

The role of monitoring, documentary and archival records for coastal shallow lake management

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The effective management and restoration of aquatic ecosystems rely on the establishment of “reference conditions,” defined as the conditions expected in the absence of anthropogenic impacts, and require a thorough understanding of the natural variability within a system. However, at least in Europe, most systematic surveys post-date the onset of human-induced pressures on aquatic ecosystems, and thus fail to capture earlier degradation to water chemistry and flora and fauna, which were already advanced. Paleolimnological methods can be used to assess a range of anthropogenic stressors, but variability within a system is often smoothed to give long-term patterns. Here, we compile monitoring, documentary, and archival records from a range of sources to extend our understanding of centennial-scale lake ecosystem change and recovery from increasing salinity. We use a case study of the Thurne Broads shallow lake coastal wetland system (Broads National Park, UK), which has been subjected to multiple pressures of anthropogenic land drainage and North Sea storm surges (primarily in 1938 and 1953 CE) that have influenced salinity. Although there are still periods with significant data gaps and the interactions with eutrophication remain unclear, we demonstrate that historical data sources can be used in combination to observe seasonal patterns and extend knowledge on past salinity change and macrophyte community structure back to the 1800s. A demonstrable change in the ecosystem is observed after the sea flood of 1938 CE, when salinity levels in parts of the Thurne Broads were close, or equivalent, to seawater. With the added anthropogenic pressures of the late 1900s, the system has failed to fully recover. Future management, whilst balancing the needs of multiple users, should focus on the current large seasonal fluctuations in salinity and the vulnerability of the system to future large salinity increases.

KEYWORDS

ecosystem recovery, eutrophication, historical records, observational records, salinity reconstruction, shallow lake management

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1 | INTRODUCTION

An understanding of the past conditions and rate of change in aquatic environments is essential to understanding future environmental scenarios and for establishing restoration targets. However, prior to the 1970s Common Era (all dates in this paper refer to the year Common Era) there was little systematic monitoring of the ecological (including chemical, hydro-morphological, and biological) quality of water bodies in the UK (Collins et al., 2012). More recently, driven by the legally binding requirements of the European Union (EU) Water Framework Directive (WFD) and Habitats Directive, there has been an increase in concern over the quality of water bodies, and the establishment of a systematic standardised monitoring regime across the UK and Europe (Joint Nature Conservation Committee, 2005; Williams, 2006). Since its conception in 2000, the WFD has required all water bodies in the EU to reach “good ecological status” or, where significant anthropogenic modification has already occurred, “good ecological potential.” All lakes over 50 hectares in area must have routine monitoring of ecological parameters and meet the requirements, originally by 2015 in the first WFD planning “cycle,” but now by 2021 in the second six-year “cycle” (European Commission, 2012; European Union, 2000). To meet good ecological status, water bodies must not be significantly altered from pre-defined “reference conditions,” which are the conditions expected in the absence of anthropogenic impacts (European Union, 2000). Monitoring of various quality indicators (e.g., water quality, macrophyte, invertebrate, and fish surveys) can be used to calculate deviation from reference conditions, and in turn to determine the effectiveness of current management (UKTAG, 2009). The establishment of reference conditions based on past ecological conditions provides evidence-based restoration targets, and information on how a site or species might respond to management (Egan & Howell, 2001; Landres et al., 1999; Swetnam et al., 1999). However, several problems exist with the use of reference conditions as a baseline for restoration. Primarily, most systematic surveys across the UK only started after the pressures on these systems, and subsequent changes to their chemistry and flora were already advanced (e.g., in the Broads National Park; George, 1992).

There is good potential to use paleolimnological methods to assess a range of anthropogenic stressors (Battarbee et al., 2011; Bennion & Battarbee, 2007; Bennion et al., 2011), including nutrient status (Bennion et al., 2004), pH (Battarbee et al., 2010), and heavy metal contamination (Yang & Rose, 2005). However, paleolimnological investigations are expensive and time consuming, meaning they are often unsuitable for conservation organisations without an academic partner institution. Furthermore, effective management decisions are limited by short-term monitoring programs and the low-resolution nature of paleolimnological methods where the variability within a system is often smoothed to give long-term patterns (Battarbee et al., 2012). However, historical biological records, which are often open access, can also be used to understand the range and variability of community structure within short periods and prior to major anthropogenic stressors. The combined use of paleolimnology and historical archival material can therefore result in robust reconstructions that can be verified and calibrated. Here, we present an approach to salinity reconstruction that can be used without, or as an initial stage in, a palaeolimnological study to provide a robust reconstruction of a lake ecosystem as a retrospective baseline for restoration.

Historical records take a variety of forms, including photographs, naturalists’ notebooks, journal articles, and excursion notes often dating back to the early 1800s. However, relatively few studies have used historical records to inform on the restoration or management of ecosystems. There are a number of studies that have investigated long-term change in lake macrophyte communities, but often combined with macrofossil analysis of sediment cores (e.g., Ayres et al., 2008; Davidson et al., 2005; Madgwick et al., 2011). A robust reconstruction of the community structure using historical data alone requires knowledge of the ecology of species, but also an acknowledgement of the bias associated with such records.

There is a wealth of literature on the use of historical biological records for mapping distribution changes and species richness for a range of animals (e.g., amphibians [Graham & Hijmans, 2006] and vascular plants [Crawford & Hoagland, 2009]). However, due to a lack of available pre-industrial water-quality data for most sites (e.g., Bennion et al., 2001; Muxika et al., 2007; Smol, 1992), relatively few studies have used quantitative historical records to inform on the changing chemical parameters of ecosystems. Where pre-monitoring data do exist, comparing historical water quality records is often hampered by changes in data quality through time. However, there are instances where historical water chemistry data have been used to inform management (e.g., Winfield et al., 2008).

When it is possible to compile datasets of historical salinity measurements and records of biological communities with known salinity tolerance, we can reconstruct salinity of shallow lakes with relative ease (e.g., Ryves et al., 2004 which uses salinity measures in combination with diatom analysis). In this paper, we present a case study of the Thurne Broads, a coastal shallow-lake wetland system in Eastern England (Figure 1), where we have compiled abundant records of historical salinity extending to the 1800s to inform the management of the modern-day system.

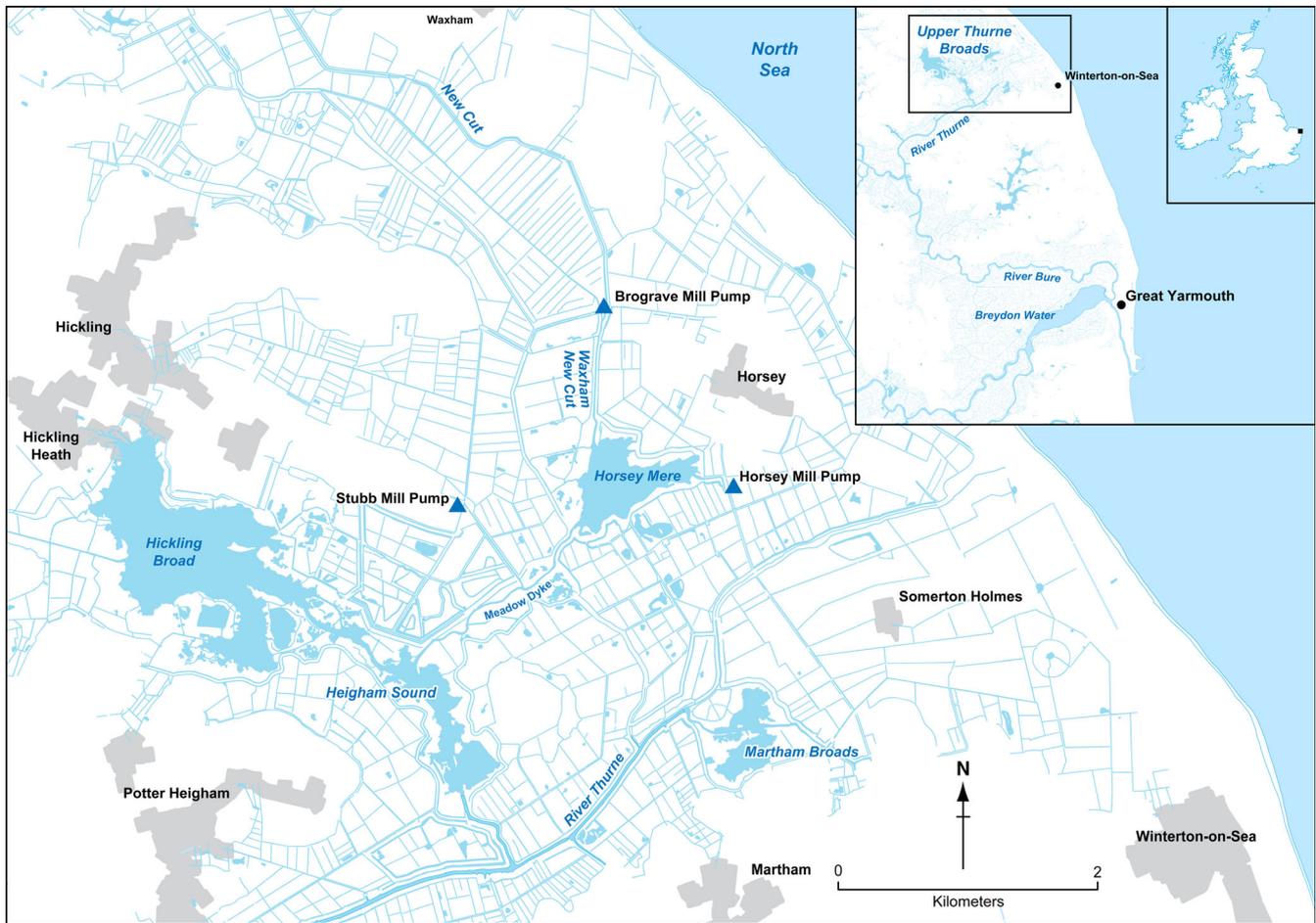


FIGURE 1 Location of the Upper Thurne Broads showing pump locations (denoted as triangles), settlements (in grey), and the drainage network (in blue).

