Broad.

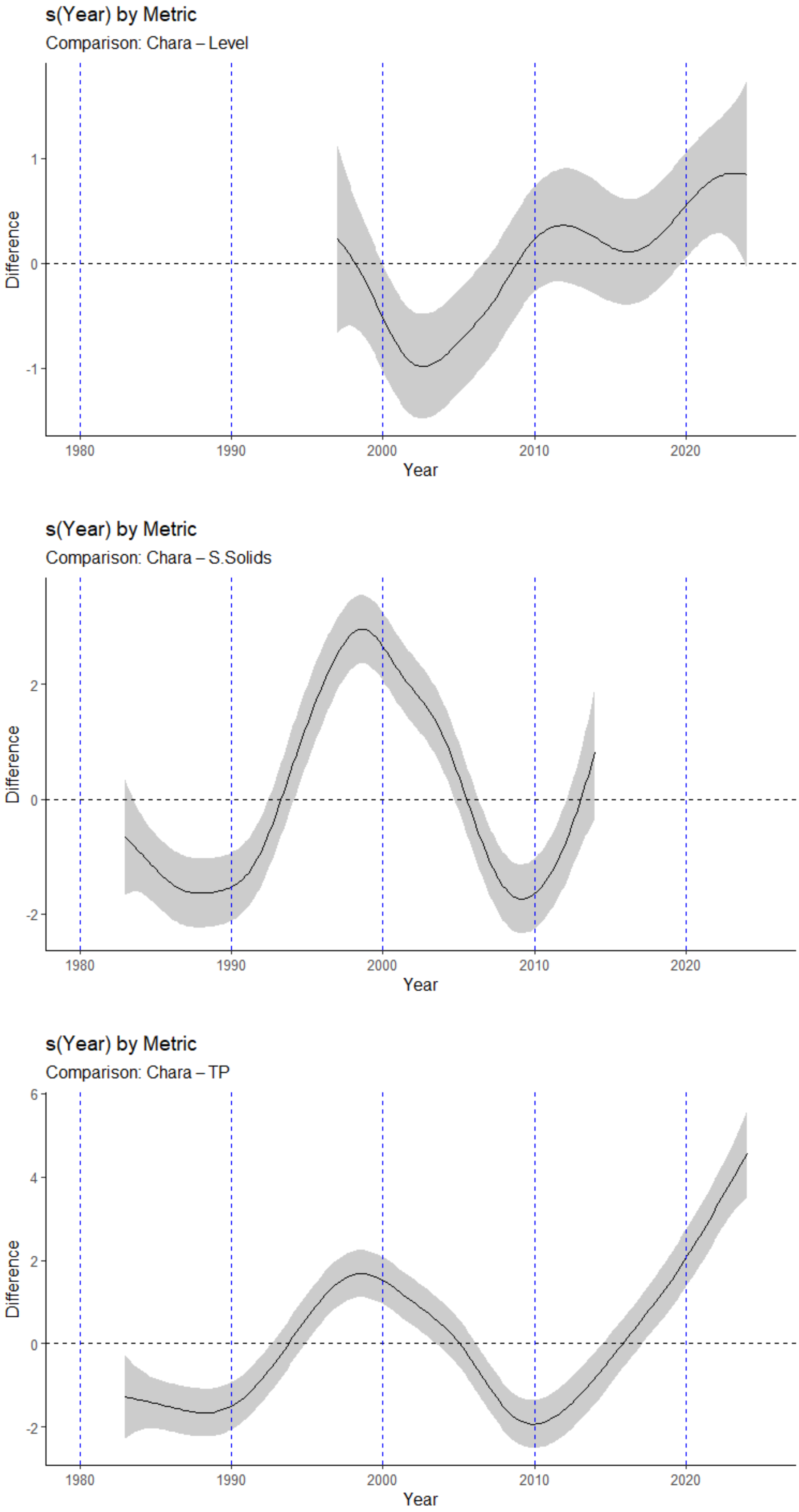

Figure 61. Comparative partial effect smooths showing the difference between the trends of Chara abundance a) water level), b) suspended solids concentration, and c) total phosphorus concentration in Hickling Broad.

11.7 Overview and Conceptual Model

The combined results from previous sections allow us to construct a plausible conceptual model and timeline of the key ecological interactions shaping the dynamics of *Chara intermedia* in Hickling Broad.

1. Declining phosphorus levels before 1990 led to reduced phytoplankton biomass and improved water transparency. The shallow depth of Hickling likely enabled submerged macrophytes—particularly canopy forming species such as *Myriophyllum spicatum*—to persist during this period.
2. In the early 1990s, *Chara* established and expanded in response to improved light availability, an increase in sunshine hours, and the fact that peak algal biomass in Hickling occurs very early in spring—prior to the main period of macrophyte growth.
3. *Chara* coverage reached its first peak around 1999. During this period, suspended solids were extremely low, reflecting reduced sediment resuspension due to the physical buffering effect of dense *Chara* beds and relatively high water levels.
4. Populations of Coot, increased in parallel with *Chara*, likely attracted by the dense vegetation. These birds exerted significant winter grazing pressure, reducing *Chara* biomass and potentially remobilising phosphorus from the plant material.
5. The resulting increase in available phosphorus supported a winter–early spring phytoplankton bloom around 2000, occurring under high water levels. The combined effect of continued grazing and elevated phytoplankton triggered the decline of *Chara*.
6. As *Chara* declined, suspended solids increased—both due to the loss of structural protection from macrophytes and enhanced wind resuspension under falling water levels. Total phosphorus also increased, primarily due to sediment resuspension. These changes were accompanied by a modest rise in phytoplankton and a decline in transparency, driven primarily by resuspended sediment.
7. By 2005, *Chara* cover had reached a minimum. By 2007, grazer abundance had declined, but low water clarity and water levels continued to prevent *Chara* re-establishment.
8. From 2012 onward, water levels began to rise again, and suspended solids declined, likely reflecting a reduction in wind disturbance. These conditions slowly re-established a habitat suitable for *Chara* recovery.
9. In 2014, *Chara* was recorded again in sheltered areas during the first of the macrophyte point surveys. Over the next five years, its distribution expanded, and by 2020, it occupied a significant portion of the broad. Although suspended solids data are unavailable after 2014, concurrent declines in total

