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Impacts of changes in sewage disposal on waterbirds wintering on the Northumbrian coast Final Report

Authors

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EXECUTIVE SUMMARY

Background

1. Over the last decade Northumbrian Water Ltd (NWL) has implemented a series of major improvements to the treatment and discharge of sewage at sites along the coast between Berwick-upon-Tweed and Saltburn (from Northumberland to Cleveland), so as to comply with the EC's Urban Waste Water Treatment Directive.

2. The impact of this directive on coastal waterbirds has raised concern as waste water discharges from outfalls can provide considerable supplies of food for bird species, either as directly edible matter or by artificially enhancing concentrations of invertebrate food through nutrient enrichment.

3. This report reviews the final results of an investigation, commissioned by NWL, begun in 1996/97 by the University of Durham and continued by the BTO from 2003/04, of the impacts of improvements to sewage discharges on waterbirds wintering on a 36 km stretch of the Northumbrian coast between the Coquet Estuary and St. Mary's Island (Figure 1.1). The area comprises extensive areas of rocky shore which are included in the Northumbria Coast Special Protection Area (SPA), which is designated for its importance for wintering Purple Sandpipers *Calidris maritima* and Turnstones *Arenaria interpres*. Between these rocky areas are the sandy (bathing) beaches of Druridge Bay, Cambois and South Blyth. The majority of improvements to sewage discharges in the area were completed by the end of the winter of 2000/01.

4. Earlier stable carbon isotope analyses indicated that a high percentage (up to 60%) of the Particulate Organic Matter (POM) in waters along this coast was derived from sewage. A repeat analysis of coastal water samples collected in September 2006 revealed significant declines in the percentage of POM attributable to sewage and thus likely the total POM available – see the separate annex to this report.

5. This report provides final analyses of the changes in the numbers of waterbirds wintering in the study area over the period 1997/98 to 2005/06. Additional analyses investigate whether changes in Turnstone numbers following the improvements to sewage treatment might be explained by changes in this species' survival rates or movements.

6. All waterbird species in the study area were included in counts, though analyses here are restricted to Purple Sandpiper and Turnstone. Analyses were carried out at two scales: firstly, for the whole coast from the Coquet Estuary to St. Mary's Island and secondly, for the Amble-Hauxley and Newbiggin areas alone. Two of the largest outfalls in the study area discharge in these areas; both discharges received improved treatment from late in the winter of 2000/01, thus potentially affecting the numbers of birds these areas could support through impacts on their food supplies. It was predicted that the impacts of the improvements to sewage discharges would be greatest at Amble-Hauxley and minimal at Newbiggin due to the differing positions of the outfalls relative to these headlands and prevailing currents.

Changes in Waterbird Numbers Following Improvements to Sewage Treatment

7. Across the study area as a whole, counts suggested that the changes to sewage treatment affected both of the species for which the SPA is designated in winter – Purple Sandpiper and Turnstone. These species showed declines following the winter of 2000/01 – which continued up to the final winter of study – having previously risen in number.

8. At a regional level, Wetland Bird Survey (WeBS) counts indicated that Purple Sandpipers were declining at least up to 2004/05, but that Turnstone numbers had stabilised after a decline

between the mid-1980s and mid-1990s. However, for neither species did it appear that changes in numbers in the study area were closely related to changes in regional numbers.

9. At the more local scale, numbers of birds changed, as predicted, according to the relative positions of outfalls. At Amble-Hauxley, where sewage discharges would have directly affected much of the intertidal zone, declines were apparent in the numbers of both study species. After a period of high stability, there was a clear decline (of 37%) in the numbers of Turnstone over the four years following the improvement to the Amble discharge, though a limited recovery in 2005/06. Purple Sandpiper numbers also fell here in 2003/04 and 2005/06, having risen prior to the improvement to the discharge.

10. In contrast, at Newbiggin where the sewage would have had less influence on intertidal communities due to the position of the outfall and thus where impacts of the improvements to the discharge were predicted to be minimal, no significant trends were apparent in the numbers of either species following the improvements.

Changes in Turnstone Survival Rates Following Improvements to Sewage Treatment

11. This study has been the first to use standard mark-recapture analyses to estimate annual survival rates for Turnstone and thus provides the most accurate estimates yet produced for the species. Analysis of data from resightings of colour-ringed Turnstone revealed that adult survival fell in 2002, from 0.769 (95% confidence limits = 0.723-0.809) to 0.611 (0.476-0.730), a rate similar to that of first-winter birds ($P = 0.0357$). No evidence was found to suggest that the decline in Turnstone numbers at Amble-Hauxley was the result of individuals increasingly using other sites in the area. Thus, it is probable that the drop in Turnstone numbers seen at Amble-Hauxley following the improvements to the Amble discharge was at least in part a consequence of a fall in adult survival.

Conclusions

12. By combining evidence from detailed count data and analyses of the movements and survival of a colour-marked population of a key species, this study has provided the best evidence to date that improvements to sewage discharges can impact invertebrate-feeding waterbird species. The changes to sewage treatment in the study area were coincident with declines in the numbers of both Purple Sandpiper and Turnstone, notably at Amble-Hauxley. For the latter species, analyses of the survival rates of colour-ringed birds suggested that the decline in numbers at that site was at least in part the result of an increase in adult mortality. The coincidence in the timing of the start of the species' declines (and the change in Turnstone survival) strongly suggests that these changes were a consequence of the improvements to discharges.

1. INTRODUCTION

Over the last decade Northumbrian Water Ltd (NWL) has implemented a series of major improvements to the treatment and discharge of sewage at sites along the coast between Berwick-upon-Tweed and Saltburn (from Northumberland to Cleveland) so as to comply with the EC's Urban Waste Water Treatment Directive (Directive 91/271/EEC and its Amending Directive 98/15/EEC) (Anon 1991, 1998). Improvements have involved secondary treatment[1], and in some cases, the installation of long offshore outfalls.

The impact of this directive on coastal waterbirds has raised concern as in many areas waste water discharges from outfalls can provide considerable supplies of food for bird species, either as directly edible matter or by artificially enhancing concentrations of invertebrate food (Green *et al.* 1990, Hill *et al.* 1993, Burton *et al.* 2002). For example, previous studies have highlighted the importance of outfalls in directly providing food for gulls (Ferns & Mudge 2000) and seaduck species such as Pochard *Aythya ferina*, Tufted Duck *A. fuligula*, Scaup *A. marila* and Goldeneye *Bucephala clangula* (e.g. Pounder 1976a, 1976b, Campbell 1978, 1984, Campbell *et al.* 1986, Campbell & Milne 1977). The changes in invertebrate and algal biomass found in the vicinity of outfalls are also likely to affect the densities of a number of other ducks and waders. Studies in Ireland (Fahy *et al.* 1975) and Holland (van Impe 1985), for example, have both linked increases in organic and nutrient inputs (from sewage, industrial and agricultural wastes) to increases in algae and invertebrates and thus to increased bird numbers.

The coast between Berwick and Saltburn comprises two Special Protection Areas (SPAs) designated under EC Directive 79/409 for their importance for birds. The Teesmouth and Cleveland Coast SPA is noted for its importance for breeding Little Terns *Sterna albifrons*, passage Sandwich Terns *S. sandvicensis* and Ringed Plover *Charadrius hiaticula*, and wintering Knot *Calidris canutus* and Redshank *Tringa totanus* and an assemblage of other wintering waterbirds. The Northumbria Coast SPA was designated for its internationally important populations of wintering Purple Sandpipers *Calidris maritima* and Turnstones *Arenaria interpres*, as well as the small, but nationally important population of Little Terns that breeds in the north of the region (Stroud *et al.* 2001). Both sites are also designated as Ramsar Sites under the Convention of Wetlands of International Importance.

This report reviews the final results of an investigation, commissioned by NWL and begun in 1996/97 by the University of Durham, of the impacts of improvements to sewage discharges on waterbirds wintering on a 36 km stretch of the Northumbrian coast between the Coquet Estuary and St. Mary's Island (Figure 1.1). The area comprises extensive wave-cut platforms and rocky headlands at Amble-Hauxley, Cresswell, Newbiggin, North Blyth and Seaton Sluice-St. Mary's Island, which are included in the Northumbria Coast SPA. These areas support high proportions of the SPA's Purple Sandpipers and Turnstones (Anthony 1999). Between these rocky areas are the sandy (bathing) beaches of Druridge Bay, Cambois and South Blyth. Mining at Lynemouth has left the beach there covered with spoil. The Coquet, Wansbeck and Blyth rivers all form small estuaries; there are also harbours at Blyth and at Amble on the Coquet. The Northumbria Coast SPA supported a peak mean of 763 Purple Sandpipers and 1,456 Turnstones between 1991/92 and 1995/96 (1.5% of the Eastern Atlantic wintering population and 2.1% of the Western Palearctic wintering population respectively: Stroud *et al.* 2001). The British wintering populations of Purple Sandpipers and Turnstones are currently estimated to be 17,530 and 49,550 respectively (Rehfisch *et al.* 2003a).

A number of improvements have been made to sewage discharges within the study area:

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[1] 'Primary treatment' entails treatment of waste water by a physical and/or chemical process involving settlement of suspended solids, or other processes in which the BOD5 (five-day biochemical oxygen demand) of the incoming waste water is reduced by at least 30-40% before discharge and the total suspended solids of the incoming waste water are reduced by at least 50-65%. 'Secondary treatment' generally involves biological treatment with a secondary settlement or equivalent process and removes 65-95% of the BOD and 60-90% of the suspended solids in the waste water (Anon 1999).

- At Amble, a new sewage works providing primary and secondary treatment was completed in January 2001. There was no change in the location of this outfall, between the Amble foreshore and Coquet Island (see Figure 1.1). This discharge is one of the three largest in the study area with a human population equivalent value of 24,143 (Environment Agency 2000).

- At Newbiggin, primary and secondary treatment was also completed in January 2001. Sewage here is discharged offshore by a long outfall completed in 1993. The discharge has a human population equivalent value of 36,231 (Environment Agency 2000).

- A new 1 km outfall was completed at Cambois in September 1997 and sewage with secondary treatment diverted through it from 2001. The discharge has a human population equivalent value of 40,607 (Environment Agency 2000).

- Sewage discharged into the Blyth Estuary and at Blyth Link House on South Beach was also diverted to the Blyth Sewage Treatment Works in January 2001 to receive secondary treatment prior to discharge into the river.

- Other changes have also affected smaller discharges in the area. Sewage previously discharged at Seaton Sluice was diverted to the Howdon Sewage Treatment Works on the Tyne in January 2001. Sewage discharged at Cresswell was diverted to the Lynemouth Sewage Treatment Works in April 2003. Secondary treatment was also provided in March 2005 for a small discharge at Hadston at the north of Druridge Bay.

Earlier stable carbon isotope analyses indicated that a high percentage (up to 60%) of the Particulate Organic Matter (POM) in waters along this coast was derived from sewage (Eaton 2001). Improvements to sewage treatment might be expected to have a noticeable impact on nutrient loading and total POM (Savage & Elmgren 2004, Costanzo *et al.* 2005) and thus primary productivity of plankton and macroalgae (Hamer *et al.* 2002). Repeated stable isotope analyses of coastal water samples collected in September 2006 revealed significant declines in the percentage of POM attributable to sewage and thus likely the total POM available – see the separate annex to this report.

Changes in primary productivity resulting from changes in nutrient loading are likely to in turn affect mussels *Mytilus edulis*, which are planktoniverous, and other invertebrate species. Mussels are important prey items for Purple Sandpipers (Feare 1966, Summers *et al.* 1990) and Eaton (2001) found that Purple Sandpiper densities on the Northumbrian coast were correlated to the densities of small mussels. Turnstones are more omnivorous (Cramp & Simmons 1983, Whitfield 1990), but as well as being affected by loss of mussels, could also be affected if the improvements to sewage discharges impact the invertebrate populations that feed on macroalgae. Alternatively, reductions in the growth of *Enteromorpha* spp. and *Ulva* spp., both of which proliferate in nutrient-enriched waters, might increase the area of foraging substrate available by Turnstones and other waders (Cabral *et al.* 1999, Lopes *et al.* 2000, Lewis & Kelly 2001). Eaton (2001) previously found that the area of suitable foraging substrate was a good predictor of Turnstone densities.

Any loss of food resources is likely to affect the numbers of birds that an area can support. Whether there is an actual reduction in bird numbers, though, will depend on whether the area was close to carrying capacity prior to the change (Goss-Custard 1985, Goss-Custard *et al.* 2002). If it was, either the increased competition for food resources will force birds to move to new feeding grounds or, if such areas do not exist or are also close to carrying capacity or if birds fail to relocate, lead to reduced survival; numbers may also decline due to reduced recruitment of first-winter birds into the local population. Impacts may also vary between species. Waders such as Knot, Dunlin *Calidris alpina* and Sanderling *Calidris alba* which may regularly move between sites to exploit varying food resources (Evans 1981, Myers 1984, Symonds & Langslow 1986, Symonds *et al.* 1984, Roberts 1991, Rehfisch *et al.* 1996, 2003b) may be less affected by reduced food supplies than more site-faithful species, such as Redshank, Turnstone and Purple Sandpiper. Both in the Northumbrian study area and elsewhere, Turnstones and Purple Sandpipers typically return to the same stretches of shore each year and remain faithful to them through the winter (Metcalfe & Furness 1985, Rehfisch *et al.* 1996, 2003b, Burton & Evans 1997, Dierschke 1998, Burton 2000, Eaton 2001).

This report provides final analyses of the changes in the numbers and survival of waterbirds wintering in the area over the period 1997/98 to 2005/06 and updates earlier results given in Burton *et al.* (2004) and Burton & Goddard (2005). Companion studies have also been undertaken by the University of Durham to look at the impacts of the improvements to sewage discharges on the breeding performance and numbers of terns in the area (Hamer *et al.* 2002, Booth & Hamer 2003).

The report has two main aims:

1. To determine whether the numbers of Purple Sandpiper and Turnstone wintering in the study area changed following the improvements to discharges.

2. To determine whether changes in Turnstone numbers might be explained by changes in this species' survival rates and / or movements.

Analyses of changes in wintering waterbird numbers were carried out at two scales: firstly, for the whole coast from the Coquet Estuary to St. Mary's Island and secondly, for the Amble-Hauxley and Newbiggin areas alone. Survival and movement analyses were undertaken solely for Turnstone caught in the Amble-Hauxley area.

As detailed above, two of the three largest outfalls in the study area discharge at Amble and Newbiggin; both discharges received improved treatment from late in the winter of 2000/01. It was predicted that the impacts of these improvements on waterbirds would vary at a local scale according to the positions of outfalls. On the Northumbrian coast, there is a net southward flow of currents. The Amble sewage outfall, which discharges at the north of the Amble-Hauxley headland, would thus have been an important influence on intertidal communities in that area. In contrast, sewage is likely to have been much less of an influence for the intertidal communities of Newbiggin headland, as the outfall there discharges to the south of that headland and further offshore. Consequently, therefore, we predicted that the impacts of the improvements to sewage discharges would be most apparent at Amble-Hauxley and minimal at Newbiggin.