The Thurne Broads are a series of connected wetlands within the north-eastern part of the Broads National Park in Norfolk. They support a range of ecosystem services, including water resources and quality, flood protection, provision of food, recreation, tourism, and education (Broads Authority, 2017; Natural England, 2009), as well as being nationally and internationally important for nature conservation. The Thurne Broads system, comprising Hickling Broad, Horsey Mere, Heigham Sound, and Martham North and South Broads, is designated as a Site of Special Scientific Interest (SSSI), Special Protected Area (SPA), Special Area of Conservation (SAC), and protected under the Ramsar convention, as part of the Broadland Ramsar site. It supports rare macrophyte and bird populations in open water, fen, and grazing marsh habitats, including the richest diversity of stoneworts in the UK with 14 out of 21 species and other UK-wide rare macrophyte species such as *Najas marina*. The system is, however, subject to multiple pressures, including eutrophication (e.g., Balls et al., 1989), boat-derived toxins (e.g., Boyle et al., 2016; Sayer et al., 2006), and saline intrusion (e.g., Holman & Hiscock, 1993). Due to these pressures, the Broads are currently designated as “poor” (Hickling Broad encompassing Heigham Sound) or “moderate” (Horsey Mere and Martham Broads) under the WFD and are currently an “unfavourable – declining” SSSI unit, primarily due to low macrophyte abundance and diversity.

Three mechanisms affect the salinity of the Thurne Broads. First, the Thurne Broads system has experienced several sea flood events since 1287. The floods of 1938 and 1953 were devastating to the East Anglia coastline and surrounding areas, with the 1953 flood being described as the worst national peacetime disaster (Baxter, 2005) (Figure 2). Sea floods have been caused by the combination of North Sea spring tides and deep Atlantic depressions, which have forced tidal storm surges southwards in the North Sea, resulting in waves breaking through the sea banks less than 10 km North of Horsey Mere.

Second, agricultural development in the Thurne catchment has led to increased drainage of marshland through a series of pumps. The pumps control the main upstream source of water into the Thurne system, pumping water drained from

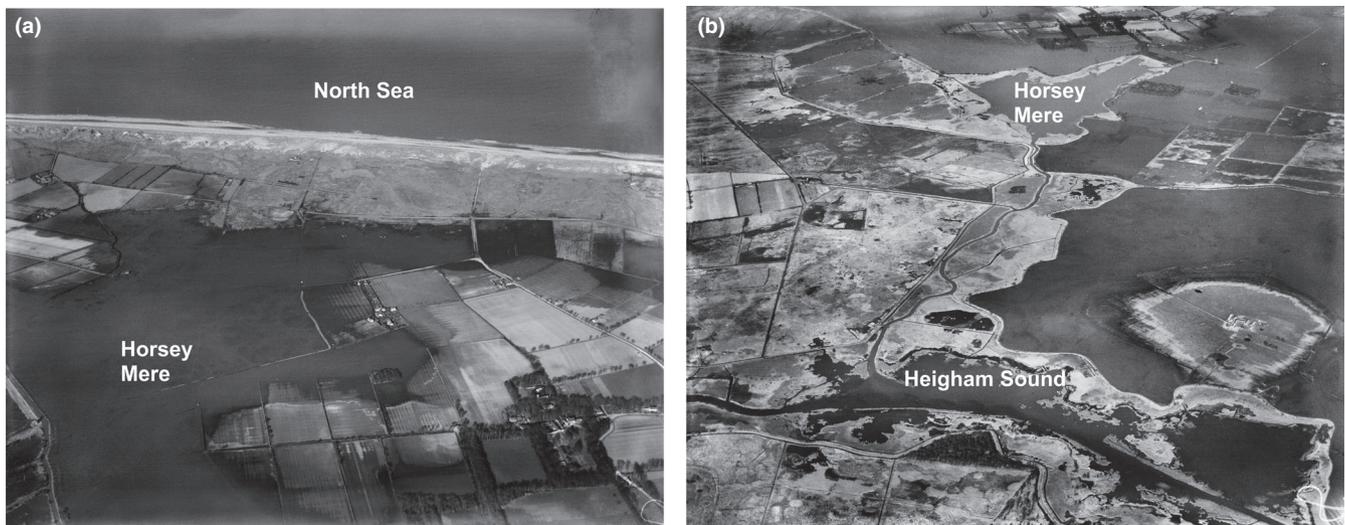


FIGURE 2 Aerial photographs of (a) Horsey Mere showing inundation from flood waters and (b) flooding between Heigham Sound and Horsey Mere during the 1938 sea floods (Image © Historic England).

surrounding land into dykes connecting the Broads. The salinity of individual ditch and pump waters varies from around 0%–60% seawater (Holman, 1994), primarily due to geologically and hydrologically controlled groundwater movement. The Brograve Level, increased efficiency of pumping, and deepening of dykes allow saline water seepage through the Holocene layers of peat, and where extensive deepening has occurred, the estuarine clay, drawing seawater into the water table. The water discharged by the drainage pumps is therefore a mixture of saline water, fresh groundwater, and direct precipitation.

Third, added to the complexities of the drainage system is the tidal influence of the River Bure. The River Thurne joins the River Bure, which discharges to the North Sea via Breydon Water Estuary at Great Yarmouth (Figure 1). However, the rivers are tidal up to 20 km inland and the combination of offshore storms and spring tides can push seawater as far upstream as the Thurne Broads. During these high tides, tidal movements allow mixing of the waters of Hickling Broad, Heigham Sound, and Horsey Mere. On a falling tide, water from Horsey Mere flows through Heigham Sound and ultimately into the Rivers Thurne and Bure, and the North Sea estuary of Breydon Water. On a rising tide, however, water is ponded back through the system so that saline water from Horsey Mere is pushed into Hickling Broad. There is little dilution by the only freshwater inflow into Hickling Broad, which is from Catfield dyke (Holman & White, 2008).

For the Thurne Broads, even in the instrumental period, fluctuations in salinity, and their ecological impact are not well understood. Sea floods to the extent of those in 1938 and 1953, which resulted in long-term increases in salinity to concentrations close, or equivalent, to those of seawater, have not occurred over the monitoring record, and therefore rates of recovery are largely unknown. It is now widely accepted that there has been a mean increase in the salinity of the River Thurne and Thurne Broads over time (Holmes et al., 2010; Moss, 2001). Anthropogenic catchment modifications (Figure 3), specifically increased efficiency of pumps for land drainage in the 1960s, have been suggested as a main underlying cause in the increased background salinity (Holdway et al., 1978; Holman et al., 1999). Yet, consistent salinity monitoring only took place when most of the drainage improvement works were taking place in the 1960s and not before this. Conflicting views exist on the salinity fluctuations during this period. Based on monitoring data from 1952–1953 and 1967–1973, Holdway et al. (1978) suggest an increase in salinity from 1968. However, this is based on relatively few data points. Other sources suggest that the greatest increase in salinity since 1892 occurred in 1970–1972 (Broadland Research Working Group, 1973). The improvement scheme on the West Somerton Level in 1979 coincided with a rise in dyke salinity and a doubling of salinity in Martham Broad (Driscoll, 1984; George, 1992). However, Bales et al. (1993) argued that chlorinity has remained the same since 1974–1975 to 1993, suggesting little increase once major drainage modifications had ceased.

The increase in salinity, along with other drivers (e.g., eutrophication, hydrological regimes, turbidity, and acidification), can have severe effects on the ecological stability of shallow lakes. These mechanisms have driven sometimes rapid degradation in shallow lakes, eventually culminating in the complete loss, or reduced growing season, of macrophytes, and ultimately a decline in overall biodiversity (e.g., Davidson & Jeppesen, 2013; Davis et al., 2003; Jeppesen et al., 2000; Ogden, 2000; Reid, 2008; Reid et al., 2007; Scheffer, 1998; Vestergaard & Sand-Jensen, 2000). Despite increases in salinity due to

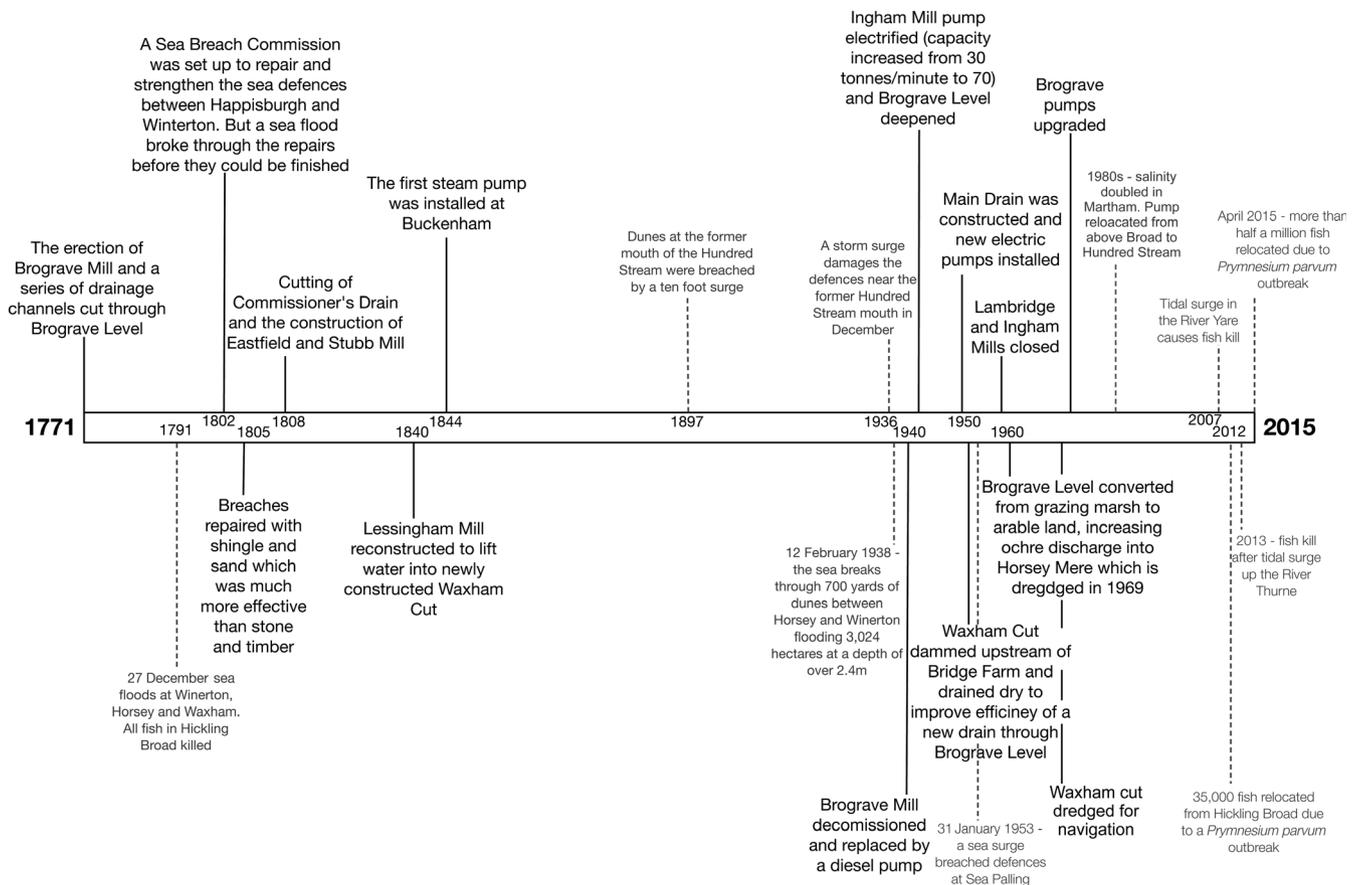


FIGURE 3 A timeline of drainage modifications in the Thurne catchment compiled from George (1992), Williamson (2005), Eastern Daily Press (2013), Environment Agency (2015), and ITV News (2015).

the mechanisms described above, since the 1970s, eutrophication has been cited as a major driver behind the turbid phytoplankton-dominated waters of the Thurne Broads. There are, however, some periods of clear water and macrophyte growth present today (Moss & Leah, 1982; Phillips et al., 2016).

