Monitoring should concentrate on three aspects:

a. Numerical comparisons with 'control' areas

In most mitigation schemes it is necessary to make comparisons with 'natural' habitats. The choice of controls is crucial. Controls should be chosen carefully and if possible replicated, avoiding the problems of pseudo-replication, ie collect data from a number of replicate areas that can be said to be independent from each other and not just many samples from one area. They should encompass a range of geomorphological and hydrodynamic conditions found in the estuary.

As seen from the examples in previous chapters, birds can use new areas for feeding or roosting and also at different times of year or tide. In a comprehensive monitoring scheme it is desirable to encompass this range of uses and count birds at different times of year and state of tide. Snapshot counts (ie counting the number of birds in one sweep) at specific times during tidal cycles at different times of year would fulfil these criteria.

It is important to standardise counts. Some of the above studies have varied the months during which counts were carried out, the state of tide between counts or even the length of time spent counting birds. The Tollesbury study was ideal in respect of the fact that counts were carried out three times monthly, thus generating sufficient data to analyse and also at the same state of tide (low, neap high or spring high) and counts took place over similar time periods on each occasion. Statistical power is increased by keeping as many factors constant. Comprehensive monitoring studies should ideally also concentrate on individual birds as well as making numerical comparisons.

b. Monitoring how individual birds use the site

As habitat loss or creation on estuaries tend to be small relative to the total estuary area, monitoring using counts may not be sufficient to determine the effects of loss or creation on individual species. The examples of habitat loss in Chapter 2 clearly show that the large natural annual variation in waterbird numbers often masks any changes caused by the loss of a section of intertidal habitat.

For this it is important to individually mark birds using colour-rings or with radio tags. Although expensive, the use of radio tags can identify the areas used by an individual bird throughout a tidal cycle. Such information is extremely valuable in determining the role of the area to be lost at all stages of the tidal cycle. This information can be fed into the decision-making process when assessing the desirable features of any habitats to be created.

Colour-ringing is extremely useful to follow the fate of individual birds. Through the use of standardised re-sighting effort, survival rates of displaced and control birds can be calculated.

c. Experimental designs

As current knowledge does not permit the reliable prediction of the outcome of restoration efforts, an experimental approach may be taken to determine the factors important in intertidal habitat mitigation. For example, experimental creeks could be dug into one section of the new area and not another. These should be carefully designed so that they have sufficient power to determine whether differences in treatments occur.

5.6 Conclusions

All the studies above indicated that waterbirds will colonise new areas of intertidal habitat. However in almost every case, differences between restored and natural bird assemblages are present. Some species, such as Redshank, occur in higher densities on new habitat than on natural habitat whereas other species such Grey Plover do not. Some of these differences are due to the immaturity of the new site, whereas others have reached an alternative stable state which is likely to persist. The causes for these differences is mostly due to habitat characteristics which impinge on food supply or some aspect of a species' behaviour. The studies also indicate that the restoration outcome is not always predictable given current knowledge but that if specific efforts are made to ensure success then the site is more likely to achieve it. Future mitigation schemes should therefore take an experimental approach and also have clear criteria determining success.

Table 5-1 Intertidal habitat creation or restoration schemes in the UK which have monitoring schemes associated with them

Site	Nature of bird data	Conclusions	Reference
UK - Retreat sites			
Seal Sands	Comprehensive bird and invertebrate surveys	Some invertebrates took longer to colonise. Bird usage has increased in relation to changing invertebrate densities. Mudflat can be restored but at least 5 years lead in time required	Evans *et al* (1998) Evans *et al* (2001)
Tollesbury Wick	Comprehensive bird monitoring in retreat and reference area	No analyses carried out	Unpublished reports
Orplands	Summer & winter bird counts in retreat and reference areas	No analyses carried out	Unpublished reports
Abbotts Hall	Summer & winter bird counts in retreat and reference areas	No analyses carried out	Unpublished reports
Northey Island	No comprehensive counts	-	
Saltram	No counts	-	
Chichester Harbour	No counts	-	
UK - Recharge sites			
Trimley	Summer & winter bird counts in retreat and reference areas	No analyses carried out	Unpublished reports
Horsey Island	Summer & winter bird counts in retreat and reference areas	No analyses carried out	Unpublished reports
Cobmarsh Island	Summer & winter bird counts in retreat and reference areas	No analyses carried out	Unpublished reports
Old Hall Point	Summer & winter bird counts in retreat and reference areas	No analyses carried out	Unpublished reports
Tollesbury Wick	Summer & winter bird counts in retreat and reference areas	No analyses carried out	Unpublished reports
Wallasea Ness	Summer & winter bird counts in retreat and reference areas	No analyses carried out	Unpublished reports

Site	Nature of bird data	Conclusions	Reference
UK – 'Unmanaged' retreat sites			
Porlock Marsh	Regular counts with reference to a small adjacent area.		Unpublished reports
Unmanaged retreat sites in Essex/Suffolk	3-year study on wintering Twite	Study of wintering Twite. Most sites unsuitable as the marshes were flat, poorly drained & highly dissected. Sites that were suitable had a higher proportion of lower saltmarsh communities, in particular Salicornia. The provision of sheltered Salicornia beds (eg shallow borrow pits) made a site suitable.	Atkinson (1998) Norris & Atkinson (2001)
Netherlands			
Dollard	PhD study 'Nature Management of Coastal Saltmarshes' in the Dollard Estuary. Covers management techniques for grazing geese & breeding Redshank.	Created marshes (using reclaim & brushwood groynes) with rigid drainage grid & grazing were suitable for grazing geese. Removal of grazing increases incursion of Phragmites which in turn reduces suitability for Redshank.	Esselink (2000)
United States			
Gog-le-hi-te wetland	Monitoring of 16 estuarine functions including birds. Weekly to monthly visits during peak shorebird and waterfowl migration period, April-September. Abundance and habitat occurrence noted. No comparison with reference areas.	Number of species using the site still increasing annually after 4 years since creation.	Simenstad & Thom (1997).
Barn Island Wildlife Management Area, Stonington, Connecticut	Birds monitored during 2 breeding seasons at five sites, both reference and restoration sites. Results are compared with a related study looking at 11 other reference sites in Connecticut.		Brawley *et al* (1998)

Site	**Nature of bird data**	**Conclusions**	**Reference**
Galveston Bay, Texas	Avian usage compared during 12 monthly counts from October 1990-September 1991 in 7 restored & 7 reference marshes. Ref marshes matches for similarities in physical similarities to restored marshes.	Shorebirds, herons & sparrows made up 63% of birds observed on natural marshes. Created marshes were made up of gulls & terns (76%) Densities of gulls & terns higher in created, shorebirds & rails in natural. Created marshes lacks microhabitats suitable for shorebirds & rails. Natural marshes also supported more diversity within species groups.	Melvin & Webb (1998)
Atlantic coast USA	Creation of ponds in saltmarsh	There was a high degree of variation in bird use of different sites. Seasonal effects were significant, but treatment (created vs. natural ponds) effect was not. Water/marsh (W/M) area ratio was positively correlated with waterfowl and black ducks, but pond number was not. Larger ponds (0.25 ha) tended to be used more than smaller ponds by most bird species.	Erwin *et al* 1991 Erwin *et al* 1994
Sarah's Creek, Gloucester Point, Virginia. 10 km north of Chesapeake Bay.	Three marshes (one restored, two reference). Birds surveyed during winter, spring and summer at high and low tide. 18 surveys in total. Perimeter walk used to record all birds on each occasion.	- Spartina band was too thin in the constructed marsh for Marsh Wrens (2m wide) and only one bird nested compared with 31 in the other marshes. Watts 1992 showed that there is a size effect - marked increase in the number of spp using the site if the area is 1-5 ha increase. - No mature saltbush communities in the constructed marsh hence use by small insectivorous passerines was lower. - Saltbush also important for Red-winged Blackbirds nesting. - A greater length of marsh/water interface in the created marsh led to increased usage by wading birds.	Havens *et al* (1995)

6. Overall assessment of mitigation and compensation schemes at meeting objectives

In this chapter, we draw overall conclusions from our review of managed and unmanaged intertidal habitat creation.

6.1 Mudflats and saltmarshes can be created. Physical stability is achievable. Equivalence with natural areas can not be guaranteed

The hydrodynamics of estuaries are sufficiently well understood to be able to design managed retreat sites at which the created environment persists. The evidence from recharge projects also indicates that mudflats can be successfully created if the geomorphological and hydrodynamic conditions are fulfilled.

The main challenge is to create or restore areas that have features similar to surrounding natural areas. The complex geomorphological features such as creek density and surface topography drive the ecological functions that natural marshes support and the linkages between habitat form and function are very poorly understood. This makes predicting the development of created or restored habitats subject to great uncertainty.

Nevertheless, areas where marshes are naturally regenerating indicate that it is possible to recreate areas of intertidal habitat that persist for at least 100 years, given time and suitable environmental conditions (tides, sediment supply etc). These restored habitats may, however, have different physical characteristics and environmental functions to those of nearby natural marshes. Restored areas may consist of over-consolidated sediments or have a different creek network pattern. These differences can be of environmental benefit, detriment or both. The science and techniques to assess these differences are not yet available.

Time is a major component in restoring intertidal habitats. If a marsh of 'natural' form is required then it is preferential to allow a natural succession from low intertidal through primary marsh to mature marsh rather than attempting to create high marsh through dredging works. Although, engineering techniques (such as placement of dredged material, planting of vegetation and digging of channels) can accelerate the rate of this process, this places artificial constraints on natural development. These marshes are often very different in form (eg topography & creek networks), and probably function, from natural marshes. Additional effort through engineering works increases the likelihood of success. However, if engineering works are used to create a creek network then the restored form should be based upon natural hydraulic laws which relate creek dimensions to features such as the size of the marsh area, and seek to replicate the form of natural habitats.

The stabilisation of new mudflats or recharged areas often takes place quicker than the establishment of 'natural' saltmarsh. The process involves dynamic interactions between erosive or reshaping forces of waves and tides, the resistive forces inherent within sediment cohesion and, finally, the binding actions of plants and micro-organisms. Sands and gravels will rapidly form well-drained deposits on placement but the cohesive nature of a muddy sediment means that it may take some time after placement before the sediments reach a 'balance' with the local tidal and wave conditions. Reaching this balance involves complex de-watering and consolidation processes as well as biological actions from micro-organisms and invertebrates.

Because of the consolidation and dewatering issues, a number of problems exist in terms of creating intertidal habitat with fine-grained sediment by hydraulic pipeline. The use of a novel pump method using high-density dredge material which is then reworked in to a slurry without additional water and transported via the pipeline to the containment site may increase the chances of success. The advantage of this method is that less dewatering is required. Invertebrates can also survive the process and no-overconsolidation has been noted. Further investigation of this method is recommended.

6.2 Geomorphological features, vegetation and some ecological functions may take a very long time to develop

Invertebrate and bird populations may not stabilise for periods in excess of five years, but in many cases presumably will stabilise. The data available on the vegetation and geomorphological characteristics of some unmanaged retreat sites raises more fundamental questions about whether success of mitigation schemes can be guaranteed.
For example, in a number of cases, vegetation on unmanaged retreat sites is very different from that on nearby natural marshes, even after a century of development and vegetation assemblages are unlikely to reach equivalence with natural marshes (Burd 1994).

In addition, the majority of restored or created sites, in the UK and elsewhere, have developed a creek network that is different in nature from that of local natural marshes, thus not restoring full ecological function. Other features such as saltmarsh pans have failed to develop on restored sites. The contrast between natural marshes and marshes on retreat sites is vividly shown by aerial photographs (see Chapter 3). It is likely that these differences are a consequence of poor drainage on the unmanaged retreat sites, that in turn results from lack of a varied surface topography and over-consolidation of underlying sediments or of placed dredged material, although further research is needed to confirm this. The implications of these vegetation and geomorphological differences for invertebrate and bird populations have not been fully characterised, but are likely to be significant. For example, Twite numbers on these sites are significantly lower than on natural marshes due to the lack of required low-marsh habitats (Atkinson 1998).

6.3 Birds are mobile and will utilise newly created habitats, if conditions are suitable

In Chapter 2, we discussed how birds distribute themselves across available resources in a way that maximises their individual rates of resource acquisition, meaning that bird distributions reflect the relative suitability of habitats. This discussion was primarily in terms of habitat loss, and showed that we would expect habitat losses to be reflected in a reduced population size. The same processes work in reverse when new habitats are created. If the new habitats offer the resources that a species needs, then the habitat will be rapidly occupied and may allow an increase in overall population size, depending on the factors controlling it. Even if factors controlling population size are acting elsewhere replacing lost habitat with a suitable alternative will provide an adequate substitute and mean that local population size is not affected. This is important when legislation in most protected area networks acts to protect a network of specific sites, rather than at a species population level.

The empirical data described in Appendix 1 and Chapter 5 supports these conclusions. The created intertidal habitats at Teesmouth, Orplands and Tollesbury have been rapidly colonised by waterbirds and evidence from the United States also concludes that birds are rapid colonisers, often reaching an equilibrium within three to five years. The resulting waterbird assemblages present on created habitats do however show some departures from expectations. At least some of these differences are predictable, in the light of information on delays in the colonisation of the sites by invertebrates. For example, larger bivalves may take time to grow to a suitable size or invertebrates without a planktonic phase may take longer to colonise. However, differences in physical (eg high walls and small size of Seal Sands enclosure) or chemical or geomorphological features (eg short *Spartina* on restored marshes makes them unsuitable for Clapper Rails) often lead to equilibrium situations where created/restored marshes are different from natural area.

These studies indicate that, if ecologically-suitable sites can be created, then bird populations should colonise. If the information on habitat and requirements of different bird species given in Chapter 4 is fed into the design phase of mitigation schemes, it should in theory be possible to create suitable replacement habitats for the bird species being displaced by a development. In practice this is, not so straightforward, as current knowledge may not be sufficient to create suitable habitats to order.

6.4 Invertebrate populations often take five years to fully establish, and can take significantly longer

It is clear from Chapters 4 and 5, that invertebrate populations can be relatively slow to establish, and this in turn can lead to delays in colonisation of sites by birds. These delays in invertebrate colonisation can be dealt with by establishing mitigation sites well in advance of development. In other cases, however, the invertebrate populations fail to establish as expected, and this is likely to lead to a failure of bird populations to establish. So a certain proportion of mitigation schemes will fail ecologically, for reasons that we do not fully understand. Differences in sediment texture, tidal height and over-consolidation of sediments have been identified as possible contributory causes of failure, but without significant additional research we are not in a position to predict whether or not a particular mitigation scheme is likely to be ecologically successful from the outset

6.5 Bird assemblages on restored and created areas are often different to natural areas

Excluding time needed for plant and invertebrate colonisation, the climax bird community in new sites may be different to those of adjacent natural areas. These differences are reported in most studies, noting particularly that newly restored areas may be dominated by generalists or only part of the assemblages supported by natural areas. Two main reasons have been identified for this. Firstly, an absence of, or unpredicted alteration to, specific habitat features on restored areas (eg *Spartina* in US restored marshes is often too short for Clapper Rails) or a feature of the created site which prevents birds using the site fully (eg high walls surrounding the Seal Sands created mudflat prevent usage by Grey Plover, despite high prey densities). Secondly, the size or dimensions of restored sites may be small compared with that of natural areas and therefore can not be expected to support a similar diversity of habitats.