2. METHODS

2.1 Waterbird Counts

2.1.1 Field methodology

Counts of waterbirds on the coast between the Coquet Estuary and St. Mary's Island were begun by the University of Durham in the winter of 1996/97. The study area was split into sections (see Figure 1.1) and the whole coast surveyed twice a month, with the aim that each section should be covered once over the low tide period (i.e. < 3.7 m OD, the median value between mean high and low tide levels: Eaton 2001) and once over the high tide period. Count sections included the lower Coquet, Wansbeck and Blyth estuaries (and the North Blyth Staithes roost site within the latter estuary). This programme of counts was continued until March 2003, with only occasional gaps during the summer months (Eaton 1997, 1998, Hamer *et al.* 2002, Fuller 2003a).

Monitoring was resumed in October 2003 by the British Trust for Ornithology (BTO), with counts of every section at high and low tide each half-month until the end of March 2004. Counts were continued following this regime in the winters of 2004/05 and 2005/06.

2.1.2 Analyses of count data

Analyses of waterbird count data were carried out for the whole coast from the Coquet Estuary to St. Mary's Island and also separately for the Amble-Hauxley (count sections A01-A08) and Newbiggin (sections A14-A19) areas. As detailed above, it was predicted that the impacts of the improvements to sewage discharges would be greatest at Amble-Hauxley and minimal at Newbiggin due to the differing positions of the outfalls relative to these headlands and prevailing currents.

Although all waterbird species in the study area were included in counts, analyses are here restricted to Purple Sandpiper and Turnstone, the two wintering species for which the Northumbria Coast SPA is designated. Due to the potential for poor quality counts in the first winter of study (M. Eaton pers. comm.), data from that winter have been excluded. Counts of sections with no intertidal area (e.g. Hauxley Haven, Druridge Pools and Cresswell Pond) were also excluded from all analyses, though St. Mary's Wetland, an important high tide roost site, was included as part of the southernmost count section.

For each species, generalized linear models (GLMs) (McCullagh & Nelder 1989; SAS Institute Inc. 2002-2003) were used to relate the number of birds on each count to the winter, month (October to March), state of tide (high or low) and count section (to take into account variation in section area and habitat) as well as the interaction between the state of tide and count section.

Models assumed a Poisson distribution for the number of feeding birds and specified a log link function. Month, state of tide, count section and winter were all treated as class variables. The problem of overdispersion caused by a combination of a large number of zero counts with several very high counts, typical of flocking species, was addressed by the application of a scale factor estimated from the square root of the Pearson's Chi-squared statistic divided by its degrees of freedom. Aside from winter, only those variables that were significant in explaining the variation in densities were retained in the final models.

Model results were used to produce and graph indices relating how the numbers of birds varied each winter relative to 2005/06. The fitted models were also used to calculate, for each species and site, the mean recorded number of birds present each winter (i.e. October to March) averaged between high and low tide. These figures are plotted on the same graphs as the indices so as to understand better how the actual numbers of each species changed over the study period and following improvements to discharges.

Similar figures were also produced from analyses using just low tide or high tide counts. As waders predominantly feed during low tide, counts from this period may better indicate how bird numbers in an area may have changed in response to changes in food supply. The majority of waders roost over the high tide period, and the flocks counted at this time may include individuals that have flown in from neighbouring sites. Furthermore, those food supplies that are utilised over high tide (notably on neap tides), such the invertebrate populations associated with tidal wrack beds (often of macroalgae), might be expected to be less impacted by changes in the nutrient supply resulting from the improvements to discharges than food supplies found in the intertidal zone. Nevertheless, as sometimes it may be difficult to fully census local bird populations present on rocky shores at low tide (Eaton 2001), high tide counts may give a better understanding of the total number of birds using some areas.

It should also be noted that as the figures plotted on these graphs are averages of the numbers recorded across the winter, they may underestimate the total numbers of some species. For example, Purple Sandpipers which forage on the Amble-Hauxley shore often roost over high tide on Coquet Island; this species' roost site at North Blyth may also be difficult to survey (though all visible parts of the piers where the species roosts were consistently surveyed across years). Also, although some species' numbers are relatively stable over the defined winter period (e.g. Turnstone), those of other species show monthly variation. The numbers of Purple Sandpipers in the study area, for example, rise to a peak in mid-winter following the arrival of a second breeding population (thought to be from Canada: Summers 1994). The total numbers of Purple Sandpipers using the study area will thus be underestimated from the mean winter value.

Although they are typically faithful to the same areas within and between winters (Metcalfe & Furness 1985, Rehfisch *et al.* 2003b, Burton & Evans 1997, Eaton 2001), the local distribution of Turnstones may be affected by the availability and productivity of tidal wrack (Fuller 2003b). Such wrack beds occur often within the study area within winter. However, no data was or had previously been collected to determine whether their availability had changed over time and so perhaps influenced the distribution and numbers of Turnstones (or other species).

The mean winter numbers in the study area are compared statistically with regional data taken from the Wetland Bird Survey (WeBS) Alerts report for winters up to 2004/05 to determine whether changes in the study area might have been due to wider trends rather than the effects of changes to sewage discharges (Maclean & Austin 2006). In this, trends in WeBS data are presented for an area corresponding to the Environment Agency's North-East Region. Mean numbers (in WeBS recording areas within the region, which include the majority of the study area) are calculated for the three month period from December to February, having first taken into account missing counts in the dataset. Regional totals for the two species for the 25-year period from 1979/80 to 2004/05 are presented graphically, together with the proportions of these totals held by the study area (as estimated from the current study) between 1997/98 and 2004/05. GLMs (with logit link functions and binomial errors) were used to determine whether these proportions showed any trend with time.

Comparison is also made with the changes reported between two surveys of the country's non-estuarine coast in 1984/85 and 1997/98 (Rehfisch *et al.* 2003c), for which the Northumbrian coast counts had also been supported by NWL.

It was not possible to determine whether changes in numbers reflected variation in annual breeding success, as data on the proportions of juveniles in the local populations had not previously been collected and broader information on the annual breeding success of these species, which mostly breed in the Arctic, is limited.

2.2 Analyses of Data from Colour-ringed Turnstones

Data from colour-ringed Turnstone were used to investigate the mechanisms underpinning any changes seen in the species' numbers following the improvements to discharges in the study area. Reduced food supplies consequent of the improvements to discharges may have impacted numbers by i. reducing the survival rates of birds; ii. causing emigration of birds out of the study area or iii. reducing the recruitment of first-winter birds into the local population. Insufficient data were collected to investigate changes in recruitment and thus analyses focus on changes in survival and movements.

2.2.1 Turnstone survival analyses

Between the winters of 1996/97 and 2000/01, 226 Turnstone were caught and individually colour-ringed on the Northumbrian coast by Mark Eaton, Rich Fuller and colleagues from the University of Durham. The information collated from the marking and subsequent resighting of these birds (collected during the course of monitoring) has provided an ideal dataset to investigate temporal and age-related variations in the survival rate of this species and so determine whether survival might explain any changes observed in Turnstone numbers in the study area.

Turnstone were caught within six different parts of the study area: Amble, Hauxley, Cresswell, Newbiggin, Blyth and St. Mary's Island. Birds were caught by cannon-netting, with the exception of one individual caught by dazzling on the night of 11 February 1997 at Blyth. Each bird was aged according to its plumage characteristics (Prater *et al.* 1977) as either adult or first-year and fitted with a metal BTO ring and an unique combination of colour-rings so that they could be subsequently identified in the field. Of the 226 birds colour-ringed, 33 were classed as first-winter birds when marked, 98 as adults and 95 unaged (Table 2.2.1.1).

Estimates of annual survival rates and recapture probabilities of Turnstone were calculated using mark-recapture methods. Data were only included from birds known to be either in their first-winter or adults, i.e. in their second-winter or older, when caught or resighted. This allowed calculation of survival rates for the period 1997/98 to 2005/06.

Following the initial work undertaken by Burton *et al.* (2004), analyses were restricted to estimating the survival rates of those Turnstone originally caught and ringed at Amble-Hauxley (though retaining resighting data from the whole study area). Comparatively few Turnstone were caught at any one other site (Table 2.2.1.1). Analyses aimed to determine whether survival rates differed between adults and first-winter birds and, for adults, following the completion of the majority of improvements to sewage discharges late in the winter of 2000/01.

To determine the survival rates of Turnstone, the study area was searched extensively for colour-ringed birds during regular counts. The use of data from resightings of colour-ringed birds is often preferable to using data from the recapture of ringed birds, as it usually provides higher 'resighting' (i.e. recapture) rates (see also Sandercock 2003, Bearhop *et al.* 2003).

Ring-resighting data were analysed using Program MARK Version 4.1 (White & Burnham 1999) in order to estimate annual survival rates 'ϕ' (i.e. the proportion of birds surviving each year) and resighting probabilities 'p'. The validity of the standard Cormack-Jolly-Seber (CJS) model (Lebreton *et al.* 1992, Seber 1982) in which ϕ and p vary fully with time depends upon a number of assumptions being upheld, notably the equal "catchability" (a bird was considered caught if it was resighted) of each marked individual. By only using data from birds caught or resighted between October and March (thus matching the analyses of count data) we restricted the numbers of passage birds that would have been included in our analyses. Inequality in catchability between birds and years was also reduced by the regular surveying of the entire study area and by excluding sightings of birds from outside the study area. (Some individuals were observed on Coquet Island and at Boulmer between 1996/97 and 2000/01, but these areas were not surveyed in latter years.) Goodness-of-fit tests (provided by the U-CARE Version 2.2 software – Choquet *et al.* 2005) indicated that model assumptions were not violated (Table 2.2.1.2). Using Program MARK, we also estimated an overdispersion parameter for the data ($\hat{c}$) to adjust the final selected model (following White & Burnham 1999).

A combination of likelihood ratio tests (LRTs) and Akaike's Information Criterion (AIC), adjusted for overdispersion and sample size ($QAIC_c$: Burnham & Anderson 1998), was used to select the model that best described the data (typically that with the lowest $QAIC_c$ value). Different models evaluated whether resighting rates p were constant or varied fully with time and whether survival rates ϕ were constant, differed between adults and first-year birds, varied fully with time or (for adults) differed in the first full year after improvements to the Amble discharge, i.e. 2002.

The estimated values for 'ϕ' will underestimate true survival if birds moved away from the study area (Sandercock 2003). The movements of colour-ringed Turnstone from Amble-Hauxley were thus examined to determine whether these birds showed any evidence of a change in use of other parts of the study area over the course of the study (see below). The impacts of changes in 'apparent survival' on the numbers of Turnstone recorded in the Amble-Hauxley area are discussed.

2.2.2 Movements of colour-ringed Turnstones

Sightings of colour-ringed Turnstone were also examined to investigate whether there was any change in the winter ranging behaviour of the species over the course of the study. For those Turnstone originally colour-ringed at Amble-Hauxley, a graphical comparison was made of the proportions of birds seen at other sites within the study area each winter (October to March). A GLM, with a logit link function and binomial errors, was used to determine whether this proportion showed any trend with time or differed between winters before and after the completion of the improvements to the Amble discharge. By doing this, we aimed to determine whether any changes in the numbers or apparent survival of Turnstones at this site following the improvement to the discharge might have been the result of an exodus of birds.

3. RESULTS

3.1 Changes in Waterbird Numbers Following Improvements to Sewage Treatment

The significance of factors considered in modelling numbers of Purple Sandpipers and Turnstones (using data from both high and low tide counts) is summarized in Table 3.1.1. Numbers of Purple Sandpipers were significantly related to all factors considered in the GLMs for the whole study area, Amble-Hauxley and Newbiggin with the exception state of tide, though the interactions between count section and state of tide were significant in each case. Numbers of Turnstones were significantly related to all factors in the models for the whole study site and for Amble-Hauxley, though not to month in the model for Newbiggin.

3.1.1 Purple Sandpiper *Calidris maritima*

Mean winter (October to March) numbers of Purple Sandpiper in the study area as a whole rose between 1997/98 and 1999/2000 – peaking at an estimated 149 birds – but fell in 2002/03, two winters after the completion of the majority of improvements to sewage discharges in the area and again in 2005/06 (Figure 3.1.1.1a). Data from low tide counts alone suggest an increase in numbers in 1999/2000, though also show the decreases in 2002/03 and 2005/06 (Figure 3.1.1.1b). In contrast, data from high tide counts suggest falls in numbers in 1999/2000 and between 2003/04 and 2005/06 (Figure 3.1.1.1c). It should be noted that standard errors using high tide counts alone were greater than when using low tide counts, perhaps due to the difficulties in counting at high tide roosts, e.g. the important roost on the piers at Blyth Harbour. Overall mean numbers of Purple Sandpipers in the study area in the five winters following the improvements were significantly lower than those in the four preceding winters ($F_{1,3624}$ = 15.11, P < 0.0001).

At Amble-Hauxley, trends reflected those over the study area as a whole, mean numbers increasing to a peak in 2000/01, but declining afterwards. There were notable declines in numbers in 2003/04 and again in 2005/06 (Figure 3.1.1.2a). A similar pattern was observed from low tide counts alone (Figure 3.1.1.2b), though was less clear using high tide count data due to a lack of high tide surveys on this stretch of coast between 1999/2000 and 2001/02 (Figure 3.1.1.2c). Mean numbers of Purple Sandpipers at Amble-Hauxley were significantly lower following the improvement to the Amble discharge than beforehand ($F_{1,927}$ = 4.55, P = 0.0331).

At Newbiggin, there was considerable fluctuation in mean numbers and no significant difference between winters before and after the completion of the improvement to the Newbiggin discharge (Figure 3.1.1.3a; $F_{1,737}$ = 0.70, P = 0.4021). Very similar results were seen from low tide counts (Figure 3.1.1.3b) and high tide counts alone (Figure 3.1.1.3c).

The regional trend in Purple Sandpiper numbers, as estimated by WeBS count data (Maclean & Austin 2006) is shown in Figure 3.1.1.4. These data suggest considerable fluctuations in the numbers in the region – however, this may be because the index is based on only three counts during the winter and for some sections, numbers will have been estimated due to missing counts. Nevertheless, it is clear that there has been a decline in regional numbers of the species since the late 1980s.

Changes in the mean winter numbers of Purple Sandpipers recorded in the study area appeared to be poorly related to changes in mean regional numbers. The proportion of region's birds (as estimated from WeBS counts) held by the study area (as estimated from the present study) actually increased over the period 1997/98 to 2004/05 (χ^2_1 = 43.90, P < 0.0001) suggesting that there were greater declines outside of the area studied.

Nationally, there was a decline in Purple Sandpiper numbers on non-estuarine coasts between 1984/85 and 1997/98 (Rehfisch *et al.* 2003c).

3.1.2 Turnstone *Arenaria interpres*

Mean winter numbers of Turnstone in the study area as a whole rose to an estimated peak of 427 birds in 2000/01 just prior to the completion of the majority of improvements to sewage discharges in the area (Figure 3.1.2.1a). As with Purple Sandpipers, mean numbers of Turnstone in the study area fell thereafter and in the five winters following these improvements were significantly lower than those in the four preceding winters ($F_{1,4162}$ = 14.70, P < 0.0001). Both low tide (Figure 3.1.2.1b) and high tide counts alone (Figure 3.1.2.1c) followed this pattern, though the former suggest some stability in numbers after 2003/04.

At Amble-Hauxley, mean Turnstone numbers were very stable over the four winters prior to the improvement to the Amble discharge, but declined in 2001/02 and in 2003/04 (Figure 3.1.2.2a). A similar pattern was observed from low tide counts alone (Figure 3.1.2.2b), though again this pattern was less apparent using high tide count data due to a lack of high tide surveys on this stretch of coast between 1999/2000 and 2001/02 (Figure 3.1.2.2c). Mean numbers in 2001/02 to 2005/06, following the improvement to the Amble discharge, were significantly lower than those in the four preceding winters ($F_{1,1061}$ = 21.22, P < 0.0001).

