Review of Biomanipulation

Appendix 4
Broads Lake Restoration Strategy

Photo by Jim Tyree

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Aims

The aims of the review are to evaluate the extent to which biomanipulation efforts have achieved the original restoration aims, which were to: -

- Shift the fish community from one dominated by roach and bream to one that is more diverse; containing a high proportion of piscivores (pike and perch), and species such as tench and rudd.
- Improve water clarity through increased zooplankton grazing pressure; and
- Re-establish and maintain submerged macrophyte populations through increasing light climate and reducing potential pressure from benthivorous fish.

Methods

Data incorporated in this review has been compiled from a number of sources. Water quality and zooplankton data has been provided by the Environment Agency and some water quality summary data (1979-89) was extracted from Moss *et al.* (1996). The large bodied grazing Cladocera encountered in the broads have been grouped together as they are distinct from the copepods and *Bosmina longirostris* in terms of maximum body size, filtering rates and feeding behaviour. The species included in this grouping includes *Daphnia* spp., *Ceriodaphnia* spp., *Diaphanosoma brachyurum* and *Sida crystallina*.

The Broads Authority has contracted ECON Ecological Consultancy to undertake annual fish surveys. For the purposes of this report, fish species have been divided into three feeding guilds (albeit crudely, with no regard for the size of individuals). These groups are zooplanktivorous (roach, roach hybrids, rudd, 10-spined and 3-spined sticklebacks), benthivorous (bream, tench, ruffe and gudgeon) and piscivorous (pike, perch and the potentially piscivorous eel). Young bream and perch and ruffe and gudgeon can all consume zooplankton, but generally have a more mixed diet than the main zooplanktivorous species, roach (M. Perrow pers. comm.). The Broads Authority has also conducted macrophyte surveys annually in August, with the sampling method outlined in Kennison *et al.* (1998).

1.0 Cockshoot Broad

Cockshoot Broad was isolated from the River Bure in 1982 due to high nutrient loading from sewage treatment works (STWs). As part of the same restoration project about 70 cm depth of surface sediment was removed from 3/5^{ths} of the broad's area (Moss *et al.* 1986). Total phosphorus (TP) and chlorophyll *a* concentrations fell rapidly and large numbers of *Daphnia* spp. were recorded (Moss *et al.* 1996). From the mid 1980s fish stocks began to increase, with a concomitant decline in the *Daphnia* spp. population, and subsequently macrophyte growth disappeared from the main basin due to increased turbidity. This prompted the near complete removal all of the fish community from the broad in early 1989. A further removal exercise was also carried out in early 1990. These initial total community removals were followed up with annual top-up removals of zooplanktivorous species such as roach and rudd, and also the potentially zooplanktivorous bream and perch. Spring spawning disruption operations have been carried out annually along favoured margins for egg laying, through setting nets that act as artificial spawning substrates. Eggs laid on the nets are destroyed by removing the nets from the water and allowed to air dry. An increase in removal effort from 2000, combined with poor roach recruitment in 2002 and 2003 successfully maintained a low roach abundance, whilst tench and rudd recruited successfully. From 2004 onwards only roach were actively removed from the broad, with comparatively low numbers actually being caught. The continued low roach abundance may be partially attributed to improved isolation of the broad from the river network, following extensive work to block off adjacent dykes (A. Kelly pers. comm.) For details of the number and biomass of fish removed each year see Table 9.1.

Results from the autumn point abundance sampling by electrofishing (PASE) surveys, undertaken from 1993 to 2005 in Cockshoot Broad, are presented. The overall mean fish species data (open water and littoral results) were extracted from the following ECON fish survey reports for the production of the figures in this chapter; Tomlinson & Perrow (2005a) for the 1993-2003 PASE data; and unpublished ECON data files for 2004 – 05 PASE data.

1.1 Effects of biomanipulation on the fish community

The abundance of roach from 1993 – 1995 (grey bars, Figure 1.1) was very low in these first few years of the PASE survey (<0.1 ind. m^{-2}) and suggests their population recovery was effectively controlled following the initial removals in 1989/90 and the subsequent annual top-up removals. This species population regained significant abundance and biomass in 1996 and 1997. Whether this increase was due to in-lake recruitment or immigration from the wider river system is unknown. Conversely, the bream population (black bars) appears to have been effectively controlled over the long-term, as both abundance and biomass (Figures 1.1 & 1.2) of this species have been reduced to negligible values since 1996. A large recruitment of 0+ roach took place in 2000, but this event appears to have been controlled through the removal work, with much reduced abundance and biomass in the years following. 2003 was another good year for roach recruitment and also for perch, with significant peaks in abundance and biomass for both species. However, the roach population in the two proceeding years was low. The relatively large number of individual roach removed from the broad in 2002 and 2003 (3,634 and 10,243 respectively) would have contributed to this decline.

The rudd population has varied during the sampling period, with strong representation in the period 2001-04. Pike and tench data are somewhat more difficult to interpret, as sampling of these often larger body-sized species is less frequent in the data (note the large pike biomass in 1995 and 2000, both caused by the capture of relatively large individual fish). However, both species have been retained in the broad following the complete fish community removals in 1989 and 1990, and their presence has been casually observed within the broad. From the large variation in roach population abundance and biomass it is clear that this species potentially has the capacity to rapidly increase its population if the relevant ecological factors controlling abundance are not operative. However, at Cockshoot, the broad has not always been completely isolated from the river, as during high tide events water from the river has bypassed the dam. There are also several small dykes that connect to the surrounding wetland areas. Those connecting with the main dyke were sealed off with plastic pilling in 2001 to prevent any potential immigration of fish from the broad. The dam structure has also been improved to prevent water getting round the sides during high water. These connections to the wider river system have not been proved to be the cause of fish immigration, but are highly probable, especially for the peak in roach abundance that occurred in 2000, prior to the improved isolation works.

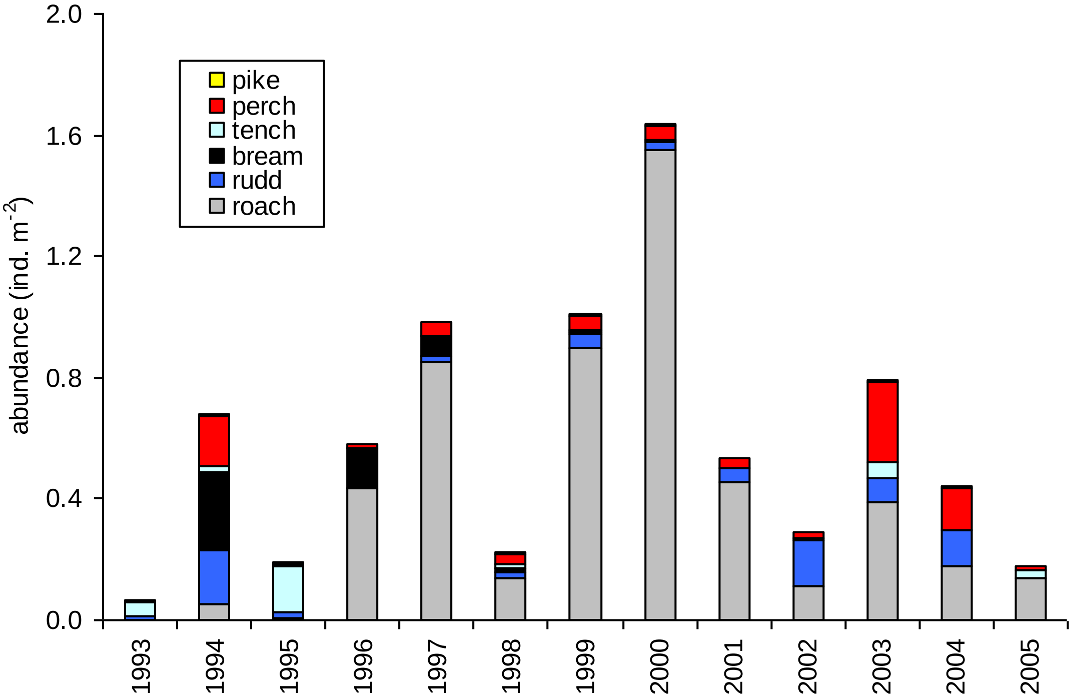

Figure 1.1 Abundance of the dominant fish species in autumn surveys at Cockshoot Broad (open water & littoral, 1993-2005)

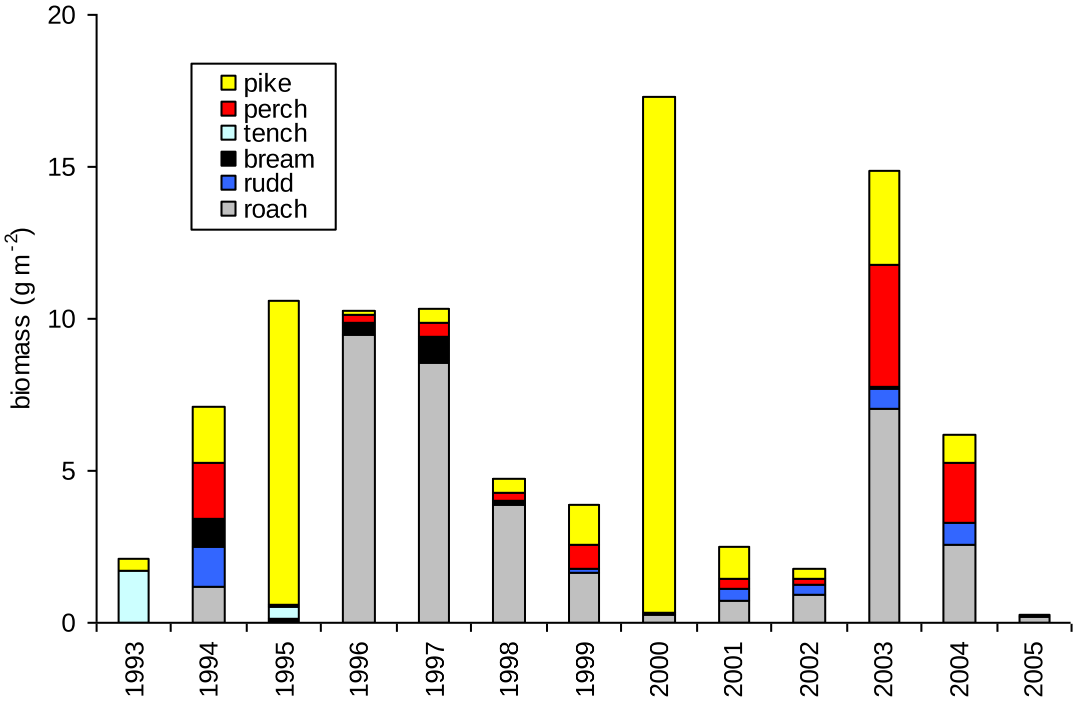

Figure 1.2 Biomass of the dominant fish species in Cockshoot Broad (open water & littoral, 1993-2005)

1.2 Water clarity and zooplankton grazing pressure

Summer water clarity in Cockshoot Broad, as determined by mean chlorophyll a concentration is highly correlated with summer mean TP concentration ($r = 0.924$, $N = 27$, $p = <0.001$). However, neither summer nor annual mean Secchi disc data were correlated with any of the other water quality variables measured, e.g. total phosphorus, chlorophyll a, total oxidised nitrate, etc. Measuring water clarity in shallow lakes is often complicated by the fact that during clear water conditions the lake bed can be seen, giving no maximum depth for traditional Secchi disc readings. In these situations the maximum water depth can only be recorded whilst using this method. This makes data analysis very difficult, as results from clear water periods are not directly comparable with more turbid periods, hence use of the chlorophyll *a* concentration in this report as a proxy for turbidity.

In shallow lakes, the lower the chlorophyll *a* /TP ratio, the lower the proportion of chlorophyll *a* to TP, indicative of the presence of factors other than P availability in limiting algal productivity, such as grazing by zooplankton. For example the years 1993 – 96 in Figure 1.3 were the years with the lowest chlorophyll *a* /TP ratios from Cockshoot Broad. This figure also shows the rapid decrease in TP from 2002, to the most recent period (2004-2006), which is characterised by annual mean TP of <50 µgl^{-1}. It could be argued that years previous to 2004, the broad was below it's potential in terms of recovery, and only the more recent dominance of macrophytes has driven TP below the 50 µgl^{-1} concentration. See the following section for a description of the macrophyte growth trends in Cockshoot Broad.

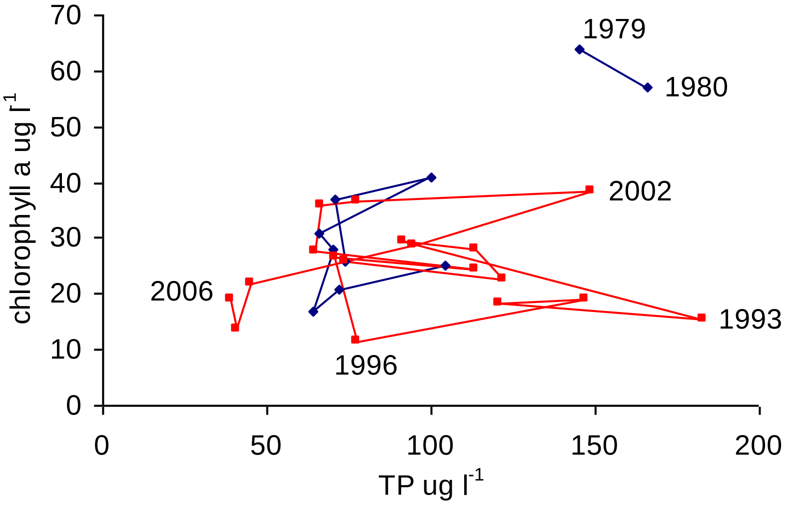

Figure 1.3 Trajectory of annual mean TP and chlorophyll a in. Blue = pre fish removal, red = post fish removal

In Cockshoot Broad, the summer mean zooplankton data (Figure 1.4a) shows that there have been distinct “high” and “low” *Daphnia* spp. biomass years. When plotted in this way, the “high” *Daphnia* summers (1990-96) appear to have a slightly lower chlorophyll a /TP ratio, although no significant negative relationship was found. Within the Cockshoot dataset there are no direct relationships between zooplankton biomass values and the zooplanktivorous fish abundance or biomass. However, roach abundance is positively related to the chlorophyll a /TP ratio values for the same year (r = 0.677, N = 11, p = <0.05) (Figure 1.4b). For example, the year with highest roach abundance, 2000, had the greatest chlorophyll a /TP value, indicating that roach were controlling zooplankton grazing rates, thus the greater chlorophyll a /TP value.

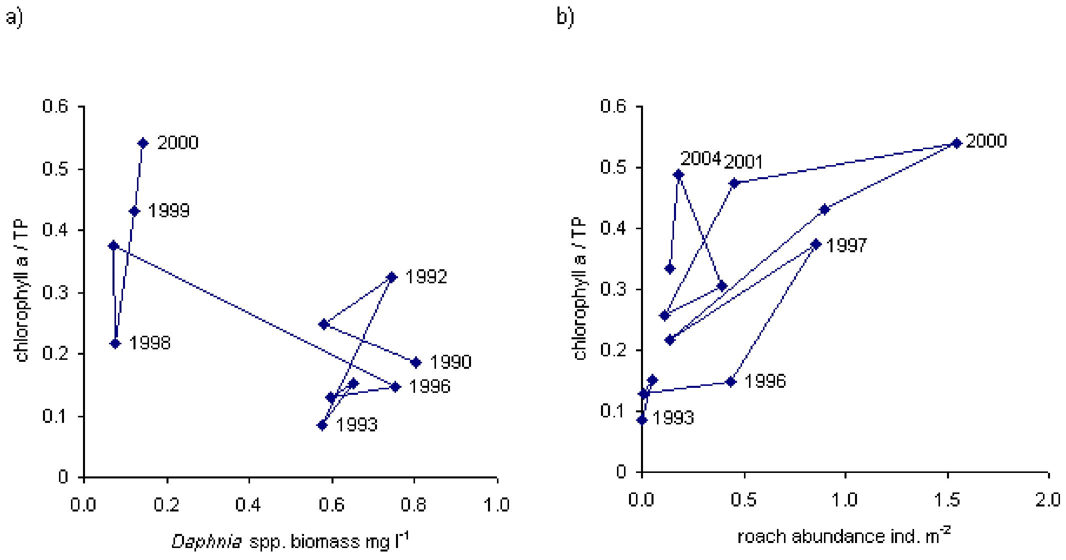

Figure 1.4 Trajectory of the relationship between chlorophyll *a* /TP ratio and a) summer mean large grazing Cladocera biomass and b) roach abundance

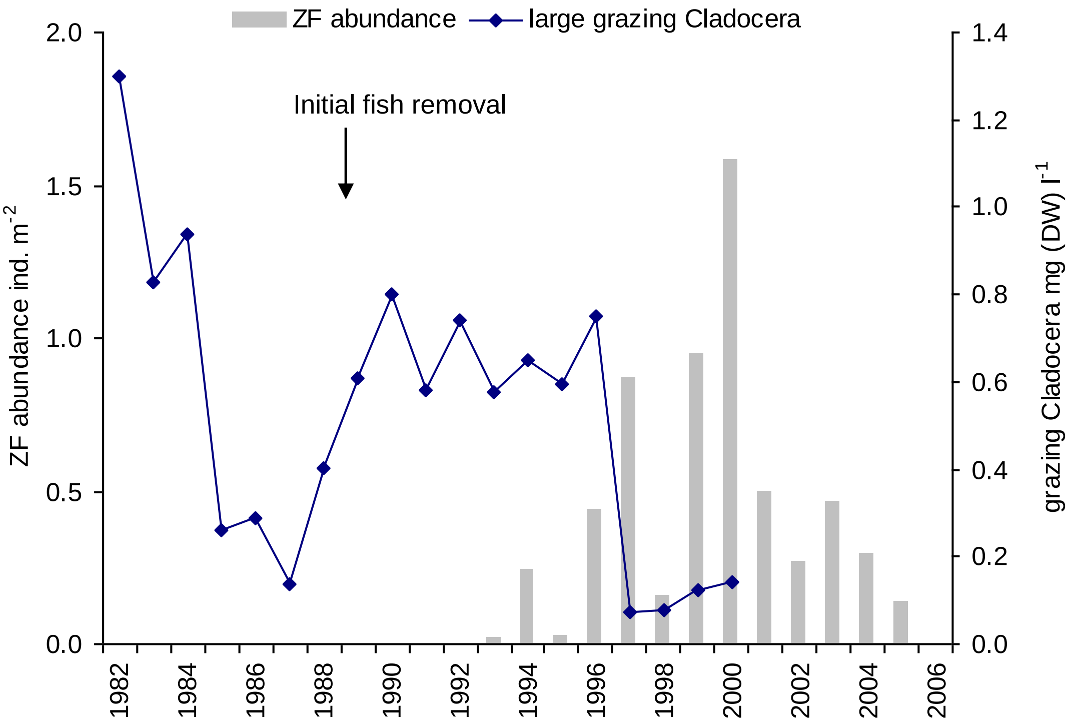

Figure 1.5 Zooplanktivorous fish (ZF) abundance and summer mean biomass of grazing Cladocera in Cockshoot Broad.

Figure 1.5 shows variation in ZF abundance and the biomass of the large grazing Cladocera. The period where monitoring of fish and zooplankton occurred simultaneously was between 1993 and 2000. The years 1993-1995 were characterised by relatively low ZF abundance (<0.3 ind. m^{-2}) and relatively high large grazing Cladocera biomass (>0.5 mg l^{-1}). The ZF population began to increase in 1996 (mainly roach), but had little immediate impact on that years mean summer grazing Cladocera biomass. The following year however, roach abundance had increased further and the large grazing Cladocera biomass was reduced to low levels. Further investigation of the fish data in the 1996 and 1997 years reveals that roach biomass during these two years was similar, but with much greater abundance in 1997. This indicates that individual roach were on average smaller in 1997, which would have lead to heavy predation pressure on the largest, more visible open water cladoceran species, such as *Daphnia spp*. It is therefore suggested that presence of many small individual roach acted to cause the decrease in large grazing cladoceran biomass in 1997. Cladoceran biomass failed to recover during 1998 – 2000, with ZF abundance also remaining relatively high.

The zooplankton monitoring data runs from 1982 to 2000 for Cockshoot Broad (Figure 1.6). The data shows that when the biomass of large grazing Cladocera was at its lowest (1985-88 and 1997-2000), there was increased biomass of both *Bosmina longirostris* and copepods. In terms of grazing pressure upon the phytoplankton, the larger grazing cladoceran species exert most pressure per biomass, especially when compared to *Bosmina longirostris* (Mourelatos & Lacroix 1990). Copepods do not generally contribute significantly to overall algal grazing pressure in a mixed zooplankton community (Wu & Culver 1991). However copepod summer mean biomass was significantly positively related to summer mean chlorophyll a concentration (r = 0.781, N = 19, p = <0.001), indicating that their abundance was less determined by fish predation compared to *Daphnia* spp.