There are many reasons for other differences between natural and restored intertidal areas which may impinge on waterbird populations. These often relate to inappropriate placement, design or management of the restored site and include physical features (eg the nature of different sediment types, different slope characteristics, topography and creek networks) as well as chemical features (eg soil oxygen level and nutrient availability) and hydrodynamic characteristics (eg tidal conditions and wave climates). Some of these may converge to have similar characteristics to natural areas whereas others may not. This, in turn, means that the degree to which given functions described in Table 1.2 may be restored is often variable.

This again emphasises that created sites will not immediately replace all the ecological functions of displaced sites and that a lead in time of at least five years is needed to determine whether the restored site is of sufficient 'value' to adequately replace lost habitat.

6.6 We either need a greater ability to predict the success of mitigation schemes or new habitat must be created and judged to be an acceptable substitute before development takes place

If we cannot determine in advance whether a proposed mitigation scheme will be successful in replacing a habitat that is being lost, it is difficult to see how the requirements of the Habitats Directive can be satisfied by mitigation that takes place at the same time as the proposed development. To remedy this deficit we would need a substantial amount of further research on existing retreat sites and some experimental work evaluating the results of a range of different management techniques on new retreat sites. The alternative, given uncertainties associated with restoration actions, is to introduce a system of mitigation banking, in which a substantial amount of new habitat is created. This habitat would then be available to be purchased to satisfy mitigation requirements. The equivalence of the mitigation site to the site being developed could then be assessed before development took place.

7. Best Practice

The recent damning review of the 20 year mitigation track record in replacing displaced wetlands in the United States (NRC 2001) highlights that where and when possible habitats designated to be of high value to society should not be destroyed with the assumption that they can readily be replaced. When, for overriding public interest (as laid down under the Habitats Directive) designated habitats are likely to be adversely affected and mitigation provided then the suggestions for best practice for wetlands habitat restoration laid down in Section 7.1 should be considered.

Each project will have different aims and therefore different measures of success. In Section 7.2, we discuss how to define and set success criteria for intertidal habitat creation/restoration schemes. First we look at how success may be defined by different interest groups and then specifically discuss the principles drafted by English Nature for the implementation of the compensatory measures required when part of the Natura 2000 network of sites is lost.

7.1 **What should be done to achieve successful wetland habitat restoration**?

Defined by the Society of Wetland Scientists[1] (2000) *wetland restoration* is: actions taken in a converted or degraded natural wetland that result in the reestablishment of ecological processes, functions, and biotic/abiotic linkages and lead to a persistent, resilient system integrated within its landscape. In order to achieve holistic wetlands restoration a number of fundamental generic scientific principles should be recognised and followed (SWS 2000):

7.1.1 Restoration is the reinstatement of driving ecological processes

The three main factors that maintain wetland ecosystems are the hydrology, geomorphological setting, physical processes (eg sediment movement), biological processes (eg competition, decomposition, predation) and biogeochemical processes (eg nutrient cycling). These interact to perform the ecological functions to produce the structure that we associate with wetlands. As actively installing the biotic structure of a system may not always be necessary (Mitsch *et al* 1998) or adequate to restore the functions of the system (Zedler 1996; Malakoff 1998), restoration needs to address these main factors first.

7.1.2 Restoration must be integrated with the surrounding landscape

Successful restoration demands that consideration be given to the landscape setting in which the system occurs. It is this landscape that underlies many of the large-scale factors and fundamental forces (eg water and sediment movement, geomorphology) that are essential to the formation and long-term maintenance of ecosystems. Restoration projects that address the effects of alterations that have occurred within the landscape as a result of human development can deal directly with the causes of degradation rather than just the symptoms. As understanding of landscape ecology and its importance to restoration develops, it becomes increasingly clear that the integration of restoration projects within the landscape context is essential to producing ecosystems that function in a dynamic and resilient manner.

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[1] The Society of Wetland Scientists is an international non-profit organisation founded in 1980 to promote wetland science and the exchange of information related to wetlands.

7.1.3 The goal of wetland restoration is a persistent, resilient system

Rather than trying to create a static mimic of a natural system, the developing field of ecological engineering is increasingly recognising that our objective should be the creation of persistent, resilient systems. These are not static but rather have enough of the physical and biological processes intact to allow the system to respond to disturbances without human intervention (Mitsch 1998). The practical realities of conducting restoration in the modern world often necessitate human involvement to maintain an ecosystem (eg prescribed burning or the removal of non-native species). In addition, implementation of adaptive management in learning how to better conduct restoration requires active management and monitoring of a site. Acknowledging these caveats and limitations in the pursuit of a wholly persistent, resilient system, the ultimate goal of restoration should be a system that is dynamic and that can function without human intervention.

7.1.4 Wetland restoration should result in the historic type of wetland but may not always result in the historic biological community and structure

The importance of maintaining the historic diversity of wetlands across a landscape requires that the geomorphology and hydrologic regime of a restored wetland match that present historically (Wilcox & Whillans 1999). However, restoration of the historic wetland type will not always lead to re-instatement of a historic or specific biological structure. While the essence of wetland restoration is ‘putting it back to a former or original state’, a variety of factors (eg successional stage, seed bank conditions, disturbance history) may prevent establishment of the communities and biological structure present prior to human disturbance even when the driving processes have been restored. Of course, natural and restored systems may be different through failure to engineer an adequate match to the historic physical, geomorphological and hydrological conditions.

7.1.5 Restoration planning should include the development of structural and functional objectives and performance standards for measuring achievement of the objectives

The planning process in which objectives and performance standards for achieving them are established is the foundation of adaptive management. It is critical that we learn from our successes and failures, particularly in the relatively new field of wetland restoration. It would be very helpful to have some element of independent assessment of the quality of post-restoration monitoring and of the overall success of individual schemes. As indicated earlier in this report, some of the post-construction monitoring uses methods that are less than satisfactory, and assessment made of the success of schemes are lacking in self criticism.

7.2 How to define success?

Defining the success of a restoration project is a difficult and controversial issue, being greatly dependant on goals and perspectives (see also discussions in the *Journal Ecological Engineering* volume 15). What may be recognised as successful restoration by one individual or organisation might be deemed a failure by another, depending on success criteria. This will continue to be the case unless clear guidelines on assessment and monitoring are put in place. Ideally, measures of success should be quantifiable. In the majority of intertidal habitat creation schemes, a lack of scientific understanding, funding availability, planning capacity or recognition of ecosystem value has often meant that only the crudest indicators representing the simplest levels of success are often assessed. This does little to advance our understanding of the systems involved.

At the simplest level successful intertidal habitat restoration has often been defined by engineers seeking to improve flood defence, or dispose of dredged material, occasionally with some added but often unspecific environmental benefits. More recently the requirements to restore or recreate habitat as either compensation or mitigation for a loss has demanded more refined and specific criteria for success. In many cases this measurement of compliance success has been through assessing certain ecological attributes such as species diversity or reestablishment of specific species. The increasing recognition of the wider benefits which the environment provides has given rise to 'functional assessment', whereby the need to restore all functions of a displaced wetland type are demanded. Criteria for determining compliance and functional success have historically focused on individual sites and are thus limited to an individual project and may not have general application (Kentula 2000). Finally, and perhaps the most difficult to satisfy, is the need to be aware of the wider implications of individual projects and recognise success or failure of specific wetland restoration actions must be placed within a greater landscape (eg watershed or biogeographical region) perspective. These aspects are discussed below.

7.2.1 Engineering success

To the engineer tasked with the creation of saltmarsh and mudflat the use of dredge material can be an environmentally friendly approach to coastal and flood defence. Benefits are accrued from working sympathetically with the environment whilst maintaining a satisfactory level of flood protection. Saltmarshes, because of their height and vegetation cover, are significantly more effective at attenuating wave energy than mudflat. Furthermore, the latter, being more susceptible to erosion, is more difficult to sustain on an open foreshore.

Because the intertidal profile is principally being enhanced as a means to improve flood defence the indicators of a successful restoration are expressed in terms of the characterisation of good flood protection. These indicators include: low (or predictable) rates of lateral erosion, vertical accretion and vegetation cover on areas at elevations suitable for halophyte colonisation. These criteria can be assessed quickly and at low cost. Environmental factors such as creek development or bird and fish utilisation (which may be different to local communities) are often consequential secondary benefits and as such are often poorly monitored. Intertidal habitats created for engineering purposes have commonly been created with whatever material was available rather than with sediment grades that match the local intertidal environment - numerous studies detailed in Chapter 4 have shown that this affects the nature of the invertebrate fauna, and hence a knock-on effect on the bird fauna. However, where environmental benefits are maximised without increasing engineering costs, suitable grade sediments are increasingly being placed on the upper intertidal zones.

7.2.2 Functional success

Functional success is a measure of whether the ecological functions of the system have been restored. These functions include, for example, the ability of intertidal habitats to support food chains, to attenuate storm action and to improve water quality, etc. Whilst maintaining ecological functioning is the key to sustaining a healthy environment, a major challenge yet to be overcome is how to determine and quantify, given constraints of time and incomplete knowledge, the functions and values of natural and restored marshes. In the US, the basis of no-net-loss is used as an attempt to ensure that wetlands lost to development are replaced by those of equivalent functional value. This functional approach is increasingly being adopted within Europe as part of measures adopted under the Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora (92/43/EEC) (known as the Habitats Directive, DG XII 2000). More research is needed to establish methods of quantifying the functional value of coastal wetlands. In the meantime, we probably need to focus on structural characteristics and assume that a created site with similar plant, animal and microbial communities and similar sediment organic content will have broadly equivalent biological and biogeochemical functions.

7.2.3 Compliance success

Compliance success reflects the need to evaluate compliance within the terms of an agreement. Currently these agreements are drawn up to reflect the need for a specific habitat created, as part of either, a mitigatory or compensatory action to replace particular attributes of the degraded or lost habitat. It is the regulators role to ensure that compliance is achieved through the assessment of habitat enhancement or replacement schemes against success criteria. Within Europe, success criteria are defined site conservation objectives developed as a requirement of the Habitats Directive.

This directive was implemented into UK domestic legislation in The Conservation (Natural Habitats, &c.) Regulations 1994. If, after an appropriate assessment, a project is deemed to have a negative effect on a site of European importance, the project may take place only if there is an overriding public interest. In this case, Regulation 53 of the Conservation Regulations deems states that 'the Secretary of State shall secure that any necessary compensatory measures are taken to ensure that the overall coherence of Natura 2000 is protected'. The term 'overall coherence' is open to interpretation.

With regard to Regulation 53 compensation, the following success criteria have been identified:

i. Creation or enhancement of a site must not damage or alter any feature of existing conservation importance.
ii. Compensation involving the creation or enhancement of coastal wetlands should be compatible with existing coastal processes.
iii. Compensation should replace habitats on a 'like for like' basis.
iv. Compensation should provide an area of habitat at least the same size as that lost.
v. Ideally, compensatory habitat for wetland birds should be established in advance of habitats loss.
vi. Compensatory habitat should be located as close as possible to the area of lost habitat.
vii. Satisfactory compensation is achieved only if all birds displaced from the destroyed area can settle and survive on the new wetland.

The concepts behind these criteria are discussed further in section 7.3 - Operational considerations.

7.2.4 Landscape success

When considering siting a restoration project it is important not only to be aware of site issues (former land use, hydrology, substrate, topography, etc.) but also to place its ecosystem functioning of the desired restoration in the wider landscape context. Subject to environmental change (rising sea-level, climatic warming, etc), the most suitable place to restore a habitat is on a site at which it once existed. Even so, a number of questions should be asked before choosing a site for restoration (Crooks & Turner 1999). What will be the effect of any coastal realignment on the wider morphodynamics of the estuary or coast? Are ecological linkages and functional benefits maximised across the site and between sites? Will the project further regional biodiversity management goals, eg National Biodiversity Action Plan targets? How is land-use expected to change and will this affect the future integrity of the site?

Wetlands are also known to have greater biodiversity value if linkages are maintained with adjacent upland ecosystems. For instance linking saltmarshes to nearby dune complexes may increase water retention in restored marshes (Broome *et al* 1988). Certain bird species may utilise intertidal areas during the day and seek shelter in upland areas at night. Not only are adjacent habitats important, individual sites may form part of a network that support a particular function, eg the support of a flyway of migratory shorebirds.

The enhancement of a wide variety of ecological functions requires that restoration projects be planned incorporating a diversity of landscape types which are connected allowing the movement of materials between them (Bell *et al* 1997; Ehrenfeld & Toth 1997). However, because of our poor current understanding of these linkages, restoration objectives have rarely been considered beyond a site-limited context. A number of steps might be considered to maximise the contribution of a restoration project to ecological functioning, eg the creation of habitats which are known to be scarce or absent, but important in local ecological functioning or restoration of a range of different habitat types on large projects although this is likely to be dependant on topographical constraints.

Further to this, it should be considered whether restoration should be restricted to replacing habitat that is being lost from the contemporary landscape, or seek to reintroduce habitats that once existed historically. A common characteristic of many estuaries is the embankment and loss of freshwater tidal and upper intertidal habitats leaving a fringe of lower intertidal habitats. It is important to strive to create all coastal floodplain features including freshwater, brackish and other marshes above intertidal areas. Given sufficient evaluation, it may be that some flexibility is required within no-net-loss objectives to allow previously displaced habitats and their functions to be restored as part of any compensatory package to offset any projected losses of the existing intertidal ecosystems.

7.3 Operational considerations for creating and restoring intertidal wetland habitat as a compensatory measure

7.3.1 Siting compensatory areas

The siting of replacement areas as compensation for a loss is not easy as it involves a complex assessment of the likelihood of success (both in terms of physical and biological success), effects on surrounding areas, economic and political cost/benefits and legislative constraints. When considering siting a restoration project, it is important not only to be aware of site issues (former land use, hydrology, substrate, topography, etc) but also to place the functioning of the desired restoration in the wider landscape context.

i. Sites at which compensatory measures are to be carried out should not damage or alter any feature of existing conservation importance (and that the new area should support the diversity and numbers of birds displaced by the development).
ii. It is often more cost-effective and ecologically beneficial to restore a degraded habitat, rather than create one where it has never existed (Crooks & Ledoux 2000). For example, the restoration of a land-claimed flood plain is more likely to be successful than the creation of a new site through the engineered re-contouring of upland areas to form wetland basins.
iii. Since designated areas are limited geographically, the suitable locations for habitat restoration or recreation are likely to be few. Ideally, the aim should be to recreate the habitat on the same site. However, sites where restoration will yield the highest benefits are not necessarily next to the original site. For example, the placing of a new intertidal habitat in an area undergoing severe erosion may not be sensible and the new habitats may be better placed in areas where they are more likely to persist or support larger-scale geomorphological processes. Although this may have a negative impact on the affected site, regional or national biodiversity targets may be better served by creating new sites elsewhere where environmental, political and economic conditions are more favourable. For example, if the primary aim is the protection of wildfowl at a national scale then offsetting the large scale habitat loss on the south-eastern estuaries of the UK may be best served by recreating areas in The Wash where conditions are more favourable to large-scale intertidal habitat creation. However, creation of wetland sites further away from centres of human population may reduce the economic value of the displaced birds to humans, and human use of the sites as a result of their landscape characteristics may also be reduced.