At Newbiggin, mean Turnstone numbers were more variable between winters with no significant difference between years ($F_{8,735}$ = 1.35, P = 0.2153) and thus between the four winters prior to the improvement to the discharge here and the five following winters (Figure 3.1.2.3a; $F_{1,742}$ = 0.03, P = 0.8731). Very similar results were seen from low tide counts (Figure 3.1.2.3b) and high tide counts alone (Figure 3.1.2.3c).

The regional trend in Turnstone numbers, as shown by WeBS count data (Maclean & Austin 2006) is shown in Figure 3.1.2.4. As with Purple Sandpiper, there was a decline in the numbers of Turnstone in the region from the late 1980s, although numbers appear to have stabilised since the winter of 1996/97.

Changes in the mean winter numbers of Turnstones recorded in the study area also appeared to be poorly related to changes in mean regional numbers. The proportion of region's birds (as estimated from WeBS counts) held by the study area (as estimated from the present study) tracked changes at the site-level (Fig. 3.1.2.4), increasing up until 2000/01, but declining following the majority of improvements to sewage discharges. There was an overall decline in the proportion of the region's birds held by the study area over the period from 1997/98 to 2004/05 (χ^2_1 = 5.34, P = 0.0209).

Nationally, the wintering numbers of Turnstone declined on non-estuarine coasts between 1984/85 and 1997/98 (Rehfisch *et al.* 2003c).

3.1.3 Other waterbird species

A total of 60 waterbird species were recorded during BTO counts over the winters of 2003/04 to 2005/06, including 31 species of wildfowl and 16 species of wader (Appendix 1).

3.2 Analyses of Data from Colour-ringed Turnstones

3.2.1 Changes in Turnstone survival rates following improvements to sewage treatment

Models describing the survival rates and resighting probabilities of Turnstone caught and colour-ringed at Amble-Hauxley are described in Table 3.2.1.1. Resighting probabilities were generally high, equal to 0.772 (95% confidence limits = 0.721-0.817) in the base model in which both ϕ and p were assumed to be constant over time. The same model suggested that apparent survival averaged 0.739 (95% confidence limits = 0.700-0.774) over the study period.

The best fit model (i.e. that with the lowest $QAIC_c$ value) indicated that resighting probabilities varied significantly between years. Values ranged from 0.618 to 1.000 over six of the seven winters, but dropped to 0.148 in the winter of 2002/03.

In this final model, two separate survival rates were estimated. Annual survival of adult Turnstones was estimated at a constant 0.769 (95% confidence limits = 0.723-0.809) over the years 1998-2001 and 2003-2005 but at 0.611 (0.476-0.730) in 2002 – a level equal to that of first-winter birds. Despite the low resighting rate in 2002/03, a likelihood ratio test confirmed that the estimated survival rate of adults in 2002 and first-winter birds was significantly lower than that estimated for adults in other years (LRT: $\chi^2_1 = 4.41$, $P = 0.0357$).

3.2.2 Movements of colour-ringed Turnstones

Turnstone are typically site-faithful during winter and, among those colour-ringed at Amble-Hauxley, only between 5 and 33% were recorded elsewhere in the study area in any one winter (Figure 3.2.2.1). Despite this, some individuals were recorded at St. Mary's Island over 30 km to the south.

The proportion of Turnstone colour-ringed at Amble-Hauxley that used other sites showed no trend across winters ($\chi^2_1 = 2.97$, $P = 0.0846$) and also did not differ between winters before and after the improvement to the Amble discharge ($\chi^2_1 = 1.02$, $P = 0.3120$).

4. DISCUSSION

Changes in waterbird numbers in relation to sewage improvements and regional trends

The influence of coastal sewage discharges on intertidal ecosystems is dependent on a number of factors in addition to the size of discharge and the treatment the sewage has received, notably the distance of outfalls from shore and currents. On the Northumbrian coast, there is a net southward flow of currents. Thus the sewage outfall which discharges at the north of the Amble-Hauxley area would have had a larger effect on intertidal communities than that at Newbiggin, which discharges to the south of this headland and further offshore. The impact of discharges on communities across the study area would thus be predicted to be highly variable and the impacts of recent improvements correspondingly localised.

Across the study area as a whole, there was evidence that the changes to sewage treatment affected both of the two species for which the SPA is designated in winter – Purple Sandpiper and Turnstone. Both these species showed declines following the winter of 2000/01 – which continued up to the final winter of study – having previously risen in number.

Purple Sandpiper and Turnstone both declined on non-estuarine coasts of Great Britain between surveys in 1984/85 and 1997/98 (Rehfisch *et al.* 2003c), in part possibly due to climate change (Rehfisch *et al.* 2004). At a regional level, WeBS counts indicated that Purple Sandpipers were still declining up to 2004/05, but that Turnstone numbers had stabilised after a decline between the mid-1980s and mid-1990s (Maclean & Austin 2006). However, for neither species did it appear that changes in numbers in the study area were closely related to changes in regional numbers. Indeed, the fact that both Turnstone and Purple Sandpiper numbers were increasing in the study area until 2000/01, suggests that local factors were important in their recent declines.

At the more local scale, numbers of birds changed as predicted according to the relative positions of outfalls. At Amble-Hauxley, where sewage discharges would have directly affected much of the intertidal zone, declines were apparent in the numbers of both study species. After a period of high stability, there was a clear decline (of 37%) in the numbers of Turnstone over the four years following the improvement to the Amble discharge, though limited recovery in 2005/06. Purple Sandpiper numbers, having risen prior to the improvement to the discharge, also fell over subsequent winters and in 2005/06 were only 50% of those recorded in 2000/01.

In contrast, at Newbiggin where the sewage would have had less influence on intertidal communities due to the position of the outfall and thus where impacts of the improvements to the discharge were predicted to be minimal, no significant trends were apparent in the numbers of either species following the improvements.

The immediate decline in Turnstone numbers at Amble-Hauxley following improvements to the discharge at that site is in contrast to an earlier study at Hartlepool. There, Eaton (2000) reported no change in Turnstone numbers between the winters of 1991/92 to 1993/94 and 1999/2000, the second winter following the diversion of discharges away from important feeding grounds on Hartlepool Headland to a new offshore outfall to the south. Purple Sandpiper numbers had fallen between these periods, though this decline followed an earlier local trend and reflected the national population trend for the species over this period (Rehfisch *et al.* 2003c). Since Eaton's study, however, numbers of Turnstone at Hartlepool have declined, suggesting that the impacts on birds of the changes in sewage disposal there were delayed (Burton *et al.* 2005). The possible reasons for the difference in the timing of the declines seen at Hartlepool and Amble-Hauxley are unclear.

Changes in Turnstone numbers in relation to changes in their survival rates and movements

This study has been the first to use standard mark-recapture analyses to estimate annual survival rates (and resighting probabilities) for Turnstone and thus provides the most accurate estimates yet produced for the species. The average survival rate of 0.739 or 74% calculated from the base model for adult and first-winter Turnstone caught at Amble-Hauxley is lower than many rates previously reported for the species, though not dissimilar to those for many other waders (Evans & Pienkowski 1984, Burton 2000, Sandercock 2003). Metcalfe & Furness (1985), for example, reported a minimum annual survival rate of 86% for a local population of colour-ringed adult Turnstone wintering in south-western Scotland and Pearce-Higgins (2001) a similar rate for adults and first-winter birds wintering in north Wales. Evans & Pienkowski (1984) recorded a return rate of 85% for colour-ringed Turnstone on the coast south of Teesmouth and Bergmann (1946) a minimum annual survival rate of 78% in a study of breeding Turnstone in Finland. In contrast, Burton & Evans (1997) reported a minimum annual survival rate of 71-72% for colour-ringed adult and first-winter Turnstone wintering at Hartlepool.

Although survival analyses only included data for birds originally caught and ringed in the study area, four other Turnstones originally ringed in the Hartlepool / Teesmouth area (between 1986 and 1993) were seen in the study area in 2001, possibly due to the changes in sewage disposal there (Burton *et al.* 2005). Despite generally high site-fidelity in the species (Metcalfe & Furness 1985, Burton & Evans 1997) some individuals may thus move wintering area between years – indeed one Turnstone colour-ringed in Northumbrian was also seen at Hartlepool only to subsequently return to its original wintering area. It is feasible, therefore, that the apparent decline in survival in 2002 could have been in part due to movements of birds away from the area following the improvements to the Amble discharge and that true survival was underestimated. The analysis of Turnstone movements, though, suggests that this is unlikely. There was no change in the proportion of Amble-Hauxley ringed birds that were seen at other sites in the study area following the improvements to the discharge and thus no reason to suppose that individuals moved to other sites outwith the study area.

Without a movement away from the area, the drop in Turnstone numbers seen at Amble-Hauxley following the improvements to the Amble discharge would have been due to either reduced recruitment of first-winter birds into the local population (see, for example, Summers *et al.* 2005) or increased mortality, as indicated for adults by the survival analyses. The results of these analyses strongly suggest that the improved treatment of sewage and subsequent loss of particulate organic matter to coastal waters had an impact on food supplies thus increasing competition between individuals and impacting their survival. Following the decline in local densities, competition would have lessened, allowing survival rates to return to their previous levels. Numbers of Turnstone at Amble-Hauxley were similar in 2004/05 to those in the previous winter and then increased in winter 2005/06. In contrast, numbers of Purple Sandpiper at Amble-Hauxley fell again between the last two winters of study and it is not possible to affirm that they have now reached a new equilibrium.

Other studies of the impacts of changes in sewage disposal on waterbirds

Reductions in the food discharged directly from waste water outfalls have previously been linked to declines in gull and duck species. On the Tyne Estuary, where five of six species of gull formerly used untreated sewage as a food source (Fitzgerald & Coulson 1973), declines of 93% and 91% were recorded in the numbers of Common Gulls *Larus canus* and Great Black-backed Gulls *L. marinus* respectively between 1969/70 and 1993/94 following improved treatment and an 86% decrease in the volume of untreated waste discharged into the river (Raven & Coulson 2001). A study in New Zealand, similarly found that following improved treatment of waste discharges into Wellington Harbour, numbers of Dominican Gulls *L. dominicanus* declined, whilst the population of Red-billed Gulls *L. novaehollandiae* became concentrated around the one outfall where sewage remained untreated (Robertson 1992). In Scotland, large declines have been noted in the local populations of several seaduck species, including Scaup, Goldeneye and Tufted Duck, following the introduction of primary treatment and the creation of new deep water outfalls (Campbell 1984, Barrett & Barrett 1985). For example, following the introduction of a primary treatment plant on the Firth of Forth, Campbell (1984) found that numbers of Scaup and Goldeneye between Leith and Levenhall fell from respective peaks of 10,280 and 2,334 in 1975/76 to 675 and 608 respectively in 1979/80 (see also Bryant 1987 for a summary).

In contrast, evidence that improvements to discharges can impact waterbird species that feed predominantly on invertebrates has previously been limited. Aside from the studies at Hartlepool discussed above, there is some useful evidence from the Clyde Estuary. Furness *et al.* (1986) reported that between 1972-77 and 1978-85 numbers of Oystercatcher *Haematopus ostralegus*, Lapwing *Vanellus vanellus*, Dunlin and Redshank on the estuary declined by 19%, 59%, 85% and 60% respectively, these declines by far exceeding national changes. Numbers of Shelduck and Pintail also declined significantly, though, in contrast, the numbers of Curlew fell by only 3% - in line with the national trend. Limited invertebrate data suggested that these declines might have resulted from a shortage of food, particularly of *Corophium volutator* and *Nereis diversicolor*, due either to the reduced nutrient enrichment of the mudflats or increased consumption of these invertebrates by fish.

5. CONCLUSIONS

In conclusion, the study has provided evidence that the changes to sewage treatment in the study area have affected both Purple Sandpiper and Turnstone – the two wintering species for which the Northumbria Coast SPA is designated.

- Both Purple Sandpiper and Turnstone declined in number across the study area following 2000/01 and the completion of the majority of improvements to sewage discharges, following earlier increases. For neither species did changes in numbers in the study area appear to be closely related to changes in regional numbers.

- At the more local scale, numbers of birds changed, as predicted, according to the relative positions of outfalls. Changes were most apparent at Amble-Hauxley where due to currents and the position of the outfall, sewage would have directly affected much of the intertidal zone. Following improvements to the Amble discharge, Turnstone numbers fell by 37% after a period of stability and Purple Sandpiper numbers by 50% after an earlier increase. The survival rate of adult Turnstone also fell in 2002, the first full year after improvements, suggesting a mechanism for the decline.

- At Newbiggin, where sewage would have had less influence on intertidal communities due to the position of the outfall and thus where impacts of the improvements to the discharge were predicted to be minimal, no significant trends were apparent in the numbers of either species following improvements.

It should be noted that these results are essentially correlative and, as it was not possible to conclude that invertebrates had not been affected by the changes to sewage discharges due to the limited quantity of data previously collected (see Burton *et al.* 2004), no direct causal link between the changes to sewage discharges and changes in waterbird numbers can be proven. However, the coincidence in timing of the start of the species' declines (and the change in Turnstone survival) strongly indicates that these changes were linked to the changes to sewage treatment.

Previous studies of the possible impacts of improvements to sewage discharges on invertebrate-feeding waterbird species have been largely based on (relatively infrequent) count data. By combining evidence from a long run of counts undertaken at an appropriate spatial scale and analyses of the movements and survival of a colour-marked population of a key species, this study has provided the best evidence to date that such impacts can occur.

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Site	1996/97			1997/98			1998/99			1999/2000			2000/01			Total
	A	1	U	A	1	U	A	1	U	A	1	U	A	1	U	
Amble	0	0	30	34	1	0	33	18	10	0	0	0	4	7	1	138
Hauxley	0	0	31	3	2	0	0	0	0	0	0	0	0	0	0	36
Cresswell	0	0	0	0	1	17	0	0	2	0	0	0	0	0	0	20
Newbiggin	0	0	0	0	0	0	0	0	0	0	0	0	3	2	0	5
N Blyth	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1
St. Mary’s	0	0	0	0	0	0	0	0	0	21	2	3	0	0	0	26
Total	0	0	62	37	4	17	33	18	12	21	2	3	7	9	1	226

Table 2.2.1.1 Numbers of Turnstone ringed at each site and in each year. A = adult; 1 = first-winter; U = unaged.

Test	First-winter birds	Adults
3.SR	$\chi^2_2 = 2.34$, $P = 0.3097$	$\chi^2_3 = 1.37$, $P = 0.7121$
3.SM	$\chi^2_2 = 0.00$, $P = 1.0000$	$\chi^2_2 = 0.81$, $P = 0.6666$
2.CT	$\chi^2_1 = 0.94$, $P = 0.3333$	$\chi^2_3 = 1.70$, $P = 0.6370$
2.CL	$\chi^2_1 = 0.94$, $P = 0.3333$	$\chi^2_1 = 0.38$, $P = 0.5392$
Total	$\chi^2_6 = 4.22$, $P = 0.6469$	$\chi^2_9 = 4.26$, $P = 0.8935$
All Groups		$\chi^2_{15} = 8.48$, $P = 0.9033$

Table 2.2.1.2 Results of goodness-of-fit tests carried out on Turnstone mark-recapture (mark-resighting) data, using information from birds caught at Amble-Hauxley.