Little attention has thus far been given to the role of repeated North Sea storm surges on the long-term salinity and ecology of the Broads, and the interactions with eutrophication. The most comprehensive reports of the floods of 1938 suggest that salinity values had declined from 32.29‰ to 12‰ by the end of 1939 and to 4‰ within six years (Buxton, 1944b). Once salinity levels had dropped after the floods, it has generally been believed that the system was ecologically stable with the return of macrophyte diversity and abundance. However, without a full assessment of all ecological datasets, it is difficult to fully assess if the ecosystem has recovered. Furthermore, increasing salinity since the drainage improvement and following the storm surges is difficult to assess due to large gaps in the data series; many of the data points are after the floods of 1938 and 1953. Very little is known on salinity prior to these events or the recovery of the ecosystem after the flooding. There is still a significant lack of knowledge on the past salinity of the system and its ecological impact prior to the instrumental period. However, due to the Broads, popularity with naturalists, a wealth of archival and documentary material exists, which has the potential to extend knowledge of past salinity variation beyond the monitoring record. This study aims to combine multiple sources, including monitoring data, published reports and naturalists' notebooks, to give a more long-term and thorough understanding of past ecological and hydrochemical conditions. In order to inform restoration targets for shallow lake ecosystems, using the Thurne Broads as a case study.

2 | METHODS

Historical chemical, ecological, and natural history records were compiled for each of the four connected Broads that comprise the Thurne Broads system: Hickling Broad, Horsey Mere, Heigham Sound, and Martham North and South Broads

(Figure 1). Due to a lack of distinction between Martham South and Martham North Broads in some records, all data for these sites were combined into “Martham Broads” and are referred to as such from here onwards.

2.1 | Monitoring data

Salinity values (as chloride mg/L) for Hickling Broad, Horsey Mere, Heigham Sound, and Martham Broads were compiled from Phillips (1976), Anglian Water from 1978 to 1981, the University of East Anglia from 1974 to 1978, and the Environment Agency routine monitoring extending from 1981 to 2012 (Phillips et al., 2015). Environment Agency monitoring data begin in 1981, at which point monthly chloride data are available. Where multiple records exist for a given year, the mean, maximum, and minimum values were calculated.

Macrophytes are one of several biological components that must be monitored under the WFD; systematic surveys have therefore provided a database of modern distributions, enabling classification of lakes based on their macrophyte assemblages (e.g., Moss et al., 2003; Willby et al., 2009). Macrophyte records from 1983 to 2017 for Hickling Broad, Horsey Mere, and Martham Broads were compiled from Broads Authority survey data and compiled to give a Thurne wide record. From 1983 to 2014, Broads Authority surveys were conducted as transect surveys and point surveys, and from 2014 to 2016 only a point survey was undertaken. To allow comparison with historical data sources, presence/absence rather than abundance data have been used.

2.2 | Documentary and archival evidence

As well as quantitative measures of salinity, numerous documentary archives can be used to infer changes in salinity in the Thurne Broads. The ecological impacts of salinity rather than actual salinity values are more likely to be recorded by users of the lake. The Thurne Broads provide a range of ecosystem services, including angling, sailing, and birding. As such, the conditions (fish stocks, aquatic plant growth, or breeding bird populations) for these activities are often recorded, which may in turn relate to salinity change in the system. For the Thurne Broads these records exist as (1) historical fish stock records, (2) old macrophyte records and descriptions, and (3) accounts of lake recovery from sea floods in 1938 and 1953; all are useful proxies for salinity in this system are discussed in detail below.

1. One of the primary concerns regarding increasing salinity has been increased abundances of *Prymnesium parvum*, a microscopic (~15 µm) biflagellate phytoplankton. *P. parvum* has a wide salinity tolerance range from 0.5‰ to 30‰, with a peak abundance at 10–15‰, making it common in brackish waters where chloride concentrations exceed 250 mg/L (Rasel & Patiño, 2017; Shilo, 1972). The species has been found to be highly abundant in the River Thurne, Hickling Broad, and Horsey Mere (Holdway et al., 1978; Wortley & Phillips, 1987) during periods of higher salinities. Due to its microscopic nature and relatively recent discovery in 1937 (Carter, 1937), historically the presence of the alga has not been consistently recorded in the Thurne Broads. However, few records of *P. parvum* or a systematic evaluation of historical fish mortalities exist. For this study, records of *P. parvum* have been compiled for pre-1996 from National Rivers Authority archives, for 1996–2008 from Environment Agency data, and post 2008 from Broads Authority data to evaluate the relationship between abundance and salinity.
2. Botany, and the collection of wild plants, have been popular with amateur and professional naturalists since the 1600s, and a wealth of historical records now exist for the Thurne Broads system, especially since the late 1800s. For example, macrophyte presence and abundance have been recorded in Hickling Broad since 1883 due to the discovery of the nationally rare plant species *Najas marina* (Bennett, 1884), and more recently due to widespread growth of European-wide rare *Chara intermedia* and the associated problems for sailing communities (McCarthy, 1999). Historical macrophyte records for Hickling Broad, Horsey Mere, Heigham Sound, and Martham Broads were extracted from a database constructed by Madgwick (2009) and again combined. Marginal emergent plant species that were not recorded in Broads Authority macrophyte surveys were removed from the historical dataset.
3. Several notebooks of amateur naturalists and journal articles have documented the Thurne Broads response, and recovery (especially in terms of fish, plants, and birds), in relation to major sea floods linked to storm surges that occurred in 1938 and 1953 (e.g., Buxton, 1939; Cable, 1991). Whilst these floods have been well documented in archival material, systematic monitoring of the system was not in place and therefore absolute salinity values are often random in location and time. For historical salinity values and accounts of ecological response, a literature review using the search terms “flood; Horsey; Hickling; Heigham; Martham; Salt; Chloride; grains per gallon” on JSTOR and the *Transactions of the Norfolk and Norwich Naturalists Society* was used alongside archival

documents from the Broads Authority and Natural England site files. As for monitoring data, where multiple records existed for a given year, the mean, maximum, and minimum values of salinity have been calculated. Where original datasets could not be obtained, values were digitised from figures if possible.

2.3 | Calculation of salinity tolerance

Ellenberg indicator values (EIVs) (Ellenberg et al., 1991) can be used as a numerical classification of habitat niches for macrophytes. The system assigns a numerical value to tolerance of species to light availability (L), temperature (T), continentality (K), soil moisture (F), soil reaction (R), nitrogen availability (N), and salinity (S). EIVs for saline tolerance of the species present in the Thurne Broads were selected from the PLANTATT database for British flora (Hill et al., 2008). The EIV scale for salt tolerance is given in Table 1.

2.4 | Conversion of salinity units

Salinity can be presented as measures of conductivity (mS/cm), practical salinity units (psu), parts per thousand (‰) of NaCl, chloride concentration or chlorinity (mg/L), and, in historical documents, grains per gallon (gpg). The conversions used here are based on the following equations:

1. Dauphinee (1980) – chlorinity (mg/L) to salinity (psu):

$$S = 0.0018066 * Cl$$

2. B. Moss (personal communication to C. Sayer, 2008) – chloride grains per gallon to chlorinity (mg/L):

$$Cl = gpg * 13.6$$

3. psu is equivalent to parts per thousand (‰).

4. Conductivity values are converted to psu following UNESCO (1983). In brief, if $C(S, t, p)$ is EC of seawater at a known salinity (S), temperature (t) and pressure (p), the conductivity ratio is defined as:

$$R = C(S, t, p) / C(35, 15, 0),$$

where $C(35, 15, 0)$ is the EC of seawater at S 35, t 15°C and atmospheric pressure (p).

These conversions are valid for waters of ionic composition similar to seawater, making them appropriate for use here, but in other waters the relationships vary with ionic composition, temperature, and ionic strength. To be consistent with the

TABLE 1 Ellenberg indicator values (EIVs) for saline tolerance. 0 indicates a species with zero saline tolerance, while nine represents species from hypersaline environments (Ellenberg et al., 1991).

EIV	
0	Absent from saline sites; if in coastal situations, only accidental and non-persistent saline spray or water can be tolerated
1	Slightly salt-tolerant species
2	Species occurring in both saline and non-saline situations, but where saline habitats are not strongly predominant
3	Species most common in coastal sites, but regularly present in freshwater
4	Species of salt meadows and upper saltmarsh, but only tolerant to occasional tidal inundation (including species of brackish conditions)
5	Species of the upper edge of saltmarsh and inundated by tides
6	Species of mid-level saltmarsh
7	Species of lower saltmarsh
8	Species permanently inundated by seawater
9	Species of extremely saline conditions of evaporation and precipitating of salts

Environment Agency monitoring record from 1981, salinity values are presented in this paper as chloride concentration (mg/L).

3 | RESULTS

3.1 | Chloride data of the Thurne System

Chloride values for Hickling Broad, Horsey Mere, Heigham Sound, and Martham Broads date back to 1892 (Figure 4). For all the Broads, there are numerous gaps in the record until routine monitoring by the Environment Agency was established in 1981. Martham Broad has the shortest record, beginning in 1934 (1,500 mg/L) and without any further data points until 1974. The record for Heigham Sound extends back to 1892, but routine monitoring ended in 1998. No data are available between 1906 and 1936, and 1945 and 1973 for any of the Broads, with the exception of individual data points for 1914, 1933, and 1952 in Hickling Broad.