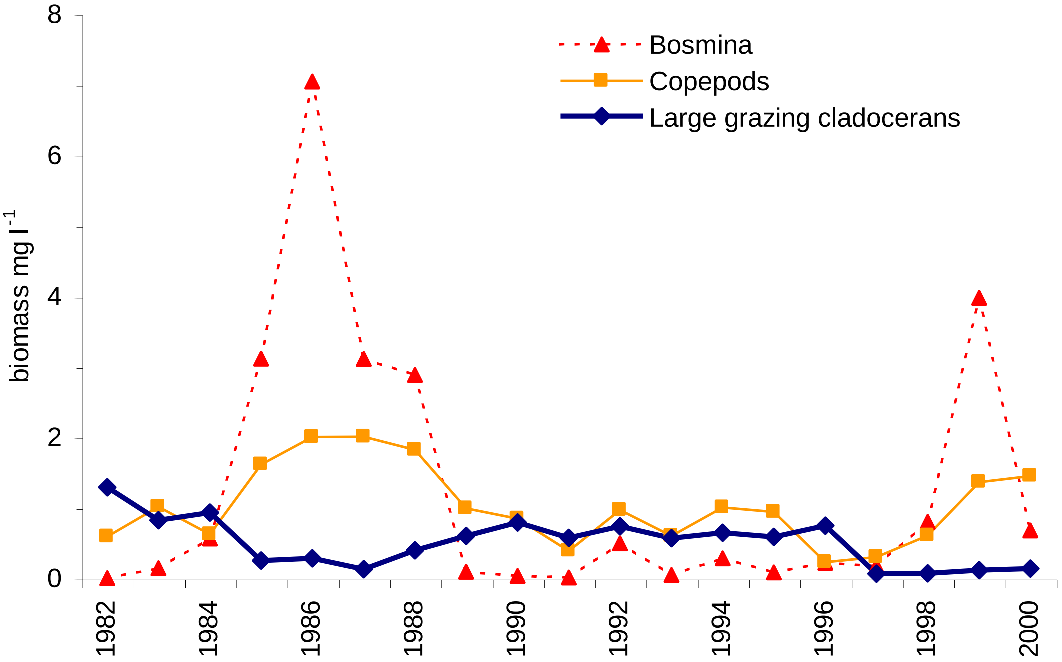

Figure 1.6 Summer mean biomass of the main open water zooplankton groups in Cockshoot Broad

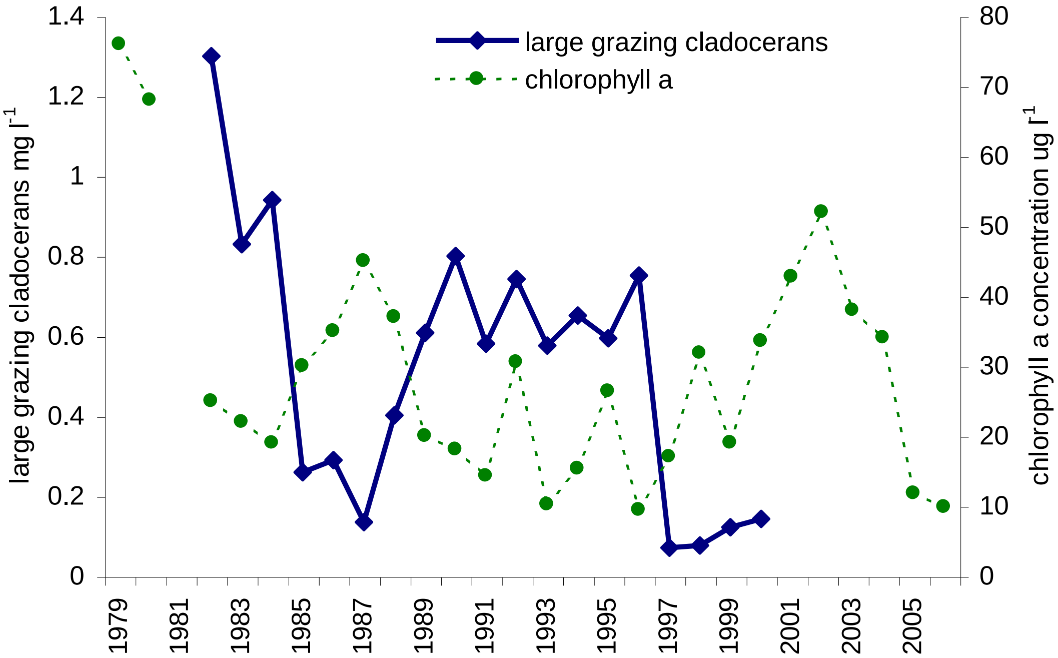

Figure 1.7 Summer mean chlorophyll a concentration and large grazing cladoceran biomass in Cockshoot Broad

1.3 Impact on macrophyte populations

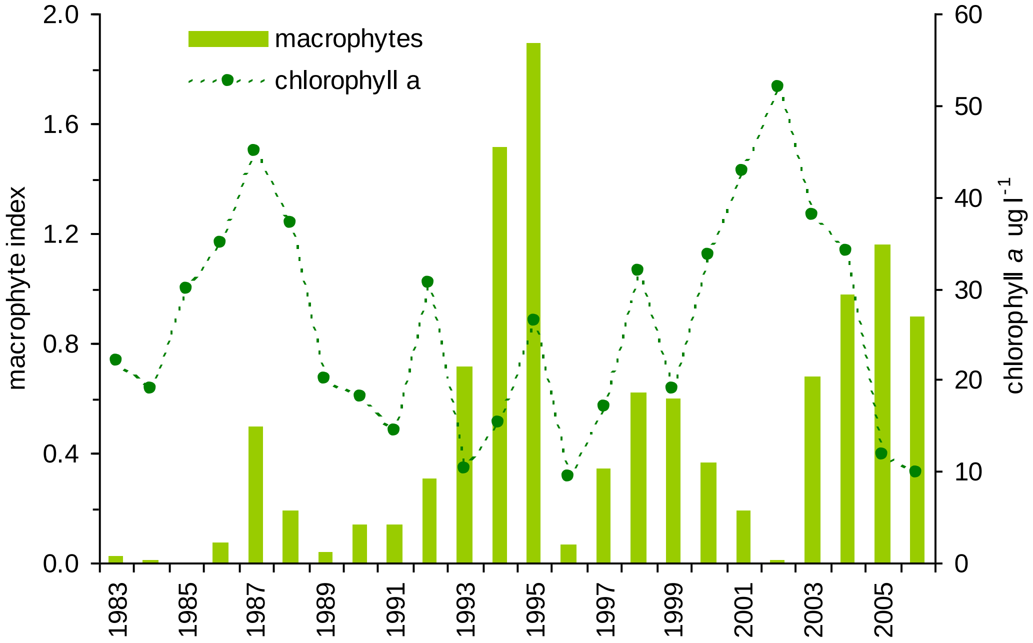

Figure 1.8 Annual macrophyte index scores and summer mean chlorophyll *a* concentration from Cockshoot Broad.

Figure 1.8 shows the annual variation in the macrophyte abundance index (excluding filamentous algae and *Enteromorpha*). Throughout the 1980s macrophyte abundance was very low, but began to increase from 1993, peaking in 1995. In 1995 the range of macrophyte species was also relatively diverse, with rigid hornwort (*Ceratophyllum demersum*) and holly-leaved naiad (*Najas marina*) dominant. Chlorophyll *a* concentrations were lower in the period of maximal plant abundance in the early 1990s. Even when macrophyte growth crashed in 1996, the summer mean chlorophyll a concentration remained low. From 1999 there was a steady increase in chlorophyll a concentration, which was mirrored by reduced macrophyte abundance. Conversely when chlorophyll *a* concentration began to decrease again after 2002, the macrophytes recovered once more.

Changes in the macrophyte community composition have also been recorded over time. In the 1980's macrophyte growth in any one year was highly variable, with often low total abundance and of limited species diversity. In the early 1990's, following biomanipulation, the abundance of fine leaved species such as the Pondweeds *Potamogeton* sp, rigid hornwort, horned pondweed (*Zannichellia palustris*), holly-leaved naiad and Canadian waterweed (*Elodea canadensis*) increased. These species have to varying degrees made up the majority of the submerged flora since then. However, since 2004, there has been a shift to greater abundance of the submerged fine leaved species in general, with early season dominance of *Potamogeton* species followed by summer dominance of holly leaved naiad. This seasonality in species presence has lengthened the period of macrophyte dominance throughout the year.

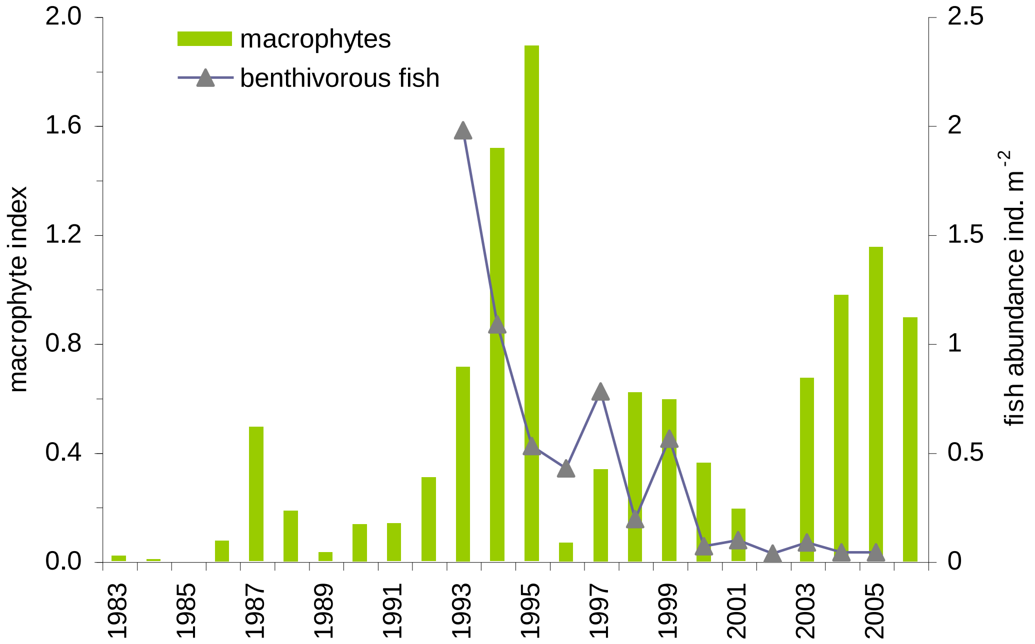

Figure 1.9 Annual macrophyte index scores and benthivorous fish abundance from Cockshoot Broad. (NB no fish data prior to 1993)

Figure 1.9 shows how the initial biomanipulation and subsequent removal efforts have reduced the abundance of benthivorous fish in Cockshoot Broad. The response of macrophytes to this gradual reduction has not been linear. However, Figure 1.9 also shows that in the last six fish survey years that potential disturbance by benthivorous fish upon macrophyte propagules and impact on sediment stability has likely to have been minimal.

1.4 Summary

Prior to sediment removal in winter 1981/82 chlorophyll a concentrations were at their greatest in Cockshoot Broad. The zooplankton community was unfortunately not recorded at this time. The initial success in increased water clarity following isolation and sediment removal, as observed through lower chlorophyll *a* concentrations between 1982 – 84, occurred at the same time as a large biomass of grazing cladocerans was present in the broad (>0.8 mg l^{-1}). The structure and abundance of the fish community was however not recorded following isolation, but predation pressure on the large grazing cladocerans was clearly low. During 1985, the large grazing cladoceran population declined markedly (see also the increase in *B. longirostris* and copepods at this time (Figure 1.6), and as a result, chlorophyll a concentrations subsequently increased, due to the reduced overall algal grazing pressure. It was not until the large grazing cladoceran biomass returned to over 0.6 mg l^{-1} in 1989 the chlorophyll *a* concentration was again driven down again.

The removal of nearly all the fish from Cockshoot in winter 1989 dramatically reduced the predation pressure upon the large grazing cladoceran population, which remained at relatively high density through until 1997. During 1997 the population crashed and remained low through until 2000, at which point no further results from analysed samples are available. Following the 1997 crash in large grazing cladoceran biomass, the chlorophyll *a* concentration began to creep up again, peaking in 2002, with a summer mean value of 52 $\mu g\ l^{-1}$. It is presumed that the relatively high abundance of roach that occurred between 1996 and 2001 drove the reduction in grazing cladoceran biomass in the late 1990s, which then allowed the chlorophyll *a* concentration to increase. The dramatic reduction in chlorophyll *a* in 2005 is matched by the lowest recorded biomass of fish (of all types) in the same year. Low fish biomass, especially that of the zooplanktivorous species is assumed to have resulted in an increased zooplankton grazing pressure on the phytoplankton and is predicted to have caused the low chlorophyll *a* concentrations observed in 2005 and 2006. Analysis of the uncounted zooplankton samples collected in this period would help confirm this presumption.

1.5 Conclusions

Despite the continued annual biomanipulation work, the fish community of Cockshoot still retains the potential to rapidly turn into one dominated by small zooplanktivorous individuals (predominantly roach). The biomanipulation has however been successful in that macrophyte colonisation of the broad has been extensive in most years following the initial fish removal. Some years the macrophytes have failed, e.g. 1996 (a year in which a relatively high ZF abundance was recorded, 0.43 ind. m^{-2}, Figure 1.5) and 2002 (when there was a high summer mean chlorophyll a, 52 µg l^{-1}, Figure 1.7). The resultant reductions in the abundance of the larger bodied *Daphnia* spp. has marked impacts upon the grazing pressure exerted upon the phytoplankton, as measured through chlorophyll *a* concentration. Biomanipulation of zooplanktivorous fish at Cockshoot Broad has been an effective technique, which reverses this situation, with clear water and macrophyte dominance the end result. The role of benthivorous fish is not clear and there is no direct relationship between their abundance and macrophyte abundance, however biomanipulation has been very successful in controlling benthivorous fish abundance.

Recent observations and surveys of Cockshoot Broad have shown that fine-leaved *Potamogeton* spp. are frequent in the early summer, which by the time of the August macrophyte survey, are beginning to senesce (Hoare & Kelly 2006). The majority of Broads Authority macrophyte surveys have been historically carried out in August, as the period of maximal overall macrophyte growth, so invariably miss-out on some early season species. These species are then followed by growth of *Najas marina*, suggesting an increased temporal stability of the macrophyte community and associated clear water state through the growing season.

Completion of the zooplankton monitoring counts from this site would help in establishing how the ecological interactions operate in controlling phytoplankton populations and thus clear water. Further calculation and analysis of the zooplankton grazing rates would reveal valuable information on which species are exerting most grazing pressure and when. However, as broads, and shallow lakes in general become more plant dominated, the open water habitat and plentiful algal food resource favoured by larger-bodied cladocerans, declines. This often results in reduced abundance of these efficient open water grazers, with a community shift to more plant associated zooplankton species. The usefulness of continued open water zooplankton monitoring in these habitats then becomes questionable in terms of quantifying grazing from the large-bodied cladocerans.

2.0 Alderfen Broad

Alderfen Broad, like Cockshoot Broad, has experienced hydrological isolation from a nutrient rich inflow, sediment removal and biomanipulation to restructure the fish community. In 1979 the stream inflow was diverted around the lake through an existing drainage dyke system, after Philips (1977) identified a large nutrient loading arising in the catchment. In 1992/3 the Broads Authority partially suction dredged the broad, thus removing the nutrients bound in the dredged sediment (Holzer *et al.* 1997). In the autumn of 1993 an initial fish removal operation was performed, mainly targeting perch, with a total fish biomass of 24 kg ha^{-1} removed. Fish monitoring results from Alderfen Broad reported by Perrow *et al.* (1994) suggest that prior to the initial removal, the fish population already had a relatively low abundance, with 1991 showing the lowest total CPUE (catch per unit effort) of all the years from 1979 when monitoring began. Subsequent annual removals were carried out in 1994 through to 1997, with the greatest numbers of perch, ruffe, roach and rudd removed in that time (in decreasing order of abundance).

The sources of fish data from Alderfen Broad used in this report were Tomlinson & Perrow (1995) for the 1994 – 2003 PASE results; unpublished ECON data files for the 2004 – 05 PASE results; Stansfield *et al.* (1999) for the 1993 – 94 removal data; and Hindes *et al.* (1999) for the 1995 – 97 removal data.

2.1 Effects of biomanipulation on the fish community

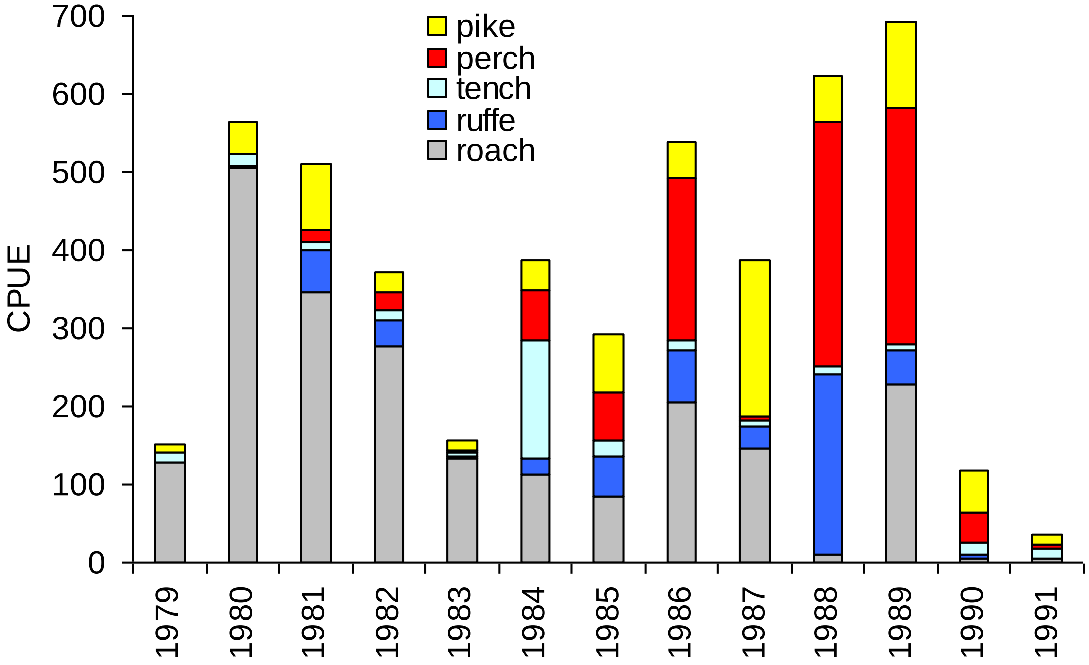

Figure 2.1 Catch per unit effort (CPUE) of the most abundant fish species in the Alderfen Broad littoral zone (not including YOY, after Perrow *et al.* 1994)

Prior to any biomanipulation work at Alderfen Broad, the littoral fish community was sampled annually in October, with results expressed as catch per unit effort (CPUE) (Perrow *et al.* 1994). Underyearling roach, bream, rudd and tench were not estimated and CPUE represents individuals >1 year old. Figure 2.1 summarises the data collected for the most abundant species. Roach were particularly abundant in the early 1980s, with a more mixed community developing by the late 1980s with perch and ruffe becoming increasingly common. In 1990 the total number of all fish species dropped dramatically and remained low in 1991. The total number and biomass of fish removed in 1993 and annually until 1997 is given in Table 9.2.

Results of the most abundant species from the autumn/winter point abundance sampling by electrofishing (PASE) surveys, performed from 1994 – 2005 by ECON, are presented.

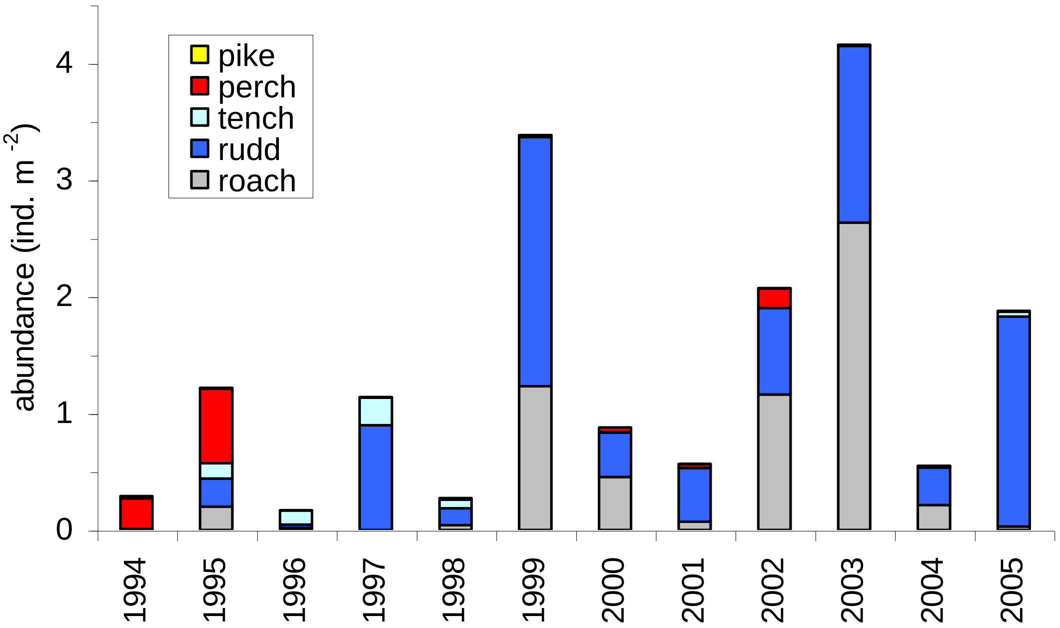

Figure 2.2 Abundance of the dominant fish species in Alderfen Broad (open water & littoral PASE, 1994-2005)

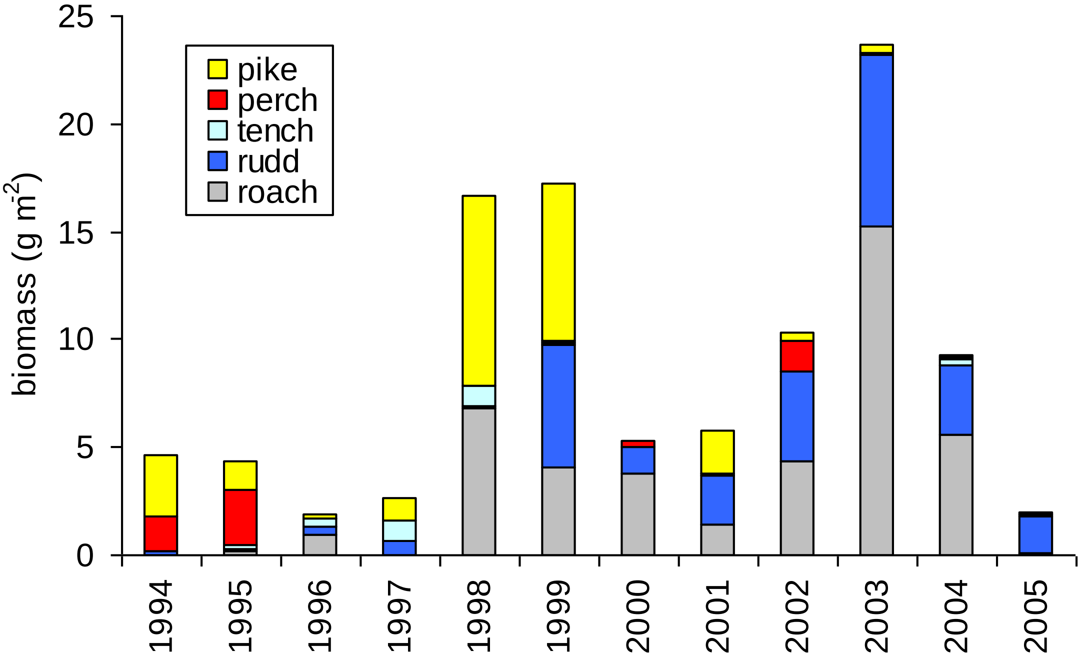

Figure 2.3 Biomass of the dominant fish species in Alderfen Broad (open water & littoral PASE, 1994-2005)

During the early years of the PASE survey, 1994 – 98, the abundance (Figure 2.2) and biomass (Figure 2.3) of was relatively low. This was in part due to the fish removal work, but mainly due to poor recruitment/high mortality in the early 1990s, following the drought in 1989/1990. This was the period when the majority of the bream population was lost (M. Perrow, pers. comm.). The biomanipulation of Alderfen was a partial experiment, rather than a long-term fish reduction operation, but none-the-less, valuable insights and results were gained. The initial removal operation removed a total of only 24 kg ha^{-1}, which was a relatively low biomass compared to other sites biomanipulated in the Broads (Stansfield *et al.* 1997). The most obvious change after the annual removals ceased was the increase in both total abundance and biomass in 1999, particularly of roach and rudd. The 1998 results appear to have remained low, presumably as populations were still low following removal, but by 1999 recruitment of roach and rudd was clearly successful. Only two captured individuals made up the pike biomass in 1999, highlighting the large variability in this species data between years. Perch abundance was high relative to other species in the first two years of the PASE survey, but the removal effort reduced their population to very low levels by 1996. Rudd presence has been a common feature in Alderfen Broad following the initial fish removals, with abundance increasing rapidly in 1999 after the biomanipulation effort ceased. Prior to the removal work however, rudd of >1 year old were only recorded up until 1983 (Figure 2.1), suggesting that this species has particularly benefited from the biomanipulation work. Reduction of competition with other fish species or environmental conditions, i.e. macrophyte growth, may have become better suited for rudd following biomanipulation. Roach abundance and biomass also increased after the regular removals stopped in 1997. Tench seem to have declined in both abundance and biomass after 1998, with only sporadic detection of this species in more recent years. The perch population has also been relatively low during the PASE survey years, with a small peak of abundance in 2000 following the addition of nearly 11,000 individuals as part of an experiment involving artificial macrophytes as refugia for their macroinvertebrate prey. The individuals introduced to the broad appear to have suffered heavy mortality and did not persist in the long term.