7.3.2 Geomorphology and coastal processes

This review and numerous previous studies have shown that geomorphology and hydrology, both within the larger coastal setting and on site, are the key parameters in determining the form and function of a natural and restored wetland. Coastal realignment provides opportunities to enhance wider estuarine functioning, improve local flood defence needs and restore intertidal habitat but also has the potential to produce adverse affects. Breaching (as well as reclamation) or relocating flood defences has impacts beyond the site, influencing, to some degree depending upon the size and location of the project, the tidal regime of the estuary and the long-term distribution of intertidal sediments. Schemes involving managed setback, sediment recharge or any other activity that may affect coastal processes should not adversely affect coastal processes.

Figure 7.1 outlines the procedure for assessing the feasibility of saltmarsh restoration, from a large-scale geomorphological perspective. This is restricted to the consideration of saltmarshes, but the flowchart can readily be generalised to include other intertidal habitats. These procedures reflect the need to consider the wider impacts of realignment beyond site specific issues.

Firstly, a review of historical saltmarsh distribution is required. If saltmarshes have not developed naturally within the estuary then this suggests that, for whatever reason, restoration actions will be problematic. Secondly, it is important to determine how the volume of tidal water entering the restored site (the tidal prism) will impact upon wider estuarine morphology. It may be that by increasing tidal prism in a section of the estuary will lead to further erosion or accretion elsewhere. If wider negative impacts can not be prevented then the project should be relocated and the site rejected. Thirdly, by and large, it is better to restore an intertidal habitat on a site where once one existed and so appropriate historical assessment is required. The fact that if saltmarshes or mudflat are found not to be formerly present at a site suggests that conditions (for instance such as wave energy) are unsuitable for habitat development. Fourthly, the shape of the restoration site has impacts upon estuarine hydraulics. Given that the natural shape of an estuary typically involves a wide floodplain hinterland adjacent to a central, migratory, channel system then these natural hydraulic features should be encouraged. Then finally, the site specific issues should be considered taking into account the topographical requirements required for habitat establishment.

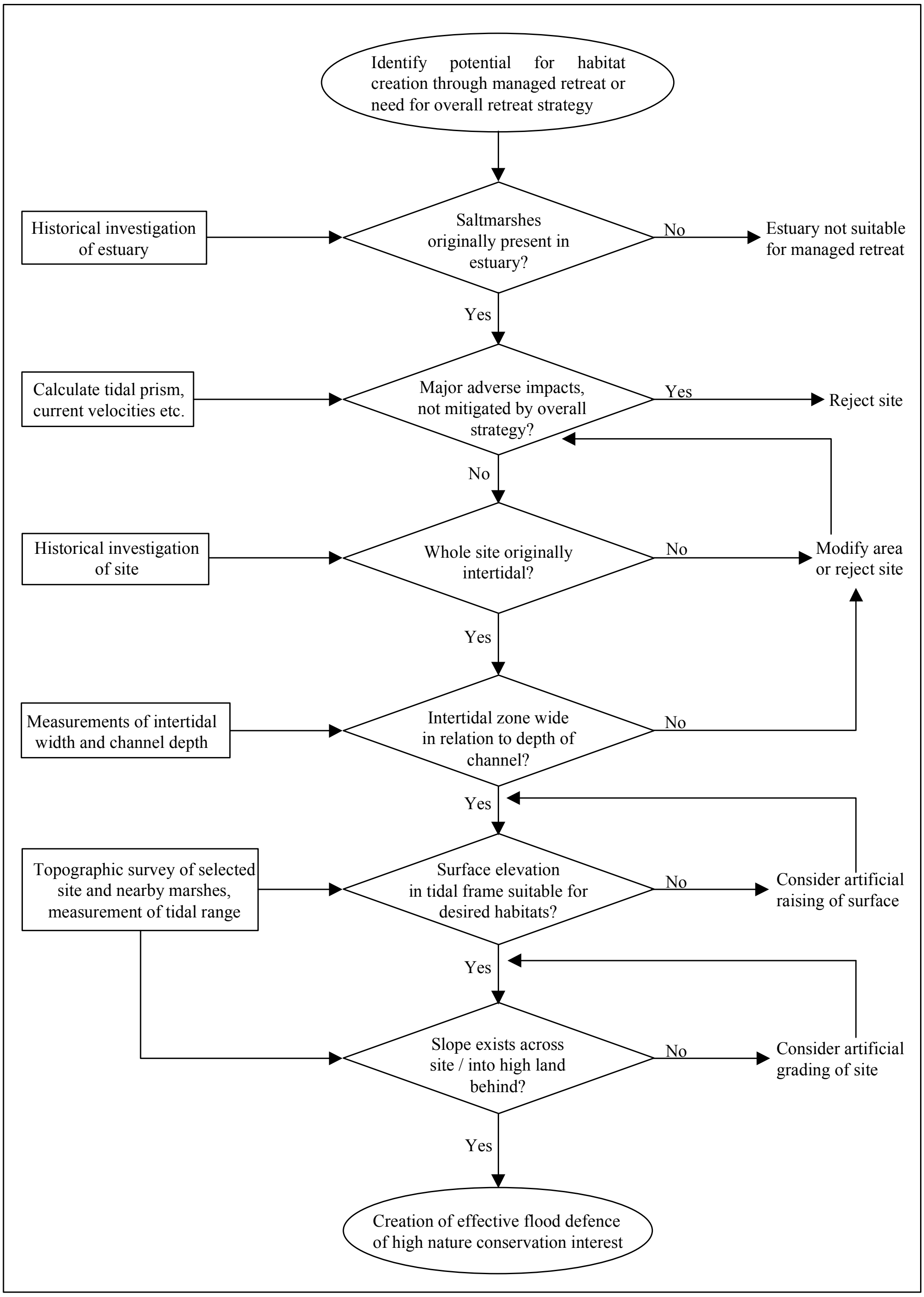
7.3.3 Is like-for-like compensation sufficient or should schemes over-compensate for any loss?

The concept of replacing lost habitat on a fully equivalent like-for-like basis is attractive but often impractical due to the uncertainty of the restoration outcome. From this global review of intertidal restoration actions, it is clear that it is possible to restore some form of intertidal habitat but may not be possible to reinstate the exact specific forms and range of functions of an established habitat, especially within a short time frame. To achieve successful habitat restoration the project must be considered in the context of space, time and degree of uncertainly (risk). This often leads to the need to over-compensate and aim to produce habitat that is either of better ‘quality’ or a greater area, or both.

Setting of mitigation ratios (ratio of the area of land created to that lost) should therefore reflect time, spatial and risk uncertainty issues. For instance, the use of dredge material to create saltmarsh tends to produce habitats of different form and function character from natural marshes. To accommodate this a high mitigation ratio should be set to minimise the loss of certain functions.

Time and space constraints often determine the magnitude of mitigation ratio but the nature of these constraints are often poorly understood. Restoration for vegetation alone can be undertaken with a very long term perspective. For example, saltmarsh vegetation on naturally regenerating marshes in south-east UK may still be different from natural marshes despite 100 years of evolution. It may be that some functions take a long time to fully establish but since these are required quickly to replace those lost, then a large mitigation ratio should be demanded. In the long-term (decades to hundreds of years) it is possible, but uncertain at the time of restoration, that the larger site will produce services and benefits the value of which will outweigh the loss of the natural site. A higher mitigation ratio may also increase the likelihood of the success of the restoration scheme although this is by no means guaranteed.

In cases where the loss involves a site of critical importance for a particular species, restoration goals may require very rapid results through creating small areas of high quality habitat, perhaps with active site management to seek to ensure the survival of the species concerned.



SOURCE: Institute of Estuarine and Coastal Studies, University of Hull

Figure 7-1 The procedure for assessing, from a large-scale geomorphological perspective, the feasibility of saltmarsh restoration

For habitats other than saltmarsh, a slope into high land is not required.

7.3.4 Timing of restoration

Many wetlands take considerable lengths of time to develop naturally, often from other landforms (eg sub-tidal, mudflat, marsh evolution). The form and function of the habitat reflects this change and is often based upon the 'imprint' of the former landform structure. The use of dredge material can accelerate this process but, in doing so, those time-dependant characteristics are overprinted and the form and function of the dredge material marsh will most likely differ from those of adjacent natural marshes.

Mitigation banking (see Box 7.1) improves the likelihood of a long-term successful restoration because the site is created in advance of the anticipated loss. There are regulatory, ecological (landscape linkages and economies of scale) and economic benefits of mitigation banking. There is however, a strong need for a clear policy framework (particularly with reference to the Habitats Directive) before this approach will become feasible (Crooks & Ledoux 2000).

Box 7.1 Mitigation Banking

In this context, a mitigation bank is a moderate size to large wetland restoration, creation or enhancement project undertaken by a single developer (public or private) or a consortium of developers not only to compensate for wetland impacts from a particular project but to act as a "bank" with credits to compensate for *future* wetland projects and impacts. Thus, the basis of the banking approach is the setting up of a *credit market*, with restored wetland values and functions being quantified as credits, which are deposited within an account, and later purchased by developers when regulators require compensation for authorised losses of habitat functions.

Overseen and regulated by a Mitigation Bank Review Team a mitigation bank offers a number of advantages over individual mitigation projects.

- Clarification of habitat substitution opportunities in that the restored habitat already exists or is in an advanced stage of restoration when required. Therefore the failure rate is reduced.
- Consolidation of small scale mitigation projects (so ensuring that incremental losses are accounted for).
- Higher ecological benefits from large scale restoration.
- Economies of scale: financial and regulatory.
- Ability to strategically plan restoration/creation schemes to meet national, regional or local biodiversity targets as opposed to merely reacting to individual site loss.

Overall, mitigation banking seeks to address some of the shortcomings of site-by-site mitigation, and proper implementation can help to avoid some of the past mistakes; however there are still some drawbacks related to it. A criticism of mitigation banks is that because they pool the mitigation needs of several sites, the wetlands lost on individual sites are not replaced at the location of the impact. Care must also be taken to ensure that the mitigation bank created is afforded adequate protection from development in future years.

7.3.5 Biodiversity targets

Within the compensatory framework of European law, it is necessary for each government to maintain the integrity of Natura 2000 sites. In essence this means that any of these habitats lost have to be replaced. The biodiversity interest contained within the part of the site to be lost is normally determined pre-loss and if possible monitored over several years to determine the range of inter-annual variation. Targets can be set in terms of both the type and extent of new habitat to be created as well as establishing populations of individual species or communities of species that (a) persist and (b) occur in densities that are within boundaries that are considered normal for that species in that site.

However, becoming increasingly widely adopted within environmental management is to plan beyond individual species and on the scale of (a) regional biodiversity targets at various scales and (b) landscape elements and biogeographical regions. Biodiversity targets may be better served through mitigation banking schemes as habitats are created in advance and possibly larger-scale than compensatory measures currently demanded under Regulation 53. Mitigation banking is not currently compatible with current European or domestic law.

Further to biodiversity targets, an extension to this approach has focused on the concepts of ecosystem ‘health’ and/or ecosystem ‘integrity’ incorporating human values with biogeophysical processes. There has been difficulty in arriving at a consensus definition of these concepts but most attempts include the belief that healthy ecosystems are ‘stable and sustainable’ and able to maintain themselves over time, displaying resilience to stress.

Critics of the ecosystem health and integrity approach to defining management goals point to the fact that ecosystems are constantly changing and do not exist in a ‘stable state’ and as such it is not possible to define an optimum condition for ecosystem preservation (Wicklum & Davies 1995). Some conservationists are also doubtful about a management approach which permits the loss of individual species as long as given ecosystem functions are not greatly altered, arguing that monitoring of certain keystone species or measures of species diversity, independent of species identity, adequately reflect the health of an ecosystem. Nevertheless, the concept of ecosystem health or integrity, interpreted broadly, is useful in that it helps to focus attention on larger systems in nature and away from the specific interests of individuals and groups.

7.3.6 Restoring intertidal areas with dredged material?

In the US early marshes created with dredged material were designed to have a static elevation. It has become increasingly recognised, over the last decade, that marsh restoration techniques should seek to accommodate a more dynamic design, incorporating plans for the evolution of geomorphology and to foster natural sedimentation (Zedler 2000). This is probably even more important for UK sites where the supply of fine sediment is usually sufficient to lead to marsh accretion if the hydrodynamic regime allows it.

A number of questions should be asked: (1) how quickly will the site naturally accumulate sediment? (2) How quickly will dredged material consolidate? (3) How compact will the consolidated material become? (4) What will be the impact of sedimentation rate and/or dredge consolidation be on the drainage characteristics of the marsh (including creek density) and how will this affect vegetation cover and desired environmental functions?

By and large, dredged material marshes which have been created too high in the tidal frame consolidate to form tabular units of high sediment strength with limited surface topography or slope and poor or absent creek drainage systems. While this may create a favourable attribute for flood defence it does not provide habitat of comparable form and function with those of local natural marshes. A number of options towards addressing the problem are available all of which involve actions to create a surface of varied topographical relief:

1. The elevation of the dredge material surface can be calculated, on dewatering, to fall well below that required for the formation of a saltmarsh. In this way the site will be covered either by a mudflat or subtidal sediments (depending upon surface elevation) from which, given suitable hydrodynamic and sedimentary conditions a marsh with a creek network will evolve (see Faber Marsh case study). In terms of the long-term development of such a site reference to local natural unmanaged intertidal restorations in the region will provide some guidance as these have commonly developed on low intertidal surfaces with little surface topographical relief.
2. Once dewatered the compacted surface may be modified to create a chosen surface relief upon which accreting marsh or mudflat sediments may develop a creek network as dictated by the underlying topography. Again, problems of drainage may ensue if the underlying surface is of a high density and an artificial creek network may require excavating.
3. Dredge material of high bulk density and viscosity may be placed upon a sloped surface. Dewatering of this material may then result in the new marsh adopting a form of topographical relief defined by the underlying topography (see North Shotley, Horsey and Trimley examples in section 3.4.3).
4. In overly consolidated sediments there may be a need to engineer a creek network system. In designing the planiform and cross-sectional morphology of creeks due consideration to the laws of hydraulic geometry should be given (Steel & Pye 1997; Zeff 1999). The construction of a creek network which mimics the natural range of creek forms will result in other environmental benefits such as fish utilisation via and diversity of invertebrate and bird utilisation. (eg Williams & Zedler 1999), although more experimental work on the engineering of creeks is required.