The tests (run through U-CARE Version 2.2 software: Choquet *et al.* 2005) involve chi-square comparisons of the number of birds caught in different time intervals. The overall comparison can be broken down into four component tests, each assessing a different facet of fit. These tests are:

Test 3.SR – of individuals caught at time *i*, how many were seen again or not seen again, this effectively tests for the presence of transient individuals in the marked population (i.e. birds only available to be caught on one or a few occasions).
Test 3.Sm - of individuals caught at time *i*, does when they were seen again depend on whether or not they were marked before time *i*? This effectively tests whether survival is different between marked/unmarked birds.
Test2.CT - are individuals equally likely to be recaptured if they were caught or not (but known to be alive) on the previous occasion, i.e. whether there is any evidence of trap-dependence.
Test2.CL – is there a difference in the time of next recapture between individuals captured and not captured at time *i*. There is no simple interpretation of this test.

Thus, Test 3 effectively tests the assumptions of equal survival, while Test 2 tests for equal catchability.

	Area	Month	Count section	State of tide	Count section * state of tide	Year
PS	Whole coast	$F_{5,3617} = 20.9$ $P < 0.0001$	$F_{28,3617} = 43.1$ $P < 0.0001$	ns	$F_{28,3617} = 22.5$ $P < 0.0001$	$F_{8,3617} = 7.8$ $P < 0.0001$
PS	Amble-Hauxley	$F_{5,920} = 14.4$ $P < 0.0001$	$F_{6,920} = 58.2$ $P < 0.0001$	ns	$F_{7,920} = 10.5$ $P < 0.0001$	$F_{8,920} = 2.8$ $P = 0.0045$
PS	Newbiggin	$F_{5,730} = 10.0$ $P < 0.0001$	$F_{5,730} = 60.1$ $P < 0.0001$	ns	$F_{6,730} = 7.1$ $P < 0.0001$	$F_{8,730} = 6.9$ $P < 0.0001$
TT	Whole coast	$F_{5,4155} = 5.2$ $P < 0.0001$	$F_{32,4155} = 47.5$ $P < 0.0001$	$F_{1,4155} = 4.1$ $P = 0.0420$	$F_{32,4155} = 17.6$ $P < 0.0001$	$F_{8,4155} = 6.0$ $P < 0.0001$
TT	Amble-Hauxley	$F_{5,1054} = 6.4$ $P < 0.0001$	$F_{7,1054} = 74.6$ $P < 0.0001$	$F_{1,1054} = 20.2$ $P < 0.0001$	$F_{7,1054} = 3.8$ $P = 0.0005$	$F_{8,1054} = 3.6$ $P = 0.0004$
TT	Newbiggin	ns	$F_{5,735} = 59.6$ $P < 0.0001$	$F_{1,735} = 8.2$ $P = 0.0043$	$F_{5,735} = 9.4$ $P < 0.0001$	$F_{8,735} = 1.4$ $P = 0.2153$

Table 3.1.1 Likelihood ratio statistics and associated probabilities for variables in models describing numbers of Purple Sandpipers (PS) and Turnstones (TT) on the Northumbrian coast study area (from 1997/98 to 2005/06).

Model	$QAIC_c$	Parameters	Model deviance
ϕ_c p_c	961.56	2	258.48
ϕ_t p_c	937.06	9	219.62
ϕ_t p_t	848.40	16	116.16
$\phi_{2002 \text{ vs other years}}$ p_t	841.16	10	121.63
ϕ_c p_t	839.65	9	122.21
$\phi_{\text{adult vs 1st-year}}$ p_t	837.61	10	118.08
$\phi_{\text{adult2002 \& 1st-year v adult other years}}$ p_t	**837.33**	**10**	**117.80**

Table 3.2.1.1 Evaluation of mark-resighting models for Turnstone originally caught and colour-ringed at Amble-Hauxley, using data from 1997/98 to 2005/06.

Different models evaluated whether resighting rates p were constant (c) or varied fully with time (t) and whether survival rates ϕ were constant (c), differed between adults and first-year birds (adult vs 1st-year), varied fully with time (t) or differed in the first full year after improvements, i.e. 2002. Bold type indicates the model that best fitted the data.

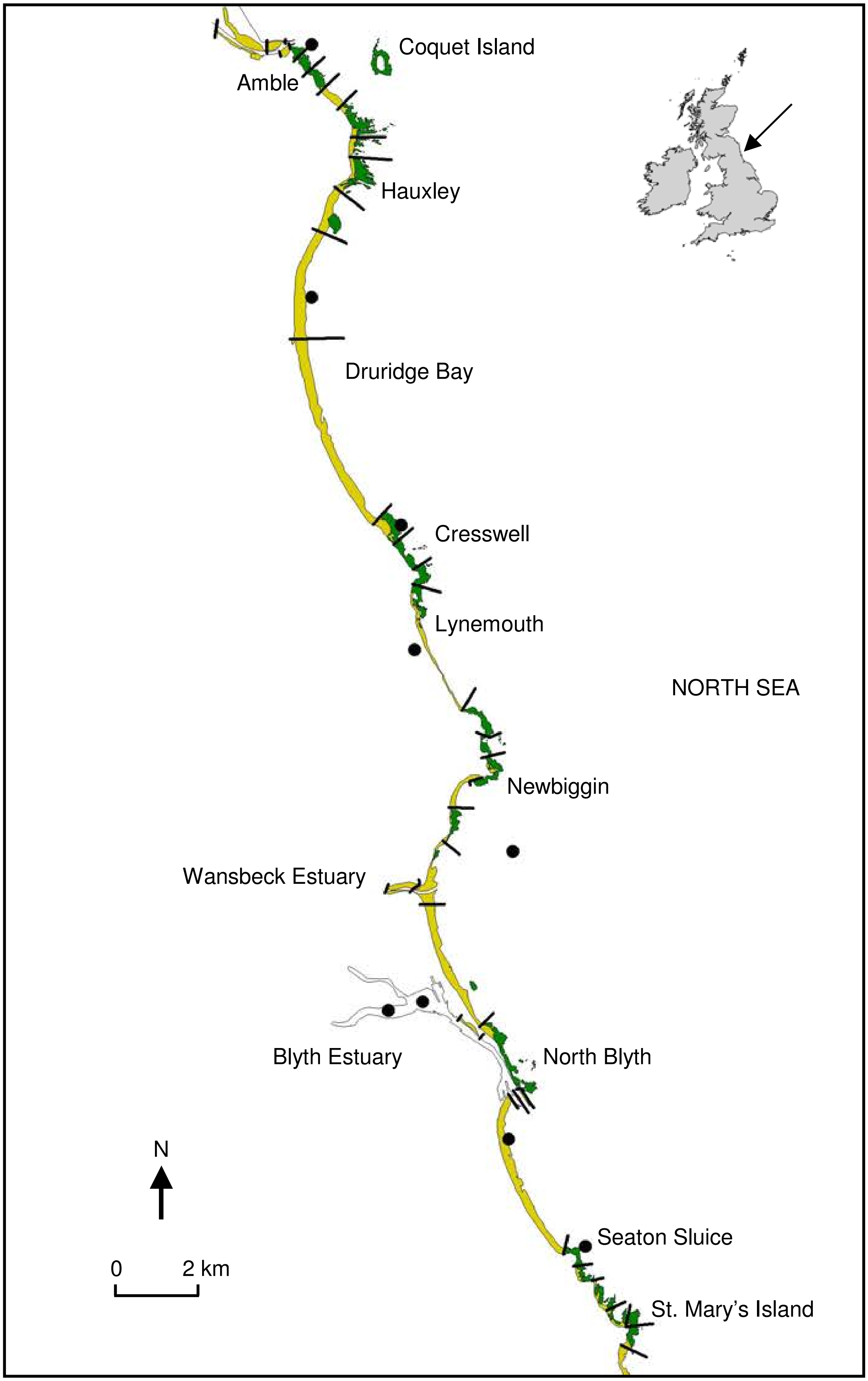


Figure 1.1 The study area, showing areas of rocky and soft sediment intertidal shore, sewage outfalls (shown by black dots) and the sections used for counting waterbirds. Amble (55˚20' N, 1˚34' W); St. Mary's Island (55˚04' N, 1˚27' W).

a.

b.

c.

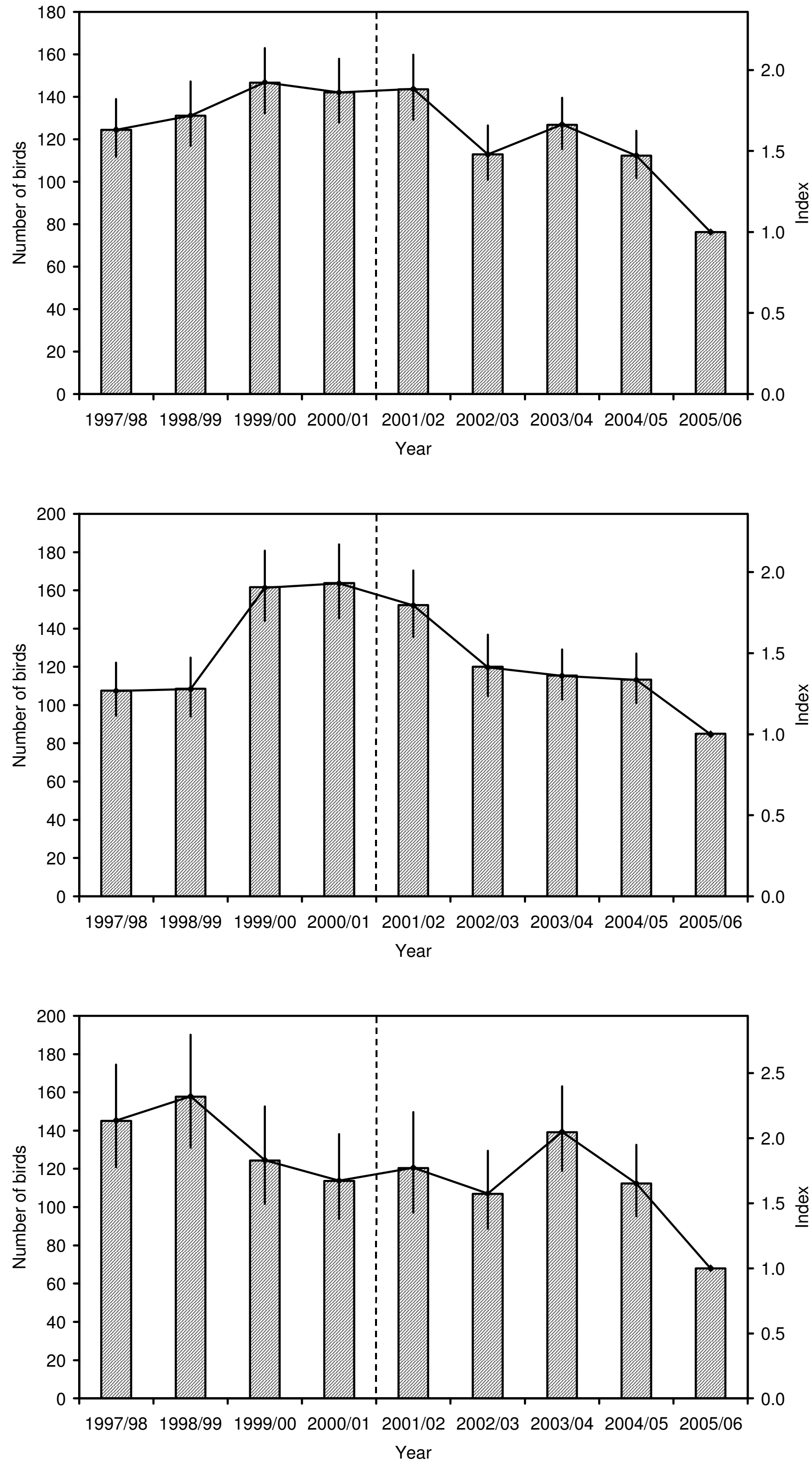
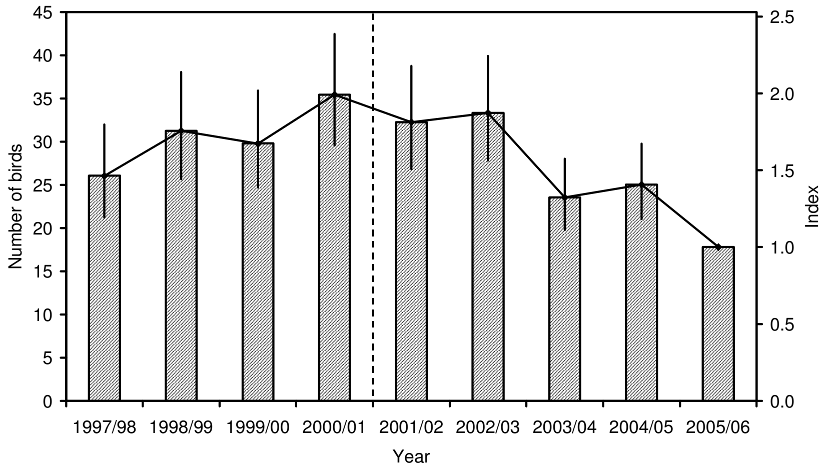
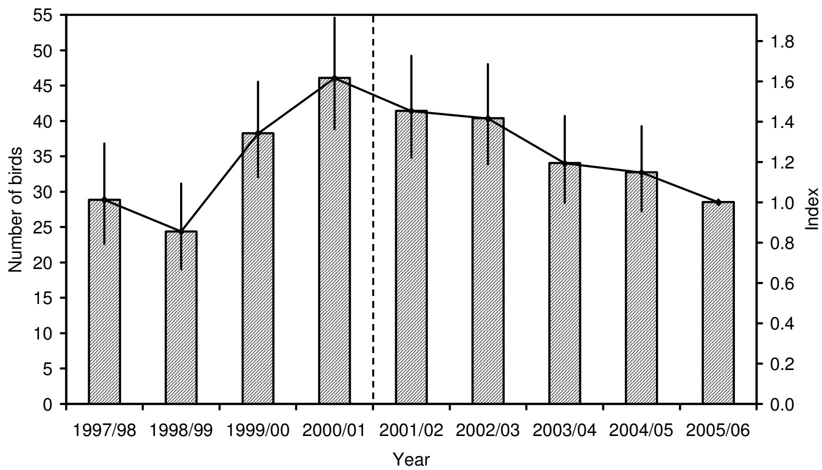


Figure 3.1.1.1 Indices (± 1 SE) and mean numbers of birds derived from models relating the numbers of Purple Sandpiper on the whole coast between the Coquet Estuary and St. Mary's Island, using data from **a.** both high and low tide **b.** low tide only and **c.** high tide only, to year, month, state of tide and count section. The dotted line indicates the date when the majority of improvements to discharges were completed.

a.



b.



c.