The average chloride concentration of Hickling Broad increased from 540 mg/L in 1892 to 1,780 mg/L in 2012, and in Horsey Mere from 701 mg/L in 1892 to 2,905 mg/L in 2012. For Heigham Sound and the Martham Broads, the earliest values of 701 and 1,500 mg/L were based on spot samples in 1892 (Figure 4). The average and maximum chloride concentrations of Heigham Sound have increased with mean values between 1,480 and 2,198 mg/L from 1981 to 1998. However, the minimum values in 1986–1990, 1994, and 1996–1998 of between 35 and 980 mg/L were comparable to those in 1892. During the 1990s, minimum values for many of the Broads are comparable to the values from the later 1800s and early 1900s. For example, in 2000 (min. 314 mg/L, max. 2,150 mg/L) and 2002 (min. 139 mg/L, max. 2,200 mg/L) the minimum salinity values of Hickling Broad were lower than the minimum values recorded before 1938 (Figure 4). The maximum values were, however, substantially higher than the period between 1892 and 1938 (ranging from 497 mg/L in 1914 to 1,121 mg/L in 1906), and consequently average values have increased. In contrast to the other Broads, the salinity of Martham Broad appears to have decreased from 1,500 mg/L in 1934 to an average value of 1,229 mg/L in 2012 (maximum value of 1,440 mg/L), although this is based on a single data point for Martham Broads (Figure 4). The average salinity of

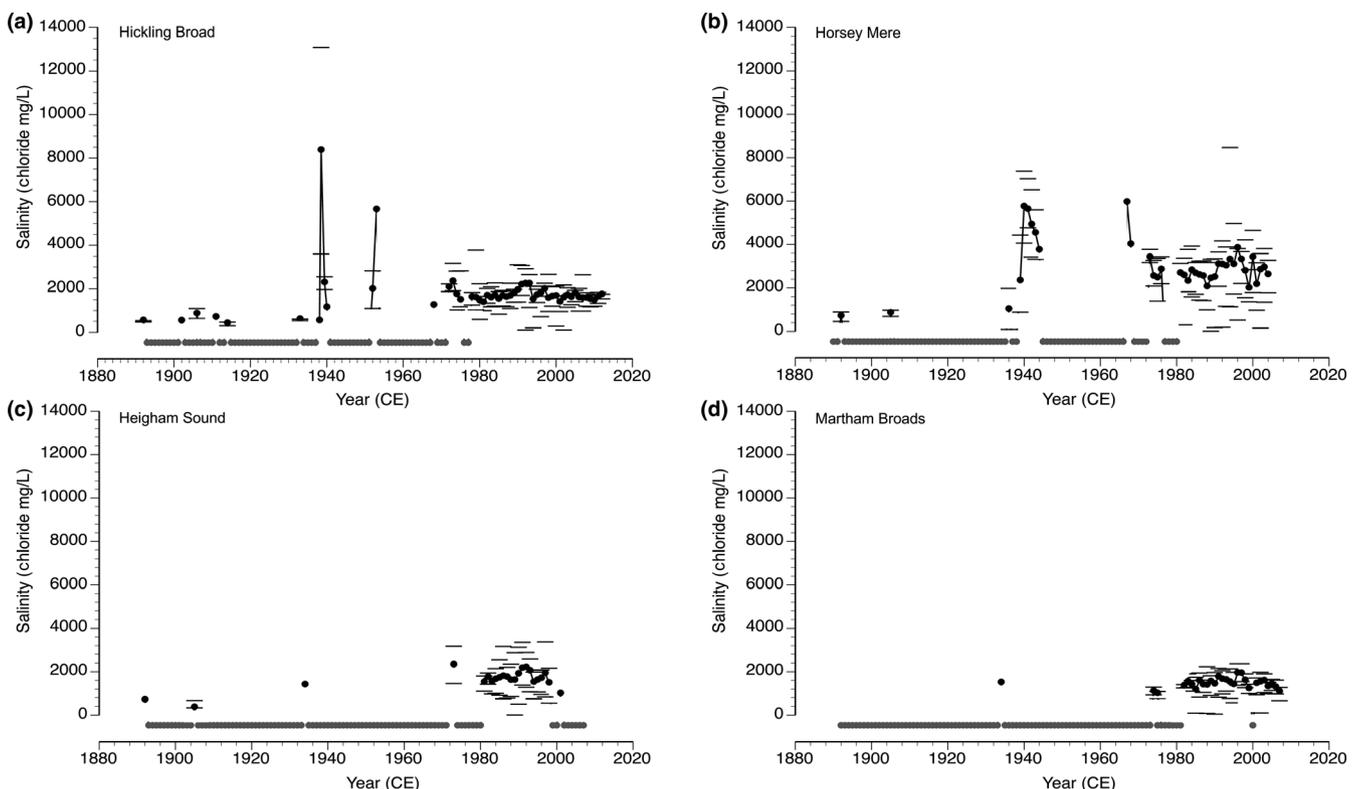


FIGURE 4 Mean, maximum, and minimum values of chloride concentration (mg/L) in (a) Hickling Broad, (b) Horsey Mere, (c) Heigham Sound, and (d) Martham Broads. Mean values are denoted by the points and black line; the minimum and maximum values are denoted by the black dashes. Gaps in the record are highlighted by the grey circles.

the Broad was below 1,500 mg/L between 2004 and 2012. In 1989, when minimum salinity values were at their lowest over the monitoring period in Horsey Mere (36 mg/L) and Heigham Sound (35 mg/L), salinity in Martham Broad was substantially higher at 927 mg/L. The low values from Heigham Sound and Horsey Mere, however, correspond to a specific day when Martham Broad was not sampled and consequently these values may not be directly comparable.

During periods when the monitoring data allow for inter-annual comparison, there was a substantial inter-annual range in salinity for all Broads (Figure 4). For Horsey Mere minimum values between 1981 to 2012 range from 36 mg/L in 1989 to 2,400 mg/L in 2009. Values in 1982 (324 mg/L), 1989 (36 mg/L), 1990 (200 mg/L), 1996 (544 mg/L), 2002 (179 mg/L), and 2006 (199 mg/L) are comparable to values recorded in 1892–1905 (min. 480 and 713 mg/L, average 703 and 847 mg/L). Maximum values range from 3,160 mg/L in 1981 to 8,500 mg/L in 1995. For Martham Broad, minimum values from 1981 to 2012 ranged between 76 mg/L in 1990 to 1,570 in 1997. Maximum values vary from 1,540 mg/L in 2004 to 2,400 mg/L in 1996 and 1997. Minimum values for Hickling Broad ranged from 137 mg/L in 1992 to 1,690 mg/L in 1997, whilst maximum values over the monitoring period ranged from 1,710 mg/L in 2001 to 3,800 mg/L in 1979. Minimum salinity of Heigham Sound ranged from 35 mg/L in 1989 to 1,450 mg/L in 1982, and maximum values from 1,830 mg/L in 1981 to 3,380 mg/L in 1991. There is no relationship between mean annual chloride values and the degree of annual variability.

In addition to substantial inter-annual variability, each Broad also experiences large annual variability (Figure 4). In Martham Broads, there was large annual variability in 1988 (ranging from 107 to 2,260 mg/L), in 1990 (ranging from 76 to 1,990 mg/L), and in 2002 (ranging from 127 to 2,000 mg/L). The annual variability for Hickling Broad was greatest in 1992, ranging from 137 to 2,950 mg/L. Greatest annual variability for Heigham Sound was in 1989, ranging from 35 to 2,900 mg/L. Largest annual variability for Horsey Mere was in 1994, ranging from 1,160 to 8,500 mg/L.

For all sites, data are lacking prior to or after the major sea floods of 1938 and 1953. Data from Hickling Broad before and during 1938 record a rise in salinity from 537 mg/L to a maximum of 13,108 mg/L. By 1940, salinity had dropped to a maximum value of 2,576 mg/L, but was still substantially higher than pre-flood maximum values. The 1938 flood was not directly recorded in Horsey Mere; however, values for 1939 suggest an increase from pre-flood values to a maximum of 4,460 mg/L, but the peak in chloride concentration was in 1940 of 7,400 mg/L. The 1953 flood is only recorded in Hickling Broad, which shows an increase from a maximum value of 2,850 mg/L in 1952 to a maximum of 5,633 mg/L in 1953. However, no data are available between the two flood events.

3.2 | Macrophyte records

Macrophyte records for the Thurne Broads extend to 1846, with Broads Authority annual surveys beginning in 1983. For years with no data, it is assumed that no sampling took place rather than the absence of species. A total of 68 species were recorded in the Thurne Broads between 1846 and 2017. The EIVs (with one suggesting slight salinity tolerance; Table 1) for species recorded in the Thurne Broads suggest that the majority of species are intolerant of salinity above freshwater, with only one occurrence of a highly tolerant species (*Ranunculus baudotii*) (Table 2). The saline tolerant species, *Potamogeton pectinatus* (EIV = 2), is present from 1865 to 2017.

Of the 13 species with an EIV of 1 or above (Table 2) present in the Thurne Broads, eight are still present in the 2000s. *Stratiotes aloides* was first recorded in 1866, but was absent by 1950 (Figure 5). *Zannichellia palustris* was first recorded in 1885 and was present until 2015. However, the species was not recorded for 80 years between 1889 and 1968, which is either two distinct periods of presence, or continuous presence with limited recording. *Chara contraria* and *Oenanthe fluviatillis*, first recorded in 1881, were also absent between 1906 and 1960. Of the other seven species recorded between 1880 and 1883, *N. marina*, *C. hispida*, and *C. aspera* were recorded almost consistently until 2017. Between 1977 and 1983, *Elodea canadensis*, *Ceratophyllum demersum*, and *Fontinalis antipyretica* (all first recorded after 1926) were not recorded in the Broads. However, during this time several other species such as *P. pectinatus*, *N. marina*, and *Hippuris vulgaris* were recorded.

The majority of years have macrophytes present that span EIVs of 0–2 (Figure 6). There are three stages of EIV in the Thurne Broads: (1) between 1846 and the late 1920s, the record is dominated by species with an EIV of 0; (2) by 1930 there was an increase in species with an EIV of 1; and (3) by the late 1960s, there was an increase in species with an EIV of 2. There is a general increase across all EIVs in the 1980s due to increased recording of species with routine macrophyte monitoring. The earliest records in 1846, 1847, and 1850 are characterised solely by the presence of macrophytes with an EIV of 0 (*Utricularia vulgaris* and *Sagittaria sagittifolia*). The highest EIV of 4 is recorded in 1897 for *R. baudotii*.