7.3.7 Management and monitoring of creation/restoration projects

Essential to reaching the targets of any habitat creation scheme are comprehensive success criteria and an effective management process and monitoring scheme. These have often been lacking or poorly carried out in past schemes, although considerable effort has been expended in monitoring some of the more recent managed retreat sites in the UK.

7.3.7.1 Comprehensive success criteria

These are required so that the goal of the creation/restoration is clear and is not open to misinterpretation (see Redmond 2000 for a discussion from a US perspective). These should be measurable and objective and the means of evaluating them should be simple and repeatable and result in monitoring that is representative of the site. They should also take into account the appropriate time needed for restoration.

Despite these recommendations, in the UK, there has been little public discussion or critical assessment of either the monitoring programmes carried out or the overall success of habitat creation/restoration schemes. Experience in the US indicates that schemes fail for a variety of reasons, and there is no expectation that things will work differently if there were a large increase in habitat creation/restoration projects for mitigation purposes in the UK and elsewhere in the EU.

There are some major research needs, covering both methods of habitat creation and methods to compare the functional equivalence of created and natural habitats. In particular, there is a need to identify the key parameters to measure to evaluate whether success has been achieved.

7.3.7.2 Monitoring

Monitoring is an important part of the compensatory process and as such should be planned pre-loss, especially for large projects. Specific desirable criteria for monitoring schemes for invertebrates and birds are summarised in Table 7.1. This summarises key desirable features, although these may be altered to the specific case. Important in any monitoring scheme are standardisation, replication and sufficient power to determine when success criteria have been met. Behind these features are a desire to monitor the characteristics of the invertebrate and bird assemblages that determine (a) base line information, (b) determine when the success criteria have been reached or whether they need to be amended and (c) provide information on effects of the habitat loss, and replacement, on the bird populations.

Although not detailed in this report, the same general issues apply to the monitoring of vegetation and other taxa. This should include site use by mobile organisms (fish and crustacea) during high tide. To establish the functional equivalence of a created habitat, monitoring would also need to encompass microbiology and bio-geochemistry of sites, both within created sites, but also including linkages between the created site and surrounding upland and marine habitats. To date little research has been carried out on this aspect and collection of basic information is necessary to determine (a) the key geo-chemical and microbiological parameters to be measured and (b) the scale of the linkages on 'natural' saltmarshes between upland, marsh and marine habitats. Collection of baseline information is required before clear monitoring guidelines can be outlined.

7.3.7.3 Management of the project

Each project will have different sets of criteria for determining success. Figure 7.2 describes a rolling management process through which success can be determined. It highlights the importance of pre-loss monitoring to determine the key features and functions of the site to be lost and the magnitude and variation in those features, whether they be sediments, plants, invertebrates or birds. From this, the scope of the restoration can be decided upon. This will depend to some degree on whether all or a proportion of functions need to be replaced and whether mitigation should take place on site or not. In some cases hydrodynamic or geomorphological circumstances may prevent it. It may be desirable to take an experimental approach to restoration to further understand our understanding of the processes involved. In either case, clear measurable criteria for success need to be drawn up and also a time frame by which these will be achieved.

Once the works have been carried out, a process of monitoring is and an adaptive management strategy through which regular assessments followed by additional corrective works (if necessary). It is important that monitoring and data analysis are carried out by appropriate competent organisations, using the best available methodology.

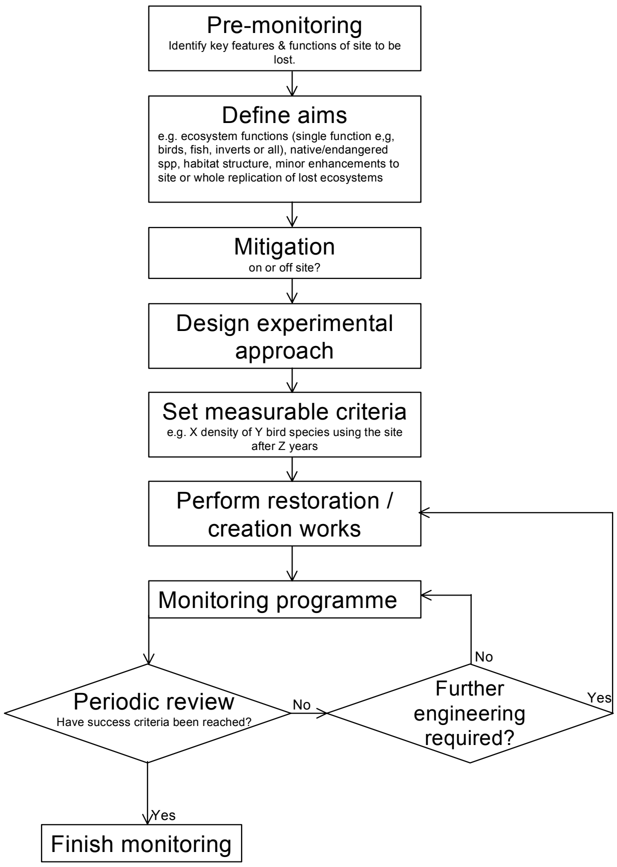


Figure 7-2 A management process through which functional equivalence can be measured

Table 7-1 Desirable features of invertebrate and bird monitoring schemes for mitigation projects

General Concept	Invertebrates	Birds
Pre-loss monitoring undertaken to establish base line	Determine abundance and annual/seasonal variation	Determine abundance at different seasons and tidal heights
Comparisons to be made with control areas	Control and created marshes to have similar geomorphic and hydrodynamic properties and be close to created area	Counts carried out at standard tidal heights or through a tidal cycle, for similar lengths of time at regular intervals throughout the year
Monitoring protocol to have sufficient statistical power to determine whether success criteria have been met	Power Analysis of pre-loss monitoring data will determine the sample size and frequency of sampling necessary	
Use suitable methods & statistical analysis	Use a replicated design in created and control areas	Use a replicated design in created and control areas
	Invertebrates to be sieved through a suitable mesh size (eg 0.3 mm will collect small individuals)	Counts to be regularly made at high and low tides during different seasons
	Measure environmental variables (height, mechanical properties of sediment) in an attempt to explain invertebrate distributions	Measure environmental variables & densities of potential food sources to assist in explaining invertebrate distributions
	Identify to species level where possible & size distribution of invertebrates	Radio-track and colour-ring individuals both pre- and post loss to determine effects of loss on displaced birds
	Ensure sampling programme effectively covers larger invertebrates (eg larger bivalves)	Follow colour-ringed birds to monitor survival in restored and natural areas
Regular reviews undertaken by competent organisations		
Monitoring to occur over a realistic time frame to determine when ecosystem functions have reached an equilibrium		
Publish the results in the peer-review literature		

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Appendix Changes in the number of birds at two managed retreat sites on the River Blackwater, Essex

A.1 Introduction

The Tollesbury and Orplands managed retreat sites are unique amongst British managed retreat sites in that standardised bird monitoring took place on the sites since intertidal inundation was restored. These surveys have allowed the changes in the bird community to be tracked over time and may provide useful insights into the speed of colonisation and total usage by waterbirds of newly created intertidal habitats. Both of these sites include saltmarsh and mudflat but are enclosed and sheltered by the remains of the old sea wall and thus may not be indicative of open coast mudflats. Bird monitoring has also been carried out on five sediment recharge sites in Suffolk and Essex which were created as part of the Harwich Haven dredging operations (Table 1.1) but these data are not in a form that can be analysed with any confidence. Counts have taken place at different times of tide, for different periods of time and not all sites have been covered during each survey period. We have therefore not included these in any further analyses.

Orplands and Tollesbury have shown similar patterns of development. Saltmarsh vegetation has developed on the higher elevations of the sites and is dominated by pioneer *Salicornia* spp. communities and areas of intertidal mud have become established on lower areas. At Orplands, there are two sites (A and B). Site B is higher than site A and has always had less bare mud and was quickly vegetated by *Salicornia*.

At Tollesbury, data were collected by Chris Tyas (RSPB Old Hall Marshes) three times each month from October 1995 to September 1999. At Orplands, monthly counts from November through to March were carried out by the Environment Agency from November 1994 to March 1998. At both sites, the size and state of tide was standardised as far as possible, although this proved difficult at Orplands. At Tollesbury counts were carried out in each calendar month and at three states of tide (low, neap high and spring high) whereas, at Orplands, counts were carried out during the winter months for approximately three hours from half an hour after low water. Counts at Orplands were carried out at approximately 1-1.5 hour intervals until the intertidal mud was covered.

This chapter analyses these data and quantifies the changes in numbers of individual species and also the community composition of the birds found on these sites. We determine whether the bird community has stabilised after four to five years of monitoring or if it is still evolving.

A.2 Methods

A.2.1 Bird usage at the Tollesbury managed retreat site

Counts of birds in the retreat site were made at low water, neap high tide and spring high tide during each calendar month from October 1995 to September 1999. Similar counts from ‘control’ areas were not made. Since September 1999, the methods used have changed and involve a count during one tidal period per month. Due to the difficulty of drawing comparisons, only the first set of data have been analysed. From May 1998 onwards, birds were noted as feeding or not feeding.

Two analytical approaches have been taken. The majority of usage by birds occurred during the winter (October to March inclusive) and data analysis has been restricted to these months only. Additionally, although Lapwing and Golden Plover, tend to use the site as a roosting area, the number of feeding and roosting birds was not available for all counts. Therefore, all counts, irrespective of activity were included. First, the factors affecting the winter usage of the site by individual species were determined using generalised linear models modelled using the SAS GENMOD procedure (SAS Institute Inc. 1996). Bird count data were modelled as a function of three factors: winter (1995, 1996, 1997 or 1998), month (October through to March), tide state (low, neap or high) and one linear variable: disturbance (1 = None, 2 = Low, 3 = Moderate/High). Likelihood ratio tests were carried out to determine whether significant differences occurred between the numbers of birds in different categories of a specific factor (eg differences between years, months or tide states).

The second approach was to determine changes in community composition over time and tide states. Correspondence analysis (CA) was performed on total bird usage data, split by winter and tide state using the computer package CANOCO (ter Braak & Smilauer 1998). Species which occurred less than five times were excluded and a 1 + log(bird usage) transformation was performed to reduce the effect of rare species or very large counts unduly influencing the analysis. Axis 1 and Axis 2 species and sample scores were plotted on a bi-plot diagram and trends in species composition were inferred over time and tide state. The bi-plot was scaled with a focus on species distances so that the resulting diagram depicts most accurately the differences between occurrence patterns of species, and the samples in which they occur are scattered around them. In simple terms, species that are close together in the plot tend to have occurred together and the samples with which they are associated are placed near them.

A.2.2 Bird usage at the Orplands managed retreat sites

Five different areas were counted at Orplands which included both the retreat site, surrounding mudflat and control areas (Map A.1; Table A.1). Counts were made from half an hour after high water for approximately three hours until the mudflats outside the retreat site were covered.

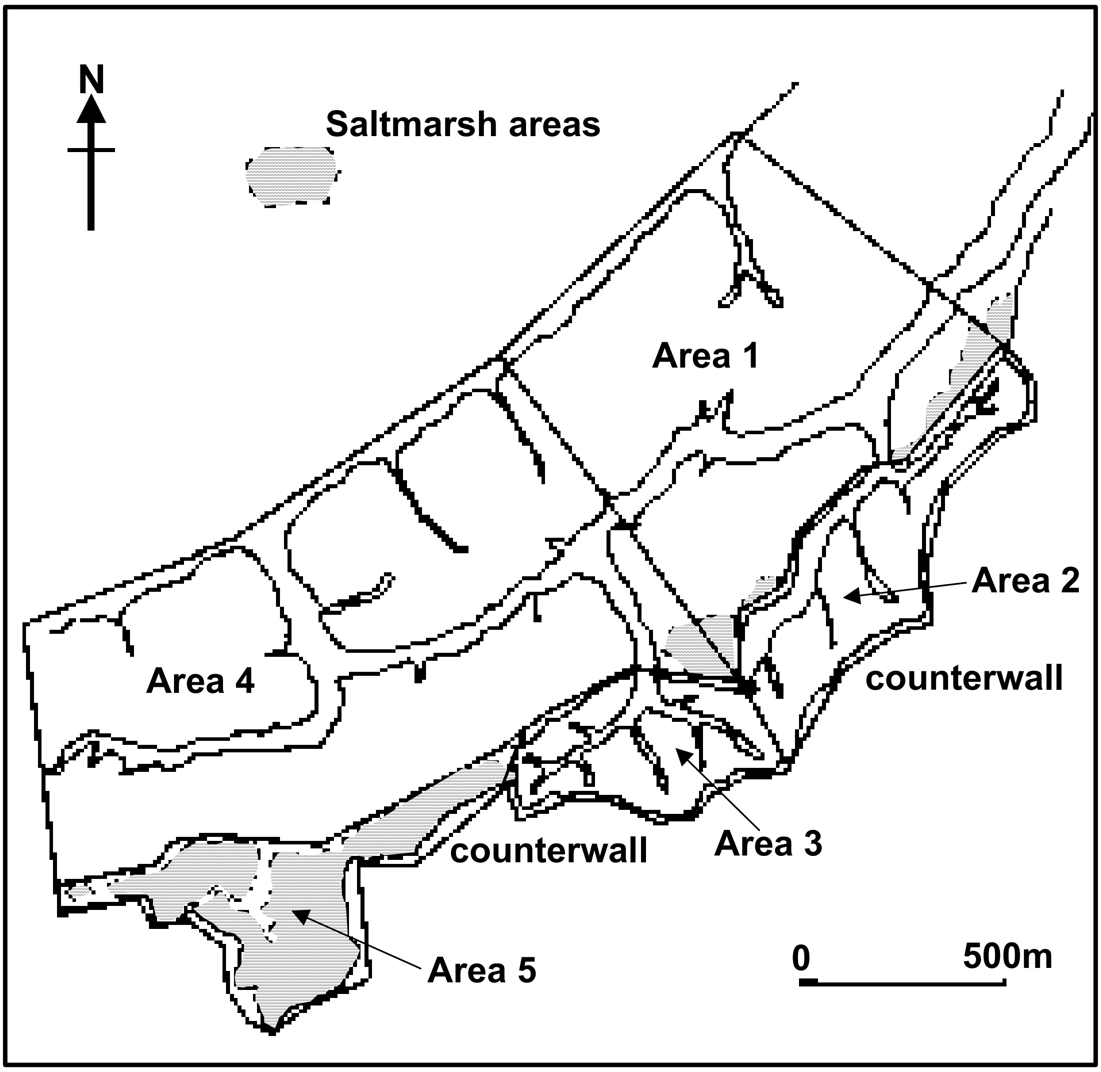
The Orplands site was difficult to cover and the number of counts carried out on each area varied between visits and, by necessity, were carried out at different states of tide. Counts were also carried out in some months during one winter and not the following winter. This has led to difficulties in analysing data and analysis has therefore been limited to analysing data collected from December through to February. An attempt to analyse species data using GLMs as above failed due to small quantities of data and so only groups of species (total number of shorebirds and wildfowl) were modelled as a function of site type (mudflat, Orplands A, Orplands B and saltmarsh) year, month and time from low water. Detailed statistical analysis for individual species has therefore not been carried out. The mean (± SE) number of birds recorded per visit was calculated for each species which had a mean count of greater than five birds during any winter. Birds have been expressed as numbers rather than densities as it was not possible to determine the extent of the mudflats at any state of tide and the area of mud on each retreat site (A or B) changed between winters.