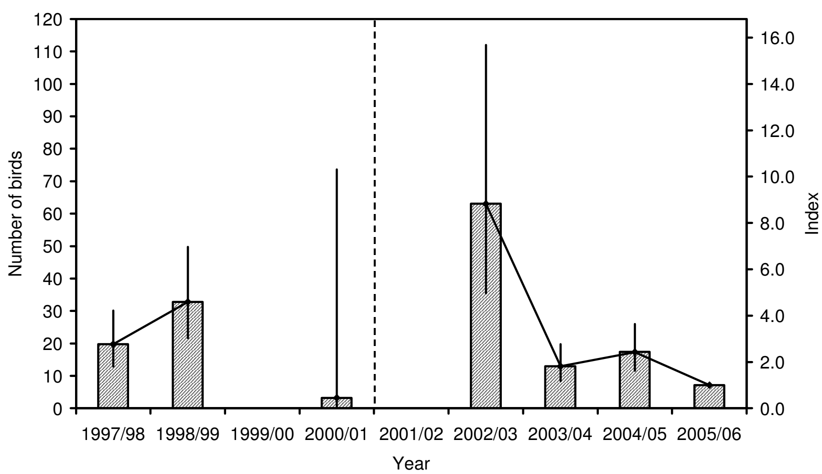
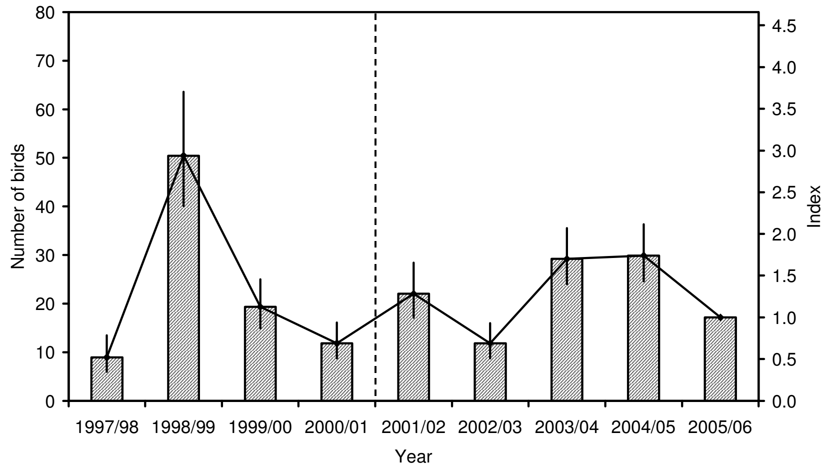
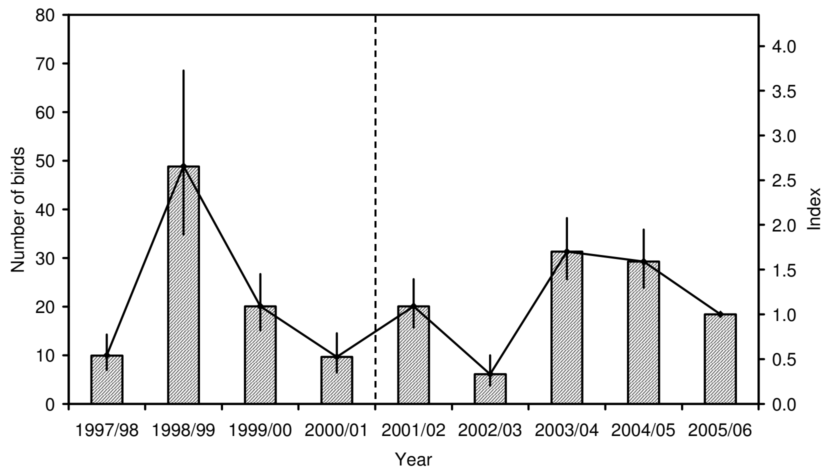


Figure 3.1.1.2 Indices (± 1 SE) and mean numbers of birds derived from models relating the numbers of Purple Sandpiper on the Amble-Hauxley coast, using data from **a.** both high and low tide **b.** low tide only and **c.** high tide only, to year, month, state of tide and count section. The dotted line indicates the date when the majority of improvements to discharges (including those at Amble-Hauxley) were completed.

a.



b.



c.

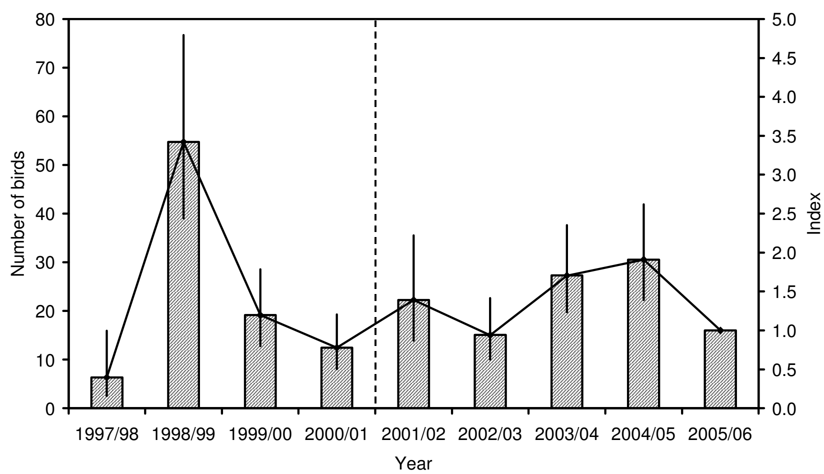


Figure 3.1.1.3 Indices (± 1 SE) and mean numbers of birds derived from models relating the numbers of Purple Sandpiper on the Newbiggin coast, using data from **a.** both high and low tide **b.** low tide only and **c.** high tide only, to year, month, state of tide and count section. The dotted line indicates the date when the majority of improvements to discharges (including those at Newbiggin) were completed.

Figure 3.1.1.4 Mean winter numbers of Purple Sandpiper in WeBS recording areas within the North-east Environment Agency region for the 25-year period from 1979/80 to 2004/05 (taken from Maclean & Austin 2006) and the proportion of this total held by the study area (as estimated from the current study) between 1997/98 and 2004/05.

a.

b.

c.

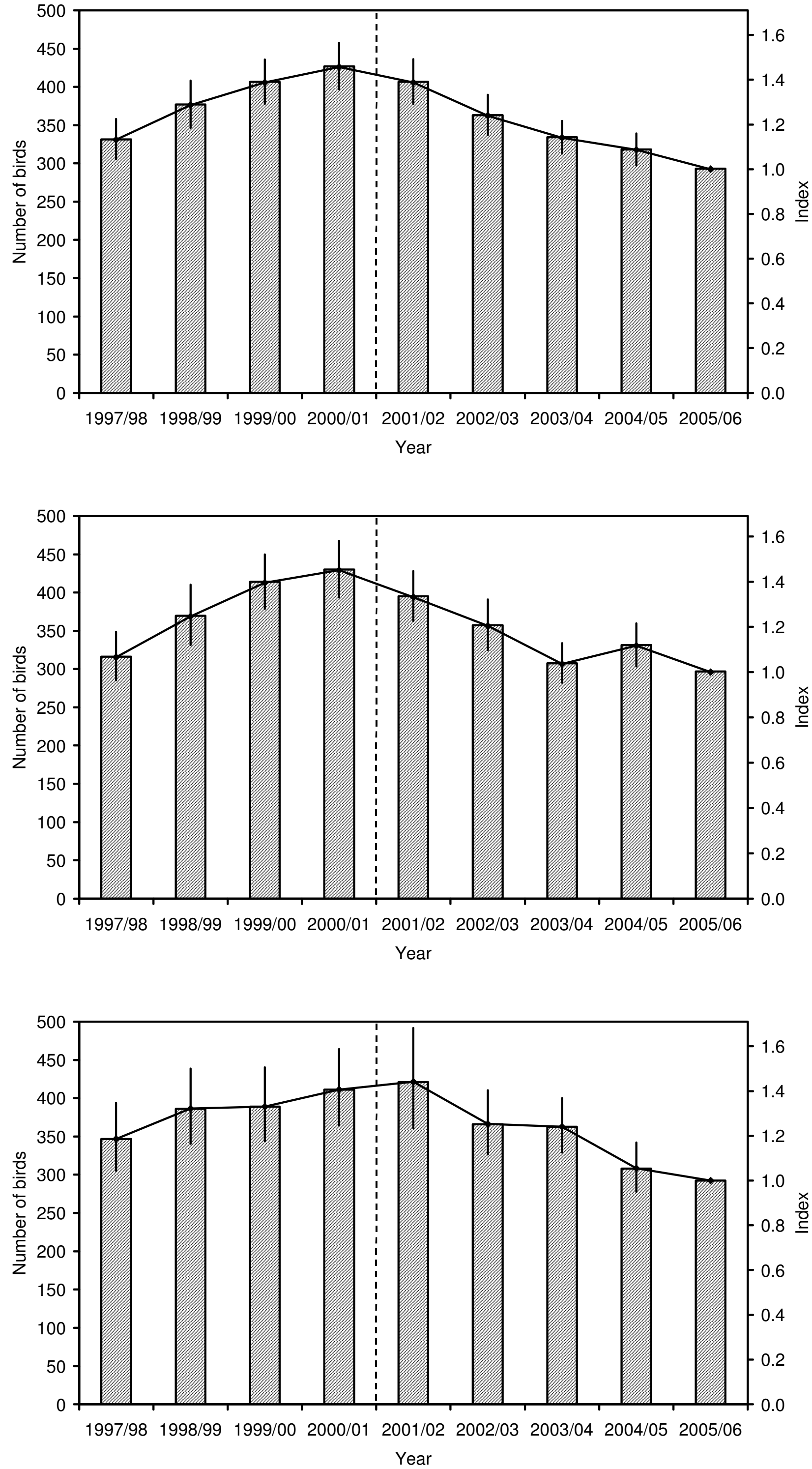
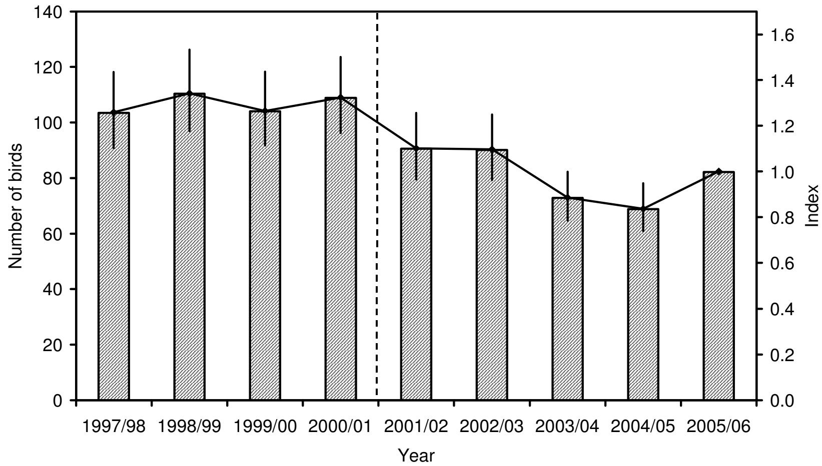
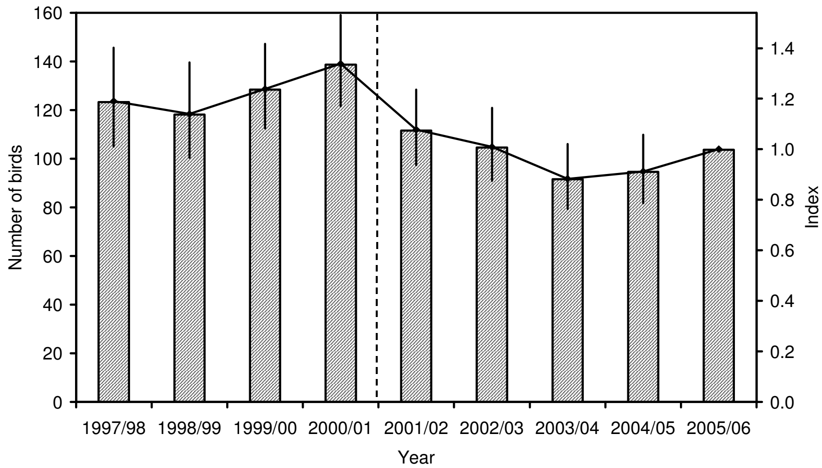


Figure 3.1.2.1 Indices (± 1 SE) and mean numbers of birds derived from models relating the numbers of Turnstone on the whole coast between the Coquet Estuary and St. Mary's Island, using data from **a.** both high and low tide **b.** low tide only and **c.** high tide only, to year, month, state of tide and count section. The dotted line indicates the date when the majority of improvements to discharges were completed.

a.



b.



c.

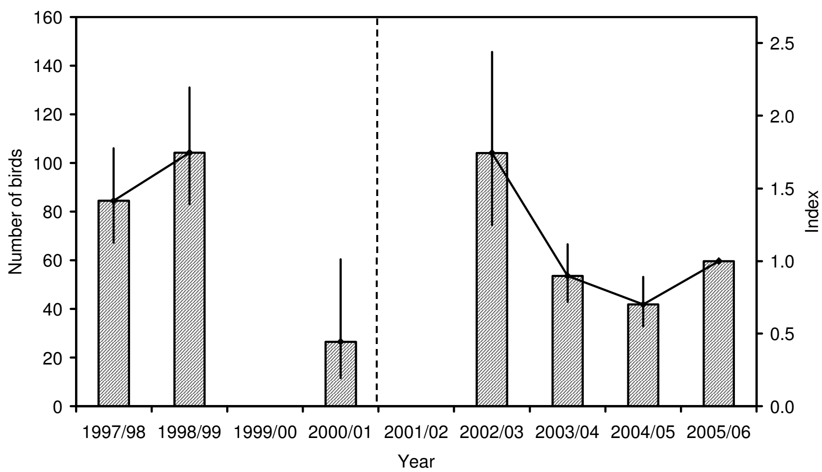
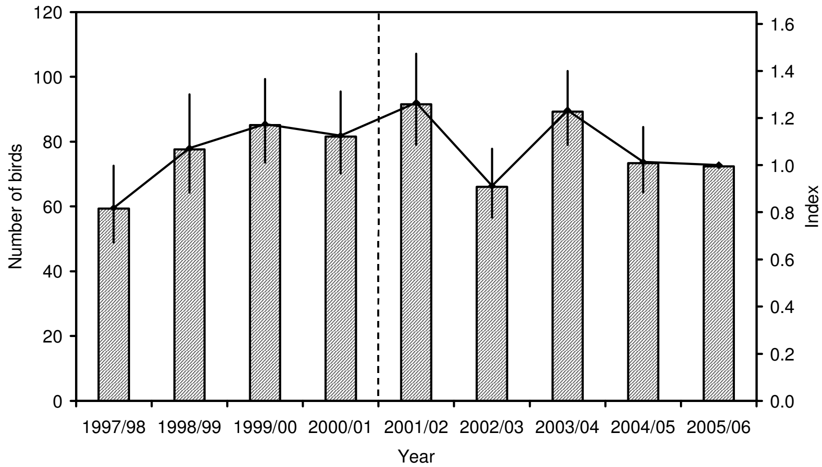
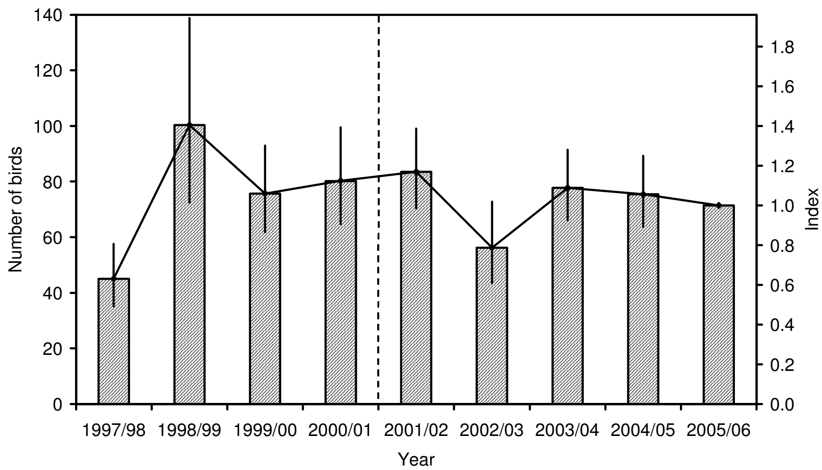


Figure 3.1.2.2 Indices (± 1 SE) and mean numbers of birds derived from models relating the numbers of Turnstone on the Amble-Hauxley coast, using data from **a.** both high and low tide **b.** low tide only and **c.** high tide only, to year, month, state of tide and count section. The dotted line indicates the date when the majority of improvements to discharges (including those at Amble-Hauxley) were completed.

a.



b.



c.