2.2 Water clarity and zooplankton grazing pressure

Unlike Cockshoot Broad, neither the annual or summer mean chlorophyll *a* concentration values in Alderfen Broad were directly correlated to annual or summer mean TP concentrations. The trajectory of the relationship between TP and chlorophyll *a* in Alderfen can be divided up into three different time periods (Figure 2.4).

The first is for water quality data gathered from 1990 – 93 (dark blue points), before the initial fish removal. 1992 and 1993 experienced very high SRP concentrations, with summer mean values of 1.34 and 1.04 mg l^{-1} respectively. These two years with large SRP release from the sediments clearly affected the chlorophyll *a* /TP ratio, as TP concentrations were proportionally greater. Zooplankton grazing may have contributed to the low chlorophyll *a*/TP ratio in these years, through grazing of algal cells, but there is no zooplankton data for these years, so this additional factor remains unquantified.

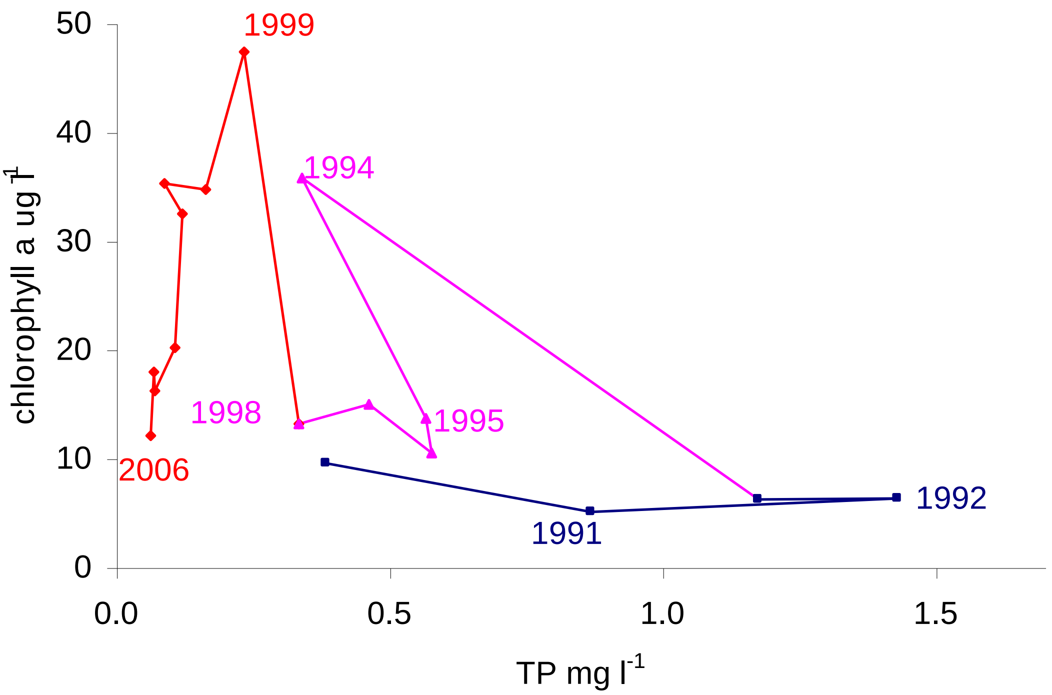

Figure 2.4 Trajectory of summer mean TP and chlorophyll *a* in Alderfen Broad (1990 – 2006).

The second period is during the fish removal years 1993 – 1997, with 1998 included as the fish community remained at a relatively low abundance for this year, before certain species naturally recovered. All years apart from 1994 in this period had relatively low chlorophyll *a* concentrations, suggesting effective control by zooplankton of the algal production. The zooplankton data only overlaps with the PASE fish surveys for three years (Figure 2.5), so there are too few data to analyse the relationship between fish and zooplankton in a robust way. The third period (1999 onwards) is that after fish removals ceased and roach and rudd abundances subsequently increased. During this period there has been a gradual decline in both the summer mean concentrations of TP and chlorophyll *a*, despite what would appear to be a fish population capable of severely limiting the grazing cladoceran population.

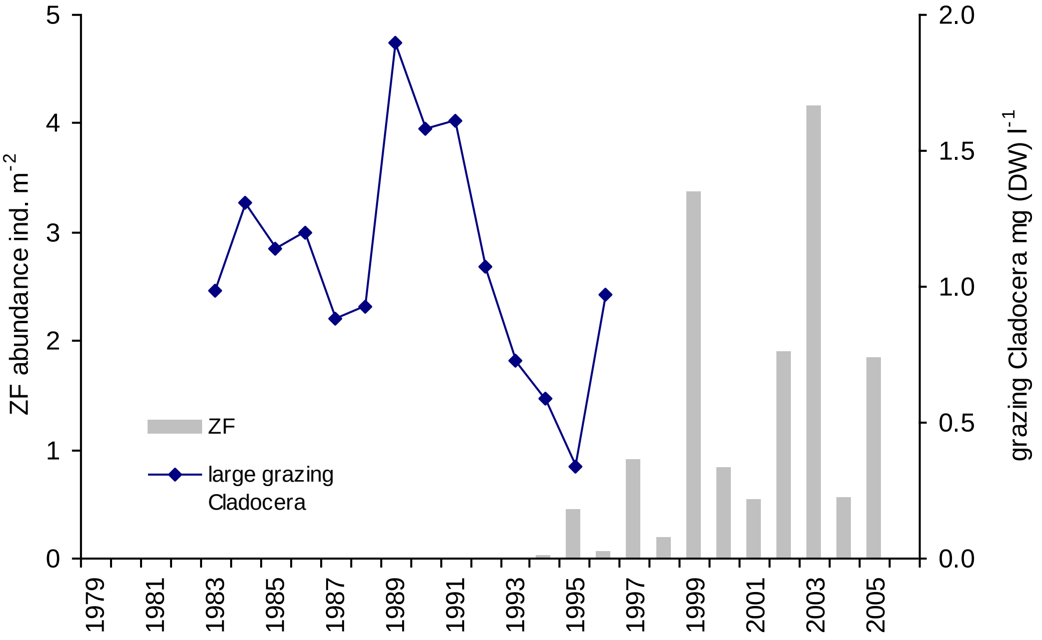

Figure 2.5 Zooplanktivorous fish (ZF) abundance (open water and littoral PASE) and summer mean biomass of grazing Cladocera in Alderfen Broad.

The CPUE fish data from Alderfen (Figure 2.6), which pre-dates the PASE surveys, overlaps with the zooplankton monitoring to a greater extent. Through the mid 1980s large grazing Cladocera biomass was reasonably stable (~1 mg l^{-1}), however this summer mean value increased to >1.5 mg l^{-1} from 1989 onwards. This period in the ZF CPUE data suggests a decrease in ZF abundance, especially in 1990 and 1991, which may have released the large Cladocerans from ZF predation pressure.

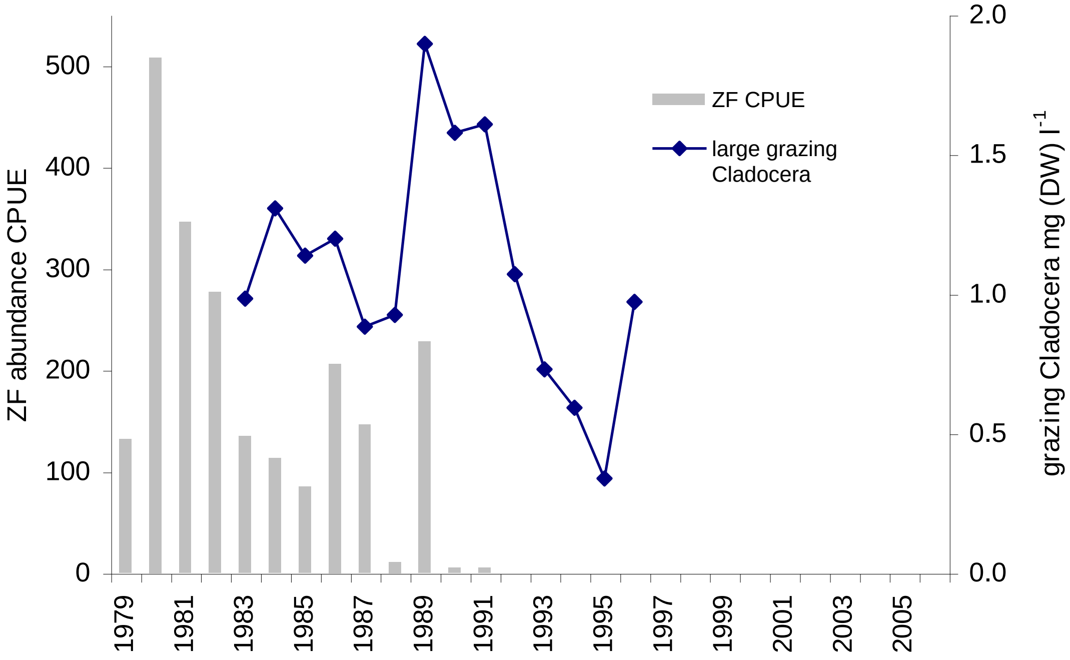

Figure 2.6 Zooplanktivorous fish (ZF) CPUE and summer mean biomass of large grazing Cladocera in Alderfen Broad.

Of the zooplankton samples that have been analysed (1983 – 96, Figure 2.7), the large grazing cladocerans were continually present, with summer mean biomass ranging from 0.3 – 1.9 mg l^{-1}. The biomass of *B. longirostris* was more variable, ranging from <0.1 mg l^{-1} for the summers of 1989 – 93, to a peak of 8.2 mg l^{-1} in 1994.

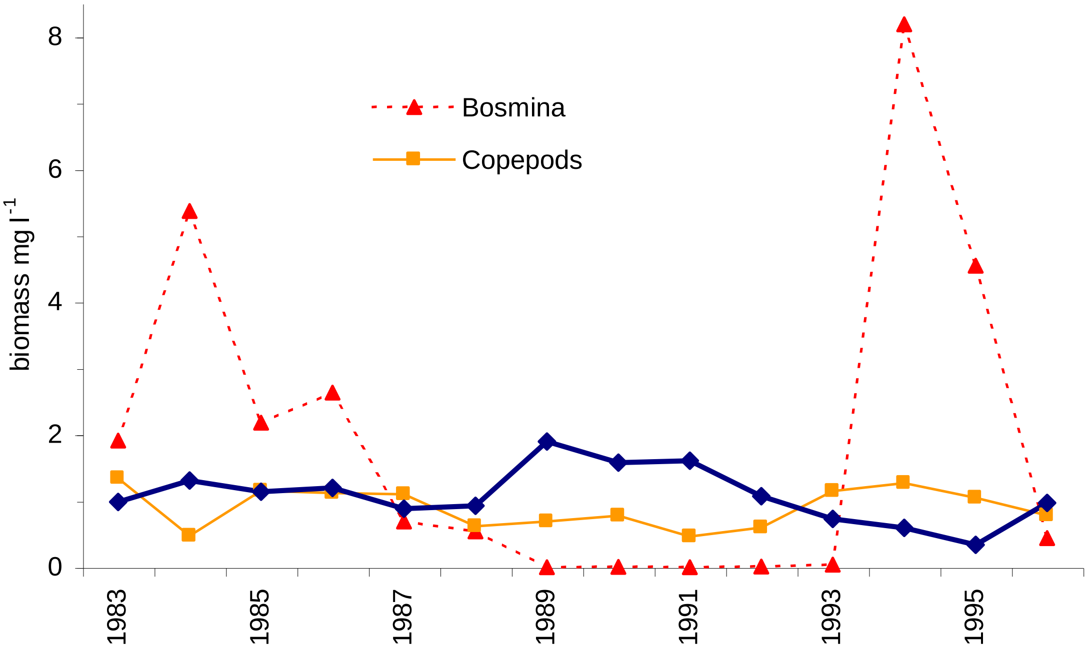

Figure 2.7 Summer mean biomass of the main open water zooplankton groups in Alderfen Broad.

No direct relationships were evident between the summer mean large grazing cladoceran biomass and water quality variables. However significant positive correlations existed between summer mean *B. longirostris* (Figure 2.8) and copepod nauplii biomass and chlorophyll *a* concentration ($r = 0.94$, $N = 7$, $p = <0.01$ and $r = 0.95$, $N = 7$, $p = <0.001$ respectively). This suggests that high *B. longirostris* abundance only occurs in years with high algal productivity, which is generally when large grazing cladoceran biomass is lower (though this is not a significant relationship within the dataset analysed).

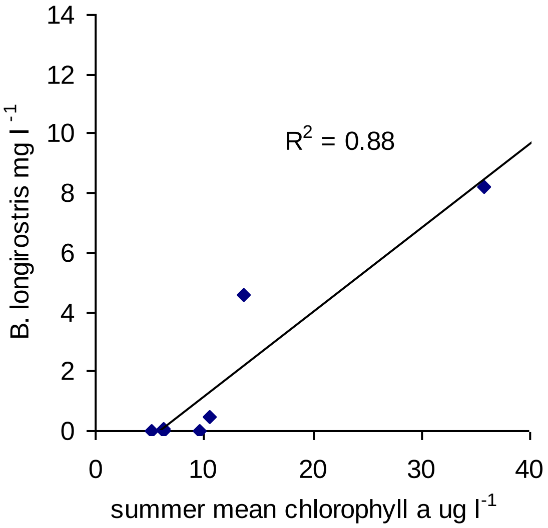

Figure 2.8 Relationship between summer mean (1990 –96) *Bosmina longirostris* and chlorophyll *a* in Alderfen Broad.

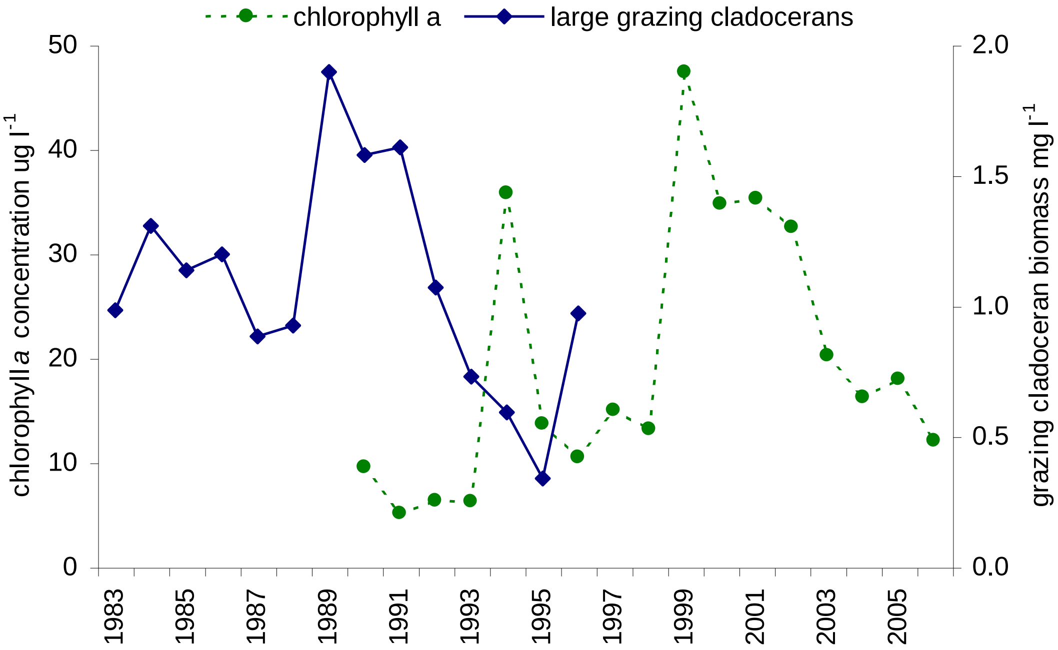

Figure 2.9 Summer mean chlorophyll *a* concentration and large grazing cladoceran biomass in Cockshoot Broad

In Figure 2.9, the 1994 peak in chlorophyll *a* can be seen to have occurred at the same time as the biomass of large grazing cladocerans declined to <0.5 mg l^{-1}. Large grazing cladocerans reached their lowest biomass in summer 1995, but a relatively high *B. longirostris* biomass in 1994 and 1995 (Figure 2.7) would have added to the grazing pressure exerted upon the algae. The reason for the decline in large grazing cladocerans from 1992 onwards is not directly obvious from the fish data, as the CPUE data (Figure 2.6) and the initial PASE surveys (Figures 2.5) suggest that zooplanktivores were of relatively low abundance at that time.

2.3 Impact on macrophyte populations

Alderfen Broad has maintained a relatively vigorous growth of rigid hornwort (*Ceratophyllum demersum*) throughout the macrophyte monitoring period and has usually been the dominant species. There has however been a cycle of failure of this species, lasting for a season or two, with an apparent return period of around seven years (Figure 2.10). The failure of macrophytes in 1999 and 2000 was accompanied by relatively high chlorophyll *a* concentrations. Filamentous algae, e.g. *Cladophora* spp. and *Enteromorpha* spp., were also present within Alderfen Broad during these years, and have been regularly present since monitoring started. The other year with a strong peak in summer mean chlorophyll *a* concentration was during 1994 when an unusually high concentration was recorded in June of 143 µg l^{-1}, matched with a Secchi depth reading of only 35 cm. The remainder of samples during mid-summer in 1994 were around 30 µg l^{-1}, which seemed not to have negatively influenced macrophyte growth as the plant index was >0.5. During the previous year, macrophyte growth was very poor, but chlorophyll *a* concentrations were also relatively low, with a summer mean of only 6 µg l^{-1}. From this analysis it is clear that turbidity caused by and direct competition for nutrients with phytoplankton are not directly related to the success of macrophyte growth in Alderfen. At its deepest point, Alderfen is roughly 1.2 m deep, so plants do not have to grow far up into the water column to reach sufficient light levels. Establishment of plant growth early in the year can mean plants can persist through the remainder of the season, despite later algal blooms, as appeared to have happened in 1994. Here the composition of the algal community is important, as blue-green algae can often persist alongside plants. However over the last six years of macrophyte monitoring, successive increases in the macrophyte index have been matched by a steady decline in the summer mean chlorophyll *a* concentration. As shown in Figure 2.4, decreasing summer mean TP concentrations have also mirrored this decline. The macrophyte community has had several years of continued stability, in the form of plant growth to the surface across the entire broad during the summer months.