To assess changes in the bird assemblage using the site, Detrended Correspondence Analysis (DCA) was carried out on the mean count per winter in each site. DCA, rather than CA, was used to reduce a pronounced arch effect which occurred using CA (ter Braak & Smilauer 1998). Rare species (recorded on less than 10 occasions) were excluded and no transformation of the species data was carried out.

Table A-1 Description of the five different areas counted at the Orplands managed retreat site

Description of the five different areas counted at the Orplands managed retreat site

Area Number	**Description**
1	Mudflats seaward of managed retreat sites
2	Managed retreat site A
3	Managed retreat site B
4	Mudflats seaward of control saltmarsh
5	Control Saltmarsh



Map A.1 Map of the Orplands study areas. Area 1 - mudflats seaward of the managed retreat sites; Area 2 - Orplands A retreat site, Area 3 - Orplands B retreat site, Area 4 - mudflat seaward of the control saltmarsh; Area 5 - Saltmarsh. (Source: Environment Agency).

A.3 Results

A.3.1 Shorebird usage of the Tollesbury site 1995 to 1999

Twenty species of shorebird were recorded on the retreat site during the four years, six of which were recorded on five or fewer occasions. These include Avocet (1 record), Curlew Sandpiper (1), Green Sandpiper (4), Common Sandpiper (4) Ruff (1) and Bar-tailed Godwit (5) and are not considered further in this analysis. In numerical terms, Golden Plover, Redshank and Dunlin were the most common with over 3,000 bird months logged. Grey Plover and Lapwing followed with 600-700 bird months and, of the remaining species, only Knot, Black-tailed Godwit and Curlew logged more than 100 bird days.

Sufficient data was available to construct models for seven species of shorebird (Table A.2; see also Figure A.1). Dunlin and Redshank showed similar patterns of annual usage; numbers were initially low in 1995 but significantly increased the following year and numbers remained stable to 1998. Grey Plover and Curlew colonised in the first year and showed no significant changes in usage over the four winters. The remaining two species, Lapwing and Golden Plover, tended to use the Tollesbury site predominantly for roosting rather than feeding area. Golden Plover, were not present in significant numbers on site in 1995 (the winter following the breach) but colonised and peaked in usage during 1996/97. There was a significant decline the following year but subsequently rose in 1998/99 to levels similar to 1997/98. From 1995, Lapwing rose to a peak in 1997 and then declined the following year.

Although the site is small, comparisons can be made of trends in numbers with the surrounding Blackwater estuary (see dashed lines in Figure A.1). None of the species for which data were available, with the possible exception of Redshank, showed similar patterns of usage between years, suggesting that the suitability of the retreat site has changed at a much faster rate than the estuary as a whole.

With the exception of Snipe and Knot, a pattern of low usage in 1995 followed by an increase in 1996 can be observed for many other species. After 1996, patterns vary. For example Ringed Plover usage increased each year whereas species such as Black-tailed Godwit, Greenshank and Spotted Redshank have declined from peaks in 1996. Oystercatcher usage remains very low. One interesting pattern is the colonisation by Knot three winters after the breach and the dramatic increase during the fourth winter.

In terms of the monthly changes in numbers, some species such as Grey Plover, Knot, Dunlin and Curlew show a similar monthly distribution to those seen in the monthly WeBS counts for the whole Blackwater Estuary (Figure A.1) whereas Oystercatcher, Ringed Plover, Black-tailed Godwit and Redshank do not. The former two species rarely use the Tollesbury site whereas Redshank showed a delay in using the site. Although present in large numbers on the wider estuary from October onwards, birds did not show a high usage of the retreat site until December, a time when energetic demands may be high.

Numbers of birds also tended to be higher during low and neap high water counts rather than spring high tides when most of the intertidal mud was covered.

A.3.2 Usage of the Tollesbury site by other waterfowl

Thirteen species of grebe, cormorant, heron, swan, goose, duck and moorhen occurred on the site during winter months, although only nine species used the site on more than five occasions. These included Little Grebe, Grey Heron, Brent Goose, Shelduck, Wigeon, Teal, Mallard, Pintail and Moorhen. Other species recorded included Great Crested Grebe (2 records), Cormorant (4), Mute Swan (2) and Red-breasted Merganser (4).

Sufficient data were available to construct models for three species. Little Grebe colonised the site in the first winter and have declined since whereas Dark-bellied Brent Goose and Shelduck colonised in the first year and numbers did not change significantly between years. Only Brent Geese showed any significant relationship with the tide state and usage was highest on neap high tides.

For other species, trends can be inferred from Figure A.1. Dabbling duck (Pintail, Mallard and Wigeon) usage has tended to decline since the first year although Teal have shown an erratic change in numbers, being absent in 1997.

Teal used the site as a roosting area, Brent Geese predominantly as a feeding area and Shelduck both fed and roosted in the area.

A.3.3 Usage of the Tollesbury managed retreat site by passerines and near-passerines

At Tollesbury, 31 species of passerine or near passerine were recorded. Five common species that provided over 87% of the total number of winter records. These species all peaked in the first year after the breach and then showed a large decline in future years (Figure A.2). With the exception of Meadow Pipit, the other common species were granivorous and most were observed feeding amongst the debris washed up on the high tide line. Smaller numbers of other species such as Greenfinch, Chaffinch and Yellowhammer also took advantage of the large amount of washed-up debris.

A.3.4 Changes in the bird assemblage using the Tollesbury plot

The ordination plot of Axis 1 and Axis 2 scores shows a clear temporal as well as tidal separation between species and samples (Figure A.3). Axis 1 describes a gradient with 1995 data on the right hand side (all tidal states), spring tides in the middle and 96-98 neap and low tides on the left hand side. There is also a clear divide in species groups with shorebirds and gulls associated with 96-98 low and neap high tide samples and wildfowl and passerines associated with the first year after inundation.

Since 1996 there has been no clear change in axis scores although the mean axis scores for 96-L and 96-N are higher than those for corresponding tidal periods in 1997 and 1998 indicating that the assemblage of birds continued to change after 1996. This is supported by the presence of Knot and Ringed Plover and Herring Gull all of which have continued to increase since this date.

Axis 2 is more difficult to interpret and generally shows much less variation compared with Axis 1 although 96-L appears as an outlier, presumably due to the relatively high usage made of the site by Spotted Redshank during this period. It has not been possible to interpret fully this axis but may be due to 'natural' variation in waterbird numbers.

In conclusion there were large changes in the species of birds that used Tollesbury between years one and two, smaller changes between two and three and little change between years three and four, although as indicated above individual species, such as Knot and Ringed Plover showed increased usage. Only a proportion of the species found on the Blackwater estuary were observed feeding and roosting in this area.

A.3.5 Bird usage at the Orplands A & B managed retreat sites and surrounding habitats

In total, 59 bird species were observed across all habitats during the December to February counts. In total, 35 species were recorded on the mudflats and 35 on the two retreat sites. Twenty-one of these species were common to the two habitats. Eighteen species were observed in the saltmarsh habitat.

Fewer waterfowl and more terrestrial passerines used the retreat site compared with surrounding mudflat (Table A.3). The waterfowl species missing from the retreat sites tended to be aquatic species such as Cormorant, Eider, Red-throated Diver and diving ducks although gulls, which are often associated with saltmarshes, were absent on retreat areas. The passerines present on the retreat site were more indicative of terrestrial rather than saltmarsh habitats (eg Blue Tit, Dunnock, Robin, and Song Thrush) and most of these species occurred during the pre-breach survey in 1994.

Compared with Tollesbury, more species were recorded overall but fewer were recorded frequently. The first species to use the site were those that prefer fine mud sediments. Redshank, Grey Plover and Dunlin colonised in the winter following the breach and have increased during the course of the study (Figure A.4). Knot first appeared during the 1996/97 winter and their usage of the site increased steadily to 1998/99. Oystercatchers, which are more typical of substrates with less mud (Table 4.2), were not observed on either of the sites during the course of the study.

Brent Geese and Shelduck were the only two wildfowl species to regularly use the site. Shelduck usage has remained approximately equal since colonisation in the 1995/96 winter and has occurred in numbers of up to 230 in the retreat site.

There were insufficient data to attempt detailed statistical modelling of the effects of time since breach, site (Orplands A or B, mudflat or saltmarsh) and the interaction with time of tide on total bird usage of the site. This was due to the large number of zero counts in the data, which violated assumptions of the General Linear Model. However sufficient data were available when considering total numbers of shorebirds (Figure A.4). Shorebirds showed a significant increase in abundance over time in Orplands A site and very low usage in

Orplands B. Usage of the surrounding mudflats was lower during the first two winters of counts and higher during the last three winters. Tide and its interaction with site type (mudflat or retreat site) were a significant factor and indicated that the number of birds on the mudflat decreased with increasing time from low tide and the reverse occurred in the retreat sites. This indicates that the retreat sites provide extra time feeding at high tide and most usage occurred during the time when the surrounding mudflats were covered.

There was a significant increase in the total number of shorebirds using Orplands A from none pre-breach to an average of approximately 200 birds in 1998. A similar pattern was observed amongst wildfowl, with a general increase to 1997 followed by a fall in numbers in 1998. This was due predominantly to the decline in the number of Shelduck using the area. Comparatively few shorebirds and wildfowl used Orplands B which was higher in the tidal frame and quickly vegetated over.

A.3.6 Changes in the bird communities at Orplands

There is a clear habitat and temporal difference in the assemblage of birds occurring at Orplands (Figure A.5). Axis 1 of the bi-plot explains most of the variation in the assemblage and runs from a shorebird-dominated assemblage on the right hand side through to one dominated by terrestrial species on the left hand side.

There is a very clear grouping of habitat types. Saltmarsh sites (coded SM on Figure A.5) and the Orplands fields pre-inundation (A94, B94) had low axis scores and Yellowhammers, Blackbirds, Skylark, Linnets, Reed Bunting and Meadow Pipits are associated with them. During the following years, Orplands A (A) moved towards a similarity with the mudflats (M, MC) and Orplands B (B) maintained an intermediate position between terrestrial and waterbird dominated assemblages.

In the 1995/96 winter, the year following the breach, both retreat sites showed a large increase in Axis 1 score but Orplands B fell back the following winter as the site vegetated over. Orplands A continued to increase at a much slower rate to the end of monitoring in 1998/99. In terms of Axis 1, the two mudflat sites show similarities with each other and little consistent change between years. In 1994, the two retreat sites had very similar assemblages that were made up predominantly of passerines.

A.4 Discussion

A.4.1 Temporal changes in the bird and invertebrate assemblages

The trends in bird numbers have been remarkably similar at both Orplands and Tollesbury, which may not be surprising as they are located on the same estuary. Since the sea defences were breached, both sites have developed areas of both mudflat and pioneer saltmarsh. Orplands A and Tollesbury are low in the tidal frame (Reading *et al* 1999) and have experienced rapid accretion since the breach (Environment Agency 1999; Reading *et al* 1999). This has led to the build up of soft muddy sediments at the seaward edge of the retreat site, which have been colonised by invertebrates (Reading *et al* 1999). The increase in invertebrate numbers in these sites has broadly been in line with the predictions made in Chapter 4. Mobile species, and those that have a planktonic larval phase such as *Nereis* and other polychaetes, and *Hydrobia* have colonised these muddy sediments and bivalves and other species that have no planktonic larval phase, such as oligochaetes, have either not colonised or did not appear for a number of years (Figures A.6 & A.7).

The first benthic invertebrates to colonise Tollesbury in appreciable numbers were *Hydrobia ulvae*, *Macoma balthica*, *Eteone longa*, *Nepthys hombergi, Nereis diversicolor*, *Pygospio elegans*, *Spio filicornis* and various unidentified oligochaetes. In following years, species such as *Mya arenaria* and *Abra tenuis* colonised. *Macoma* spread across the site and increased rapidly in numbers during the fourth winter after the breach. Fewer species occurred at Orplands but *Hydrobia* and *Nereis* continued to increase and were consistently the most common and widespread species in both sites. Small individuals of these two species initially colonised Tollesbury but increased in size during the following years (Reading 1999). However during the winter 1998/99, four winters after the breach, *Nereis* were still significantly smaller in the setback site compared with those in the surrounding control mudflat, whereas *Hydrobia* were significantly larger. Species diversity in the Tollesbury retreat site was initially low (14 species) in 1995 but increased to 19 in 1998. Compared with the surrounding mudflat (11-13 species), this is higher and probably reflects the greater diversity of sediment types within the set-back area as the control site was very small in area when compared with any of the realignment sites. For comparisons of bird assemblages, this is not a good control site. Control sites should ideally be larger and include a number of independent replicates.

The waterfowl that exploit these newly created retreat sites are typical of the sediments and invertebrates that have colonised the areas. There were five common species that used the two retreat sites in the years after breach. Brent Geese have exploited the build-up of algae on the sites whereas Shelduck, Dunlin, Grey Plover and Redshank are likely to have exploited the polychaetes and *Hydrobia* that initially colonised the sites. Common amongst the major prey items of Grey Plover, Dunlin and Redshank are *Nereis*, *Hydrobia* and *Macoma* which, by 1998 were in the top four most widespread intertidal invertebrates in the Tollesbury retreat site. Knot started to use the sites as bivalves, in particular *Macoma*, colonised. The large increase in Knot coincided with the large increase in *Macoma* in 1998. Other species such as Oystercatcher, which feed on larger bivalves, tended to show very low usage of the site.

Intertidal areas provide roosting as well as feeding areas and Tollesbury has been used by large numbers of Golden Plover and Lapwing. Although we are unable to comment on how critical the provision of a safe roosting area at Tollesbury is (birds may be able to roost elsewhere), large numbers of birds are using it.

There are often high densities of passerines breeding and wintering on United Kingdom saltmarshes although these vary seasonally, between saltmarsh habitats and between regions (Brown & Atkinson 1996; Kaljeta-Summers 1997). The community of passerine birds using the Tollesbury site in the first winter was atypical compared to two other United Kingdom saltmarsh sites in that Corn Bunting and Goldfinch were common and other species found on saltmarshes such as Twite, Snow Bunting, Greenfinch and Linnet were either absent or less common. In the following years, the passerine community diversity was lower and was dominated by Meadow Pipits and Skylarks, more typical of east coast marshes (Brown &

Atkinson 1996). No initial influx of passerines during the first winter was observed at Orplands and the passerine assemblage was typical in that Skylarks and Meadow Pipits dominated. At this site, numbers of Meadow Pipits remained similar between winters but Skylarks increased in both Orplands A and Orplands B as the cover of *Salicornia* increased.