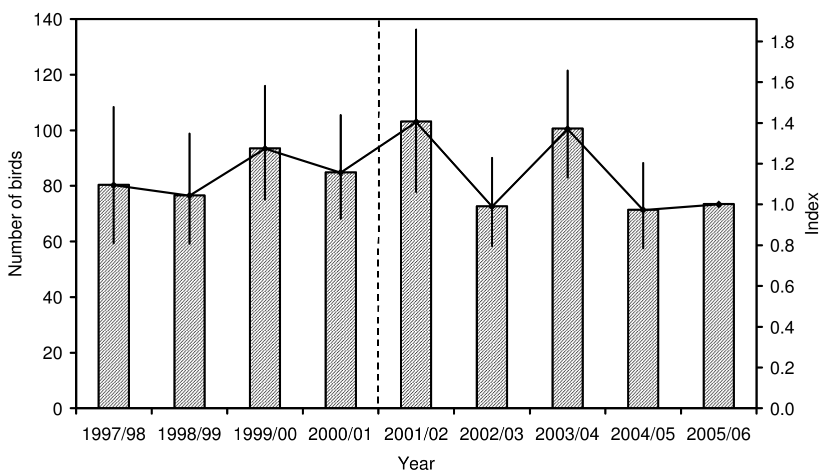


Figure 3.1.2.3 Indices (± 1 SE) and mean numbers of birds derived from models relating the numbers of Turnstone on the Newbiggin coast, using data from **a.** both high and low tide **b.** low tide only and **c.** high tide only, to year, month, state of tide and count section. The dotted line indicates the date when the majority of improvements to discharges (including those at Newbiggin) were completed. Note, year was not significant in these models.

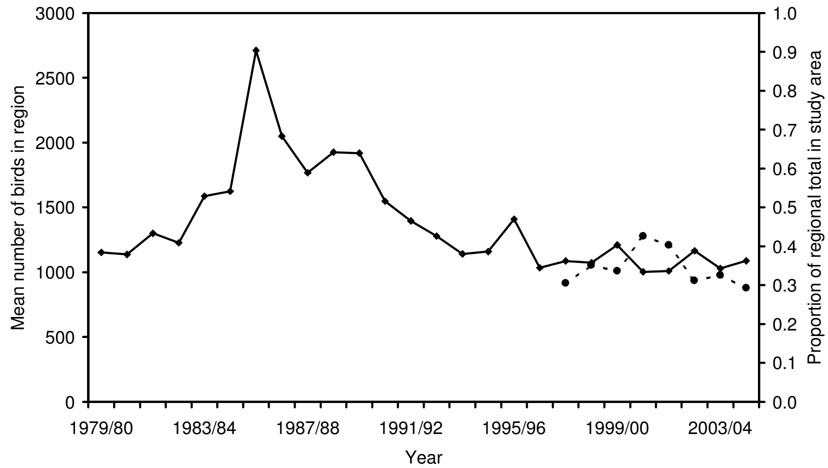


Figure 3.1.2.4 Mean winter numbers of Turnstone in WeBS recording areas within the North-east Environment Agency region for the 25-year period from 1979/80 to 2004/05 (taken from Maclean & Austin 2006) and the proportion of this total held by the study area (as estimated from the current study) between 1997/98 and 2004/05.

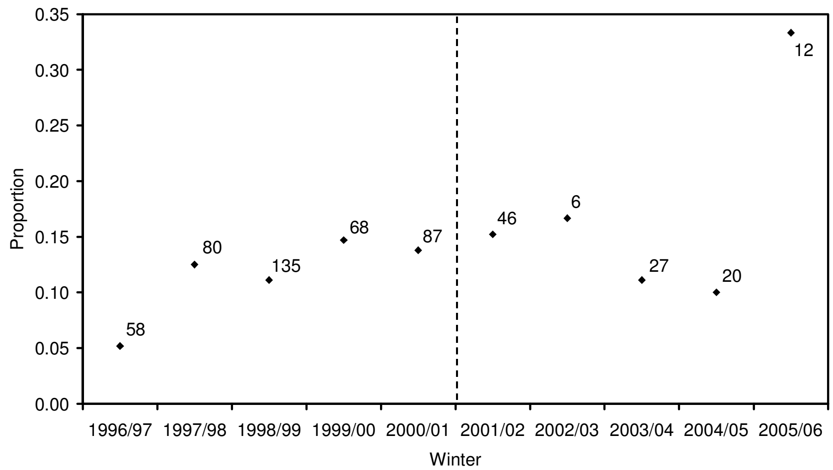


Figure 3.2.2.1 The proportions of Turnstone colour-ringed at Amble-Hauxley that were seen at other sites in the study area each winter. The total numbers of Turnstone colour-ringed at Amble-Hauxley that were seen each winter are indicated by data points. The dotted line indicates the date when the improvements to the Amble discharge were completed. Note the large proportion of individuals seen in other areas in 2005/06, though also sample sizes.

Appendix 1. Waterbird species recorded in the study area during the winters of 2003/04 to 2005/06 (1 = present).

Species	2003/04	2004/05	2005/06
Red-throated Diver *Gavia stellata*	1	1	1
Black-throated Diver *Gavia arctica*	1	1	1
Great Northern Diver *Gavia immer*	1		
Little Grebe *Tachybaptus ruficollis*	1	1	1
Great Crested Grebe *Podiceps cristatus*	1	1	1
Red-necked Grebe *Podiceps grisegena*			1
Black-necked Grebe *Podiceps nigricollis*	1		
Cormorant *Phalacrocorax carbo*	1	1	1
Shag *Phalacrocorax aristotelis*	1	1	1
Little Egret *Egretta garzetta*	1		
Grey Heron *Ardea cinerea*	1	1	1
Mute Swan *Cygnus olor*	1	1	1
Bewick's Swan *Cygnus columbianus bewickii*	1		
Whooper Swan *Cygnus cygnus*	1	1	1
Pink-footed Goose *Anser brachyrhynchus*	1	1	1
European White-fronted Goose *Anser albifrons albifrons*	1		
Greylag Goose *Anser anser*	1	1	1
Ross's Goose *Anser rossii*	1		
Canada Goose *Branta canadensis*	1	1	1
Barnacle Goose *Branta leucopsis*	1		
Brent Goose (dark-bellied) *Branta bernicla bernicla*	1	1	
Brent Goose (light-bellied) *Branta bernicla hrota*	1		
Shelduck *Tadorna tadorna*	1	1	1
Mandarin *Aix galericulata*		1	
Wigeon *Anas penelope*	1	1	1
Gadwall *Anas strepera*	1	1	1
Teal *Anas crecca*	1	1	1
Green-winged Teal *Anas carolinensis*		1	
Mallard *Anas platyrhynchos*	1	1	1
Pintail *Anas acuta*	1	1	1
Shoveler *Anas clypeata*	1	1	1
Pochard *Aythya ferina*	1	1	1
Tufted Duck *Aythya fuligula*	1	1	1
Scaup *Aythya marila*		1	1
Eider *Somateria mollissima*	1	1	1
Long-tailed Duck *Clangula hyemalis*	1	1	1
Common Scoter *Melanitta nigra*	1	1	1
Velvet Scoter *Melanitta fusca*	1	1	
Goldeneye *Bucephala clangula*	1	1	1
Red-breasted Merganser *Mergus serrator*	1	1	1
Goosander *Mergus merganser*	1	1	1
Ruddy Duck *Oxyura jamaicensis*	1		
Moorhen *Gallinula chloropus*	1	1	1
Coot *Fulica atra*	1	1	1
Oystercatcher *Haematopus ostralegus*	1	1	1
Ringed Plover *Charadrius hiaticula*	1	1	1
Golden Plover *Pluvialis apricaria*	1	1	1
Grey Plover *Pluvialis squatarola*	1	1	1
Lapwing *Vanellus vanellus*	1	1	1
Knot *Calidris canutus*	1	1	1
Sanderling *Calidris alba*	1	1	1

Appendix 1 Continued

Species	2003/04	2004/05	2005/06
Little Stint *Calidris minuta*		1	
Purple Sandpiper *Calidris maritima*	1	1	1
Dunlin *Calidris alpina*	1	1	1
Ruff *Philomachus pugnax*	1	1	1
Snipe *Gallinago gallinago*	1	1	1
Woodcock *Scolopax rusticola*	1		
Black-tailed Godwit *Limosa limosa*			1
Bar-tailed Godwit *Limosa lapponica*	1	1	1
Curlew *Numenius arquata*	1	1	1
Redshank *Tringa totanus*	1	1	1
Turnstone *Arenaria interpres*	1	1	1
Kingfisher *Alcedo atthis*	1	1	1

BTO Research Report No. 442

Impacts of changes in sewage disposal on waterbirds wintering on the Northumbrian coast
Final Report – ANNEX

Authors

N.H.K. Burton & A.P. Goddard

Report of work carried out by
The British Trust for Ornithology
under contract to
Northumbrian Water Ltd

May 2007

British Trust for Ornithology, The Nunnery, Thetford, Norfolk IP24 2PU
Registered Charity No. 216652

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EXECUTIVE SUMMARY

1. Measurement of carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) stable isotope analyses were used to quantify the relative contribution of sewage to the particulate organic matter (POM) in the water column of inshore waters before and after the improvements to sewage treatment along the Northumbrian coast. Results update those of Eaton (2001) who quantified the relative importance of sewage to POM in 1999 prior to improvements.

2. Coastal water samples were collected from a total of seven sites between North Amble and Seaton Sluice in mid-September 1999 and 2006. Reference sewage samples were taken at the pumping stations prior to discharge from major outfalls at Amble and Cambois and macroalgae samples from four sites. With knowledge of the isotope ratios of these two sources of POM, it was possible to estimate the relative contribution of each to the particulate matter in the water samples.

3. Due to significant variation in the $\delta^{15}N$ values of sewage between 1999 and 2006, it was not possible to use $\delta^{15}N$ values to determine the proportional contribution of sewage to the POM in coastal water samples. The change in the $\delta^{15}N$ values of sewage probably reflected the improvements to treatment.

4. Estimation of the proportional contribution of sewage to POM was thus based on comparison of the $\delta^{13}C$ values for coastal water samples relative to means for sewage and macroalgae. $\delta^{13}C$ values of sewage samples ranged from -26.4‰ to -24.8‰, whereas those of macroalgae were considerably greater, ranging from -19.1‰ to -11.3‰.

5. $\delta^{13}C$ values of coastal water samples varied by site, year and the interaction between site and year. $\delta^{13}C$ values in 2006 were typically heavier than those in 1999, i.e. more similar to those of macroalgae, though this varied by site. Significant increases in $\delta^{13}C$ values and thus decreases in the estimated proportional contribution of sewage to POM were recorded at four of the seven sampling sites.

6. Results indicate that the improvements to discharges will have appreciably reduced the total amount of POM in the water column. This in turn is likely to have impacted filter-feeders such as mussels and so potentially the birds, such as Turnstones and Purple Sandpipers, that prey on them.

6. INTRODUCTION

This Annex describes how stable isotope analysis have been used to quantify the relative contribution of sewage to the particulate organic matter (POM) in the water column of inshore waters before and after the improvements to sewage treatment along the Northumbrian coast. Results update those of Eaton (2001) who quantified the relative importance of sewage to POM in 1999 prior to improvements.

In these analyses, the ratios of different isotopes of an element in a sample are compared to those from potential sources, so as to identify and quantify its origin. Here, analyses look at the isotope ratios of carbon and nitrogen to quantify the relative contribution of sewage and marine algae to the particulate matter in inshore waters. Improvements to sewage treatment might be expected to have a noticeable impact on nutrient loading and total POM (Savage & Elmgren 2004, Costanzo *et al.* 2005) and thus primary productivity of plankton and macroalgae, in turn affecting planktoniverous mussels *Mytilus edulis* and other invertebrate species and thus the bird species that feed on them.

Typically the ratio of ^{13}C to ^{12}C (known as $\delta^{13}C$) is higher in marine plants and algae than in terrestrial plants, because of the higher $\delta^{13}C$ found in bicarbonate (the source of carbon fixed by marine algae) than in carbon dioxide (the source of carbon fixed by terrestrial plants) (Lajtha & Michener 1994). Sewage particulates result from the consumption by man of plants of predominantly terrestrial origin, which are not digested and have a relatively low $\delta^{13}C$ value. A difference in the ratio of the two stable isotopes of nitrogen (^{15}N to ^{14}N, referred to as $\delta^{15}N$) may also be found between marine and terrestrial sources, as may differences in the ratios of stable isotopes of sulphur and hydrogen. Samples enriched in ^{13}C or ^{15}N are referred to as being "heavier" than non-enriched samples.

A number of studies have successfully used stable isotopes to demonstrate the dispersal and uptake of sewage in marine ecosystems, using carbon (Burnett & Schaeffer 1980, Rogers 1999, 2003, Waldron *et al.* 2001, Thornton & McManus 1994), nitrogen (Sweeney & Kaplan 1980, Costanzo *et al.* 2001, 2005, Gartner *et al.* 2002, Savage & Elmgren 2004) and sulphur (Sweeney & Kaplan 1980). Some studies have used stable isotopes to trace sewage inputs through food chains, as the stable isotope composition of an organism reflects that of its diet (although some fractionation of isotopes normally occurs during assimilation). For example, by measuring carbon and nitrogen stable isotopes, Spies *et al.* (1989) calculated that 15-20% of the diet of fish (such as Dover sole *Microstomus pacificus*) near an outfall in California was derived from sewage particulates. Other studies have also found evidence for the uptake of sewage particulates by marine organisms e.g. Gearing *et al.* (1991), Van Dover *et al.* (1992), Moore *et al.* (1996), Kwak & Zedler (1997), Rogers (1999, 2003) and Waldron *et al.* (2001).

The present study aimed to determine the proportion of the suspended POM in inshore waters of the Northumbrian coast study area that originated from sewage both before and after improvements to sewage treatment. This approach was used by Tucker *et al.* (1999) to trace sewage through Boston Harbour and Massachusetts Bay (using nitrogen stable isotopes).

A range of intertidal seaweeds (macroalgae) was sampled in order to measure $\delta^{13}C$ and $\delta^{15}N$ values for this major natural source of POM in inshore waters. In addition, sewage samples were taken from the coastal outfalls at Amble and Cambois in order to measure the isotope ratios for the sewage particulate matter. With knowledge of the isotope ratios of these two sources of particulate matter, it was possible to estimate the relative contribution of each to samples of particulate matter taken from the inshore waters of the study area.

Prior to improvements to discharges, sampling was conducted in both spring and autumn 1999 (Eaton 2001). Autumn sampling was repeated in September 2006 in order to establish the change in the relative contribution of sewage to the POM in the study area's inshore waters.

7. METHODS

7.1 Sampling locations and collection

Coastal water samples, sewage and algal samples were collected in mid-September 1999 and 2006.