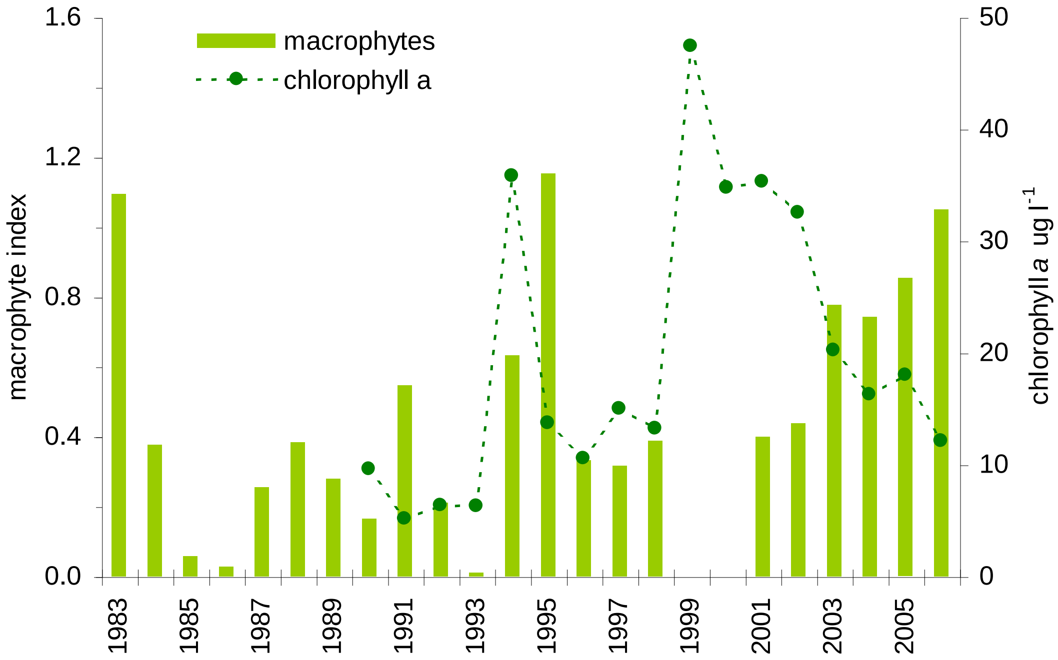

Figure 2.10 Annual macrophyte index scores and summer mean chlorophyll *a* concentration from Alderfen Broad. (No plants found in 1999 & 2000)

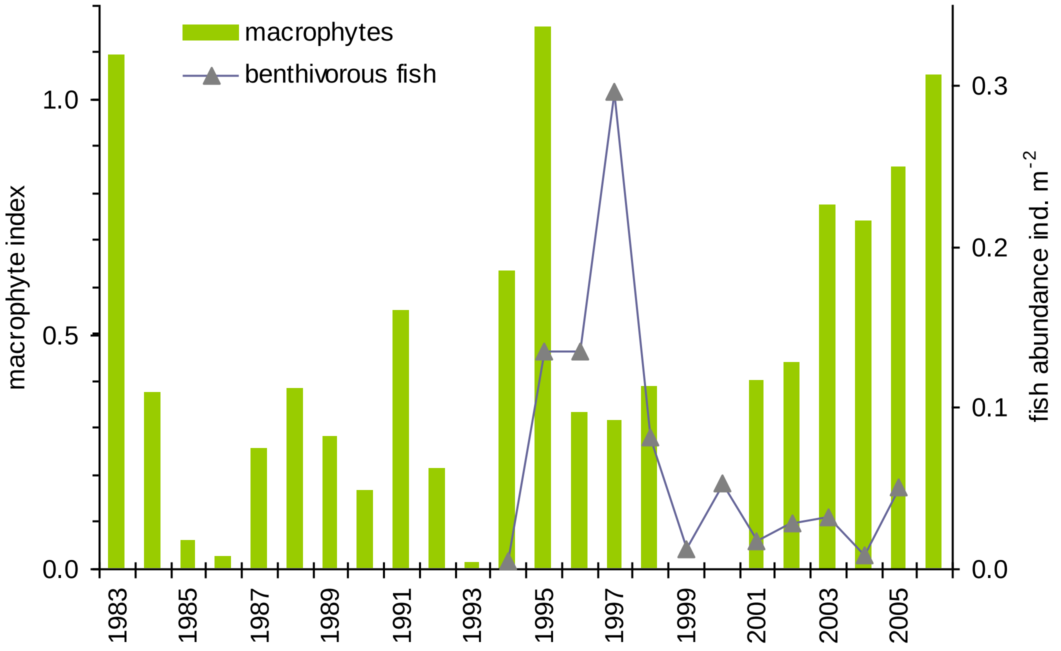

Figure 2.11 Annual macrophyte index scores and benthivorous fish biomass (from PASE survey) from Alderfen Broad.

Benthivorous fish numbers have been reduced to such an extent in Alderfen Broad that these species, principally bream, have had very little influence on lake functioning since the PASE surveys began (Figure 2.11). A reduced presence of benthivorous fish can be seen as part of the process towards recovery of macrophytes in shallow lakes. Clearly in Alderfen a sustained recovery has occurred once these conditions have been generated (2001 onwards). The macrophyte survey method may also be rather insensitive to years with a very large biomass of plants, thus reducing the amount of relative change between “good” and “bad” plant growth years. The extent of plant growth in the more recent years (2003-2006) may therefore be under represented in Figure 2.11.

2.4 Summary

The initial fish removal event in Alderfen Broad was a response to a large abundance of young perch, which are zooplanktivorous when they have a small body size. The large benthivorous species population was also targeted and was successfully reduced. However, the fish removal work in Alderfen was more of an event than a full biomanipulation exercise. The removal years only lasted from 1993 to 1995, with targeted roach and rudd removal in 2000. All the fish removals were carried out in a broad that had already experienced a large fish kill event, which had significantly reduced the overall biomass and altered the community composition. The contribution of rudd to the overall fish community in Alderfen Broad has successfully increased, this is a species previously identified as a desired member of a restored fish community. The presence of relatively high abundances (>1 ind. m-*2*) of this species in some years has not negatively influenced macrophyte growth or water clarity. The beneficial influence of fish removals was obvious on the overall community abundance and biomass, as in the second season following the cessation of these removals; roach and rudd recruitment was highly successful.

It would appear that during the summer the majority of TP becomes locked up in the macrophyte growth, which in recent years has been very abundant. The potential for phosphorus to be released from the surface sediments has been minimised by two factors in Alderfen Broad. The first is due to reduction in sediment disturbance by benthivorous fish, which feed in the surface sediments, mixing the top layers with the overlying water. The second is the fact that the dominant macrophyte species, rigid hornwort, is not directly rooted into the sediment, this also reduces exchange of nutrients between the two compartments.

Establishing the role that the fish population has played in structuring and controlling the abundance of the zooplankton community has not been possible over the decadal time scale at Alderfen, as the years of simultaneous reportable data do not exist. In turn, explanation of the variation in chlorophyll *a* concentration at this site is limited, but what data there is suggests that grazing zooplankton have a strong capacity to reduce algal populations and increased water clarity. The near continuous presence of macrophytes is also shown to be an important factor in maintaining clear water conditions.

2.5 Conclusions

Through natural mechanisms and active management the fish community in Alderfen Broad now has the features of a community suited to macrophyte dominated conditions. Following suction dredging and the fish removals water clarity has been good, with periods of continual macrophyte dominance. This initially appeared to be cyclical in nature with frequent crashes in plant abundance. In recent years phosphorus release events that characterised the early years following the restoration process have ceased. Over the last 6-7 years macrophyte dominance has been stable, with clear water throughout the summer months. Water quality in terms of TP and chlorophyll a has shown a very positive trend over this period, suggesting the broad has entered a new phase of ecological stability. The current presence of holly leaved naiad within the broad also suggests an increase in the stability of the macrophyte dominated state, as diversity of plant species offers greater plant cover over the season and less susceptibility to crashes in abundance.

3.0 Pound End

In-lake restoration work at Pound End began in 1990 with suction dredging removing the nutrient-rich surface sediment from Pound End and the neighbouring Hoveton Little Broad. Pound End is an embayment of Hoveton Little Broad, which enabled effective isolation through installation of a barrier across the narrow opening between the two areas. The first fish removals were in spring 1990. Initially, nets were used to isolate Pound End from Hoveton Little Broad, until winter 1991/92, when a steel structure was put in place, with 1 mm mesh screens incorporated in the piles to allow water exchange. This structure was however also found to allow small fish to enter the broad during high water periods. The last fish removals were performed in 1999, with the barrier finally removed in 2002. In addition to isolation from the fish community in Hoveton Little Broad, several “carousel” type bird exclosures were constructed in Pound End to encourage macrophyte growth.

The sources of fish data from Pound End used in this chapter were Hindes *et al.* 1999 for the 1994 – 98 PASE results; unpublished ECON data files for the 1999 - 2000 PASE results; Stansfield *et al.* (1997a) for the 1994 removal data; and Hindes *et al.* (1999) for the 1995 – 98 removal data.

3.1 Effects of biomanipulation on the fish community

The initial removal in 1990 yielded many 0+ roach captured through seine netting. Bream were also targeted through a spawning disruption operation. In winter 1990/91 large concentrations of young fish were removed through netting and electro-fishing. In spring 1991 thousands of 0+ roach, 233 roach to 0.5 kg and 344 bream to 2.5 kg were removed. Further large bream were removed later that year, followed by 528 individuals in 1992. By 1993 only 4 individual bream were captured, indicating a successful reduction of the adult bream population to just a few individuals (Holzer *et al.* 1997). The efficacy of the steel piling barrier in preventing fish movements lasted for several years after 1993, as far fewer fish were removed from 1994 – 96.

The results from the annual autumn PASE surveys conducted by ECON (Figure 3.1) show that a relatively low roach abundance was maintained from 1994 – 98 with a subsequent increase in abundance experienced in 1999 and 2000. The PASE surveys in Pound End were conducted in the autumn, so the last fish removal in spring 1999 was clearly followed by either a very successful recruitment of roach or a large immigration around the barrier, the latter being the more likely cause. The cessation of fish removals enabled the increased roach abundance to remain around the 2 ind. m^{-2} level in the 2000 survey. Perch were the only other species that contributed significantly to the overall abundance in Pound End, with maximum abundance reached in 1995 of 0.4 ind. m^{-2}. Bream were also present from 1995 - 2000, though at very low abundances.

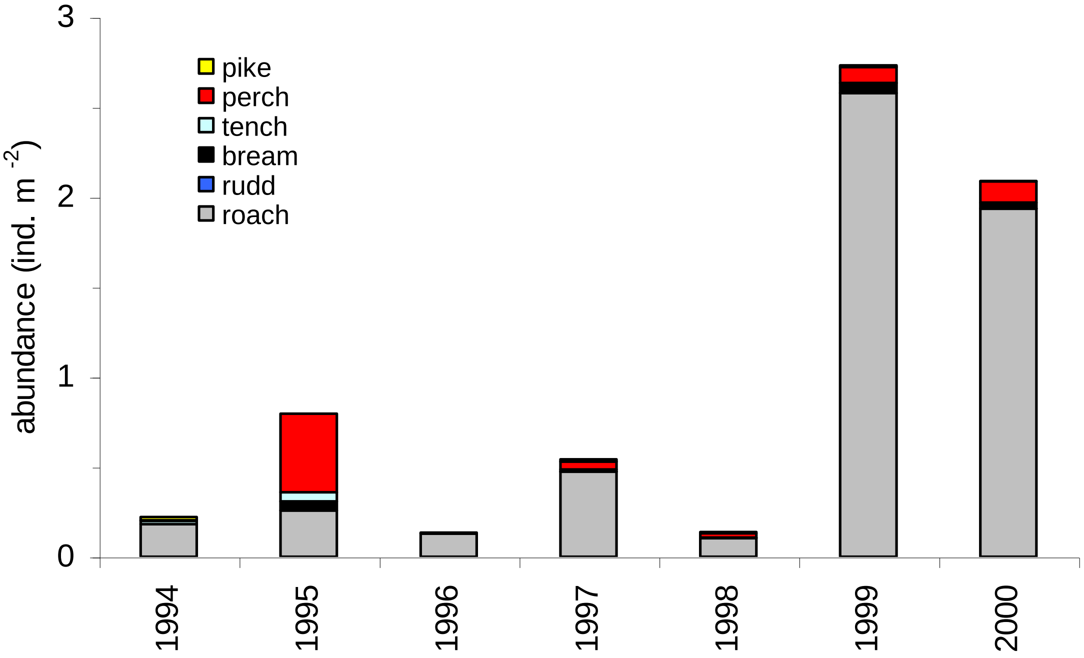

Figure 3.1 Abundance of the dominant fish species in Pound End (open water & littoral, 1994-1998)

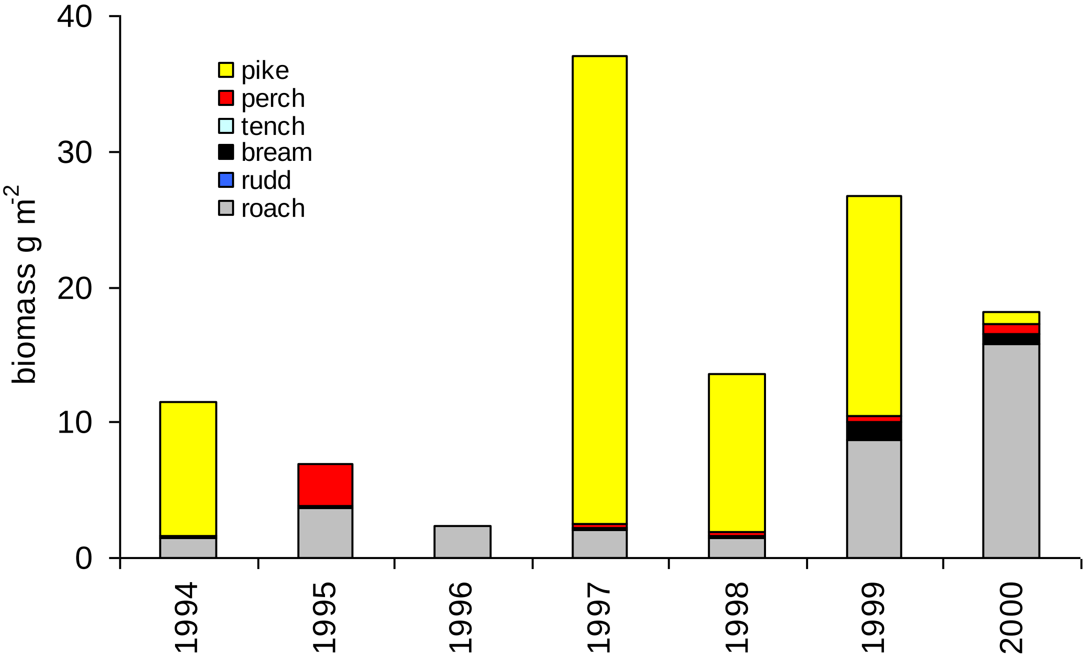

Figure 3.2 Biomass of the dominant fish species in Pound End (open water & littoral, 1994-1998)

As in Figure 3.1, the biomass of roach (Figure 3.2) also displayed a marked increase in 1999 and 2000, increasing to greater than 8 g m^{-2}. Pike biomass was the greatest of any fish species, though this was usually made up of a just a few individuals in any one sampling occasion. Perch biomass was greatest in 1995 (3 g m^{-2}) with a small peak in 2000. Bream biomass also increased slightly in 1999 and 2000 compared to previous years, increasing to more than 0.8 g m^{-2}.

3.2 Water clarity and zooplankton grazing pressure

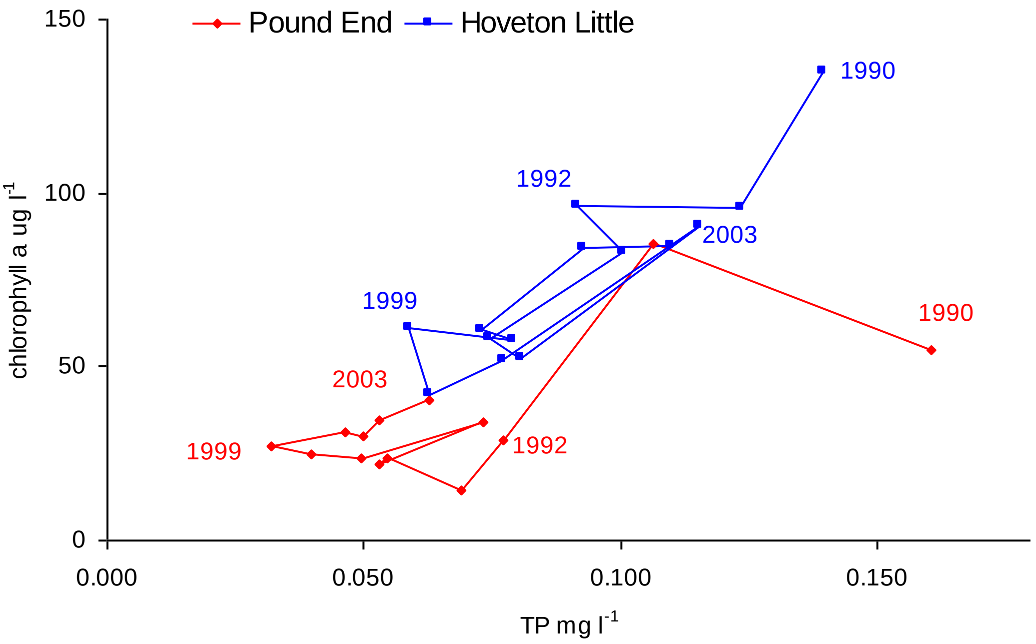

Figure 3.3 Trajectory of summer mean (1990 – 2003) TP and chlorophyll *a* in Pound End (red) and Hoveton Little Broad (blue)

During the first two years of EA water quality monitoring data in Pound End (1990 – 91), gathered following the mud pumping and biomanipulation work, the TP and chlorophyll *a* concentrations were relatively high compared to subsequent years (Figure 3.3). There then ensued a period from 1992 – 2003 of relative stability in the relationship between TP and chlorophyll *a* in Pound End. Hoveton Little Broad also experienced a similar decline in TP and chlorophyll *a* immediately following mud pumping, but failed to reach the lower levels of both measures attained in Pound End. The Pound End data generally follows a slope of lower chlorophyll *a* to TP (chlorophyll *a* /TP ratio) than Hoveton Little, indicating greater control of algal production in Pound End. This could be through control of the algal production by zooplankton grazing or direct competition for nutrients by the filamentous algae that were present. This suggests that the extra restoration effort in terms of biomanipulation helped achieve the goals of lower available nutrients and reduced turbidity from algae.

A PASE survey comparing the fish community between Hoveton Little Broad and Pound End was conducted in 1994 (Stansfield *et al.* 1997b). This study revealed that the total fish abundance was similar between the two sites, but in the autumn, roach was dominant in Hoveton Little Broad (90%), whereas perch was dominant in Pound End (66%). In Pound End seven years of simultaneous fish and zooplankton community monitoring have occurred from 1994 – 2000 (Figure 3.4). In the first four years following sediment and fish removal from Pound End (1990 – 94), no fish community data was available from PASE surveys, but removal data gives an indication of the relative abundances of the dominant fish species. From 1990 – 94 the summer mean biomass of large grazing Cladocera was relatively high, except during 1991, a year when immigration of thousands of small roach was observed (Holzer *et al.* 1997). This increased abundance of zooplanktivorous roach had dramatic negative impacts upon the large grazing Cladocera. Successful design and construction of a more robust steel barrier and annual fish removals in 1992 and 1993 allowed the large grazing Cladocera biomass to recover and increase to above 1 mg l^{-1}. However through the mid 1990s Cladocera biomass gradually declined whilst the ZF abundance remained relatively stable. The increase in ZF abundance in 1999 and 2000 had little impact on Cladocera biomass as it remained around the 0.2 mg l^{-1} level. This pattern suggests that the large grazing Cladocera population was already limited in the late 1990s, with the low macrophyte cover providing little refuge from the ZF population present.

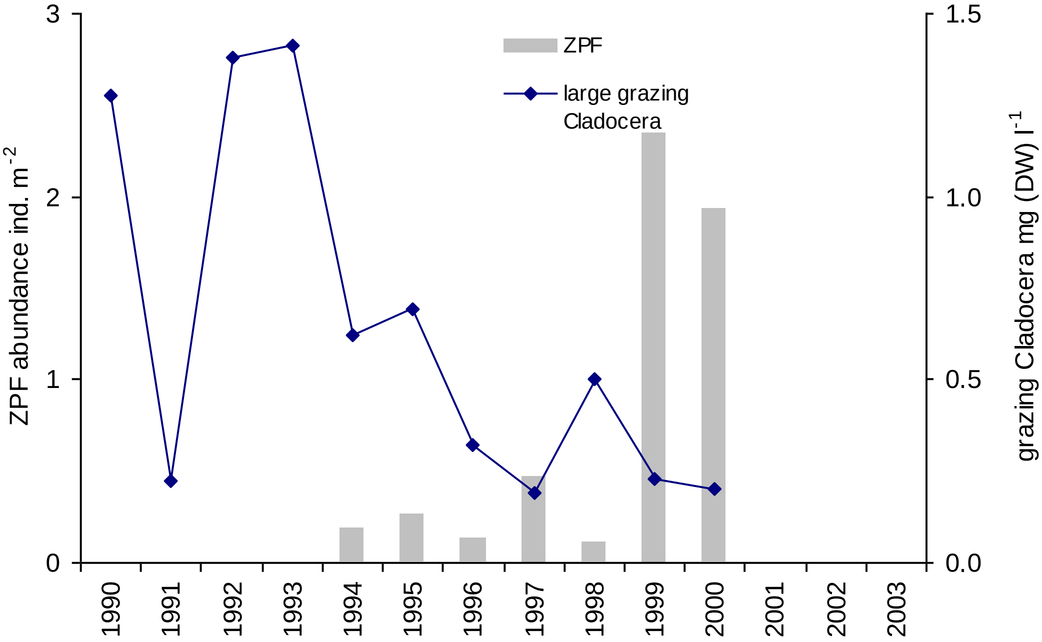

Figure 3.4 Zooplanktivorous fish (ZF) abundance (1994 – 2000) and summer mean biomass of large grazing Cladocera (1990 – 2000) in Pound End.

Figure 3.5a shows that there was a significant positive correlation between grazing Cladocera biomass and water clarity (r = 0.685, *N* = 11, p = <0.05). Such direct relationships were not observed in the data from Alderfen and Cockshoot Broads, probably because the clarity in Pound End was not usually greater than the maximum water depth. In the very clear water sites, the datasets were truncated by the methodological limitation of Secchi depth, which in turn reduced the statistical power, and validity of such relationships. In Pound End it is evident that that fully clear water (i.e. Secchi depth greater than total water depth) was never achieved for prolonged periods, with the summers of 1992 – 94 having the greatest mean Secchi depths (>1.1 m), with a few sampling occasions in those years having clear water to the bottom. The effect of variation of grazing Cladocera biomass upon water quality is demonstrated in Figure 3.5b, as a significant negative association with the chlorophyll *a* /TP ratio exists in the data (r = -0.781, *N* = 11, p = <0.01). This relationship means that at greater Cladocera biomasses, the amount of chlorophyll *a* (algal production) is depressed relative to the total amount of phosphorus available for such growth, i.e. Cladocera grazing effectively controlled algal production.