A.4.2 Rates of change in bird species abundance in the retreat sites and implications for future mitigation schemes

Both sites saw large changes in the bird species using the sites during the first year after breach and a general shift towards a fauna dominated by waterbirds. At Tollesbury, there were large numbers of passerines recorded in the first winter as seed-rich debris was washed up on the tide line. After the second winter, once a waterbird dominated assemblage had established, there were only smaller changes as species such as Ringed Plover and Knot started to colonise and increase in number. A remarkably similar pattern was seen at Orplands with the quick establishment of a waterbird community followed by smaller annual changes from the second winter onwards as sediments and the number and size of benthic invertebrates changed.

As far as can be told from these limited data from a small number of sites, the waterbird assemblage at both sites is similar to that using similar habitats in the surrounding estuary. The lack of sandy habitats in the retreat areas is likely to be responsible for the low usage of the areas by Oystercatchers and Knot. Although further work would be required the delayed usage of the Tollesbury site by Redshank (ie only used in mid- to late-winter) does suggest that habitats outside the retreat areas are preferred at other times. The relatively enclosed nature of the site may be associated with a higher perceived predation risk but further detailed comparisons would need to be made.

Despite four or five years of monitoring, the waterbird and invertebrate assemblage on these sites was still evolving. This has important implications for future mitigation projects in terms of timing, size and quality of area and extent of post creation monitoring. If a no-net-loss principle is applied then there is a strong argument for the provision of new habitats at least five years (and preferably longer) before existing habitat is removed and a case for long-term, although not necessarily constant, monitoring.

At both the retreat sites, saltmarsh and intertidal mud has formed as a result of the setback. Although small in area and not typical of more exposed mudflats, colonisation of new intertidal mud by invertebrates and birds can occur and the species that use the sites will depend on the substrate and the invertebrates found within it. Although both these sites are typical of muddy estuaries, there is no reason to believe that the general principles from these case studies can be applied to sandier systems although, due to their more dynamic nature, the time taken for stability to be reached may well be different. That the assemblage is different from the whole estuary indicates that the habitats in the retreat sites are less diverse than the surrounding estuary and that they can only support a proportion of the species found on the estuary. This concept is discussed further in Chapter 5.

Table A-2 Total bird usage (measured as bird months) of the managed retreat site at Tollesbury by selected species and groups of species for the winter period (October to March). Values relate to the total usage, measured in bird months, for different states of tide and for each winter. Where a factor was significant in the GLM model this is denoted in the P column - *** $P < 0.001$, ** $P < 0.01$, * $P < 0.05$, NS $P > 0.05$. Values that are not significantly different to each other are followed by the same superscript letter. Disturbance was modelled as a linear factor : + denotes a positive correlation with disturbance (a higher bird usage with disturbance); - denotes a negative correlation with disturbance. Further interpretation can be made from Figures A.1 and A.2. (Data source: C.J. Tyas)

	Tide				**Winter**					**Disturbance**		**Month**
	P	**Low**	**Neap**	**Spring**	***P***	**1995**	**1996**	**1997**	**1998**	***P***		
Little Grebe	NS	54	53	75	***	111^{a}	53^{b}	4^{c}	14^{c}	**	+	***
Dark-bellied Brent Goose	***	140^{a}	1391^{b}	381^{ab}	NS	405	834	379	294	*	-	***
Shelduck	NS	77	138	219	NS	131	36	110	157	NS		NS
Golden Plover	***	1809^{a}	868^{a}	0^{b}	***	2^{a}	1494^{b}	295^{c}	886^{bc}	***	-	***
Grey Plover	NS	232	240	149	NS	95	263	132	131	***	+	***
Lapwing	***	533^{a}	73^{b}	38^{b}	***	52^{a}	161^{abc}	349^{b}	82^{ac}	***	-	***
Dunlin	***	514^{a}	587^{a}	435^{b}	**	148^{a}	316^{b}	440^{b}	632^{b}	**	+	***
Curlew	***	61^{a}	66^{a}	13^{b}	NS	39	55	14	32	NS		NS
Redshank	NS	1243	1441	837	***	182^{a}	955^{b}	1159^{b}	1225^{b}	NS		***
Total shorebirds	***	5448^{a}	4507^{a}	1521^{b}	***	447^{a}	4082^{b}	3202^{b}	3745^{b}	NS		***
Total shorebirds (exc. Golden Plover & Lapwing)	**	3106^{a}	3566^{a}	1483^{b}	***	393^{a}	2427^{b}	2558^{b}	2777^{b}	NS		***
Total wildfowl	***	259^{a}	1869^{b}	902^{c}	NS	1041	970	518	501	**	+	***
Total grebes	NS	54	53	77	***	111^{a}	54^{b}	5^{c}	14^{c}	**	+	***
Total gulls	NS	514	587	435	***	148^{a}	316^{ab}	440^{b}	632^{b}	NS		***
Total granivorous passerines	NS	713	597	736	***	1523^{a}	203^{b}	105^{b}	209^{b}	NS		***
Total insectivorous passerines	NS	74	57	118	***	140^{a}	20^{b}	15^{b}	75^{c}	NS		***

Table A-3 Species found during the 1994/95-1998/99 December to February counts at the Orplands A & B managed retreat and surrounding mudflat sites.

Data source: Environment Agency.

Species common to mudflats & retreat sites	Species found only on mudflat habitats	Species found only in the retreat sites
Black-headed Gull	Cormorant	Blue Tit
Black-tailed Godwit	Common Gull	Chaffinch
Carrion Crow	Eider	Dunnock
Curlew	Great Black-backed Gull	Green finch
Dark-bellied Brent Goose	Great Crested Grebe	Goldfinch
Dunlin	Goldeneye	Linnet
Golden Plover	Herring Gull	Magpie
Grey Plover	Oystercatcher	Robin
Grey Heron	Rook	Skylark
Knot	Red-throated Diver	Snipe
Lapwing	Sparrowhawk	Song Thrush
Little Grebe	Shoveler	Wren
Mallard	Wigeon	Yellowhammer
Meadow Pipit		
Reed Bunting		
Redshank		
Red-breasted Merganser		
Ringed Plover		
Shelduck		
Teal		
Turnstone		

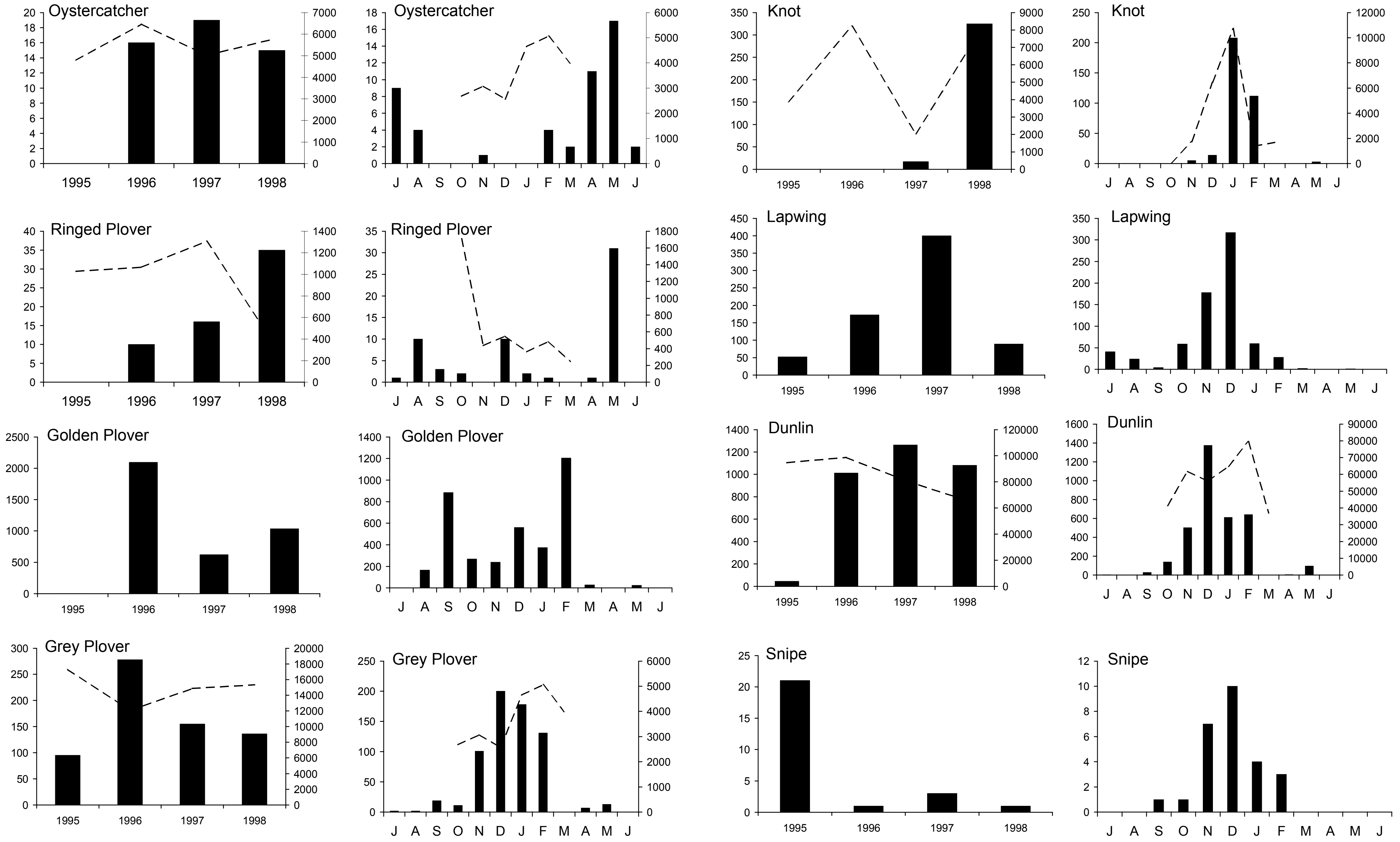


Figure A.1 Changes in the total usage of the Tollesbury managed retreat site by shorebirds between 1995 and 1998. Usage is measured as the total number of birds seen on each of the surveys each year or month. Data source: C.J. Tyas.