Although there is an increase in species with an EIV of 1 during the 1930s and 1940s, the record is still dominated by species with an EIV of 0 in the 1930s (Figure 6). However, by 1943, and continuing until 1954, there was a decrease in species recorded with an EIV of 0. From 1954, the number of species recorded with an EIV of 0 remains high. Despite a

TABLE 2 Species present in the Thurne Broads with an Ellenberg indicator value (EIV) for saline tolerance above 1

Species	EIV	Sites
<i>Ranunculus baudotii</i>	4	Horsey Mere
<i>Potamogeton pectinatus</i>	2	Hickling Broad, Horsey Mere, Heigham Sound, Martham Broads
<i>Zannichellia palustris</i>	2	Hickling Broad, Horsey Mere, Heigham Sound, Martham Broads
<i>Callitriche stagnalis</i>	1	Hickling Broad, Martham Broads
<i>Ceratophyllum demersum</i>	1	Hickling Broad, Horsey Mere, Heigham Sound, Martham Broads
<i>Hippuris vulgaris</i>	1	Hickling Broad, Horsey Mere, Heigham Sound, Martham Broads
<i>Lemna gibba</i>	1	Hickling Broad
<i>Lemna minor</i>	1	Hickling Broad, Horsey Mere, Heigham Sound, Martham Broads
<i>Nuphar lutea</i>	1	Hickling Broad, Horsey Mere, Heigham Sound, Martham Broads
<i>Potamogeton crispus</i>	1	Horsey Mere, Heigham Sound and Martham Broads
<i>Potamogeton perfoliatus</i>	1	Hickling Broad, Horsey Mere, Martham Broads
<i>Potamogeton pusillus</i>	1	Hickling Broad, Horsey Mere, Heigham Sound, Martham Broads
<i>Stratiotes aloides</i>	1	Horsey Mere, Martham Broads

general trend for increased species with an EIV of 2 since the late 1960s, between 1998 and 2002, the number of species with an EIV of 2 decreased to one species per year. This was also the case in 2015, 2016, and 2017. In 2014, there was also a decrease in species with an EIV of 1, and the majority of species have an EIV of 0. In other years, the numbers of species with an EIV of 0 or 1 are similar.

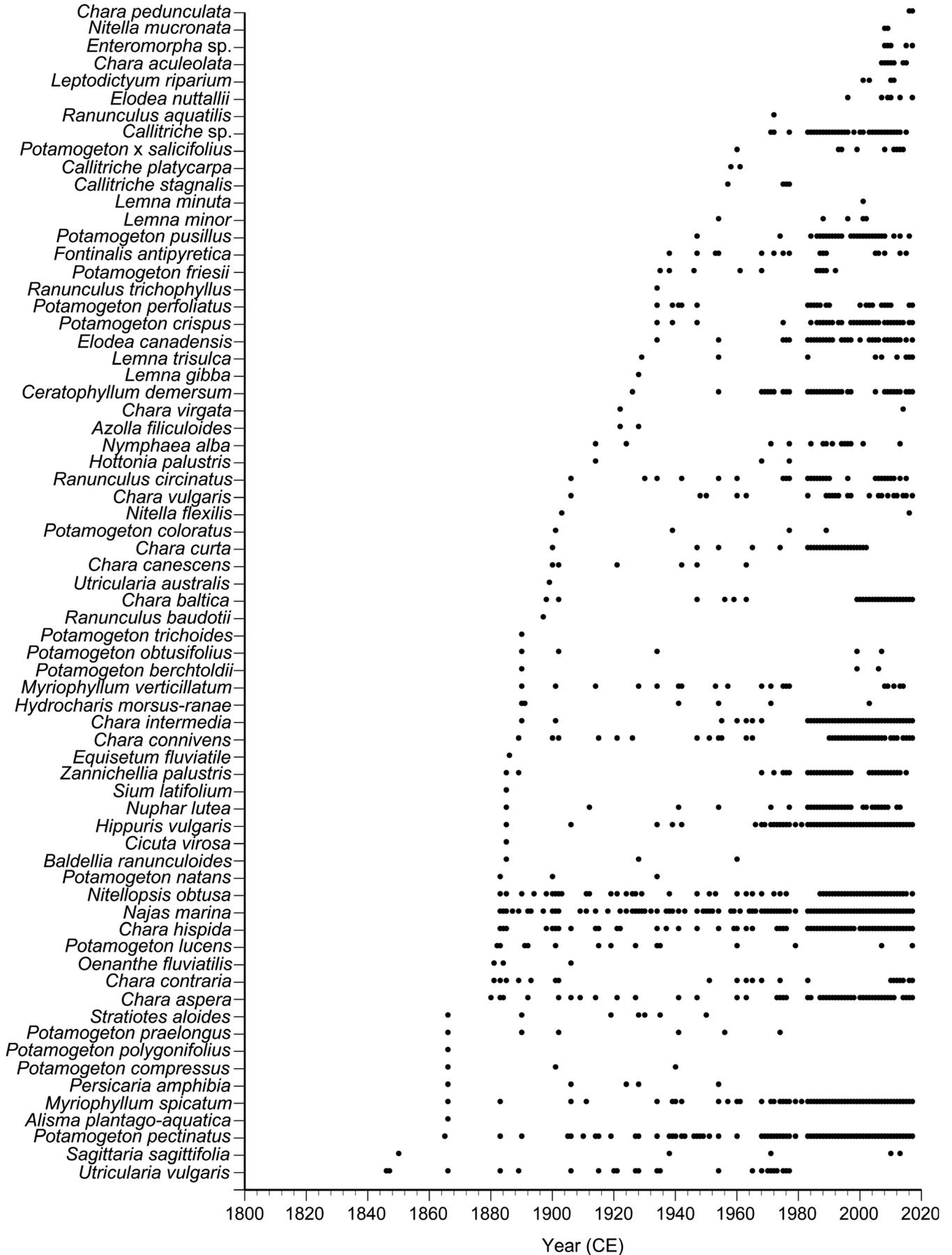
3.3 | Observational accounts of salinity

The observational evidence of salinity in the Thurne Broads is primarily focused around the flooding by seawater resulting from North Sea storm surges. Sea floods have been recorded in the Norfolk Broads since 1287, with 25 recorded in total (Table 3). There are detailed accounts of the ecological response to the 1938 sea flood with reports by Anthony Buxton (Buxton, 1939, 1941, 1943, 1944a, 1944b; Ellis, 1944) providing detailed seasonal and inter-annual response of marsh vegetation, submerged macrophytes, fish, and wildfowl to the floods (Table 4).

For the 1938 floods, Buxton (1939) reports *N. marina* (EIV = 0), *Hottonia palustris* (EIV = 0) and *U. vulgaris* (EIV = 0) as disappearing from the Thurne Broads after the flood event (Buxton, 1939). In addition to the disappearance of some submerged macrophytes, marginal vegetation, associated with surrounding marsh habitats, demonstrated a shift to more saline-tolerant species with the first marsh plants to return including *Juncus gerardii* (EIV = 3) and *Juncus maritimus* (EIV = 5). Buxton (1939) reported that filamentous algae, *Cladophora sauteri*, and *P. pectinatus* (EIV = 2) had returned to Hickling Broad, and *N. marina* (EIV = 0) had returned to Heigham Sound by 1939. In addition, reports from 1939 suggest that *P. pectinatus* was “growing rapidly” and that *Myriophyllum spicatum* (EIV = 0), *Callitriche obtusangula* (EIV = 1), *H. vulgaris* (EIV = 1), and *Lemna minor* (EIV = 1) had begun to return to dykes surrounding Horsey Mere (Ellis, 1944), where salinity was lower (0.6‰ compared with 3.51‰ or 322 mg/L compared with 1,885 mg/L). *Myriophyllum spicatum* and *C. sauteri* were also present in Horsey Mere by 1939. *Najas marina* had not returned to Horsey Mere by 1940, but was 10 times more abundant than it was pre-flood in Hickling Broad (Vincent, 1941). *Utricularia vulgaris* (EIV = 0) had not returned to Hickling Broad by 1940. By September 1941, *C. aspera*, *C. hispida*, *P. perfoliatus*, *P. praelongus*, *M. verticillatum*, *Hydrocharis morus-ranae*, and *N. lutea* had all returned to Horsey Mere (Buxton, 1943).

In March 1938, a 2 lb (0.9 kg) cod (*Gadus morhua*) was caught in Horsey Mere, a grey mullet (*Mugil cephalus*) in Hickling Broad, and a number of smelts and herrings in the dykes (Buxton, 1939). Sprat (*Sprattus sprattus*) and unknown species of crab (*Brachyura*) and brine shrimp (*Artemia*) were all noted to be present. However, it is suggested that the estuarine fish only remained in the system for a short time and freshwater fish began to return to the River Thurne, but not the actual Broads, by May 1938 (Buxton, 1939). Brine shrimp, however, remained in the system until at least November 1938

FIGURE 5 Macrophyte presence across the Thurne system. Broads Authority surveys begin in 1981, therefore increased recording after this period is due to increased recording effort. Abundance of species is not recorded. Species are listed in order of appearance in the records.