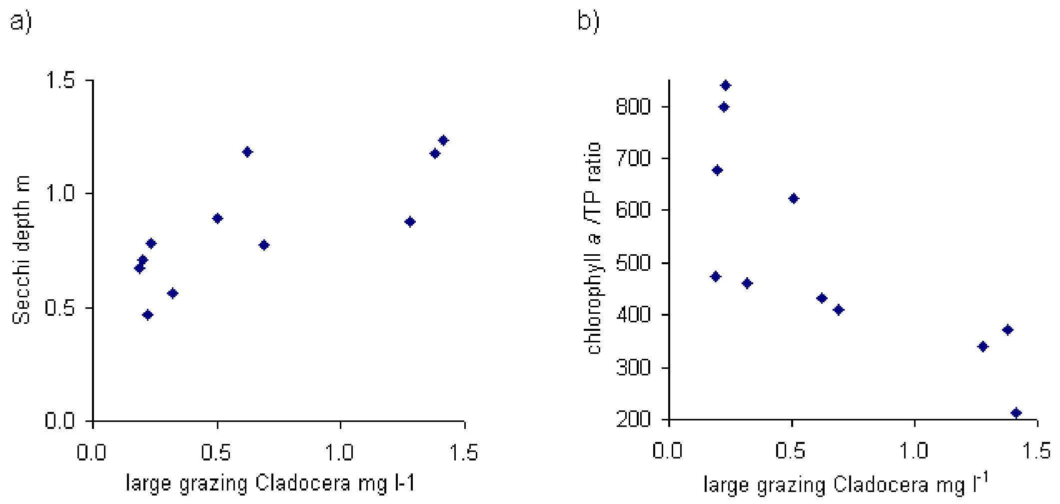

Figure 3.5 Relationship between summer mean large grazing Cladocera and a) summer mean Secchi depth and b) chlorophyll *a* /TP ratio.

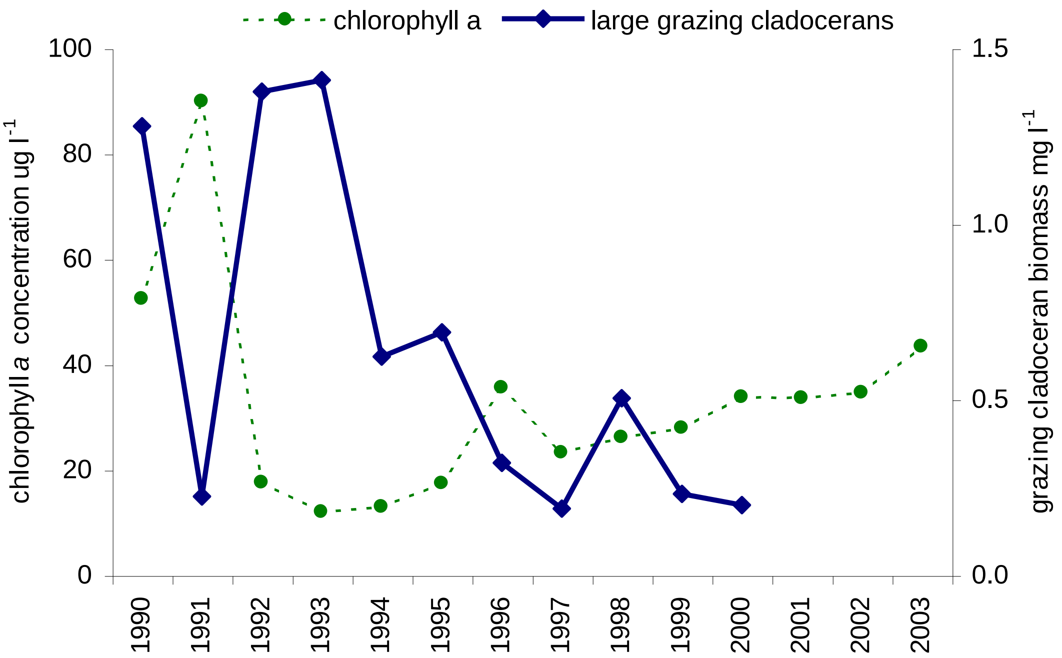

Figure 3.6 Summer mean chlorophyll *a* concentration and biomass of large grazing Cladocera in Pound End.

The direct nature of the relationship between large grazing Cladocera biomass and chlorophyll *a* concentration in Pound End is shown in Figure 3.6. In 1991 the large influx of small roach that entered the then isolated Pound End, appear to have reduced Cladoceran biomass with a concomitant increase in chlorophyll *a* concentration. Subsequent years with relatively high grazing Cladocera (1992 – 95) had low chlorophyll *a* values, which have steadily increased as Cladocera biomass has declined.

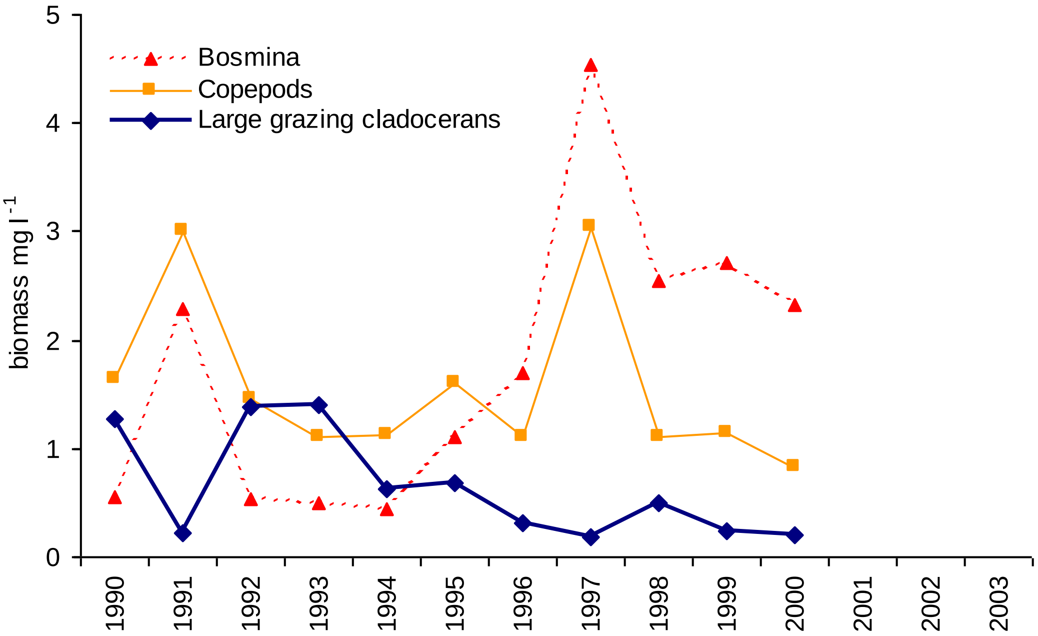

Figure 3.7 Summer mean biomass of the main open water zooplankton groups in Pound End.

Figure 3.7 shows that in the early years of fish removals, large grazing Cladocera were relatively abundant (excluding 1991). As in Cockshoot and Alderfen Broads, during years with the lowest grazing Cladocera biomass, the relative proportion of Bosmina and copepods increased, resulting in a significant negative relationship between the large grazing Cladocera and Bosmina (r = -0.767, *N* = 11, p = <0.1).

3.3 Impact on macrophyte populations

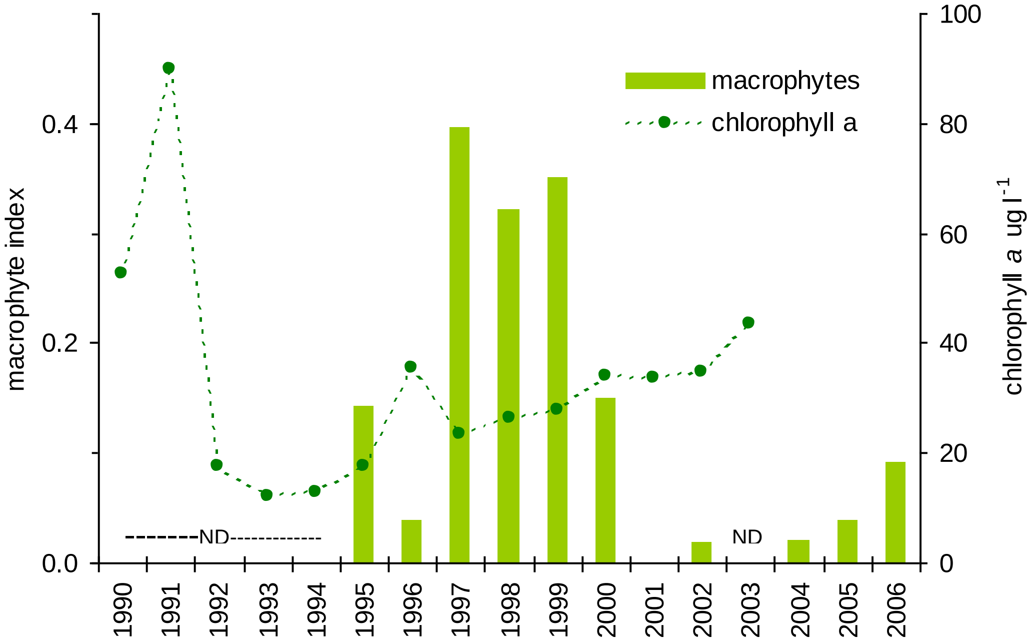

Figure 3.8 Annual macrophyte index scores (not including filamentous algae) and summer mean chlorophyll *a* concentration from Pound End.

The rake-based transect surveys for macrophytes have been conducted in Pound End from 1995 to 2006. None of the annual macrophyte index scores have ever exceeded 0.4, with a mean score of 0.14 (1995 – 2006 surveys). Values below 0.5 are generally considered to represent sites with low macrophyte abundance, with those sites above 0.5 considered to be abundant in macrophytes. Cockshoot and Alderfen have had mean scores of 0.65 and 0.54 over the same period. As this macrophyte index represents the sum of all species scores, years with greater diversity can have inflated index scores, so the number of species needs to be considered to some extent. Pound End however has remained a low diversity site for macrophyte species, never exceeding four species in any one year. The peak years for macrophytes in Pound End were 1997 – 99, with *Ceratophyllum demersum* and *Najas marina* regularly recorded, though distributed in discrete patches rather than an extensive coverage of the lakebed. Macrophyte index scores were not significantly associated with any other measured water quality or ecological variable in Pound End. The presence of the UK BAP species *Najas marina* is significant in conservation terms, as the only other sites in the River Bure valley were this species has been regularly recorded over the last ten years are Cockshoot and Upton Broads. Sporadic individual specimens have also been recorded from Hoveton Little, Hoveton Great, Ranworth, Decoy and Wroxham.

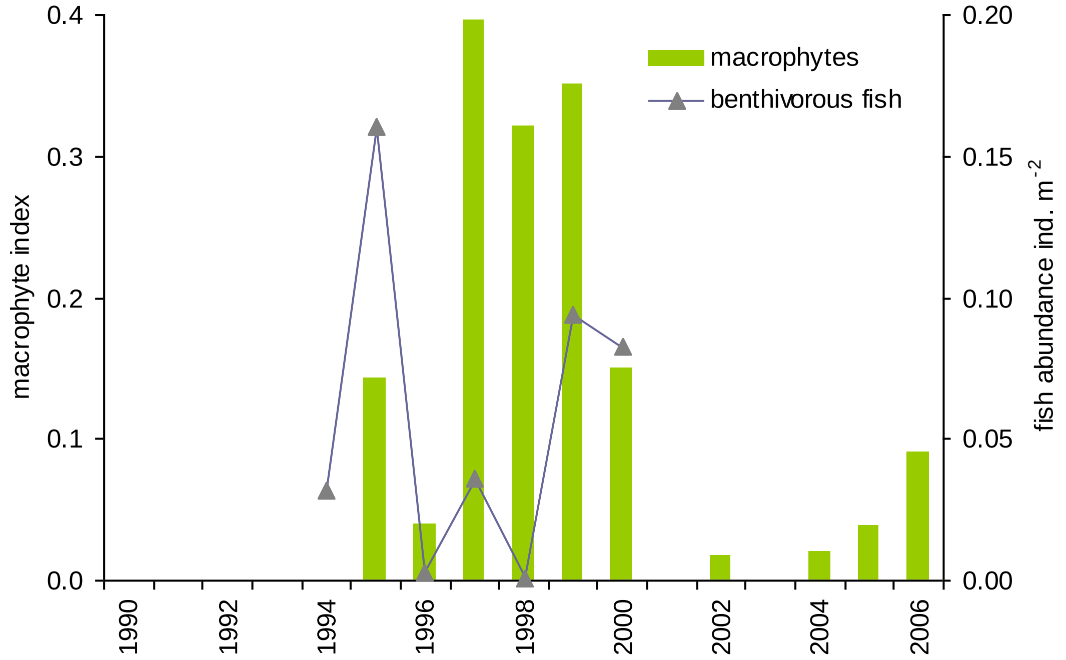

Figure 3.9 Annual macrophyte index scores and benthivorous fish abundance (PASE, 1994-2000) from Pound End.

Figure 3.9 shows the abundance benthivorous fish following the removal of the vast majority of the population. As in Alderfen, the removal of benthivorous fish can be seen a vital stage in the restoration process, which in this case delivered several years of increased macrophyte growth that, would have not otherwise occurred.

3.4 Summary

The fish removal in Pound End has been successful in delivering a reduced abundance of zooplanktivores, with roach abundance often below 0.2 ind. m^{-2}. The value of such removal efforts is highlighted by the 1999 and 2000 PASE data, which shows how quickly high roach abundances can return. The physical difficulties of isolating a waterbody subject to tidal variation in water height have also been demonstrated at this site. Numerous over toppings and breaches during high water periods have allowed fish ingress and led to the eventual abandoning of Pound End as an actively biomanipulated site.

In contrast to the isolated Cockshoot and Alderfen, Pound End has continuously shared water with the relatively enriched River Bure. The steel fish barrier would have reduced the flushing rate of Pound End, compared to when the two basins had been in open hydrological connection, but the mesh grills ensured some mixing. The two basins however became ecologically separated, as demonstrated by the parallel monitoring in Hoveton Little Broad. The main broad showed the higher average TP and chlorophyll *a* concentrations as it still had a direct riverine connection (Figure 3.3). During the biomanipulation period the fish community between the two basins was different, however, the effect of removing the steel barrier, is that the water quality variables of both sites are slowly return to similar values as they become ecologically more similar.

3.5 Conclusions

The positive impacts that biomanipulation can have upon water quality was successfully demonstrated in the Pound End case study. Increased water clarity and reduced chlorophyll/TP values were both observed in the years with greater biomasses of large grazing Cladocera. The restoration of stable macrophyte dominance within the waterbody was however harder to achieve. In the exclosure areas, water lilies thrived, but on the whole, Pound End has been rather species poor in terms of submerged macrophytes and of generally low abundance. Without a source of macrophyte seeds and recruitment of species, it is likely that restoration of clear water conditions in waterbodies connected to the open river system will have to wait for further reductions in nutrient inputs in the catchment. At lower average TP concentrations the success of fish removal efforts are likely to be greater in terms of delivering clearer water. More wide-scale reduction of fish biomass may also help achieve clear water. Experience in The Netherlands has shown that year on year commercial fish “harvesting” from connected waterways has worked in reducing benthivorous fish abundances (H. Hosper pers. comm.). Such management at the catchment scale is more sustainable in the longer (decadal) timescale, which can be supported by more intense restoration effort at priority sites where greater improvements are required.

4.0 Ormesby and the Trinity Broads

This chapter summarises a detailed evaluation of the changes in the water quality, zooplankton, macrophyte and fish data available for the Trinity Broads (Ormesby, Rollesby, Ormesby Little & Filby Broads) carried out by Dr Geoff Phillips, National Ecology Team Leader, Environment Agency. This summary focuses on changes in Ormesby Broad following biomanipulation in the winter of 1995, but considers both the impacts of this on the other Trinity Broads and the longer term changes in the system. Full details of fish removal work and surveys are detailed in (Tomlinson & Perrow 2005b). A summary of the manipulation and follow up work are detailed in Table 4.1 below.

Table 4.1 Summary of biomanipulation work carried out at Ormesby Broad

Period	Events	Isolation	Fish Management	
1996 – 1999	1995 (winter) Barrier installed	Effective barrier	Annual fish removal	Control bream spawning
2000 - 2002	2000 (spring) Barrier replaced, large ingress fish during 2001	Barrier less effective	Annual fish removal	Control bream spawning
2003 - 2005			No removal	
2005 - 2006	2005	New Barrier	No removal	Control bream spawning

Data are drawn from Environment Agency water quality and zooplankton monitoring, annual Broads Authority macrophyte surveys, annual winter fish surveys carried out by ECON for the Trinity Broads Partnership. The analysis covers the period 1983 – 2005 (where data are available). For analysis, data were summarised as annual and seasonal means. The year was divided into three seasons, the preceding winter, and the growing season, itself split into two, divided by the clear water maxima, which generally occurs in June.

a) winter preceding growing season November – February
b) spring of growing season March – June
c) summer of growing season July – October

Statistical analysis[1] was carried out on these summary data with values categorised into 5 year periods ending in 1980,1985,1990,1995,2000 & 2005 variables were transformed to ensure normality for ANOVA (using logarithmic or square-root transformations) where appropriate.

4.1 Effects of biomanipulation on the fish community

There are insufficient pre-biomanipulation surveys to test the significance of the change in fish biomass. However it can be seen that the biomass of fish was substantially lower in all of the Trinity Broads in comparison to that found in Ormesby (Figure 4.1) in the winter of 1995/96 prior to the installation of the dam isolating Ormesby from the rest of the system.

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[1] Using SPSS v14

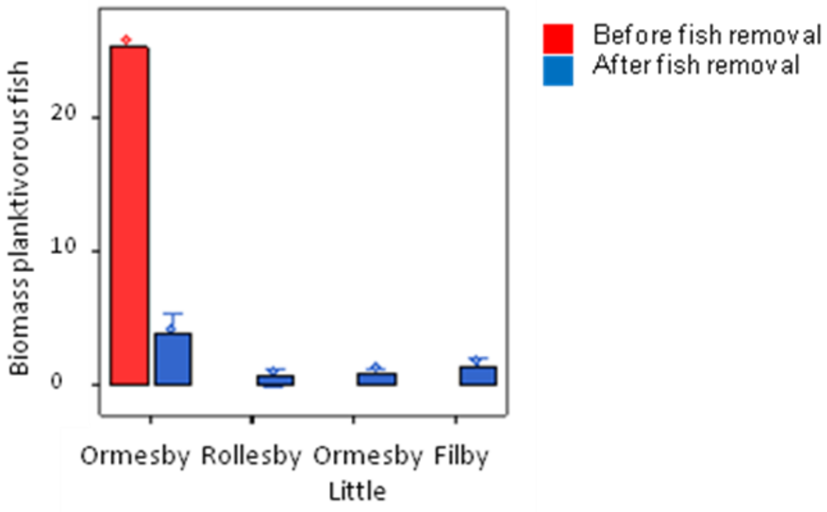

Figure 4.1 Mean planktivorous fish biomass in winter of 1995/96 prior to biomanipulation in Ormesby Broad (red) and in the following decade (1996-2005) for each of the Trinity Broads (blue).

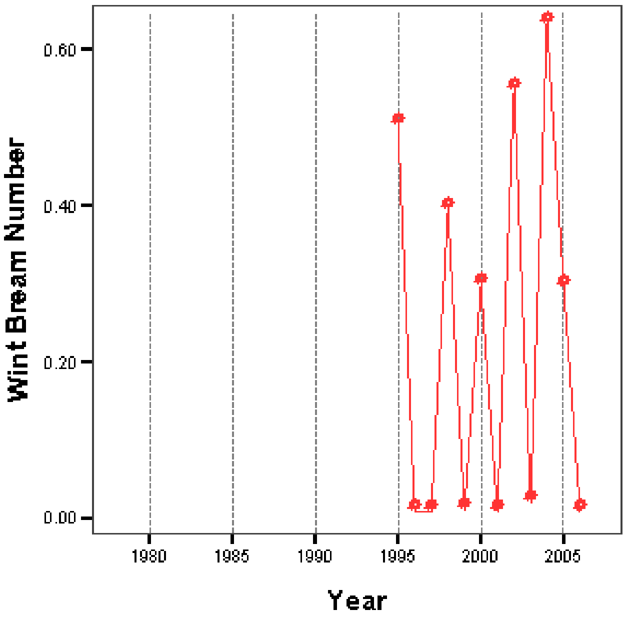

Figure 4.2 Number of bream recorded in Ormesby Broad in the winter preceding the growing season.

It was also noted that the number of Bream recorded in winter surveys of Ormesby Broad also showed very marked alternate year cycles (Fig 4.2). Surveys were carried out during the winter of each year and the convention followed in this report has been to link these values to the following year. However, as they are dominated by 1+ fish they also represent the bream recruitment for the previous summer.

4.2 Water clarity and zooplankton grazing pressure

During the spring, in the decade prior to biomanipulation, the chlorophyll *a* concentration was not significantly different in each of the Trinity Broads. During the summer, the chlorophyll *a* in Ormesby was significantly lower (Figure 4.3) than in the other Broads (F=3.4 p=0.022). Following biomanipulation the only significant reduction in chlorophyll *a* occurred in Rollesby and Filby during the summer (F=13.4

p=0.002 & F=5.9 p= 0.026 respectively) and there was no significant reduction of chlorophyll *a* in Ormesby Broad. Thus, during the summer, following biomanipulation, the differences between the chlorophyll *a* concentration in the individual Trinity Broads were no longer significant.