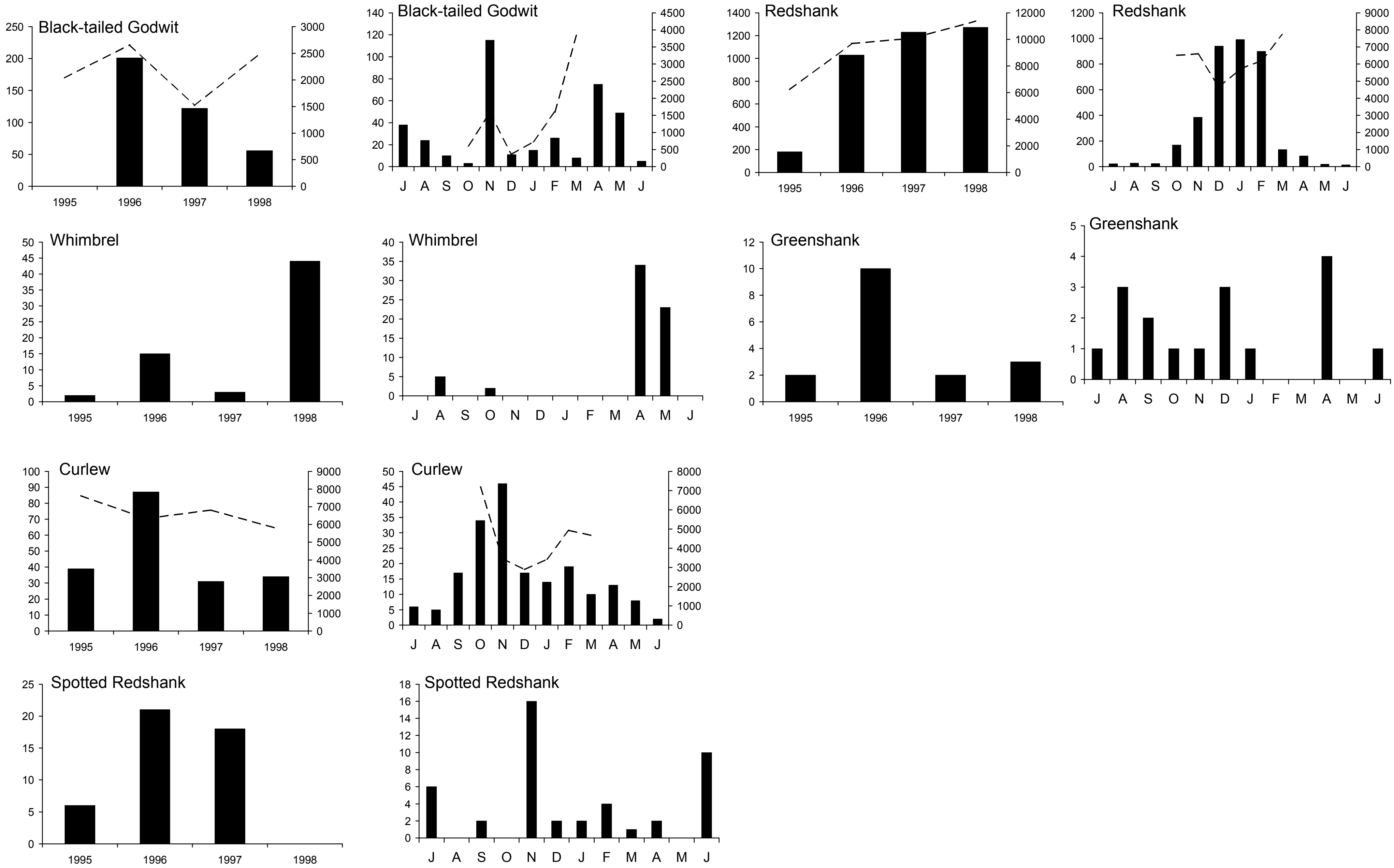
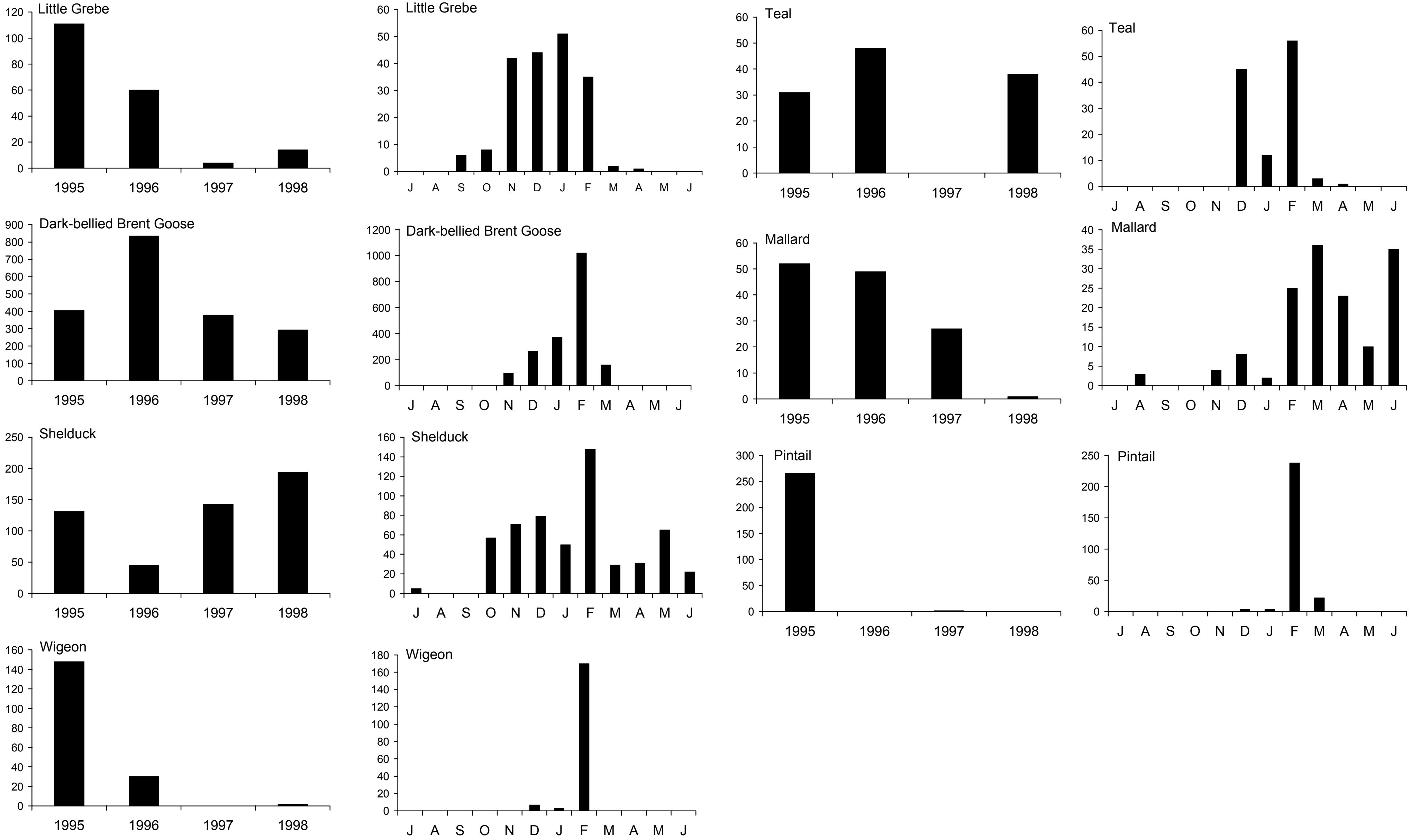
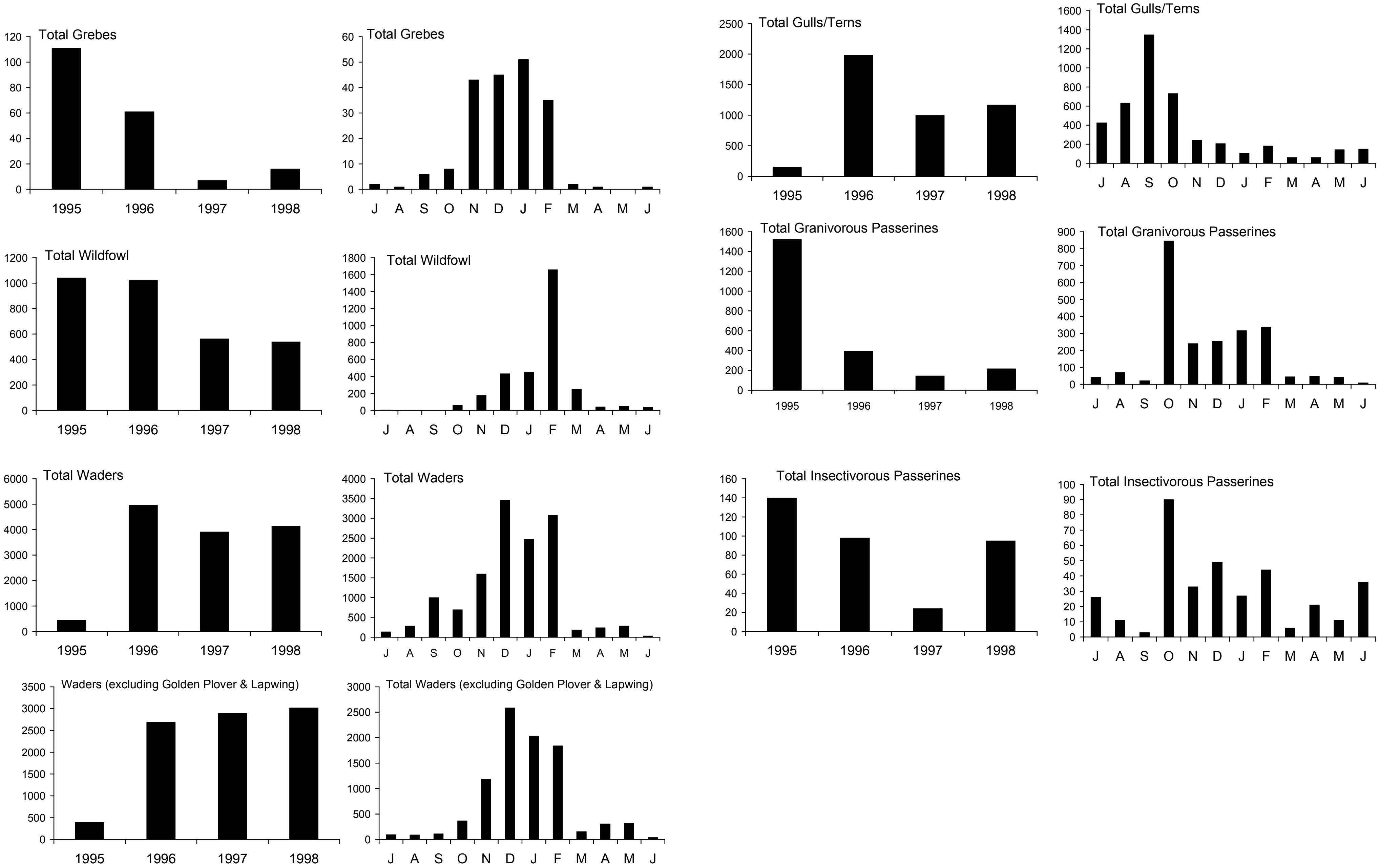


Figure A.1 (cont.)

Figure A.1 (cont.)





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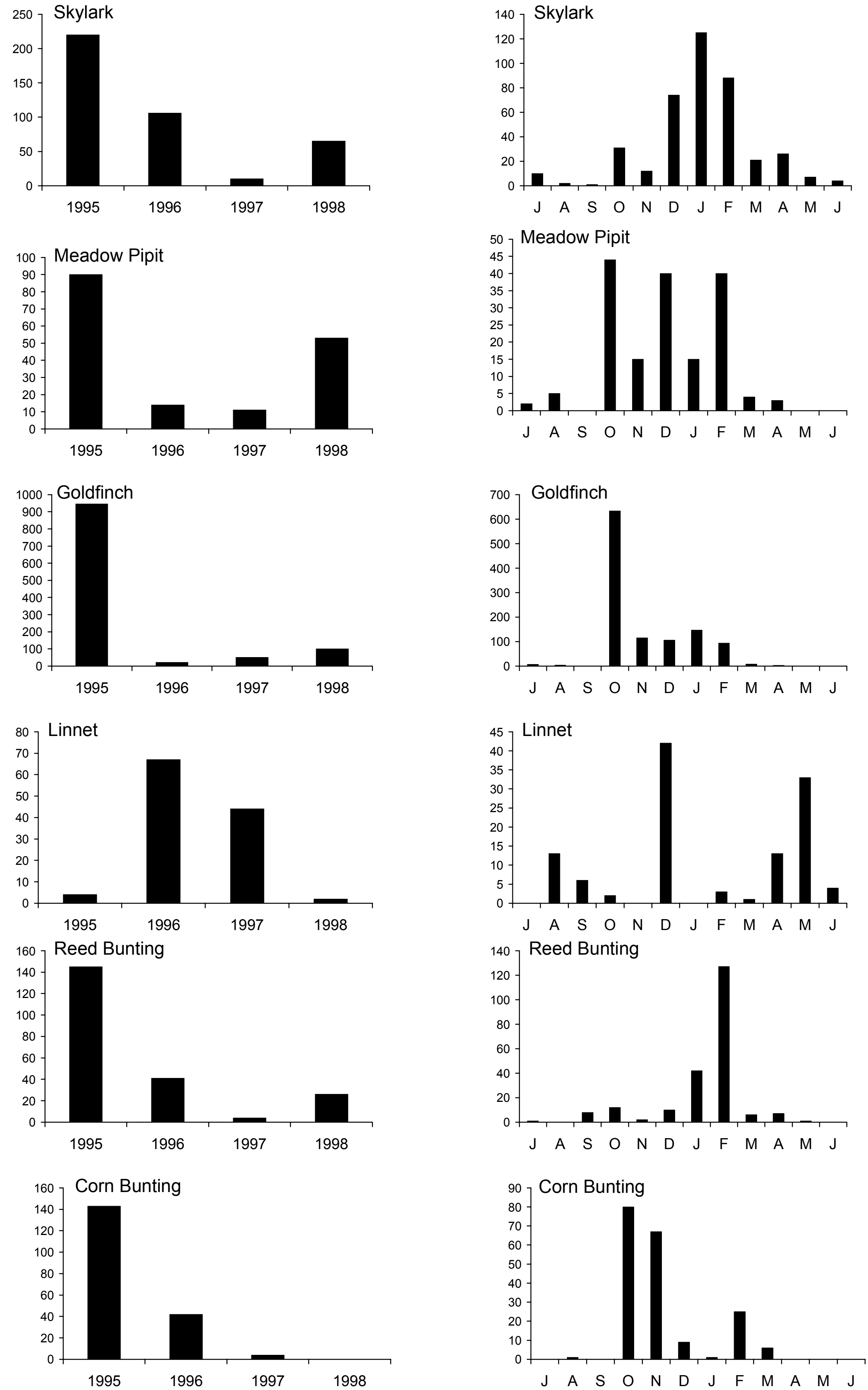


Figure A.2 Changes in the number of passerines at the Tollesbury managed retreat site between 1995 and 1998. Data source: C.J. Tyas.

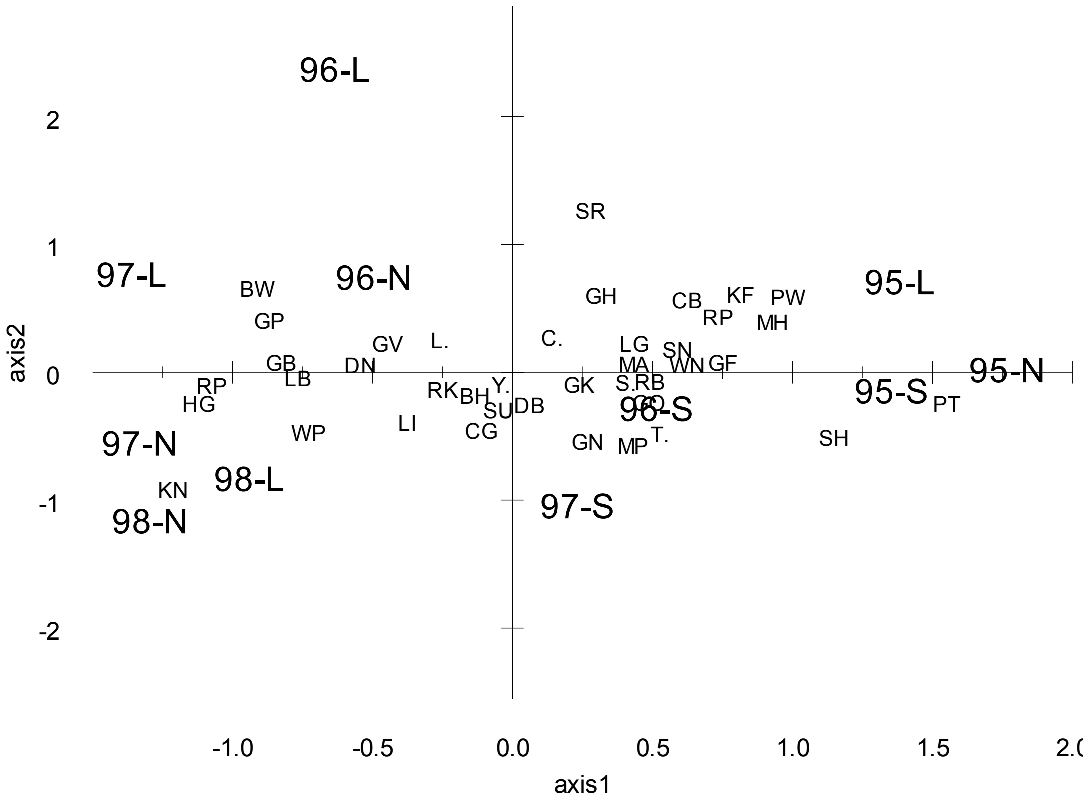


Figure A.3 Correspondence Analysis species and site bi-plot of axis 1 and axis 2 scores from Tollesbury species data. N=neap tide, S=spring tide, L=low tide. The two digit numbers correspond to each winter - 97= 1997/98 winter.