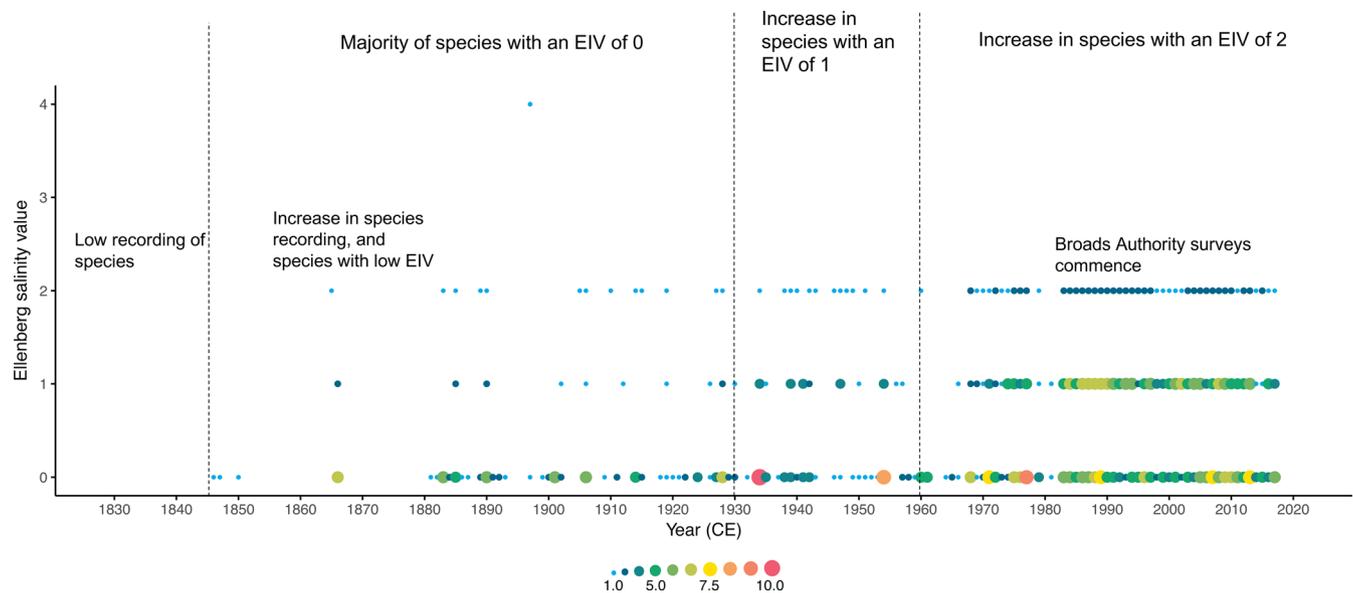


FIGURE 6 Ellenberg indicator values (EIVs) for salinity tolerance in species across the Thurne system. Colour and size of the points (shown in key) are proportional to the number of species recorded with that EIV in a given year (e.g., in 1897 one species was recorded with an EIV of 4 and in 1934 10 species were recorded with an EIV of 10, three with an EIV of 1, and one with an EIV of 2). The figure shows an increase in species with an EIV of 1 or 2 in the late 1960s, and as Broads Authority surveys commence, an increase in the number of species recorded.

(Buxton, 1939; Ellis, 1939), with Buxton (1944a) noting its disappearance by 1939. By 1941, rudd (*Scardinius erythrophthalmus*) were still scarce in the Thurne system (Buxton, 1942).

A wealth of ornithological records are also associated with the floods and can potentially inform on available grazing habitat (Table 4). Immediately after the floods of 1938, only 15 coot (*Fulica atra*) remained on the Thurne Broads in November compared with the 100–200 coot usually found at that time of year (Buxton, 1939). By November 1939, this number had increased to around 1,000 coot on Hickling Broad, but there were only four individuals on Horsey Mere (Buxton, 1944a). Coot continued to be rare on Horsey Mere until 1943, when numbers started to increase (Buxton, 1944b). There were no teal (*Anas crecca*) or shoveler (*Anas clypeata*) on Horsey Mere in 1939, but they were found overwintering alongside coot on Heigham Sound and Hickling Broad (Buxton, 1939). Coot and diving ducks feeding on *C. aspera* and *C. hispida* (Buxton, 1943; Ellis, 1944) returned to Horsey Mere in September 1941, coinciding with the return of charophytes. Despite the return of wildfowl grazing, breeding wildfowl numbers were still reduced in 1941 (Buxton, 1943). In 1942 bearded tit (*Panurus biarmicus*), mute swans (*Cygnus olor*), water rail (*Rallus aquaticus*), teal, and shoveler all bred at Hickling Broad, but not at Horsey Mere as usual. By September 1944, grey heron (*Ardea cinerea*) had returned to the area for the first time (Buxton, 1949). By 1946, teal and shoveler had still not returned to Horsey Mere since the flood (Buxton, 1947). By 1950, breeding bearded tits had still not returned to the Horsey Mere (Buxton, 1951).

Between 1953 and 1960s, high tides bringing seawater in to the system were described as annual events occurring “year after year” in “autumn and winter” by Cable (1991, p. 199, p. 105) (Table 4). However, substantially fewer documentary records exist for the 1953 and 1978 floods. Most records focus on the effects on the nearby coastline rather than on the ecology of the Thurne Broads. For the 1953 flood, however, Cable (1991) suggests that the system had recovered “within forty-eight hours” (Table 4). Considerably more detail is given to the floods of 1956, with accounts of 1 lb (0.5 kg) sea trout (*Salmo trutta*) and smelt (Osmeridae), which at that time were rare, in the River Thurne (Cable, 1991). With the introduction of regular monitoring, it has become less common for observational notes to be made; however a few exist for the 1980s and 1990s. Cable (1991) notes that although smelt were rare in 1953, it was a common component of the fish fauna in the Thurne system by the 1980s and 1990s (Phillips et al., 2016).

3.4 | Records of *Prymnesium parvum* and fish mortality

Prymnesium parvum was first documented in Norfolk in 1969 in relation to the mortality of 250,000 fish in the Thurne system (Bowler, 1971). Since then, populations have been recorded annually or biannually (Wortley & Phillips, 1987) with

TABLE 3 History of floods in the Thurne Broads from archival and documentary evidence

Year	Details	References
1287		
1601	Water breaking through the sea banks by the Hundred Stream	Cornford (1979)
1608	Water breaking through the sea banks by the Hundred Stream	Cornford (1979)
1625	Water breaking through the sea banks by the Hundred Stream	Cornford (1979)
1651	Water breaking through the sea banks by the Hundred Stream	Cornford (1979)
1655	High tide swept the dunes between Horsey and Winterton	
1665		Sainty (1939)
1683		N.R.O. (1792)
1715	Water breaking through the sea banks by the Hundred Stream	Cornford (1979)
1717	Water overtopped the banks, but no account of such widespread flooding	Cornford (1979)
1718	Water overtopped the banks, but no account of such widespread flooding	Cornford (1979)
1720	Water overtopped the banks, but no account of such widespread flooding	Cornford (1979)
1791	Water breaking through the sea banks by the Hundred Stream	Cornford (1979)
1792		
1805	Tide reached highest level on record at 10.6 ft above Newlyn datum	Mosby (1939)
1897		
1907		
1938	Water breaking through the sea banks by the Hundred Stream. Total flooded area of 7,469 acres. Water was pumped off flooded land into New Cut, Meadow Dyke, and Candle Dyke	Mosby (1939)
1953	106,161 acres flooded, 307 people died	Steers (1953)
1970	Major drying of some smaller broads likely to increase salinity. The Thurne reached up to 60% seawater	Wortley (1976)
1976	Drought and major drying of some smaller broads likely to increase salinity	Wortley (1976)
1978	Tidal surge	Steers et al. (1979)
1993	Tidal surge	Kindleysides (1993)
2007	Tidal surge	Eastern Daily Press (2013)
2013	Tidal surge	Eastern Daily Press (2013)

Droughts are highlighted in italicised text as they were likely to lead increased salinity through a reduction in freshwater inputs.

the majority linked to fish mortality. However, since the importance of *P. parvum* in fish kills was only first established in 1969, few quantitative historical records of *P. parvum* concentrations of the Thurne Broads exist prior to this date. In September 1974 to November 1976, *P. parvum* did not account for any of the algae present in the Thurne Broads (Watson, 1981). There are no further data on counts of *P. parvum* cells until 1996 when sporadic but more regular monitoring began for Hickling Broad and Horsey Mere. However, for times when salinity was also recorded, there is no significant relationship between chloride concentration and *P. parvum* cell concentration (Figure 7). Even where documented fish kills may, therefore, indicate an increased concentration in *P. parvum*, there is insufficient evidence to suggest that these outbreaks coincide with periods of increased salinity as previously proposed (Brown, 2017; Holdway et al., 1978; Wortley & Phillips, 1987).

4 | DISCUSSION

4.1 | Salinity reconstruction of the Thurne Broads system

The wealth of information on the ecology of the Thurne Broads provides the opportunity to improve knowledge on the past salinity of the system. Based on the data presented here, the average salinity of the system has increased since the 1970s

TABLE 4 Observational evidence of salinity in the Thurne Broads from documentary and archival evidence

Year	Comment	References
May 1938	Salinity had dropped, but by this point Hickling was more saline than Horsey	Buxton (1939)
Jan 1939	Sharp decline in salinity following rain	Buxton (1944a)
1939	Brackish water hydroid (<i>Cordylophora caspia</i>) now disappeared from Horsey Mere	Buxton (1944a)
Sep 1941	<i>Chara aspera</i> , <i>Chara hispida</i> , <i>Potamogeton perfoliatus</i> , <i>Potamogeton praelongus</i> , <i>Myriophyllum verticillatum</i> , <i>Hydrocharis morsus-ranae</i> , and water lilies recorded for first time since flood in Horsey Mere	Buxton (1943)
Autumn 1943	Fungus, <i>Hebeloma mesophaeum</i> , abundant after the flood is now only prevalent in a few isolated spots	Buxton (1944b)
Jan 1953	“... within forty-eight hours the river was back to normal”	Cable (1991, pp. 26–27)
1953	“Year after year the salt encroached further and further upstream ...”	Cable (1991, p. 199)
Oct 1956	“... salt water had poured into the River Bure, pushing well up into and beyond the River Ant, into the River Thurne...” Sea trout of 1 lb caught in the Lower Thurne Smelt from the lower Thurne was very rare, but plentiful by 1991	
1959	“The annual Autumn and Winter salt tides continued to take their toll on the fresh hatch of fry”	Cable (1991, p. 105)
Early 1960s	“... samples throughout the year on Hickling Broad and Horsey Mere ... remained almost constant at 15% salt content on Hickling and 20% on Horsey”	Cable (1991, p. 125)
Early 1960s	“After the salt tide, they always returned, and then in the early sixties most of the roach population mysteriously disappeared. It was thought that some disease had killed them, but if this were the case I never saw any dead”	Cable, 1991, p. 143
July 1968	Surface sodium content 840 ppm	Cable (1991, p. 76)
1971	Horsey dredged	Morgan and Britton (1969)
1980s	Although not plentiful, the pike had recovered from the 1969 Prynnesium outbreak	Bailey (1984)

due to (1) an increase in the maximum values of salinity (minimum values remain relatively constant), and (2) the slow rate of the recovery of the system from sea floods.

Increases in background salinity were thought to be due to increased efficiency of agricultural drainage in the Thurne catchment to the upgrade to diesel-powered pumps (Holman & Hiscock, 1993). Currently, maximum values of chloride concentration in the Thurne system are recorded during October–January in Hickling Broad, Horsey Mere, and Heigham Sound, and during April–June in Heigham Sound and the Martham Broads. The average monthly water discharge from the Brograve, Catfield, Stubb Mill, and Horsey Mill pumps (Figure 1) is highest between November and February, while the Somerton (North and South) pumps have a less seasonal influence (I. Holman, personal communication, 2018). Increased maximum values of chloride concentration during the winter months are, therefore, likely caused by the pumping of saline water into the system from the drainage pumps. This constant input of saline water has undoubtedly hindered the recovery of the system from the 1938 sea flood, in agreement with the mesocosm experiments of Barker et al. (2008).