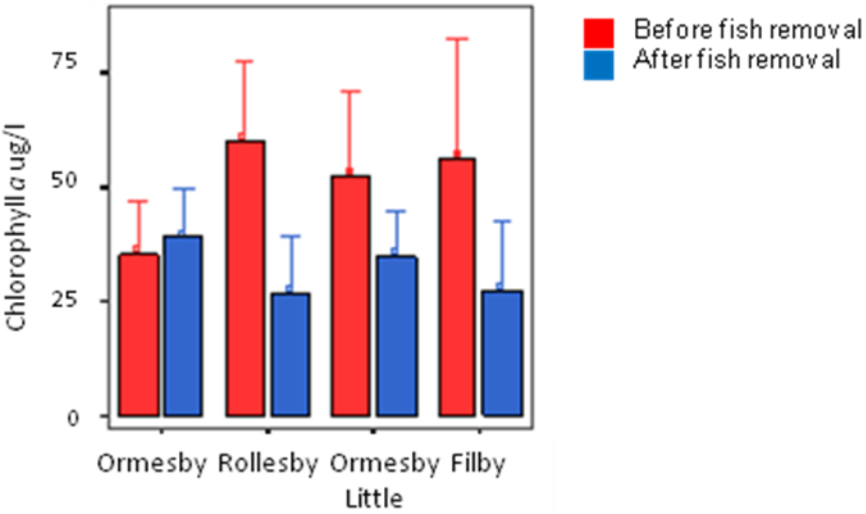

Figure 4.3 Mean summer phytoplankton biomass (chlorophyll a concentration) for the decade prior to (1986 -1995) and following (1996-2005) biomanipulation for each of the Trinity Broads.

Following the reduction in phytoplankton biomass in Rollesby, Ormesby Little and Filby, Figure 4.4 shows that there was a significant increase in transparency during the spring in Rollesby (F=13.2 p=0.002) and during the spring and summer (Figure 4.4) in Filby (F=5.9 p=0.026, F=6.8 p=0.018).

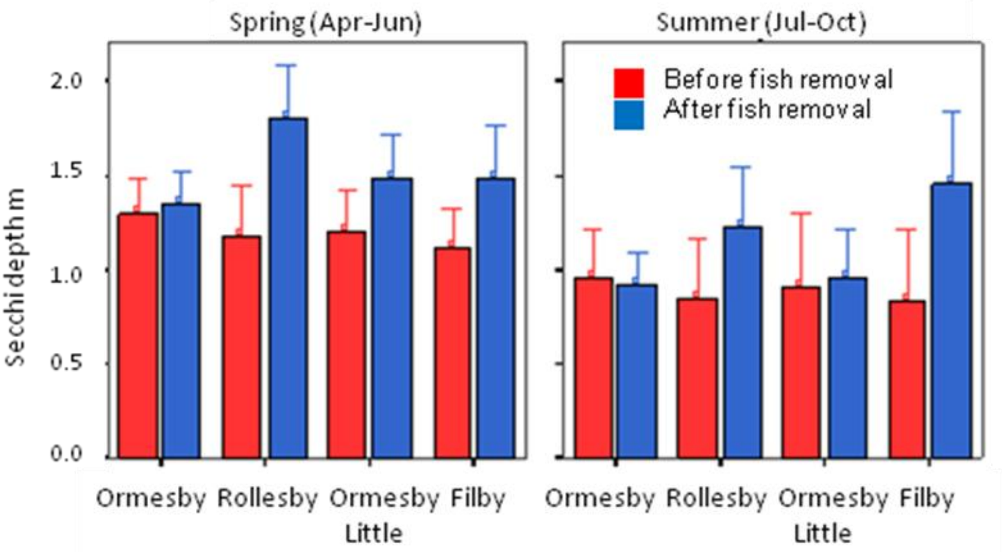

Figure 4.4 Mean spring and summer Secchi disc depth for the decade prior to (1986 -1995) and following (1996-2005) biomanipulation for each of the Trinity Broads.

Prior to biomanipulation the biomass of zooplankton was similar in all of the Trinity Broads. In general, following biomanipulation, there was a significant reduction in the biomass of small bodied cladocerans (*Daphnia hyalina*, *Ceriodaphnia pulchellum*, *Bosmina longirostris*) in the decade following biomanipulation. During the spring this was highly significant in Ormesby, Rollesby and Ormesby Little (F=26.1 p<0.001, F=17.2 p=0.003, F=51.3 p<0.001). Significant decreases during the summer in Rollesby, Ormesby Little and Filby (F=9.9 p=0.01, F=6.4 p=0.025, F=30.9 p=<0.001).

This was accompanied by a significant increase in the biomass of large bodied cladocerans (*D. magna, D. pulex*) (Figure 4.5). Significant increases occurred during the spring, summer and winter in Ormesby (F=18.3 p=0.001, F=7.5 p=0.017, F=32.2 p=<0.001). Ignoring season there was a significant decrease in the biomass of small cladocerans in all of the Trinity Broads following biomanipulation (p<0.001) and a significant increase in the biomass of large cladocerans in Ormesby Broad (p=0.002).

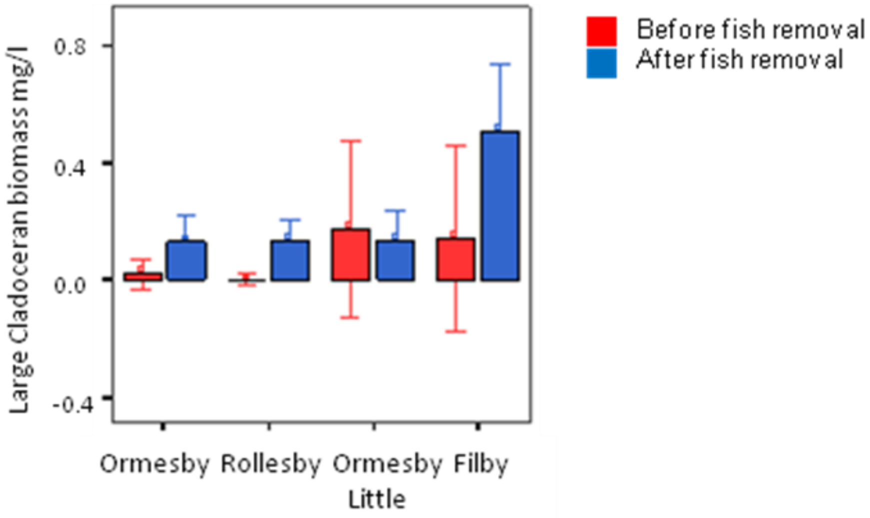

Figure 4.5 Mean summer large cladoceran biomass (mg/l) for the decade prior to (1986 -1995) and following (1996-2005) biomanipulation for each of the Trinity Broads

Prior to biomanipulation the TP concentration shows a clear gradient, increasing from Ormesby to Filby (Figure 4.6). There was a significant increase in the annual concentration of TP following biomanipulation in Ormesby Broad (F=13.5 p=0.001). This occurred in the spring and summer in Ormesby (F=6.8 p=0.018, F=25.0 p=<0.001) but there was no evidence of an increase during the winter in Ormesby. In contrast there were slightly significant increases in the winter in Rollesby, and Filby (F=6.8 p=0.018, F=6.2 p=0.024) but not at other times of year.

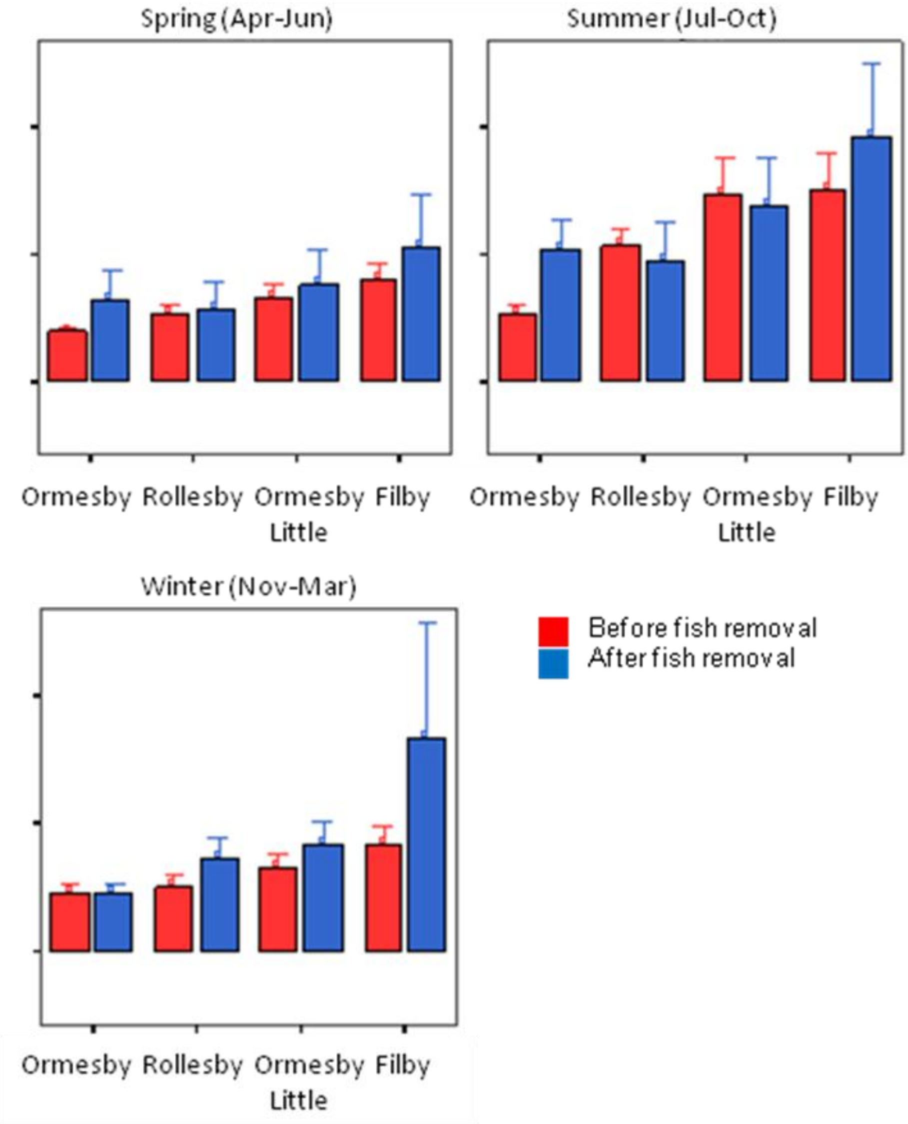

Figure 4.6 Mean seasonal total phosphorus concentration (ug/l) for the decade prior to (1986 -1995) and following (1996-2005) biomanipulation for each of the Trinity Broads

Figure 4.7 shows there was a significant reduction in the annual chlorophyll *a*/TP ratio in all of the Trinity Broads following biomanipulation (Ormesby F=10.4 p=0.002, Rollesby F=12.6 p=0.001, Ormesby Little F=5.6 p=0.021, Filby F=13.6 p=0.001). This was most marked in Ormesby and Filby Broads, in Ormesby differences were significant in spring and summer (F=6.8 p=0.018, F=8.9 p=0.008).

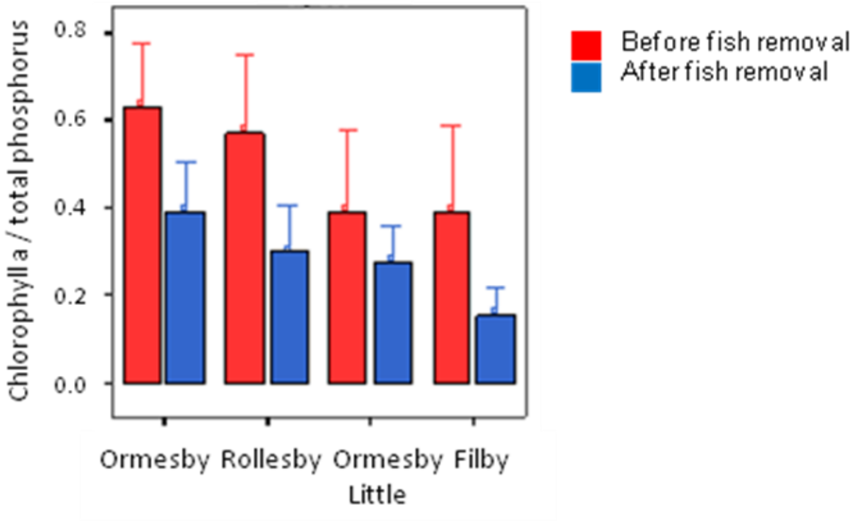

Figure 4.7 Mean summer chlorophyll *a* / total phosphorus ratios, for the decade prior to (1986 -1995) and following (1996-2005) biomanipulation for each of the Trinity Broads.

4.3 Impact on macrophyte populations

There was a highly significant increase in the cover of macrophytes in all of the Trinity Broads following biomanipulation (Figure 4.8a) (Ormesby F=19.2 p=<0.001, Rollesby F=13 p=0.001, Ormesby Little F=52.8 p=<0.001, Filby F=110 p=<0.001). There was also a highly significant increase in the number of taxa recorded in all of the broads except Rollesby (Figure 4.8b) (Ormesby F=64.7 p=<0.001, Ormesby Little F=15.8 p=<0.001, Filby F=29.7 p=<0.001).

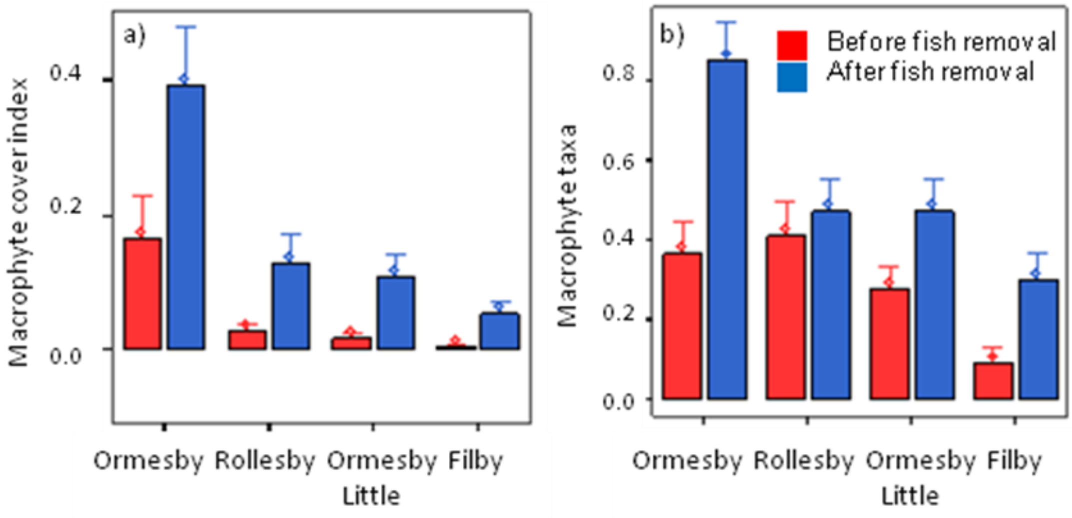

Figure 4.8 Mean macrophyte cover index a) and number of taxa found b), for the decade prior to (1986 -1995) and following (1996-2005) biomanipulation for each of the Trinity Broads.

4.4 Summary

- Changes found in the decade following the biomanipulation can be seen in all of the Trinity Broads, not only in Ormesby.
- There was a significant change in the size distribution of cladoceran grazers, with more large bodied grazers present in all of the broads in the decade following biomanipulation.
- Despite this, following biomanipulation, there was only a significant reduction in phytoplankton biomass (chlorophyll a) in Rollesby & Filby broads. There were similar changes in water transparency, with no significant change in Ormesby.
- However, there was a significant increase in total phosphorus (TP) during the spring and summer, but not in the winter, in Ormesby following biomanipulation. As a consequence the ratio of chlorophyll *a* to TP decreased significantly.
- The macrophyte cover and number of taxa increased significantly in all broads. This was most marked in Ormesby Broad, despite this broad showing the least change to either the light climate or phytoplankton biomass.

From these observations it is likely that in the decade following biomanipulation planktivorous fish biomass was significantly lower in all of the Trinity broads. This resulted in a shift in the size distribution of cladocerans, which increased the grazing rate and reduced the biomass of phytoplankton in relation to phosphorus. However, an increase in phosphorus concentration, meant that the overall biomass of phytoplankton was not reduced. Despite this there was a significant increase in macrophyte abundance and diversity.

3.5 Conclusions

- There is evidence of changes in the food web, indicative of top down control in the Trinity Broads in the decade following biomanipulation

- There has been a substantial increase in TP. This occurred after biomanipulation , and as it was restricted to the summer is likely to be derived from the sediment. It is probable that this is linked in some way to the biomanipulation.

- There is minimal evidence that the increase in macrophyte abundance and diversity is directly influenced by this, as neither was water transparency.

5.0 Barton Broad

Prior to the biomanipulation work in Barton Broad, extensive mud-pumping was carried out to remove the nutrient-rich surface sediments and to increase the water depth. The result of this work was a reduction in water concentrations of total phosphorus and chlorophyll *a* (Broads Authority 2006). There has also been a steady decrease in the phosphorus load entering Barton Broad since the late 1970's, following phosphate stripping and effluent diversion at waste-water treatment works in the River Ant catchment (Phillips *et al* 1999). The improved water quality has been a vital step in the restoration of this broad.

A further phase in the restoration process has been through the biomanipulation effort, which has entailed fish removal from several exclosure areas around the perimeter of the broad. As Barton Broad has a wide navigation channel that is required to be kept open to enable free passage of boats, biomanipulation was not feasible in the whole of the lake area, as at Ormesby Broad. Isolated bays or "exclosures" were separated from the main lake by flexible PVC-coated, polyester barriers supported in the water column by two rows of floats. There are currently four main exclosure areas, two of which (Turkey Broad West (2.4 ha) & Neatishead Arm South (1 ha)) have been in place since 2000. Two smaller exclosures Turkey Broad east (TBe) (0.7 ha) and Neatishead Arm north (NAn) (0.8 ha) were installed in 2003.

In addition to the fish removal, the Turkey Broad west (TBw) barrier also had artificial macrophyte structures (plastic brushes) placed within it to act as a refuge for zooplankton. The brushes were originally enclosed in March 2001 within a 0.3 ha mesocosm within the TBw barrier. Due to regular fish incursions into this particular barrier, the main TBw barrier and mesocosms were converted into one large mesocosm (not in contact with the lake edge) in early March 2003.

This case study aims to present the data collected from the fish-exclosure areas to date with interpretation as to the successes and limitations of the partial-lake biomanipulation approach adopted.

Methods

Details of the point abundance sampling by electrofishing (PASE) survey methods carried out by ECON Ecological Consultancy have been previously reported (Perrow *et al.* 1996). Environment Agency and Broads Authority staff has conducted water quality monitoring since the Clearwater 2000 project began, with water samples collected at least on a monthly basis. The SCUBA macrophyte surveys conducted by Jane Harris are based on the methodology adopted for the Hickling macrophyte monitoring (Harris 2001).

5.1 Effects of biomanipulation on the fish community

Data from the autumn point abundance sampling by electrofishing (PASE) surveys, undertaken by ECON Ecological Consultancy from 2000 to 2007 in the Barton Broad exclosures, are presented. The overall mean of fish species data (open water and littoral results) is used throughout. Tables 2 to 5 give the number and biomass of all fish species removed from the fish exclosures.