phosphorus and improved water clarity suggest that conditions for macrophyte growth had strengthened during this period.

10. Grazing pressure increased again during the *Chara* recovery phase, but this time dominated by Mute Swans rather than Coot, especially after 2020.
11. By 2024, another winter phytoplankton peak was observed, coinciding with high water levels. We have seen this combination of factors previously and this could mark the beginning of another decline phase for *Chara*, echoing the cyclical pattern seen in previous decades.

These observations indicate that the long-term reduction in phosphorus has helped shift Hickling Broad toward a clearer, more stable, macrophyte-dominated state. There is adequate statistical support for the positive feedback loop, in which *Chara* enhances water clarity through sediment stabilisation and nutrient uptake, reinforcing its own growth.

While there is also evidence supporting negative feedbacks—particularly the impact of bird grazing on *Chara* decline—the additional role of winter phytoplankton blooms as a destabilising factor remains more speculative. This mechanism is supported by coincident patterns in the data but lacks strong statistical confirmation.

Water level fluctuations appear to interact with these biological feedbacks, amplifying their effects. Together, these processes give rise to the observed decadal cycle of *Chara* expansion and collapse.

Continued long-term monitoring will be critical to confirming the persistence of these cycles and understanding the evolving dynamics of the system.

12 Management

In the previous version of this dossier three main groups of future management options were proposed.

- *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.*
- *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.*

It is beyond the scope of this review to assess catchment-level management actions in detail, but there is no evidence of long-term change in phosphorus or chlorophyll-a concentrations in Horsey Mere that would suggest any significant extensification of the Horsey/Brograve Levels as proposed in the first management option, although hydrological intervention to reduce saline intrusion, has been tested at Hempstead Marshes through the installation of a tilting weir, which has demonstrably reduced salinity levels downstream and further investigations are ongoing to evaluate broader pumped catchment management strategies as part of the Brograve pump replacement..

While a notable reduction in total phosphorus was observed in discharge from the Catfield pump, this is believed to reflect dilution effects due to increased precipitation rather than targeted source control as proposed in the second management option. The potential scheme for increasing freshwater discharge from Catfield pump to Hickling was ruled out on environmental grounds (rather than lack of implementation). It became apparent that the option was not feasible due to water abstraction constraints and impacts on neighbouring fen habitats sensitive to groundwater level variations.

The third option, sediment removal from Hickling Broad was undertaken between 2015 and 2021, a period during which both total phosphorus concentrations declined, and Chara abundance increased. Given the overlap in timing, it is difficult to isolate the specific effect of sediment removal. However, it can be concluded that the removal had no detrimental impact on the redevelopment of *Chara* beds outside the marks channel. Moreover, trend analysis shows a continued decline in total phosphorus in Hickling, reaching the lowest levels recorded (Figure 60c), suggesting that sediment removal may have contributed positively to these improvements by removing a nutrient store.

Looking ahead, the future management of Hickling Broad presents a complex challenge. There are calls to increase the cutting of submerged vegetation to expand open water space for recreation. While routine channel cutting has continued, the last widespread vegetation management occurred between 1994 and 1998, targeting *Myriophyllum spicatum* and *Potamogeton* species. These species declined steadily during that period, with a slight acceleration (Section 7 Figure 40), and the first major collapse of the *Chara* beds occurred in 2000. While no previous analysis has suggested that plant cutting contributed to the collapse, and this review attributes it instead to grazing pressure, the findings presented here reinforce the stabilising role of dense and widespread *Chara* beds in maintaining the clear-water state of Hickling Broad. In general, management should aim to preserve and support the persistence of *Chara* beds wherever possible.

The north-west area of the Broad, where *Chara* is less dominant and *Myriophyllum* more prevalent, may offer an opportunity to undertake limited vegetation cutting to increase recreational access, without destabilising the broader ecological balance. However, the analysis also suggests the possibility of a decadal cycle in *Chara* abundance. If this pattern continues, the coming years may represent a natural return to the non-*Chara* phase of the cycle.

It is important to acknowledge that, while the concept of decadal-scale ecological cycling is supported by both the data and theoretical understanding, it remains somewhat speculative. Although the monitoring record for Hickling Broad spans over 40 years, the first decade reflects a phase of nutrient recovery, limiting our ability to confidently characterise long-term periodicity. Continued monitoring and evidence-based adaptive management will be essential for interpreting and responding to these evolving dynamics. Moreover, if water levels continue to rise due to increased winter rainfall and sea level rise, nutrient concentrations will need to be tightly managed to avoid ecological imbalances. Since adjusting the fundamental water level or bathymetry is neither feasible nor desirable, proactive nutrient management becomes even more critical.

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