The importance of the combination of sea floods and catchment drainage in shifting baseline (average background) values of salinity is evident in Martham Broads. These broads have not experienced increased salinity, but are less affected by the tidal influence of the River Thurne and therefore receive little, if any, salt water from high spring tides. Spot salinity measurements taken by the Environment Agency during the 2013 storm surge suggest that saline water penetrates the River Thurne as high as Martham Dyke, but this is still some distance downstream to the Martham Broads (Figure 1). Whilst also positioned away from the main drainage activity, it has been suggested that the Martham Broads are still affected by changes in the drainage from Somerton North and South pumps (e.g., the doubling of salinity associated with the drainage of the Somerton levels; George, 1992). However, our data suggest that average salinity has not increased in these Broads,

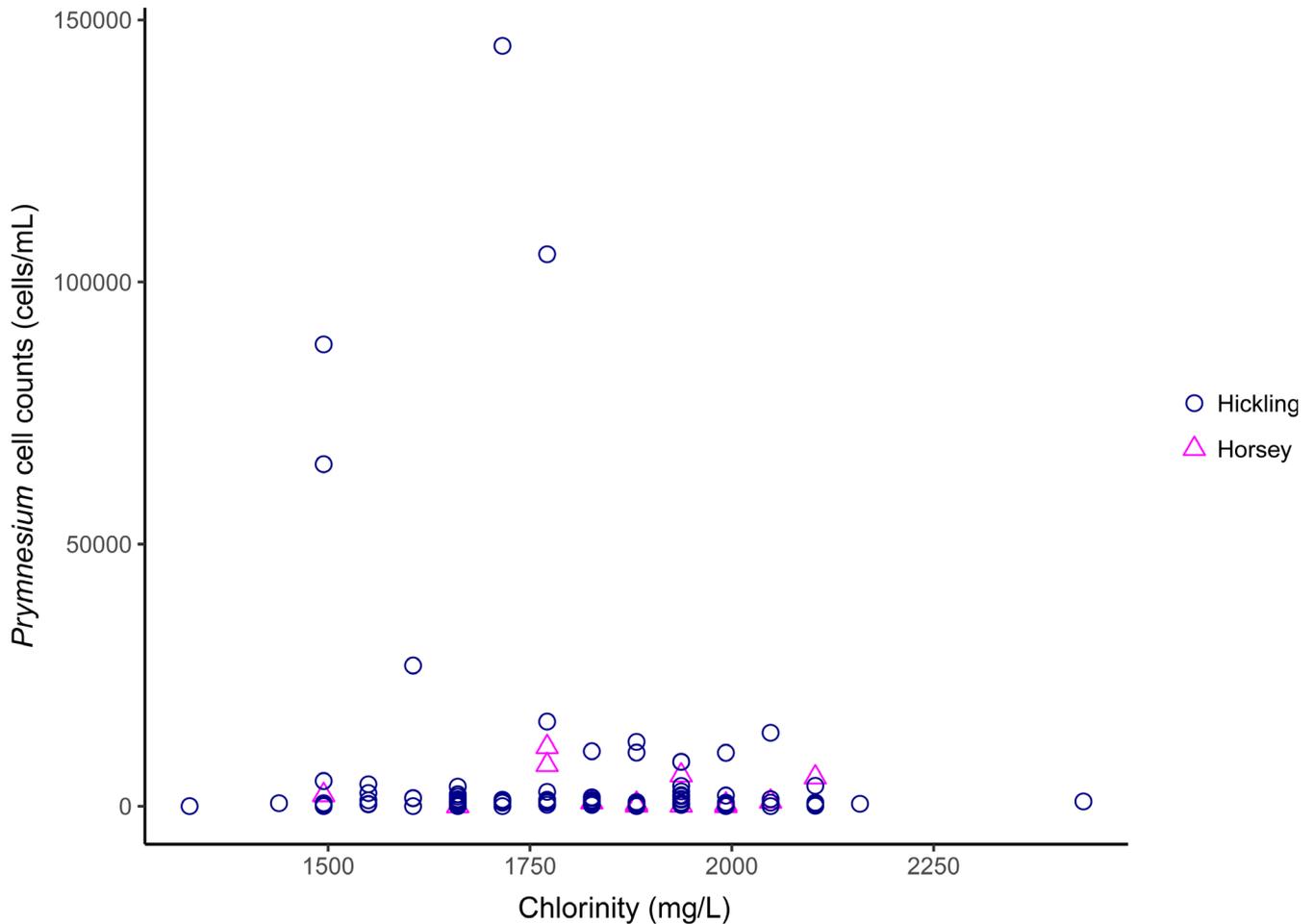


FIGURE 7 Relationship between chlorinity and *Prymnesium parvum* concentration in Hickling Broad (blue circles) and Horsey Mere (pink triangles) from Environment Agency and Broads Authority monitoring records since 1995. Records for Horsey Mere only extend to 2000.

potentially indicating that land drainage has a limited influence. It is likely that it is the combination of floods, lack of flushing, and drainage that has affected the salinity of the other broads.

Using the salinity values alone, it is difficult to confirm that the increased average salinity in the Thurne Broads is solely an outcome of more powerful and deeper drainage regimes. Very few data points exist between the time of the two major sea floods (1938 and 1953) and the improvements to the drainage network between 1940 and 1960. Any changes in salinity between 1945 and 1973 are therefore difficult to relate to specific natural or anthropogenic events. Furthermore, the first drainage scheme began in 1940, increasing the efficiency of the Brograve pump by upgrading it to diesel, before the system had fully recovered from the effects of the 1938 flood (e.g., Buxton, 1941).

There are no measured salinity values for Hickling Broad or Horsey Mere that suggest salinity ever returned to its pre-flood values after 1938. In fact, the average salinity between 1944 and 1973 for Horsey Mere (3,750 and 3,416 mg/L) and Hickling Broad between 1940 and 1968 (1,147 and 1,250 mg/L) are remarkably similar. Despite the lack of measured salinity, during this period there is some archival and documentary evidence of salinity. Two major episodes of high salinity are recorded over this period; during the sea flood of 1953 and the summer droughts of 1970 (Steers, 1953; Wortley, 1976). Between 1952 and 1973, three saline-tolerant species (*P. pusillus*, *L. minor*, and *C. stagnalis* with EIVs of 1) were first recorded in the system.

It is evident that a combination of sea floods and anthropogenic catchment modifications have increased salinity in the Thurne system as a whole. However, the primary barrier to disentangling the impacts of sea floods and changes in drainage is the unknown rate of recovery of the system from the 1938 and 1953 events. The detailed accounts of Buxton (1939, 1941, 1943, 1944a, 1944b) clearly document the recovery of the system. However, the observations are sometimes limited in their wider relevance to salinity. The Buxton reports document the staggered return of macrophytes to the Thurne system. Whilst the initial species to return to the system have a clear salinity tolerance (e.g., *Juncus gerardii* with an EIV of 3), species with

EIVs of 1 or 0 have varying rates of recovery. Rather than being related to declining salinity, the recovery of the macrophyte community structure may be related to germination rates of species from seed. For example, *P. pectinatus*, which returns to the system within a year of the 1938 flood (Buxton, 1939), germinates rapidly from seed whereas *P. natans* seeds can, in some cases, remain dormant for up to three to four years (Sauvageau, 1894). Germination may, therefore, account for some of the delayed recovery. However, *P. perfoliatus* and *P. praelongus*, which did not return to Horsey Mere until 1941, germinated within a year in culture experiments (Sauvageau, 1894). The slow recovery of Horsey Mere is discussed extensively in the documentary evidence of the floods. Sustained high salinity in Horsey Mere until 1968 is suggested by similar values recorded in 1944 and 1973.

4.2 | Interactions with eutrophication

Aside from salinity, eutrophication can also lead to the loss of macrophyte abundance and diversity in shallow lakes (Scheffer, 1989). Moss and Leah (1982) disregarded salinity as the main factor responsible for loss of macrophytes and the development of phytoplankton dominance in Hickling Broad, citing eutrophication as the main agent of change. However, they recognised the importance that salinity played in the survival of *P. parvum* and *Neomysis integer*, a common brackish to marine water shrimp that predates on micro-crustaceans, including Copepods and Cladocera. When the models of Moss and Leah (1982) were developed, *N. integer* was not abundant and not considered to be a major disturbance in the system. Since fish are major predators of *N. integer*, fish mortality led to an increase in *N. integer* and a decrease in Cladocera (Ellis, 1965; Irvine et al., 1993). It is possible, therefore, that the annual fish mortalities along with brackish water conditions (large-bodied *Daphnia* decline rapidly at ~2‰; Jeppesen et al., 2007) decreased numbers of zooplankton grazing on algae, thus inducing macrophyte loss. Barker et al. (2008) confirmed the decline of daphnids at high salinities alongside fish predation. The primary mechanism for macrophyte loss could therefore be increased salinity either long term or episodically, rather than nutrient enrichment, or both mechanisms together.

Since the Thurne Broads receive relatively little nutrient-rich effluent and have less intensive agricultural practices in their catchment than other Broads, the source of nutrient enrichment is unclear (Irvine et al., 1993). Although Moss and Leah (1982) dismissed salinity as a major cause of the loss of macrophytes and increase in phytoplankton in the 1970s, the interactions between eutrophication and salinity must be fully explored. With increasing salinity and the influence of increased ionic concentration, the mobilisation of PO_4^{3-} bound in sediments leads to N and P eutrophication acting simultaneously to the impacts of salinisation. Furthermore, increasing salinity reduces the phosphate-sorption capacity of the sediments, increasing available phosphate (Sundareshwar & Morris, 1999). If the salinity remained high following the 1938 floods, the capacity of the sediments to hold phosphate could have been limited. In addition, although boat use has now decreased (with a 20% decline in small private motor boats between 2008 and 2015; Broads Authority, 2018), the remobilisation and erosion of sediment by boat propellers has previously been a problem in the Broads (Davies, 1883; Mason & Bryant, 1975; Moss, 1977). A particular increase in boat use occurred in the 1960s and 1970s, with a threefold increase in boats from 1947 to 9,247 boats registered by 1964 (Moss, 1977). Furthermore, the widespread loss of macrophytes, which would act to stabilise sediment, coupled with the coastal winds contributes to constant and high turbidity. Along with the increased eutrophication in the 1970s, the remobilisation of these sediments by boats and wind would have led to increased N and P eutrophication. The high internal loading of P from sediments has been confirmed by others (Moss, 2001; Phillips & Jackson, 1990; Phillips et al., 2016), but little attention has been given to the role that increased baseline salinity may have on phosphate availability.