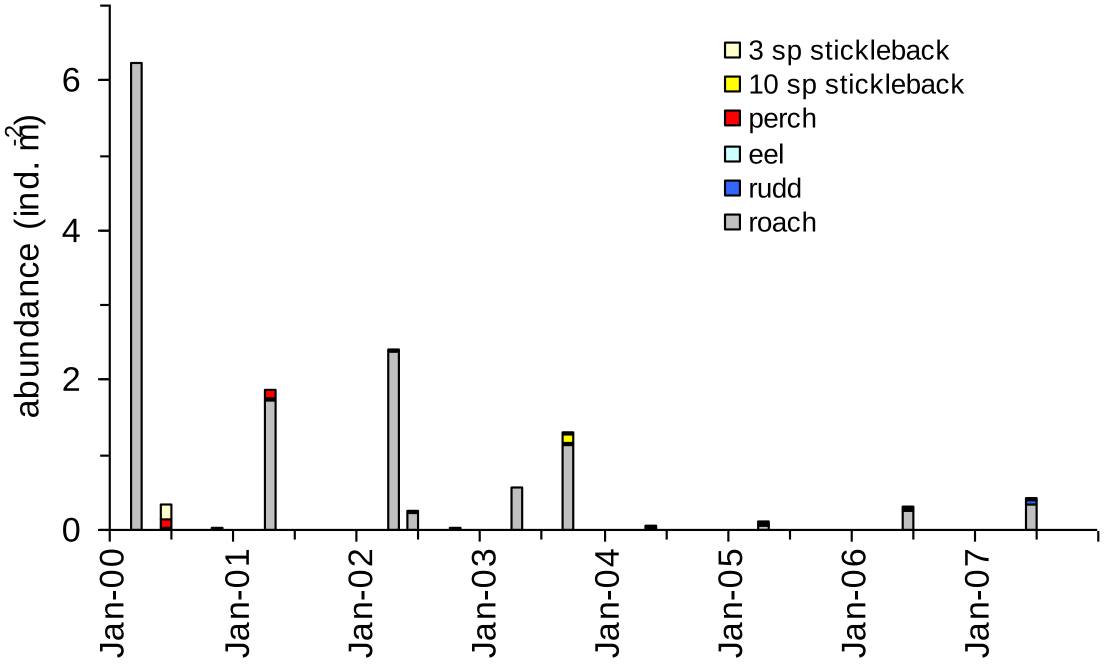

Figure 5.1 Abundance of the dominant fish species in all PASE surveys in Neatishead Arm south (NAs) exclosure (open water & littoral, 2000 - 2007)

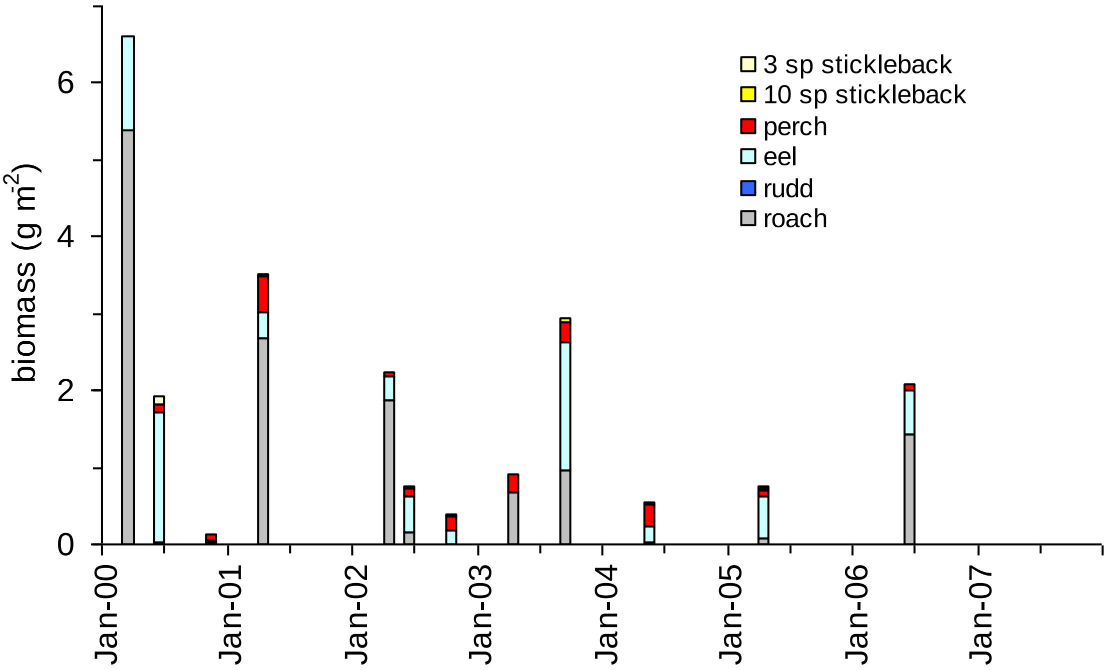

Figure 5.2 Biomass of the dominant fish species in all PASE surveys in Neatishead Arm south (NAs) exclosure (open water & littoral, 2000 - 2007)

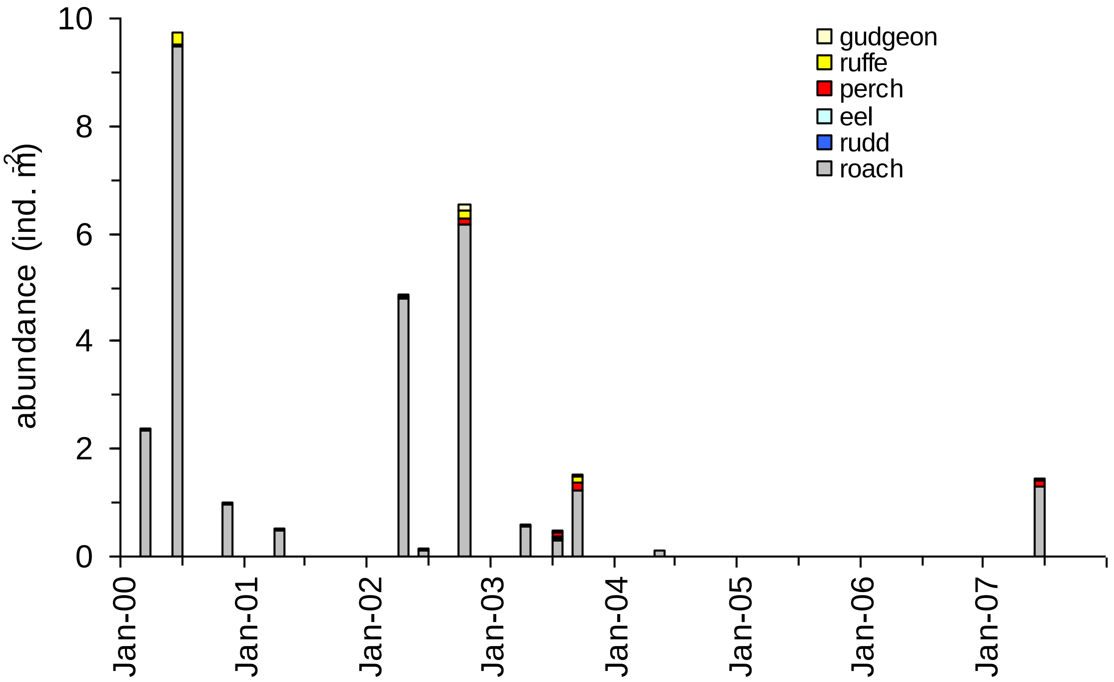

Figure 5.3 Abundance of the dominant fish species in all PASE surveys in Turkey Broad west (TBw) exclosure (open water & littoral, 2000 - 2007).

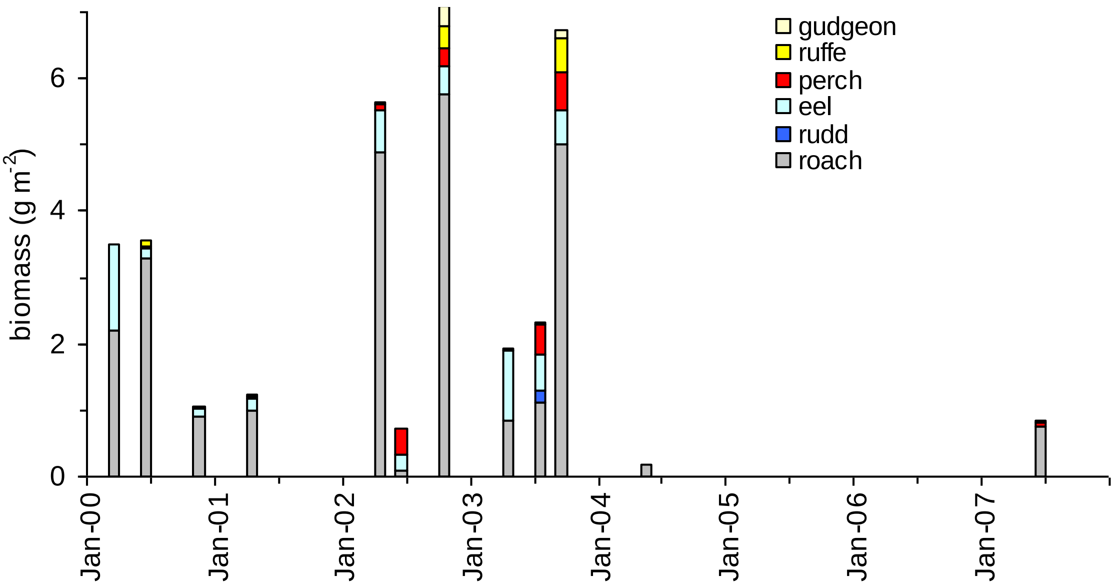

Figure 5.4 Biomass of the dominant fish species in all PASE surveys in Turkey Broad west (TBw) exclosure (open water & littoral, 2000 - 2007).

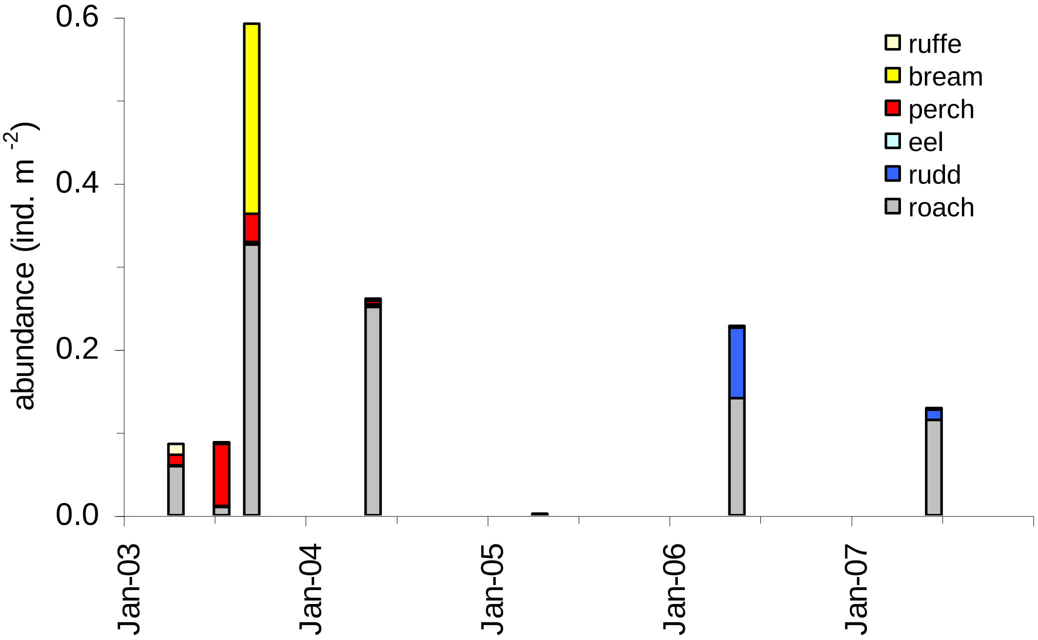

Figure 5.5 Abundance of the dominant fish species in all PASE surveys in Neatishead Arm north (NAn) exclosure (open water & littoral, 2003 - 2007).

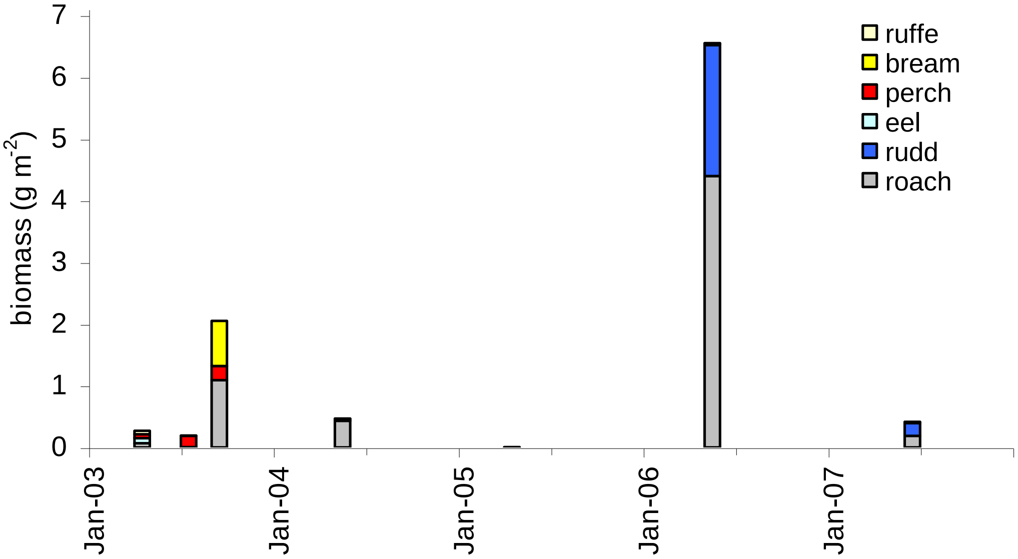

Figure 5.6 Biomass of the dominant fish species in all PASE surveys in Neatishead Arm north (NAn) exclosure (open water & littoral, 2003 - 2007).

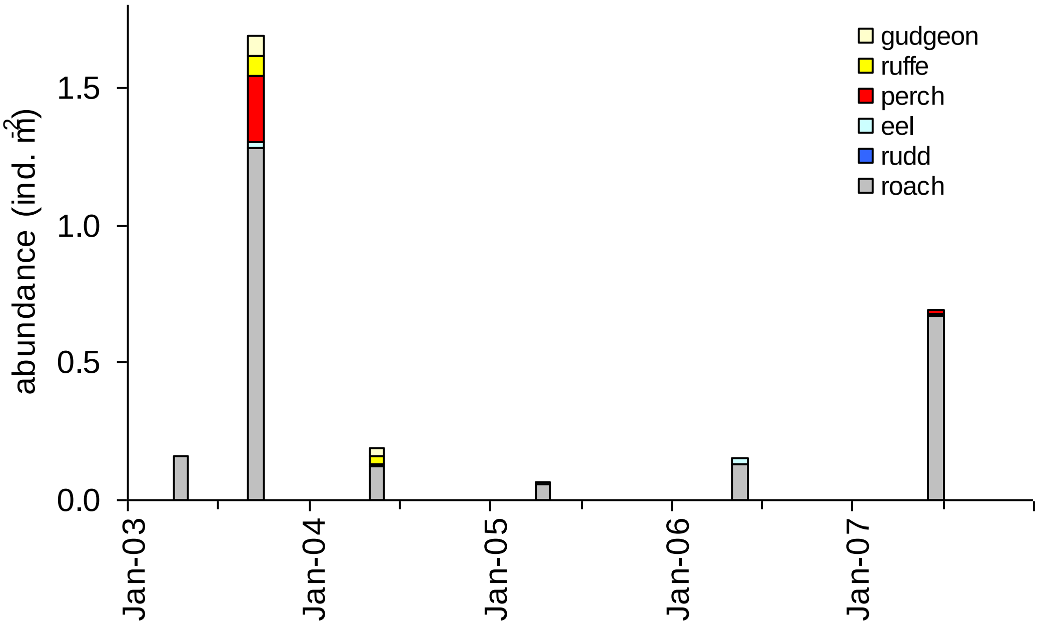

Figure 5.7 Abundance of the dominant fish species in all PASE surveys in Turkey Broad east (TBe) exclosure (open water & littoral, 2003 - 2007).

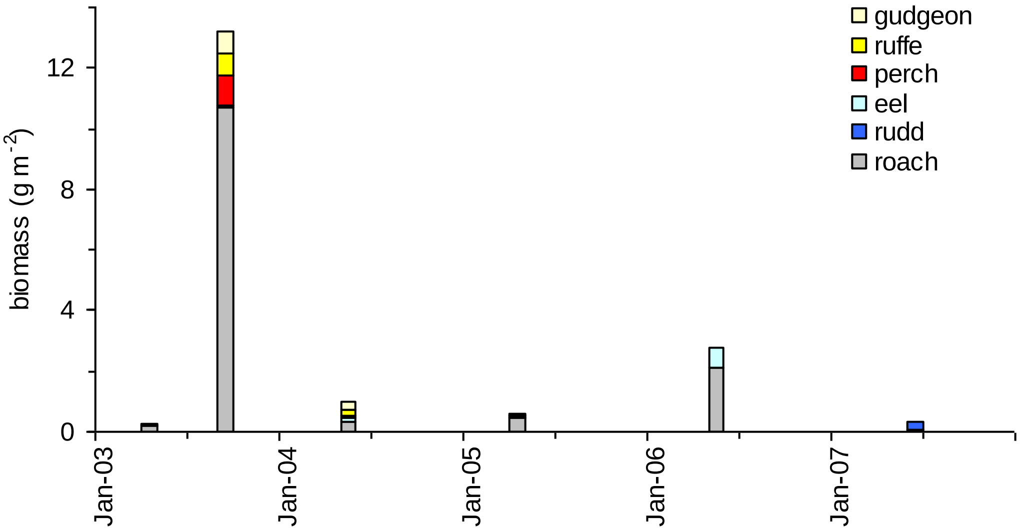

Figure 5.8 Biomass of the dominant fish species in all PASE surveys in Turkey Broad east (TBe) exclosure (open water & littoral, 2003 - 2007).

Of the exclosures currently present in the broad (Sept 2007), NAs and TBw have been in place the longest, both installed originally in February 2000. The fish removals and subsequent maintenance of relatively low fish abundance and biomass has been particularly successful in the NAs exclosure. All large benthivorous species have been removed and the roach population has been effectively controlled over the period (Figure 5.1). Roach abundance has been consistently below 0.4 ind m^{-2} since spring 2004. Encouragingly, eel and perch have made a noticeable contribution to the total community biomass within the exclosure (Figure 5.2). Both of these species are deemed to be desirable members of a restored, macrophyte-dominated shallow lake ecosystem.

TBw has been the harder of the two oldest barriers to keep fish out of, with several incursions having occurred. See Figures 5.3 and 5.4 for the peaks in fish abundance and biomass that occurred in 2002 and 2003. This has led to several alterations to the layout of this exclosure. TBw has also had various other experimental exclosures installed within it, such as one with plastic brushes suspended on strings, aimed to simulate macrophyte structure and provide underwater refuge for invertebrates. Lack of data in 2005 and 2006 make interpretation of the single survey in 2007 difficult, but it appears that fish removal work in TBw has reduced the overall abundance and biomass of roach over time.

The fish surveys conducted in NAn since 2003 have shown a relatively low abundance of all fish species. The bream that were present in autumn 2003 appear to have been completely removed. In the last two years rudd has made a much greater contribution to the biomass. Again this species is particularly suited to macrophyte-dominated conditions. The biomass of roach and rudd in 2006 was not matched by a proportionately great numerical abundance. This indicates that individual fish of greater size were present, rather than lots of small fish. This is a positive shift in the community as fewer mouths of these zooplanktivorous fish means less predation on the zooplankton.

The abundance of zooplanktivorous fish has also been successfully controlled since 2004 in TBe, with the exception of an incursion of small roach in early summer 2007. No bream have been captured in this exclosure since May 2004.

Overall the barriers have worked very well in excluding fish from the biomanipulation areas. Where failures have occurred, especially in TBw, these have been due to high water allowing fish around the back of the swamp/carr and into the exclosure; tidal and/or wave action undercutting the base of the barriers; and occasional rips between the uPVC panels of the barriers as they wear with age. Exclusion of the shoals of larger benthivorous fish such as bream has been almost total. The stabilisation of the sediment through lack of feeding disturbance, in addition to the physical shelter provided by the barriers themselves, will have increased the survival rates of rooted vegetation. Roach incursion and spawning remains a constant risk, but with adequate monitoring and targeted removal exercises, their populations can be effectively controlled.

5.2 Water clarity and zooplankton grazing pressure

The best measure of water transparency in the exclosures has been found to be that of the concentration of chlorophyll *a*, the predominant pigment used in the cells of microscopic algae, or phytoplankton. As such, this chemical measure obtained from several years of monitoring within the NAs and TBw exclosures reveals the relative clarity of water within the fish removal areas. Figure 5.9 shows the lower average summer chlorophyll *a* concentrations within both fish exclosures when compared to the main broad. The other exclosures, NAn and TBe have not been monitored for changes in water quality.

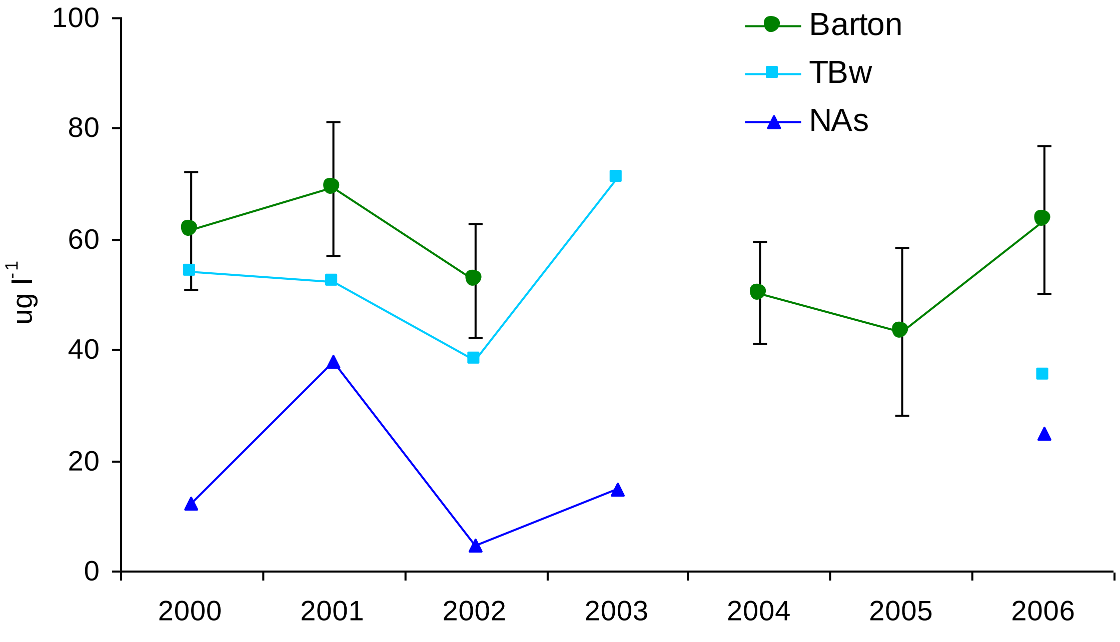

Figure 5.9 Summer mean chlorophyll *a* concentrations within Barton Broad (± 1 standard error) and the NAs and TBw fish exclosures.

Previous studies in the Broads and elsewhere in Europe have shown that the direct feeding links between small fish such as roach, and their zooplankton prey, are important elements in determining water quality in shallow lakes. The rate of predation upon the algal-grazing zooplankton in biomanipulated areas is therefore reduced to such an extent that their populations flourish and control the growth of algae. At the same time as the zooplankton reduce the algal concentration, the rooted macrophytes which grow through the cleared water lock up a significant proportion of the nutrients that would otherwise be available to the algae. The data presented in Figure 5.9 supports this general scheme of lake functioning, and also the value of utilising fish removal as a tool to increase water clarity.

The small amount of zooplankton data from Turkey Broad west and Neatishead Arm south indicate an increase in the populations of *Daphnia* sp, but are too few to statistically derive any clear relationships with water quality variables.

5.3 Impact on macrophyte populations

The ecological monitoring that most usefully describes change in water clarity and overall success of shallow lake restoration efforts is that of the water plants. Submerged macrophytes are responsive to environmental conditions, but also help shape the overall ecological structure, so are an ideal indicator group.