Sewage samples were taken at the pumping stations prior to discharge from major outfalls at Amble and Cambois (eight samples in 1999 and 10 in 2006).

Water samples (from which particulate matter was filtered) were collected from a total of seven sites:

North Amble (NU274049)
South Amble (NU278043)
North Hauxley (NU288031)
South Hauxley (NU287018)
Cresswell (NZ296938)
North Blyth (NZ320819)
Seaton Sluice (NZ339769)

The water sampling sites effectively formed two transects running south (down current) from two of the three main sewage outfalls in the study area: the first five sites to the south of Amble outfall and the latter two to the south of the Cambois outfall.

Coastal water and sewage samples were collected in litre (1999) or half litre bottles (2006). Six samples were taken from each site in 1999 and 12-14 in 2006. Water samples were collected at half-tide on the rising tide, by wading out to approximately 0.5 m water depth and collecting water from below the surface. Care was taken to avoid areas with high wave action as such samples contained large amounts of sand. This method ensured that samples represented water that flowed over intertidal rock on the incoming tides and was filtered for food by feeding mussels and other invertebrates.

Algae were sampled from four rocky shore sites – Amble, Cresswell, North Blyth and Seaton Sluice. At each site, we collected two whole, healthy-looking specimens of three algal species – *Laminaria hyperborea* (lower shore), *Fucus vesiculosus* (mid-shore) and *Enteromorpha linza* (uppershore).

7.2 Stable isotope analyses

Details of the analyses used in 1999 are provided in Eaton (2001).

In 2006, sewage and water samples were filtered using a vacuum pump through 25 mm diameter Whatman glass fibre filters. Between 210 and 500 ml (typically 250 mm) of each sample was filtered before the papers became clogged. Filter papers were subsequently air-dried at 60°C. Analysis used half of a single filter paper for each sample.

Algal species were washed thoroughly in de-ionised water and dried at 60°C. Samples were then finely ground with a pestle and mortar so as to be homogenised for analysis.

Measurements of stable isotope ratios were made at the Stable Isotope Laboratory in the School of Environmental Sciences at the University of East Anglia, Norwich. Samples were measured using an Europa 20-20 continuous flow mass spectrometer, coupled with an ANCA elemental analyser. Samples were combusted with oxygen, at 1000C, the resulting N_2 and CO_2 were separated by a GC column and sent to the mass spectrometer for the isotope analysis.

Stable isotope values are reported as δ values in permil (‰), with respect to their international standards – for carbon, Pee Dee Belemnite (δPDB, a marine limestone fossil; Craig 1953) and for nitrogen, atmospheric air (δAIR; Mariotti 1983):

δ (‰) = $[(R_{sample}/R_{standard}) - 1] \times 1000$

where δ is the sample isotope ratio (^{13}C or ^{15}N) relative to the standard and R is the ratio of heavy to light isotopes ($^{13}C/^{12}C$ or $^{15}N/^{14}N$) in the sample or standard.

The precision of the measurements was to 0.25 ‰ for both carbon and nitrogen.

7.3 Statistical analyses

Stable isotope values for sewage samples were first compared to determine whether there were differences between sites (Amble or Cambois) or years (1999 or 2006). Likewise, values for macroalgae samples were compared to determine if there were differences according to species (*Laminaria hyperborea*, *Fucus vesiculosus* or *Enteromorpha linza*), site (Amble, Cresswell, North Blyth or Seaton Sluice) or year.

In 1999, replicates were analysed for several samples of sewage, macroalgae and coastal water particulates (Eaton 2001). In the current study, replicated values were averaged prior to analyses to avoid pseudoreplication. Current analyses also include valid unreplicated results from 1999 which were not included in Eaton (2001).

These and subsequent analyses were undertaken using generalised linear models (GLMs; PROC GLM in SAS: SAS Institute Inc. 2002-2003). Models considered the following class factors: site, year, the interaction between site and year and for analyses of stable isotope values for macroalgae, algae species and the interaction between species, site and year.

The proportion of particulate matter in a sample that originated from sewage inputs was calculated using a two-source mixing model:

$$P_{sewage} = (\delta_{sample} - \delta_{algae})/(\delta_{sewage} - \delta_{algae})$$

Where P_{sewage} = the proportion of particulates in a given sample that are derived from sewage, δ_{sample} = the stable isotope ratio of the sample and δ_{algae} and δ_{sewage} = the respective stable isotope ratios of algae and sewage for the sample site.

Analyses were undertaken to determine whether $\delta^{13}C$ and $\delta^{15}N$ values of coastal water samples and P_{sewage} were related to site, year and the interaction between site and year. It is predicted that following the improvements to sewage discharges the $\delta^{13}C$ and $\delta^{15}N$ values of coastal water samples would become more similar to those of macroalgae and thus that the proportion of particulates attributable to sewage would decline.

8. RESULTS

8.1 Isotope ratios of sewage and macroalgae samples

Mean $\delta^{13}C$ and $\delta^{15}N$ values for sewage and macroalgae species by site and year are shown in Tables 8.1.1 and 8.1.2 respectively.

8.1.1 Sewage

$\delta^{13}C$ values of sewage samples ranged from -26.37‰ to -24.77‰. Significant relationships were found between $\delta^{13}C$ values and year ($F_{1,32}$ = 34.99, P < 0.0001) as well as the interaction between site and year ($F_{2,32}$ = 43.60, P < 0.0001), though not site on its own ($F_{1,32}$ = 1.18, P = 0.2848) – probably as samples collected at the same site in the same year showed little variation in their signatures. The mean $\delta^{13}C$ value of sewage from the Amble outfall was significantly heavier (less negative) in autumn 1999 than autumn 2006, whereas mean value for the Cambois outfall was significantly less heavy in 1999 than in 2006.

$\delta^{15}N$ values of sewage samples ranged from 1.00‰ to 16.50‰. $\delta^{15}N$ values of sewage were significantly related to site ($F_{1,32}$ = 69.60, P < 0.0001), year ($F_{1,32}$ = 133.73, P < 0.0001) and the interaction between site and year ($F_{1,32}$ = 65.68, P < 0.0001). At both Amble and Cambois, $\delta^{15}N$ values were significantly heavier in 2006 than in 1999, though the difference was greatest at Amble where the mean value increased from 2.36‰ to 12.95‰.

8.1.2 Macroalgae

$\delta^{13}C$ values of macroalgae samples were heavier than those of sewage, ranging from -19.08‰ to -11.26‰. $\delta^{13}C$ values of macroalgae were significantly related to species ($F_{2,36}$ = 14.44, P < 0.0001) and the interaction between species and site ($F_{9,36}$ = 2.57, P = 0.0213), but not site alone ($F_{3,36}$ = 2.07, P = 0.1210), the interaction between site, year and species ($F_{12,24}$ = 1.39, P = 0.2367) or year ($F_{1,24}$ = 0.00, P = 0.9940). $\delta^{13}C$ values of *F. vesiculosus* were on average the lightest and those of *L. hyperborea* the heaviest. Those of the latter species also differed significantly between sites ($F_{3,12}$ = 6.04, P = 0.0095), those from Amble and Cresswell in the north of the study area being lighter than those at North Blyth and Seaton Sluice in the south ($F_{1,14}$ = 12.97, P = 0.0029). There were no significant differences between sites in the $\delta^{13}C$ values of *F. vesiculosus* ($F_{3,12}$ = 1.78, P = 0.2044) or *E. linza* ($F_{3,12}$ = 1.14, P = 0.3735).

$\delta^{15}N$ values of macroalgae samples ranged from 3.03‰ to 9.60‰ and were significantly related to species ($F_{2,45}$ = 17.14, P < 0.0001), but no other factor (interaction between site, year and species: $F_{21,24}$ = 1.51, P = 0.1635; site: $F_{3,24}$ = 1.01, P = 0.4049; year: $F_{1,24}$ = 0.63, P = 0.4346; interaction between site and species: $F_{1,24}$ = 0.97, P = 0.4658). $\delta^{15}N$ values of *E. linza* were on average the heaviest and those of *L. hyperborea* the lightest.

8.1.3 Use of $\delta^{13}C$ and $\delta^{15}N$ values of sewage and macroalgae samples for estimating the proportional contribution of sewage to POM

Due to the variation in $\delta^{15}N$ values of sewage between 1999 and 2006, and notably because of the change in the mean value for the Amble outfall relative to that for macroalgae between 1999 and 2006, it was not possible to use $\delta^{15}N$ values to determine the proportional contribution of sewage to the POM in coastal water samples. The change in the $\delta^{15}N$ values of sewage probably reflects the improvements to treatment – secondary and tertiary treatment may increase $\delta^{15}N$ values, at least of dissolved inorganic nitrogen (Savage & Elmgren 2004) – though the reason for the discrepancy in values at Amble and Cambois in 2006 is unclear.

Estimation of the proportional contribution of sewage to POM was thus based on comparison of the $\delta^{13}C$ values for coastal water samples relative to means for sewage and macroalgae. Due to the variation in $\delta^{13}C$ values for sewage it was necessary to use different means for different sites and years. $\delta^{13}C$ values for water samples collected at North Amble, South Amble, North Hauxley, South Hauxley and Cresswell were referenced to the mean value for sewage from the Amble outfall, with different values for 1999 and 2006. For samples collected at North Blyth and Seaton Sluice, $\delta^{13}C$ values for the Cambois outfall were used, again with different values for 1999 and 2006.

The relative importance of the different types of algae sampled, in terms of volume of input into inshore waters, is not known. Therefore, the $\delta^{13}C$ values for algae used in calculations were derived by averaging those for *F. vesiculosus*, *L. hyperborea* and *E. linza*, though (due to the variation between sites in the values of *E. linza*) separately for sites in the north and south of the study area. $\delta^{13}C$ values for water samples collected at North Amble, South Amble, North Hauxley, South Hauxley and Cresswell were thus referenced to the mean value for macroalgae sampled at Amble and Cresswell, whereas for water samples collected at North Blyth and Seaton Sluice, analyses referenced the mean $\delta^{13}C$ value for macroalgae sampled at those same sites.

8.2 Isotope ratios of coastal water samples and the proportional contribution of sewage to POM

Mean $\delta^{13}C$ and $\delta^{15}N$ values for coastal water samples by site and year are shown in Table 8.2.1.

$\delta^{13}C$ values of coastal water samples ranged from -12.05‰ to -24.16‰. Values varied by site ($F_{6,116}$ = 22.47, P < 0.0001), year ($F_{1,116}$ = 118.94, P < 0.0001) and the interaction between site and year ($F_{6,116}$ = 15.71, P < 0.0001). $\delta^{13}C$ values in 2006 were typically heavier than those in 1999, i.e. more similar to those of macroalgae, though this varied by site. Significant increases in $\delta^{13}C$ values were recorded at North Amble ($F_{1,16}$ = 87.27, P < 0.0001), North Hauxley ($F_{1,16}$ = 50.00, P < 0.0001), Cresswell ($F_{1,17}$ = 26.03, P < 0.0001) and North Blyth ($F_{1,17}$ = 27.70, P < 0.0001), but no significant changes at South Amble ($F_{1,16}$ = 1.98, P = 0.1782), South Hauxley ($F_{1,16}$ = 1.59, P = 0.2250) or Seaton Sluice ($F_{1,18}$ = 0.92, P = 0.3494).

Small quantities of nitrogen meant that reliable results could not be obtained for some samples in both 1999 and 2006 and that there were possible inaccuracies in those results retained for statistical analyses. $\delta^{15}N$ values of coastal water samples varied by year ($F_{1,90}$ = 10.81, P = 0.0014), but not site ($F_{6,84}$ = 1.23, P = 0.3002) or the interaction between site and year ($F_{6,78}$ = 0.47, P = 0.8309). $\delta^{15}N$ values were on average heavier in 2006 than in 1999.

Estimates of the proportion of POM derived from sewage at each sampling site, prior to improvements to discharges in 1999 and afterwards in 2006, are shown in Fig. 8.2.1 and Tables 8.2.2. (The mixing model used assumes that POM comes either from sewage or macroalgae). The proportion of POM derived from sewage, as estimated from $\delta^{13}C$ values, varied by site ($F_{6,116}$ = 27.41, P < 0.0001), year ($F_{1,116}$ = 107.24, P < 0.0001) and the interaction between site and year ($F_{6,116}$ = 15.98, P < 0.0001). Significant decreases in the proportion attributable to sewage were recorded at North Amble ($F_{1,16}$ = 90.51, P < 0.0001), North Hauxley ($F_{1,16}$ = 52.15, P < 0.0001), Cresswell ($F_{1,17}$ = 28.73, P < 0.0001) and North Blyth ($F_{1,17}$ = 19.75, P = 0.0004), but no significant changes at South Amble ($F_{1,16}$ = 1.67, P = 0.2153), South Hauxley ($F_{1,16}$ = 2.06, P = 0.1705) or Seaton Sluice ($F_{1,18}$ = 0.01, P = 0.9390). Decreases of 31 to 47% were recorded where differences between years were significant.

Estimates of the possible decrease in available POM shown in Table 8.2.2 have been calculated assuming that the quantity of organic matter derived from macroalgae has remained unchanged (see discussion).

9. DISCUSSION

9.1 $\delta^{13}C$ and $\delta^{15}N$ values and their use in the estimation of the proportion of POM in coastal water samples attributable to sewage

Previous studies indicate that values of $\delta^{15}N$ from sewage may show considerable variation. For example, although Tucker *et al*. (1999), Hunt *et al*. (1996) and Rogers (1999, 2003) reported values of 3.3‰, 3.2‰ and 2.3‰ respectively for sewage in Massachusetts, New York and New Zealand, Van Dover *et al*. (1992) reported values ranging from -1.1‰ to 7.2‰ for sewage from a number of sites in north-east USA. Savage & Elmgren (2004) reported much higher values of 24 and 38‰ for treated sewage discharged in the Baltic, noting that secondary and tertiary sewage treatment may enrich $\delta^{15}N$ signatures.

The $\delta^{15}N$ values of sewage in the present study in 1999 were in similar to those found in the above U.S. studies. However, in 2006, the $\delta^{15}N$ values for sewage were enriched, particularly at Amble where the mean value was 13.0‰. The changes between 1999 and 2006 (particularly relative to macroalgae) and the differences between the sewage measured at Amble and Cambois precluded the use of $\delta^{15}N$ values in investigating the changing contribution of sewage to the POM in inshore waters of the study area.

$\delta^{15}N$ values of macroalgae may themselves be affected by sewage, due to the increased urea and ammonia in seawater nitrogen being taken up by the algae (Rogers 1999, 2003). Use of these values would thus clearly be misleading in attempting to estimate the relative contribution of sewage to inshore POM, though they may be used as a trace for sewage influence (Costanzo *et al.* 2001, Gartner *et al.* 2002, Savage & Elmgren 2004). In the present study, $\delta^{15}N$ values of macroalgae samples showed considerable variation and, though they were significantly related to species, did not differ between sites or years. $\delta^{15}N$ values for macroalgae were lighter than those found by some previous studies e.g. compared to values for *Enteromorpha* of 8.1 and 14.4‰ reported by Tucker *et al*. (1999) and 11.9 and 11.4‰ by Kwak & Zedler (1997), but in the range of those for *Fucus* in the Baltic reported by Savage & Elmgren (2004).