Furthermore, many of the macrophyte species that are tolerant of increasing salinity are also tolerant of increasing trophic status. For example, the deeper mean depth and higher P concentration of Horsey Mere result in poor light penetration, which contributes to the low macrophyte abundance and diversity of this waterbody, along with the high salinity. The Trophic Ranking Scores (TRS; Palmer et al., 1993) for species with an EIV of 1 or above are listed in Table 5. However, there are a number of species that have low EIV values for salinity, but a high TRS (e.g., *Myriophyllum spicatum* EIV = 0 and TRS = 10.0). Several of these species returned to, or are first recorded in, the system between 1940 and 1960. During this period, *M. spicatum*, *L. triscula* (EIV = 0, TRS = 10.0), *E. canadensis* (EIV = 0, TRS = 8.5), and *P. friesii* (EIV = 0, TRS = 10.0) are all present. Each has a very low salinity tolerance, but a high trophic tolerance, and although the abundance of each is unknown, the high TRS suggests eutrophication may have been an important control on the macrophyte community structure. However, between 1940 and 1960, all four species were present in Heigham Sound while *M. spicatum* and *P. friesii* were recorded in Hickling Broad. Limited records exist for Martham Broads at this time. Although records are sporadic across the period, none of the species were recorded in Horsey Mere, suggesting that salinity was still

too high here for *M. spicatum*. Given the higher salinity at this site, it is not surprising that eutrophication would have a minor influence on ecosystem regime.

The increased remobilisation of sediment from increased boat use and wind, but relative reduction in salinity from peak flood levels, would account for the presence of species with a low EIV for salinity, but high TRS. Ertsen et al. (1998) demonstrated that Ellenberg indicator values for salinity can be successfully calibrated to the Cl concentration of groundwater. The EIV values presented here are therefore a reflection of chloride concentrations (i.e., seawater input), rather than conductivity (i.e., nutrient and seawater input), which could be related to eutrophication.

The succession of macrophyte presence in the Thurne Broads does not reflect previously suggested patterns observed with eutrophication-driven changes in macrophyte diversity in shallow lakes. Previous work (e.g., Sayer et al., 2010) suggests a shift in dominant species from *Chara/Myriophyllum* → *Ceratophyllum* → *Potamogeton* (particularly *P. pectinatus* and *P. pusillus*). In the 1960s and 1970s species of *Myriophyllum*, *Ceratophyllum*, and *Potamogeton* are recorded in unison, suggesting other controls on community structure, which may be attributed to the brackish water conditions. However, following the fish mortality events of 1969, the system shifted to a *Ceratophyllum/Potamogeton*-dominated system, suggesting an increased influence of eutrophication with the increase in predation on Cladocera from brackish-water shrimp.

Despite the clear combined effects of salinity and eutrophication, once the black gull population decreased in the 1980s, Hickling Broad returned to a clear water state (Armitage et al., 2001) without any reductions in salinity. The role of eutrophication in increasing nutrient availability is therefore a major cause of the phytoplankton-dominated state of the Thurne Broads. However, it remains unclear how heavily the shift from macrophyte abundance also relies on high salinity.

5 | IMPLICATIONS FOR MANAGEMENT

Effective conservation of an area must be informed by a range of data sources (Dawson et al., 2011). Here we have improved the understanding of the functioning of the Thurne system by assembling records not previously used to assess causes of change and consequent ecosystem recovery, or lack of recovery, in the Thurne Broads. Effective management of the Thurne Broads and its catchment requires balancing the needs of multiple uses, including navigation, conservation, recreation, and agriculture.

Previous management suggestions for the Thurne Broads have been to reduce salinity to promote macrophyte growth and a return to clear water (Barker et al., 2008). From the data presented here, it is likely that new baseline chloride concentration after the 1938 sea flood is still above the “Favourable Condition” criteria (600 mg/L chloride concentration for Hickling Broad; Broads Authority, 2006). By analysing monitoring, archival, and documentary evidence, it is now evident that the subsequent drainage improvements have caused greater fluctuation of salinity of the system. The primary concern for the management of the system should be the increase in maximum chloride concentrations, and its ecological consequences for species and ecosystem functioning.

TABLE 5 Trophic Ranking Scores (TRS) (Palmer et al., 1993) for species in the Thurne Broads with an Ellenberg indicator value (EIV) (Ellenberg et al., 1991) for saline tolerance above 1. A TRS of 10 indicates species with a high tolerance to eutrophic waters.

Species	TRS	EIV	Sites
<i>Ranunculus baudotii</i>	10.0	4	Horsey Mere
<i>Potamogeton pectinatus</i>	10.0	2	Hickling Broad, Horsey Mere, Heigham Sound, Martham Broads
<i>Zannichellia palustris</i>	10.0	2	Hickling Broad, Horsey Mere, Heigham Sound, Martham Broads
<i>Callitriche stagnalis</i>	7.7	1	Hickling Broad, Martham Broads
<i>Ceratophyllum demersum</i>	10.0	1	Hickling Broad, Horsey Mere, Heigham Sound, Martham Broads
<i>Hippuris vulgaris</i>	7.7	1	Hickling Broad, Horsey Mere, Heigham Sound, Martham Broads
<i>Lemna gibba</i>	N/A	1	Hickling Broad
<i>Lemna minor</i>	9.0	1	Hickling Broad, Horsey Mere, Heigham Sound, Martham Broads
<i>Nuphar lutea</i>	8.5	1	Hickling Broad, Horsey Mere, Heigham Sound, Martham Broads
<i>Potamogeton crispus</i>	8.5	1	Horsey Mere, Heigham Sound and Martham Broads
<i>Potamogeton perfoliatus</i>	7.3	1	Hickling Broad, Horsey Mere, Martham Broads
<i>Potamogeton pusillus</i>	8.5	1	Hickling Broad, Horsey Mere, Heigham Sound, Martham Broads
<i>Stratiotes aloides</i>	N/A	1	Horsey Mere, Martham Broads

The sea floods had substantially more influence on the system than previously considered, and with North Sea storm surge activity predicted to become more extreme by the end of the century (Weisse et al., 2012; Woth et al., 2006), protection against future floods is essential; further flooding may push the baseline of the system higher. However, it must also be recognised the system is naturally brackish (Pallis, 1911) and management should not aim to achieve a freshwater environment. Instead the system should be managed in line with seasonal fluctuating salinities of over 2,000 mg/L in the Thurne Broads. Barker et al. (2008) suggest that diversion of some saline water to the sea could result in salinity reduction to between 1,600 and 1,800 mg/L. Whilst a maximum value of salinity would place a threshold for ecological tolerance, it is not sustainable in a dynamic environment influenced by sea floods.

The understanding of shifts in macrophyte diversity and ecosystem recovery to stressors relies on the ability to test and fully understand past patterns in the system. However, the inability to remove a pressure (e.g., one that is a natural phenomenon such as tidal incursions in the Thurne Broads) that creates shifts in baseline conditions limits our ability to effectively manage and restore these systems.

6 | CONCLUSIONS

From the evidence presented here, it is likely that a combination of both increasing salinity (from storm-induced sea floods and anthropogenic catchment modifications) and eutrophication caused the widespread loss of macrophytes from much of the Thurne Broads system in the 1970s. However, there is still limited information on the recovery of the system and the baseline salinity prior to the 1960s. Without robust high-resolution understanding of the salinity prior to the monitoring period, past salinity is still difficult to fully assess.

It is likely that the Thurne system did not fully recover from the effects of the 1938 sea flood. In combination with drainage modifications to the catchment currently driving maximum chloride values, the recovery of the system has been hindered. A combination of sea floods and anthropogenic catchment modifications have shifted baseline salinity values, particularly for Horsey Mere and Hickling Broad. Due to the more direct effects of the floods and the ongoing influence of the Brograve pump, the recovery rate both ecologically and chemically of Horsey Mere has been longer than for the other Broads. However, all Broads have been adversely affected by eutrophication before full recovery.

The consolidation of records for the Thurne Broads has improved our understanding of the interactions occurring in the system between 1940 and 1970. However, the use of the Thurne Broads to discuss reference conditions is problematic since Hickling Broad (including Heigham Sound) and Horsey Mere are designated heavily modified under the WFD; moreover the entire Broads system is anthropogenic in origin. However, the Thurne Broads have acted as a test-bed for the approach, and as a low-cost (often open access) (Appendix S1), effective method, it could be applied elsewhere in situations where pre-anthropogenic reference conditions can be more clearly assigned.

Historical records can be problematic by their fragmentary and random nature, selectively recording species or events. Common species can therefore be under-represented in the records and rare species, highly prized by naturalists and conservation bodies, given greater prominence. Furthermore, observations are often anthropocentric, focusing on crop success after flooding, the more visible and aesthetic marsh vegetation, the presence of wildfowl, or the effects on industrial land (e.g., Buxton, 1939, 1941; Steers, 1953). Despite this, while technology has improved the precision and resolution of records, field notes and observations remain a primary research method to produce narratives of changes (Yeh & Klemmer, 2004). The primary benefit of observational notes is their tendency to record apparently irrelevant, non-linear, anomalous results that when assembled may result in a series of events that improves understanding of long-term ecological change (Boero, 2013).

When using historical data to help define WFD targets, it is therefore necessary to account for several uncertainties, namely (1) the methodology cannot distinguish between absence of a species or lack of recording; (2) the presence of a species is often recorded, but not abundance; (3) increases or decreases in species may be related to sampling intensity and frequency, and therefore inferred changes in community structure are largely uncertain; and (4) the method cannot distinguish between drivers of change. In sites such as the Thurne Broads where multiple drivers are influencing the biodiversity of the system, the method can be used to determine reference conditions, but it cannot be used to inform the management of individual environmental drivers. To improve understanding of the interactions between eutrophication and salinity throughout the system, high-resolution, fully quantitative records are required on a site by site basis. The approach of combining archival and documentary evidence with paleolimnological records has been effectively achieved for Barton Broad (e.g., Madgwick et al., 2011), and could effectively be employed here to improve these records.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

Appendix S1. Open access sources used in the analysis. These sources may be used for further analysis, and include data for other locations.

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