Table 5.1 Locations and years of SCUBA macrophyte surveys in the Barton Broad fish exclosures.

	Neatishead Arm North	Neatishead Arm South	Turkey Broad East	Old Barrier C (north of broad)
2000	✓	✓		✓
2001	✓			✓
2002	✓			✓
2003	✓			✓
2004	✓			✓
2005	✓	✓	✓	✓
2006	✓			✓
2007	✓	✓	✓	✓

Of the biomanipulated fish exclosures, NAn has been monitored for macrophyte growth every year since its installation (Table 1). The trial fish barrier installed in 1996 in the north of Barton Broad (Barrier C) was removed before the first SCUBA macrophyte survey in 2000. SCUBA surveys subsequently conducted in this part of the broad have therefore served as a useful control area for evaluation of plant growth in the main broad, outside of the actively biomanipulated exclosures.

Figure 5.10 shows a pattern of increased macrophyte growth in the barrier C area from 2004 onwards, a pattern which matches closely that reported in the longer macrophyte dataset from the annual transect survey of the main broad (Hoare & Kelly 2006). In the north of the broad, lilies were the only plants present in any abundance prior to 2004. However, before the dramatic increase in hornwort, *Elodea* species and filamentous algae was observed across the broad in 2004, the NAs exclosure had already experienced profuse plant growth for several years (Figure 5.11).

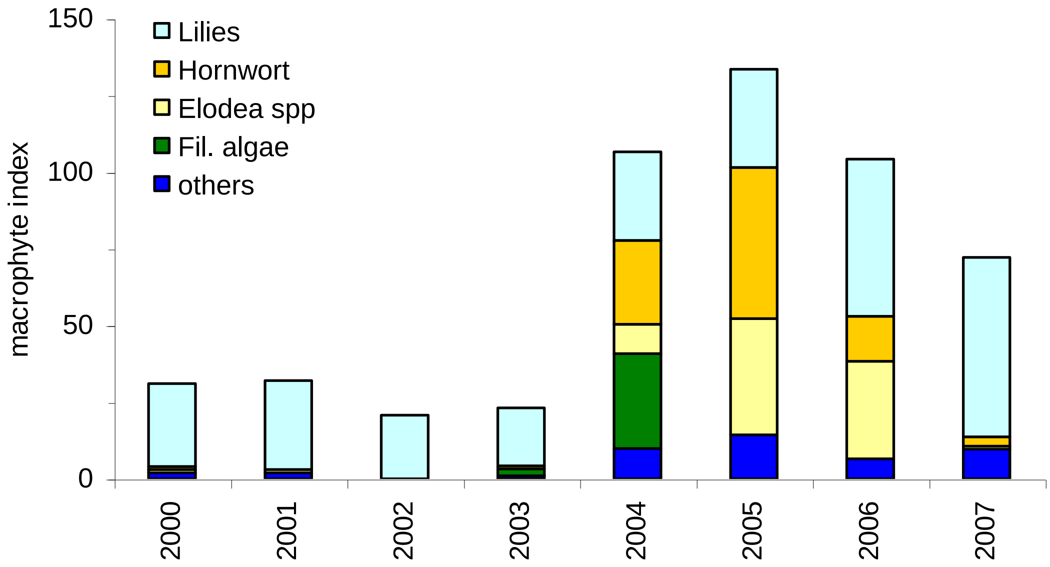

Figure 5.10 Annual SCUBA macrophyte survey results from the Barrier C area (northern part of the broad)

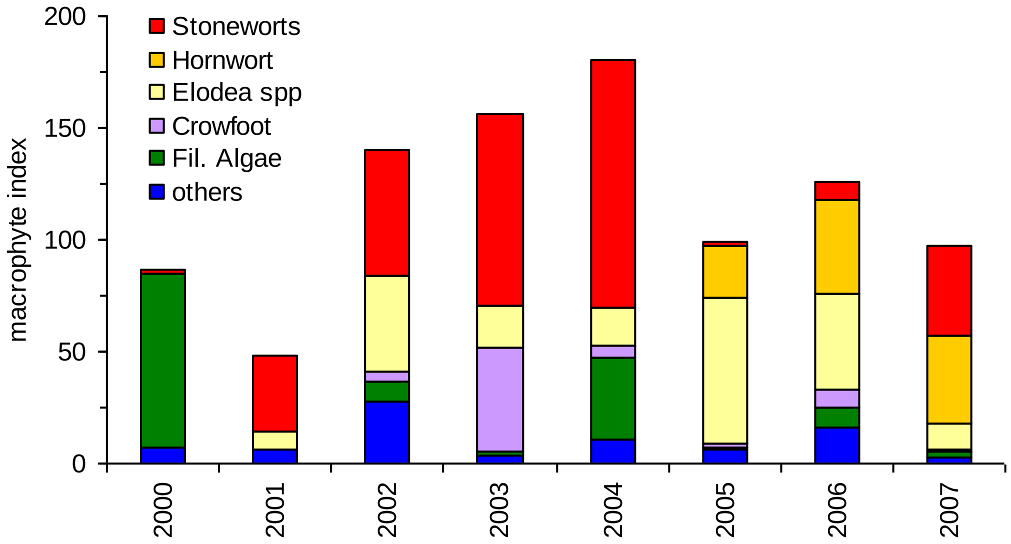

Figure 5.11 Annual SCUBA macrophyte survey results from Neatishead Arm south (NAs) exclosure

Figure 5.11 shows that during the first summer (2000) after exclosure and fish removal within NAs, a large amount of filamentous algae was present, but with only a small amount of other macrophyte species. In the second year this had changed to a moderate abundance of plants, but this time dominated by stonewort species, namely *Chara globularis*, *C. vulgaris* and *C. hispida*, which represent a far more sensitive group of plants. By the third year, 2002, stoneworts were again prevalent, but so were *Elodea* species, e.g. Canadian pondweed, and ivy-leaved water-crowfoot. Since 2002, NAs has been typified by abundant plant growth, with up to nine species present, often growing up to the water surface.

A comparison of the variation on plant types present and relative abundances within the different fish-exclosure areas is presented in Figure 5.11.

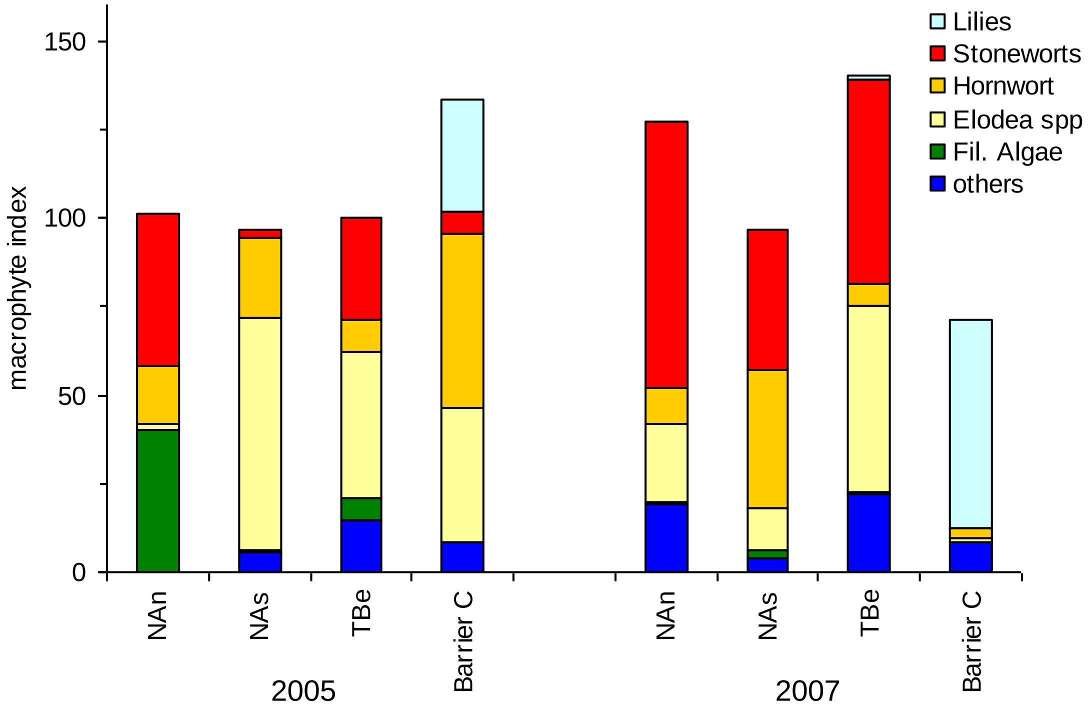

Figure 5.12 SCUBA macrophyte survey results from four fish exclosure areas in Barton Broad during 2005 and 2007.

The barrier C area is not enclosed by a barrier or biomanipulated, so it's relatively similar plant abundance to the other three actively biomanipulated areas during 2005 demonstrates how widespread the growth of plants was along the broad's littoral margins that year, especially along the northern and western sides. Conversely, in 2007, the old barrier C area was depauperate in plant growth compared to the actively biomanipulated areas.

5.4 Summary

Removal of fish from the exclosures has been very successful, with reduced abundance and biomass in all cases. Roach have been the most numerous species removed, with benthivorous species, particularly bream, being almost totally excluded from all exclosures throughout the period. The experimental arrangements of barriers, mesocosms and artificial refuges (plastic brushes) installed at Turkey Broad west have demonstrated the need for the presence of marginal vegetation within the exclosures; a good fish-tight seal where the barrier meets the land; and continued fish removals to prevent population increases.

The change in ecological structure within the barriers has lead to increased clarity compared to the main broad. Zooplankton data from the Barton exclosures is limited to a small amount of data from Turkey Broad west and Neatishead Arm south. This shows increases in the potential grazing rate of the zooplankton species released from such intense predation pressure.

The greatest indicator of the success of the lake restoration process has been the clear water periods generated within the barriers, which have been accompanied by abundant macrophyte growth. All biomanipulated areas have experienced greater plant growth than that observed in the broad as a whole. In 2005 extensive plant growth was observed chiefly along the western edge of the main broad, but this did not include the variety of species found in the exclosure areas.

5.5 Conclusions

Annual selective fish removals have gradually shifted the fish community from one where zooplankton was heavily predated, to one with a more even balance of mixed fish species. Within the exclosures clearer water and a diverse macrophyte community has been achieved.

The dramatic increase in macrophyte growth within the exclosures following biomanipulation may have been assisted by a fertile seed bank being present, either being exposed during the dredging operation or arising from plants upstream of the broad. Whatever the mechanism of recolonisation, the regrowth has been rapid and of a species diversity not seen in the broad as a whole. The gradual improvement of water quality within the main broad and the presence of dense plant stands within the exclosure may have been the triggers that led to the increased abundance of plants along the western margin of the main broad in 2003 (Hoare & Kelly 2006).

It is clear that whilst Barton Broad remains turbid and generally free of plant growth, maintenance of the fish free exclosures is necessary for the continuation of the clear water and macrophyte diversity that now resides within them. This will involve maintaining the integrity of the barriers and continued monitoring and removal of fish species, along with macrophyte surveys to assess the future success of the restoration technique.

6.0 Hoveton Great Broad

There are currently six barriers of the same design as those used in Barton Broad installed in Hoveton Great. Three have been in place since 2001, with a latter three put in place in early 2002. Monitoring of these exclosures has been minimal in terms of the full range of ecological and water quality variables recorded at other Broads sites. The exclosure and fish removal technique has been used in this Broad, purely as a management tool, without the need for intensive monitoring of the outcomes.

Fish surveys and removals in spring 2003 showed that three of the smaller barriers had low overall fish populations likely to be conducive for macrophyte growth and clear water. The larger exclosures revealed a larger fish population, mainly roach, of which roughly half was removed. (Tomlinson & Perrow 2003).

Subsequent fish removals in spring 2004 identified that the integrity of the barriers to fish passage was generally poor and that less desirable elements of the fish community, especially large adult bream, where finding their way into several of the exclosures. Problems with over-topping, under-cutting and rips in the barrier panels were threatening the success of the fish removals.

As such, improved water clarity and recovery of macrophytes in the Hoveton Great Broad exclosures has been minimal. Favourable results have occurred during the periods that fish have been successfully excluded from the barriers, but this condition has been difficult to sustain at this site. Macrophyte species that have been recorded within the exclosures has included rigid hornwort, white water-lily and horned pondweed, although not a densities significantly different to the main broad.

7.0 Overall conclusions

The examples of fish removal presented in this report demonstrate that there is always some benefit to be gained from fish community manipulation work in eutrophic shallow lakes such as the Broads. The inter-annual success of the removals, in achieving clearer water, is often dependent upon the characteristics of the physical barrier preventing return of fish into the manipulated area. In a navigable, riverine environment such as that found in the majority of the broads, the technical feasibility of this task is a major challenge. Recovery of a diverse macrophyte community is also highly dependent upon the presence of a viable seed-bank contained in the sediment.

As the greatest nutrient inputs from point source discharges have reduced due to effective legislative control and new technologies employed by the water companies, the likelihood of biomanipulation achieving its aims have increased. The nutrient concentration threshold, principally phosphorus, at which recovering lakes will switch from algae to macrophyte dominance, is not always clear, as variation in plant colonisation may hold back certain lakes. However, lower potential for algae to proliferate, in terms of the nutrients available, is a key foundation for long-term lake restoration. Further reduction in the inputs of nutrients within the river catchments as a whole will also support this effort.

The aim of lake restoration is to achieve self-sustaining and resilient ecosystems that support the diversity of aquatic life and associated ecological functions that they once did prior to the degradation witnessed in modern times. When viewed in the longer-term, the recent fish manipulations carried out in the Broads may be seen as single events, which kick-start such sites into a macrophyte-dominated condition with clear water. With this view, the current management work undertaken in the Broads, is in many cases, still at the stage of getting macrophytes established and ensuring the water is clear enough for long enough, for successful plant growth to continue. Once the macrophyte community is established in sufficient condition for intervention to be unnecessary to maintain it, then the restoration process will be complete. As this final stage has not been realised, the continuation and expansion of restoration techniques, such as biomanipulation, are required if The Broads are to be successfully restored. Fish removals clearly are an effective tool in this effort, as demonstrated by the case studies presented in this report, and thus the continuation of such work is recommended.

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Table 9.1 Fish removal data from Cockshoot Broad

COCKSHOOT	1989	1990					1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
Spawning interference	?	?	?	?	?	?	?	nd	?	?	?	?	✓	✓	✓	✓	✓	✓
Removal effort (days)																2	2	
Number																		
Roach	43682	15125					509	nd	11068	1592	1891	6678	770	3634	10243	108	123	1231
Bream	20869	826					0		112	145	12	37	2	28	18			
Rudd	1259	676					0		84	149	0	309	0	130	66			
Gudgeon	1828	2898					7		2	1	0	0		13	42			
Perch	1808	2154					617		197	72	0	0						
Ruffe	3628	870					76		77	182	1	217						
Tench	74	781																
Pike	113	72																
Stickleback sp.	17	0													2			
Roach hybrids	17	1													2			
Eel	655	50																
Biomass																		
Roach	283	87					3.96	nd	14.96	9.45	3.00	16.26	4.38	7.41		0.56	0.04	1.01
Bream	311	241					0		6.57	1.85	7.16	3.62	0.01	0.11				
Rudd	7.2	4					0		0.51	2.76	0	0.84	0	0.12				
Gudgeon	4	27.6					0.04		0.004	0.01	0	1.82	0	0.01				
Perch	45	31.7					8.03		3.62	1.08	0							
Ruffe	5.6	7.3					1.41		0.49	1.33	0.006	?						
Tench	45	16.2																
Pike	105	41																
Stickleback sp.	0.005	0													?			
Roach hybrids	0.06	0.3													?			
Eel	69.5	5.3																

Table 9.2 Fish removal data from Alderfen Broad

ALDERFEN BROAD	1993	1994	1995	1996	1997	1998	1999	2000
Spawning interference operation	?	?	?	?	?	?	?	x
Removal effort (days)	?	?	?	?	?	?	?	?
Cost	?	?	?	?	?	?	?	?
Number								
Bream	0	2	0	0	0	?		
Roach	1	1	810	0	0	?		1500
Rudd	158	51	48	0	0	?		6600
Tench	55	8	0	0	0	?		
Perch	776	19485	7340	0	0	?		added 10900
Ruffe	176	27	773	0	0	?		
Eel	20	49	0	0	0	?		
Pike	52	0	0	0	0	?		
Biomass								
Bream	0.0	5.1	0	0	0	?		
Roach	0.3	0.3	0.43	0	0	?		?
Rudd	4.1	1.3	0.67	0	0	?		?
Tench	1.5	2.8	0	0	0	?		
Perch	77.0	31.3	54.58	0	0	?		?
Ruffe	1.5	0.5	6.72	0	0	?		
Eel	16.4	33.9	0	0	0	?		
Pike	23.1	0.0	0	0	0	?		

Table 9.3 Fish removal data from Pound End

POUND END	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999
Spawning interference operation	?	?	?	?	?	?	?	?	?	?
Removal effort (days)	?	?	?	?	?	?	?	?	?	?
Number										
Bream	?	?	?	?	10	17396	545	17757	2145	?
Roach	?	?	?	?	29752	2835	200	350	81	?
Hybrid	?	?	?	?	0	19	7	28	4	?
Rudd	?	?	?	?	193	164	120	200	19	?
Gudgeon	?	?	?	?	0	156	68	133	21	?
Perch	?	?	?	?	2772	12636	1524	2676	0	?
Ruffe	?	?	?	?	431	6338	492	6104	697	?
Tench	?	?	?	?	43	?	?	?	?	?
Eel	?	?	?	?	239	?	?	?	?	?
Pike	?	?	?	?	?	?	?	?	?	?
Biomass										
Bream	?	?	?	?	10.45	194.06	16.05	82.87	22.53	?
Roach	?	?	?	?	259.05	8.94	1.69	73.61	8.96	?
Hybrid	?	?	?	?	0	0.27	0.21	2.22	2.03	?
Rudd	?	?	?	?	2.75	5	4.12	4.64	0.78	?
Gudgeon	?	?	?	?	0	1.01	0.54	1.62	0.21	?
Perch	?	?	?	?	12.65	95.61	27.45	21.22	0	?
Ruffe	?	?	?	?	4.4	56.84	5.8	60.06	10.41	?
Tench	?	?	?	?	5.7	?	?	?	?	?
Eel	?	?	?	?	30.25	?	?	?	?	?
Pike	?	?	?	?	?	?	?	?	?	?

Table 9.4 Fish removal data for Neatishead Arm south

Neatishead Arm south (NAs)	2000	2000	2000	2001	2002	2003	2004	2005	2006	2007
	March	Apr/May	July		Apr/May	May	May	May	May	Jul
Removal effort (days)	0.5	1	1.5	?	2	1	0.5	2		
Number										
Bream	0	0	810	?	0	0	1	8		
Roach	1086	1718	25324	?	12928	903	11	326	70	117
Rudd					1	0		25		11
Ruffe	0	6	5	?	0	0		6		2
Perch										2
Gudgeon					1	0		8	2	2
3 spined stickleback	0	26	5277	?	0	0		4	8	
10 spined stickleback	0	0	567		1	0		15		
Biomass (g)										
Bream	0	0	5660	?	0		4	3015		
Roach	942	1516	451	?	9749		18	1321	927	452
Rudd					1			68		33
Ruffe	0	45	29.3	?	0			17		
Perch										13
Gudgeon					2			43	5	7.2
3 spined stickleback	0	20.8	2135	?	0			9	2	0.7
10 spined stickleback	0	0	285		1			18		

Table 9.5 Fish removal data for Turkey Broad west

Turkey Broad west (TBw)	2000	2000	2000	2001	2002	2003	2004	2005	2006	2007
	March	Apr/May	July		Apr/May	May	May	May	May	July
Removal effort (days)	0.5	2	1.5	?	3.5	1.75	1.5			?
Number										
Bream	0	34	890	?	13	3	4	No fish	No fish	2
Roach	606	8002	20761	?	12586	4160	21	detected	detected	83
Hybrid	0	7	0	?	0					
Rudd					2					
Gudgeon	0	0	65		351					
Perch										1
Ruffe	0	3	100	?	9	1				2
3 spined stickleback	0	12	108	?	24	7				
10 spined stickleback	0	0	5		28					
Biomass (g)										
Bream	0	47.9	2164	?	12047		9174			17
Roach	526	8615	2944	?	12362		70			134
Hybrid	0	326	0	?	0					
Rudd					1					
Gudgeon	0	0	34		363					
Perch										1.3
Ruffe	0	19	87	?	63					1.6
3 spined stickleback	0	9.6	38	?	43					
10 spined stickleback	0	0	2.8		41					

Table 9.6 Fish removal data for Neatishead Arm north

Neatishead Arm north (NAn)	2003	2004	2005	2006	2007
	May	May	May	May	July
Removal effort (days)	1.5	0.5	1.5	1	
Number					
Bream	3	6		2	
Roach	1129	1263	4	1592	1013
Rudd		3	9	55	17
Ruffe	3				
Gudgeon			18		
3 spined stickleback	1		6	1	
10 spined stickleback	1		12		
Biomass (g)					
Bream		23	0	2	
Roach		2279	5	1592	1334
Rudd		4	27	55	168
Ruffe			0		
Gudgeon			40		
3 spined stickleback			11	1	
10 spined stickleback			18		

Table 9.7 Fish removal data for Turkey Broad east

Turkey Broad east (TBe)	2003	2004	2005	2006	2007
	May	May	Apr	May	July
Removal effort (days)	1.25	2	0.5	1	
Number					
Bream	0	2	0	2	
Roach	159	469	496	259	26
Hybrid	1	0	0		
Rudd	0	0	1	1	6
Ruffe	0	31	0	12	
Gudgeon	0	8	1	102	
3 spined stickleback	0	0	0		
10 spined stickleback	0	0	0		
Biomass (g)					
Bream		6	0	2	
Roach		1579	2016	528	55
Hybrid		0	0		
Rudd		0	1	1	75
Ruffe		169	0	46	
Gudgeon		22	2	220	
3 spined stickleback		0	0		
10 spined stickleback		0	0		