There were significant differences in the $\delta^{13}C$ values for sewage between sites and years, though in actuality they only ranged in value by 1.6‰, differences likely being due to samples being collected on single occasions, thus reducing variation. The $\delta^{13}C$ values found for the Amble and Cambois outfalls (which ranged from -26.4‰ to -24.8‰) were similar to those previously reported for sewage, e.g. -23.5‰ (Spies *et al*. 1989), -23.7‰ (Kwak & Zedler 1997) and -22.8‰ (Van Dover *et al*. 1992) for studies in the U.S.A. and -26.5‰ for a study in New Zealand (Rogers 1999, 2003).

$\delta^{13}C$ values for macroalgae were considerably heavier than those of sewage, ranging from -19.1‰ to -11.3‰. Rogers (2003) found a small but significant increase in the $\delta^{13}C$ values of macroalgae following closure of a sewage outfall in New Zealand. However, in the present study, $\delta^{13}C$ values did not differ between years, suggesting little sewage influence. Comparison of the carbon isotope ratios in water samples to those of sewage and macroalgae thus provided a reliable estimate of the proportion of POM attributable to sewage. In previous studies (e.g. Rogers 2001, 2003), $\delta^{13}C$ values have also proved valuable in tracing the incorporation of POM through the food chain, for example in filter-feeders such as mussels, and estimation of the proportion of POM attributable to sewage may thus provide a useful measure of its influence on the foraging environment of waders such as Turnstones and Purple Sandpipers.

9.2 Estimates and changes in the proportion of POM derived from sewage

Analyses assumed that POM in the inshore waters of the study area was derived from two sources: sewage or macroalgae. In actuality, other sources may also have had some influence on the isotope ratios recorded. Marine phytoplankton, for example, may have contributed to POM in inshore waters, though probably to a relatively insignificant extent in comparison to sewage or macroalgae. Previous studies have found $\delta^{13}C$ values for phytoplankton to be heavier than for macroalgae (e.g. Gearing *et al*. 1984, Rau *et al*. 1981 and c.f. Waldron *et al.* 2001) and so any contribution from phytoplankton to POM would have led to an underestimate of the relative input from sewage.

More notably, POM from rivers may potentially have contributed to POM in inshore waters, particularly at sites immediately south of the mouths of the Coquet and Blyth rivers. The POM from these estuaries is likely to have similar isotope ratios to those measured in sewage and inclusion of such inputs in samples would have led to the overestimation of the relative importance of sewage. This may have been the case at Blyth and, notably, Seaton Sluice, where high unchanging estimates of the proportion of POM derived from sewage might have been biased by the influence of Seaton Burn. It was not possible to quantify the extent of the contribution from freshwater sources – and thus any overestimation of that from sewage – though at sites other than Blyth and Seaton Sluice, results suggest that it was not likely to be great. Estimates of the relative contribution of sewage to the POM in inshore waters at the four sites immediately south of the Coquet, for example, were all exceeded by the value obtained at Cresswell, where there are no major freshwater inputs (49% of the POM in inshore waters at Cresswell was estimated to be derived from sewage prior to improvements). The similarity between $\delta^{13}C$ values for POM at North Amble and North Hauxley and macroalgae following improvements also suggests that the influence of freshwater at these sites was probably small at this time. Given that sampling was undertaken in the same month before and after the improvements, at a similar state of tide and at the same locations, it is reasonable to suppose that the influence of freshwater was similarly small before the improvements and thus that the changes recorded do reflect the changing influence of sewage inputs.

The increases in $\delta^{13}C$ values between 1999 and 2006 at four of the seven sample sites show that, as expected, sewage has had a decreasing influence in the study area. Rogers (2003) reported similar changes in the $\delta^{13}C$ signatures found in mussels *Mytilus galloprovincialis* following closure of an outfall in New Zealand (Rogers 2003). As in that study, the proportional contribution of sewage to POM and thus the likely total amount of POM decreased following the changes to outfall discharges. This in turn is likely to have impacted filter-feeders such as mussels and so potentially the birds, such as Turnstones and Purple Sandpipers, that prey on them.

Results in the present study varied considerably between sites, with no significant changes recorded in three cases. Only 15 and 22% of POM at South Amble and South Hauxley was estimated to originate from sewage in 1999, prior to the improvements to discharges, in comparison to 39 to 62% at other sites. It is possible that this discrepancy could have been related to the tendency for detached wrack to accumulate on the beaches at these sites (Eaton 2001). The microbial decomposition of wrack may result in local increases in the amount of marine POM in the water column, thereby temporarily reducing the relative input of sewage (but not the absolute amount of POM originating from sewage). Decreases in the average relative contribution of sewage to POM at these locations will have been hidden if wrack was present during sampling in 1999, but not 2006.

The estimates of possible decrease in available POM shown in Table 8.2.2 were calculated assuming that the quantity of organic matter derived from macroalgae has remained unchanged. Actual changes in the macroalgae and presence of wrack in the study area during sampling are unknown, though it is possible that, as macroalgae may utilise dissolved nitrogen from sewage, there may have been a decrease in macroalgae biomass following the improvements to discharges. If this were the case, the decreases in POM calculated would be underestimates.

10. CONCLUSIONS

In conclusion, the study has shown that the proportional contribution of sewage to the particulate organic matter (POM) found in the inshore waters of the study area significantly decreased following the improvements to sewage treatment.

- Estimation of the proportional contribution of sewage to POM was based on comparison of the $\delta^{13}C$ values for coastal water samples relative to means for sewage and macroalgae. $\delta^{13}C$ values of sewage samples ranged from -26.4‰ to -24.8‰, whereas those of macroalgae were considerably greater, ranging from -19.1‰ to -11.3‰.

- $\delta^{13}C$ values of coastal water samples varied by site, year and the interaction between site and year. $\delta^{13}C$ values in 2006 were typically heavier than those in 1999, i.e. more similar to those of macroalgae, though this varied by site. Significant increases in $\delta^{13}C$ values and thus decreases in the estimated proportional contribution of sewage to POM were recorded at four of the seven sampling sites. The lack of recorded change at the other three sites may potentially have been due to the unquantified influence of wrack deposits or (at least in one case) freshwater inputs.

Results indicate that the improvements to discharges will have appreciably reduced the total amount of POM in the water column. This in turn is likely to have impacted filter-feeders such as mussels and so potentially the birds, such as Turnstones and Purple Sandpipers, that prey on them.

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			$\delta^{13}C$			$\delta^{15}N$	
Site	Year	Mean (‰)	s.e. (‰)	n	Mean (‰)	s.e. (‰)	n
Amble	1999	-25.38	0.10	8	2.36	0.24	8
Amble	2006	-25.61	0.05	10	12.95	0.91	10
Cambois	1999	-26.09	0.07	8	2.23	0.22	8
Cambois	2006	-25.05	0.05	10	4.09	0.15	10

Table 8.1.1 Mean $\delta^{13}C$ and $\delta^{15}N$ values for sewage samples by site and year.

			$\delta^{13}C$			$\delta^{15}N$		
Site	Year	Species	Mean (‰)	s.e. (‰)	n	Mean (‰)	s.e. (‰)	n
Amble	1999	*E. linza*	-15.22	0.06	2	8.60	0.36	2
Amble	2006	*E. linza*	-13.65	1.00	2	7.19	0.14	2
Cresswell	1999	*E. linza*	-13.86	0.06	2	8.75	0.11	2
Cresswell	2006	*E. linza*	-16.17	0.60	2	8.41	0.08	2
N Blyth	1999	*E. linza*	-15.35	0.05	2	6.78	0.02	2
N Blyth	2006	*E. linza*	-16.15	1.63	2	8.99	0.37	2
S Sluice	1999	*E. linza*	-18.58	0.17	2	7.77	0.22	2
S Sluice	2006	*E. linza*	-14.74	1.96	2	8.94	0.66	2
Amble	1999	*F. vesiculosus*	-16.34	0.73	2	7.63	0.19	2
Amble	2006	*F. vesiculosus*	-17.35	0.12	2	6.03	1.07	2
Cresswell	1999	*F. vesiculosus*	-17.31	0.77	2	7.47	0.47	2
Cresswell	2006	*F. vesiculosus*	-17.69	1.40	2	6.71	0.85	2
N Blyth	1999	*F. vesiculosus*	-15.58	0.24	2	6.50	0.47	2
N Blyth	2006	*F. vesiculosus*	-16.16	0.66	2	7.52	0.49	2
S Sluice	1999	*F. vesiculosus*	-16.86	1.13	2	8.17	0.27	2
S Sluice	2006	*F. vesiculosus*	-16.03	0.62	2	7.00	0.38	2
Amble	1999	*L. hyperborea*	-14.91	0.40	2	5.97	0.11	2
Amble	2006	*L. hyperborea*	-15.80	0.82	2	6.55	0.43	2
Cresswell	1999	*L. hyperborea*	-15.65	1.55	2	4.92	1.89	2
Cresswell	2006	*L. hyperborea*	-14.46	0.76	2	5.90	0.87	2
N Blyth	1999	*L. hyperborea*	-11.45	0.19	2	6.16	0.32	2
N Blyth	2006	*L. hyperborea*	-12.50	0.90	2	6.89	0.09	2
S Sluice	1999	*L. hyperborea*	-13.40	0.14	2	6.16	0.72	2
S Sluice	2006	*L. hyperborea*	-13.89	1.75	2	7.15	0.14	2

Table 8.1.2 Mean $\delta^{13}C$ and $\delta^{15}N$ values for macroalgae samples by species, site and year.

Site	Year	δ13C Mean (‰)	δ13C s.e. (‰)	n	δ15N Mean (‰)	δ15N s.e. (‰)	n
N Amble	1999	-19.47	0.29	6	6.30	0.31	5
N Amble	2006	-15.75	0.24	12	12.93	3.97	5
S Amble	1999	-17.11	0.52	6	4.74	0.55	5
S Amble	2006	-17.64	0.09	12	9.38	1.75	11
N Hauxley	1999	-20.25	0.33	6	5.61	0.30	6
N Hauxley	2006	-15.72	0.42	12	8.55	1.61	10
S Hauxley	1999	-17.85	0.22	6	4.74	0.39	3
S Hauxley	2006	-17.49	0.17	12	6.57	1.45	10
Cresswell	1999	-20.42	0.15	6	5.13	1.33	5
Cresswell	2006	-18.23	0.28	13	6.77	1.08	10
N Blyth	1999	-21.88	0.25	6	4.11	0.28	2
N Blyth	2006	-17.83	0.50	13	9.74	1.24	10
S Sluice	1999	-20.57	0.21	6	6.38	0.61	2
S Sluice	2006	-20.08	0.31	14	8.87	2.03	8

Table 8.2.1 Mean $\delta^{13}C$ and $\delta^{15}N$ values for coastal water samples by site and year.

	1999			2006			Estimated decrease in POM
	Mean	s.e.	n	Mean	s.e.	n	
N Amble	0.39	0.03	6	0.00	0.02	12	-38.61
S Amble	0.15	0.05	6	0.20	0.01	12	6.23
N Hauxley	0.47	0.03	6	0.00	0.04	12	-46.95
S Hauxley	0.22	0.02	6	0.18	0.02	12	-5.04
Cresswell	0.49	0.02	6	0.26	0.03	13	-31.21
N Blyth	0.62	0.02	6	0.28	0.05	13	-47.17
S Sluice	0.50	0.02	6	0.50	0.03	14	0.78

Table 8.2.2 Estimated proportions of particulate organic matter (POM) derived from sewage at seven sites on the Northumbrian coast in 1999 and 2006 and estimates of the decrease in available POM. Proportions derived from a two-source mixing model using $\delta^{13}C$ values.

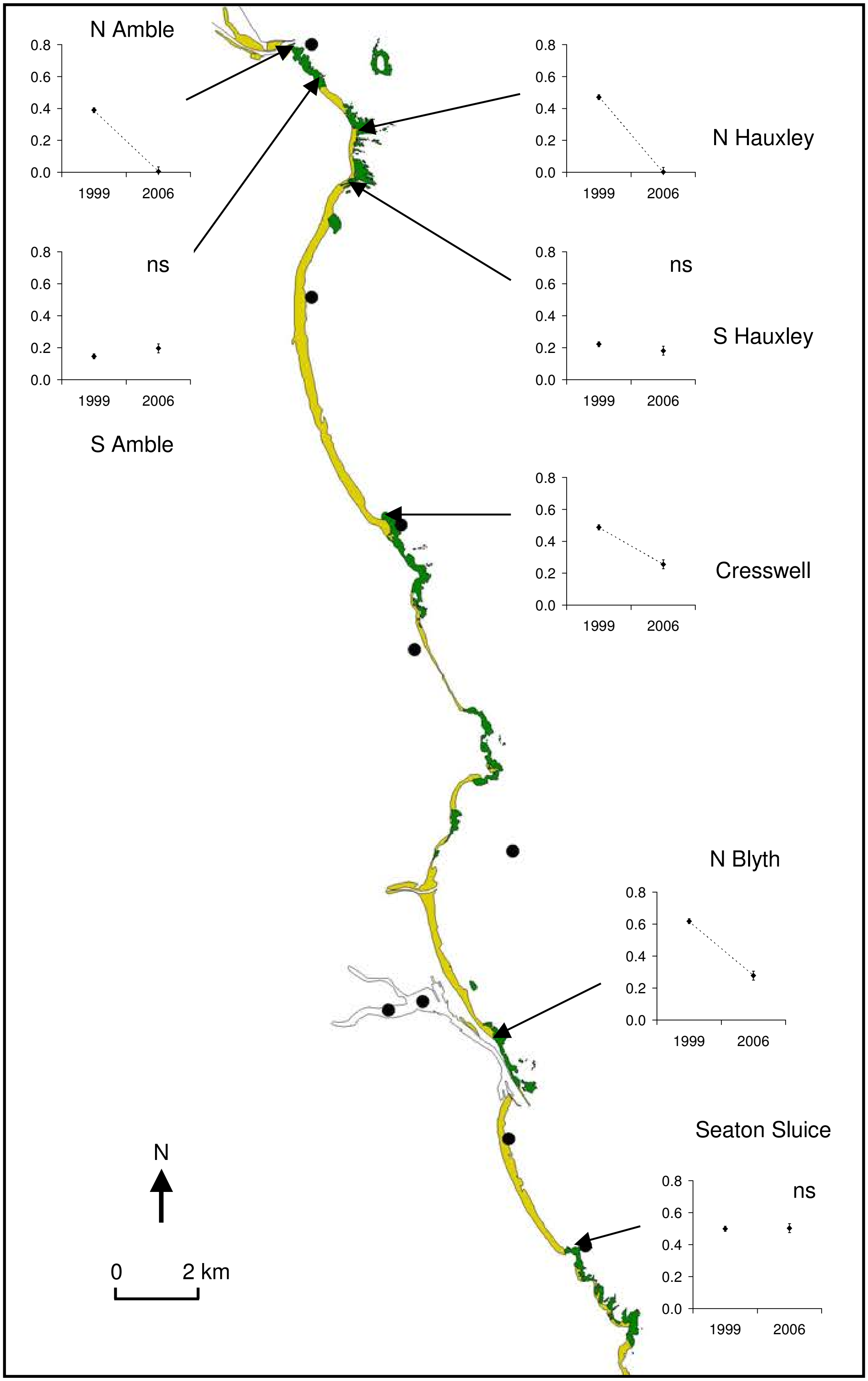

Figure 8.2.1 Changes in the estimated proportion of particulate organic matter (POM) derived from sewage at seven sites on the Northumbrian coast between 1999 and 2006, calculated using (autumn) $\delta^{13}C$ values. Black dots show the positions of outfalls. 'ns' indicates an insignificant change.