Species Codes: KN - Knot; HG - Herring Gull; RP - Ringed Plover; BW - Black-tailed Godwit; GP - Golden Plover; GB - Great Black-backed Gull; LB - Lesser Black-backed Gull; WP - Woodpigeon; DN - Dunlin; GV - Grey Plover; L. - Lapwing; LI - Linnet; RK - Redshank; BH - Black-headed Gull; Y. - Yellowhammer; SU - Shelduck; CG - Canada Goose; DB - Dark-bellied Brent Goose; C. - Carrion Crow; SR - Spotted Redshank; GH - Grey Heron; GK - Greenshank; GN - Goldeneye; LG - Little Grebe; MA - Mallard; S. - Skylark; RB - Reed Bunting; GO - Goldfinch; MP - Meadow Pipit; T. - Teal; CB - Corn Bunting; SN - Snipe; WN - Wigeon; GF - Geenfinch; RC - Ringed Plover; KF - Kingfisher; MH - Moorhen; PW - Pied Wagtail; SH - Sparrowhawk; PT - Pintail;

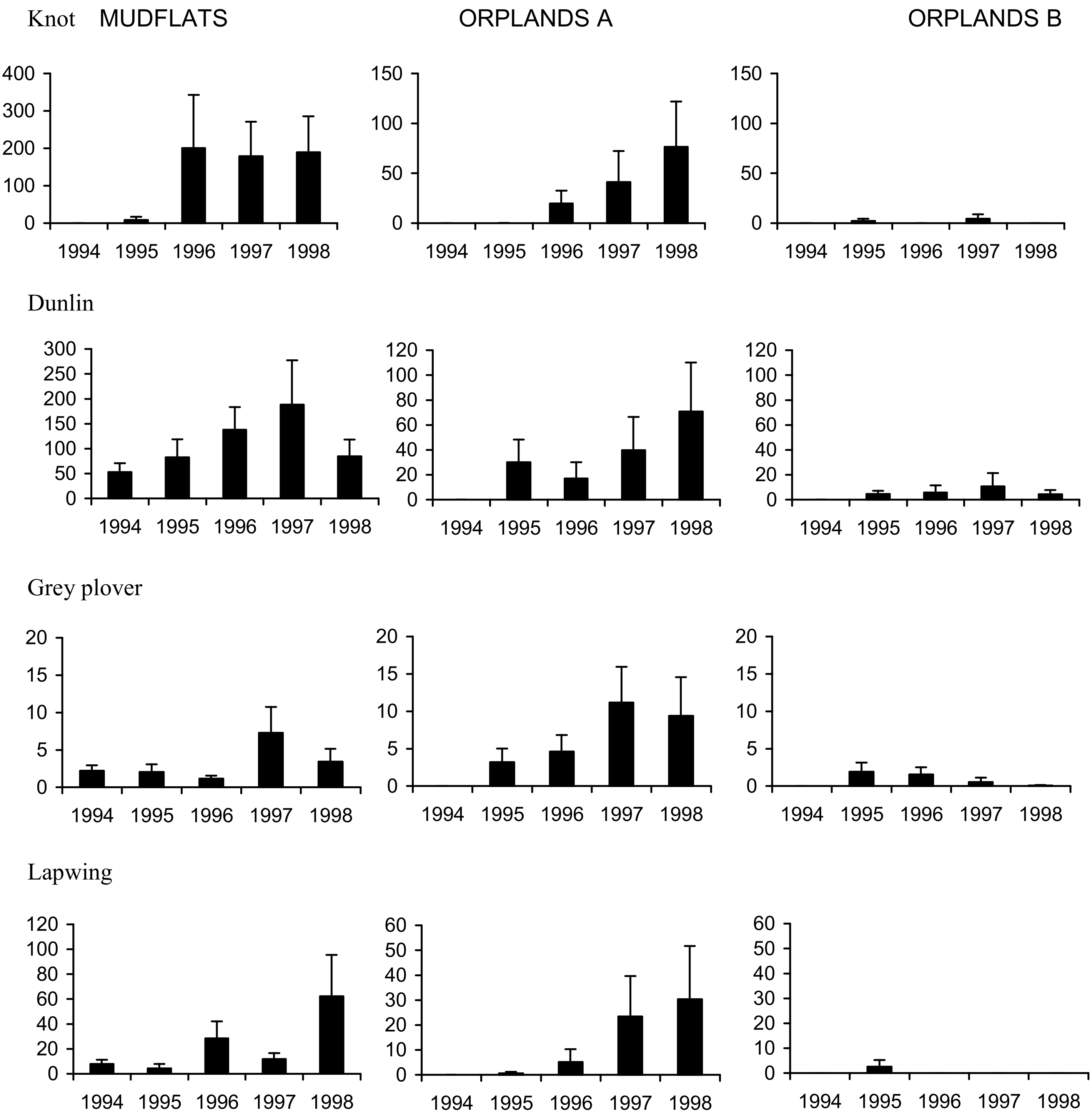


Figure A.4 Mean (± SE) number of birds recorded on both Orplands managed retreat sites and the surrounding mudflats. Data source: Environment Agency.

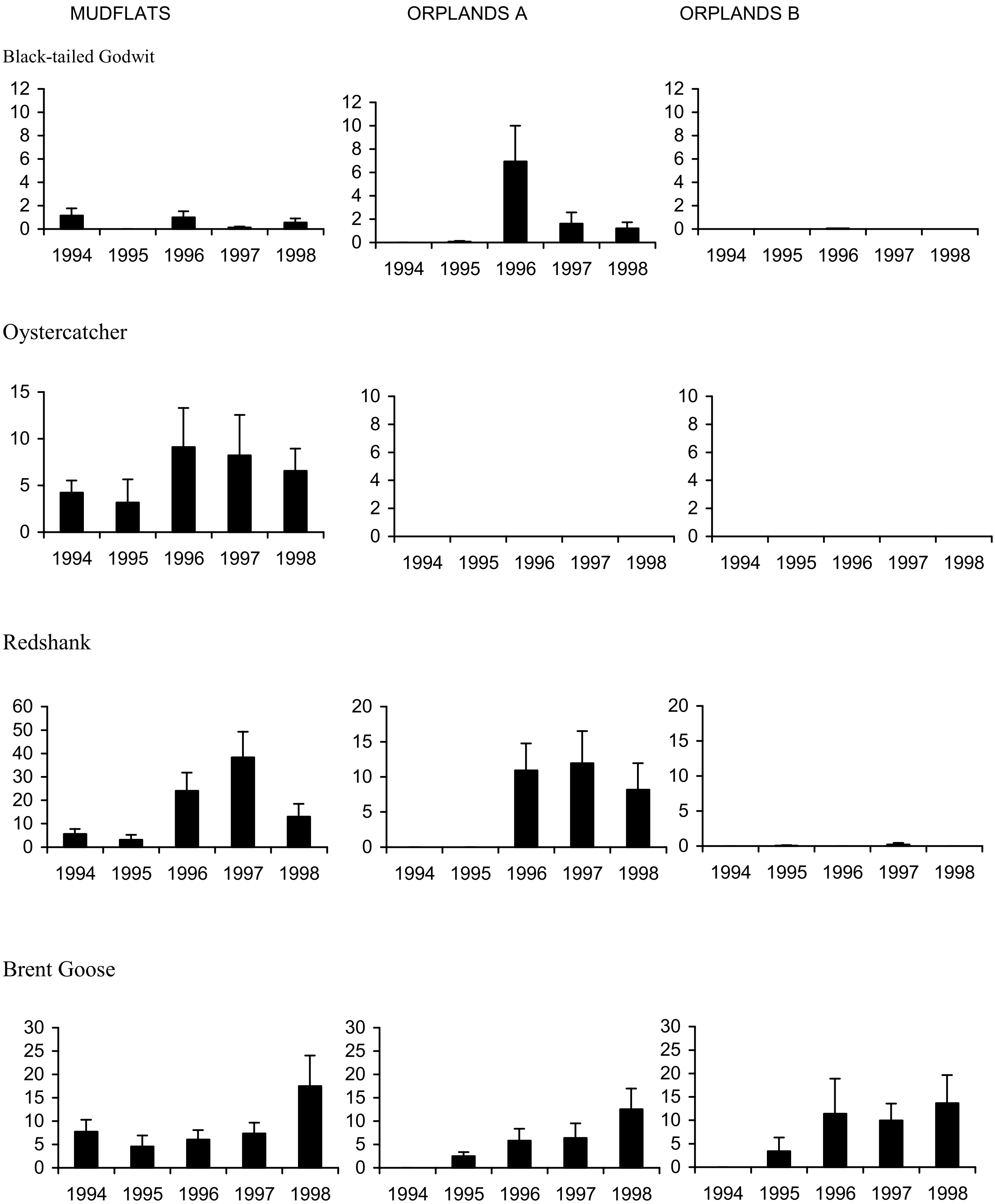
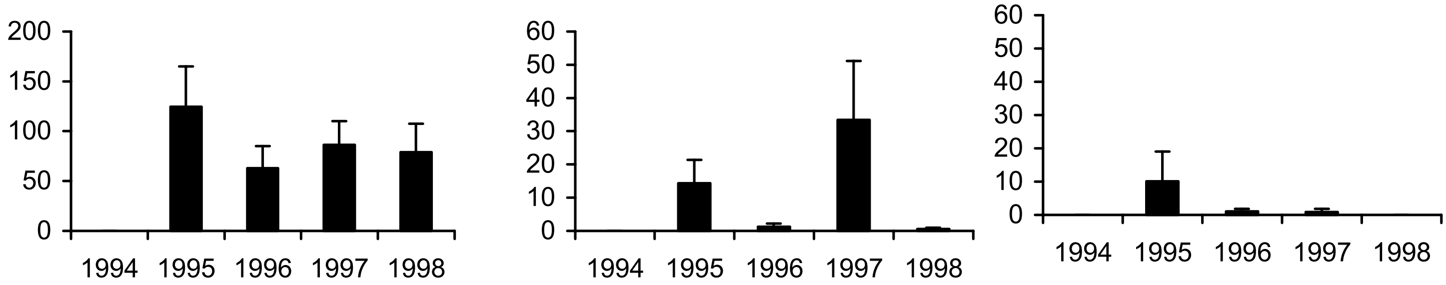


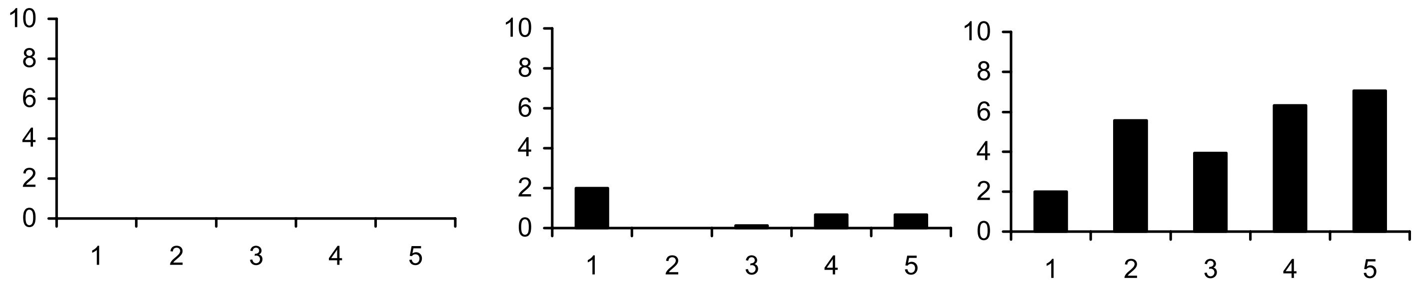
Figure A.4 (cont.)

MUDFLATS ORPLANDS A ORPLANDS B

Shelduck



Skylark



Total shorebirds

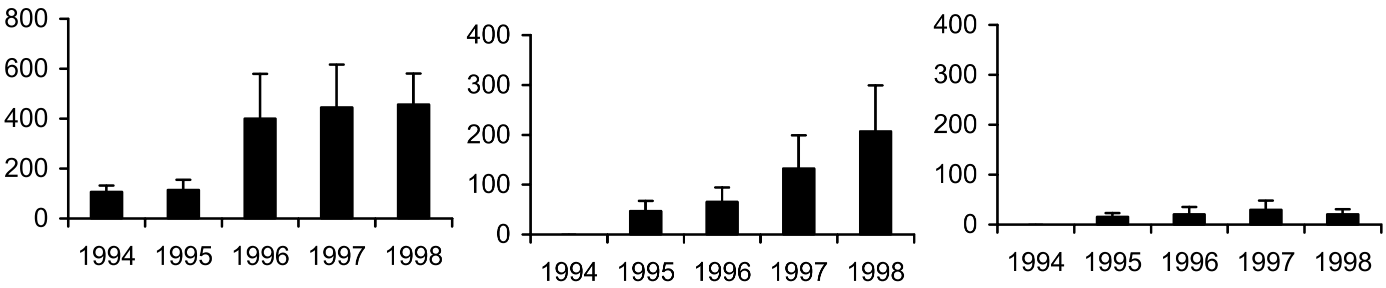


Figure A.4 (cont.)

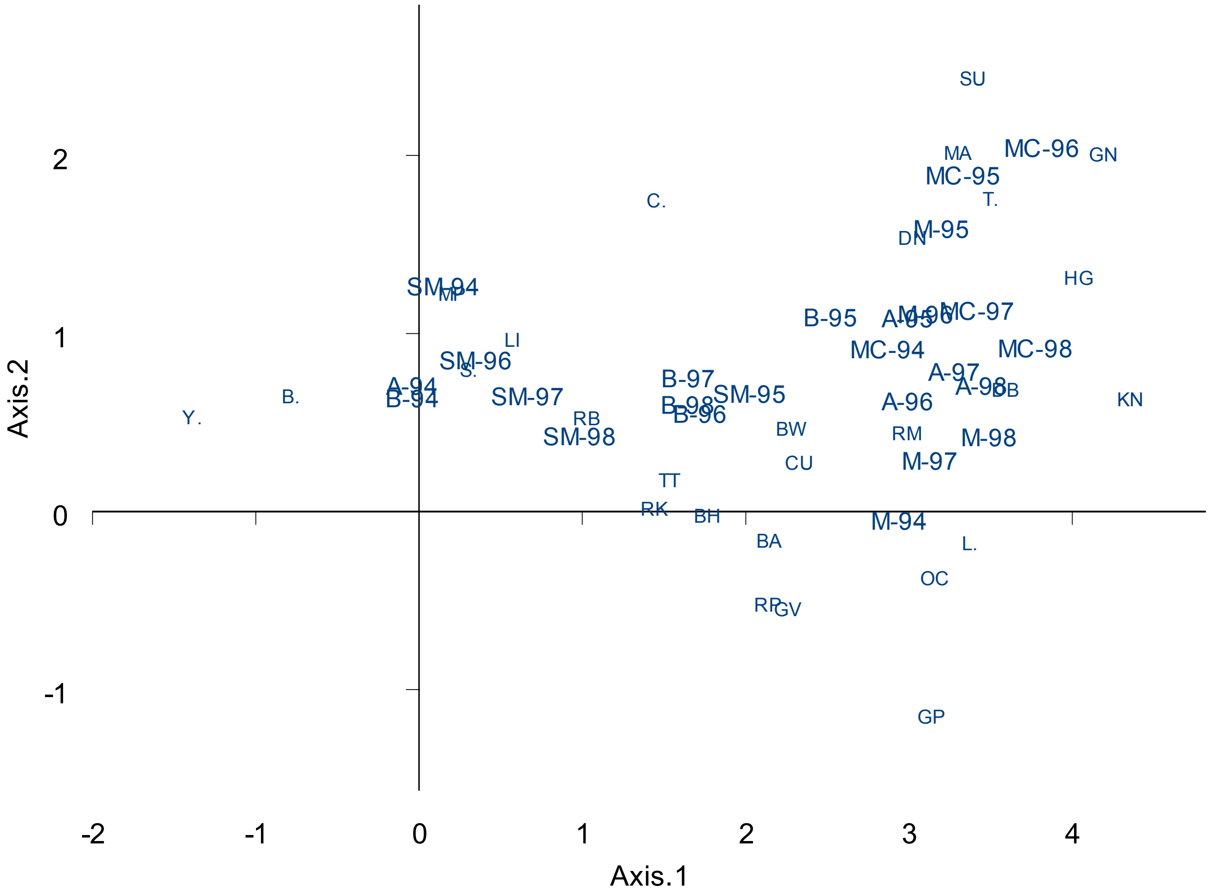


Figure A.5 DCA bi-plot of species and year and site axis 1 and axis 2 scores. Axis 1 describes both a change in time and in habitat. Detrended Correspondence Analysis species and site bi-plot. SM - Saltmarsh; A - Orplands A; B - Orplands B; M - Mudflat; MC - control mudflat. Species codes appear in Figure A.3 legend.

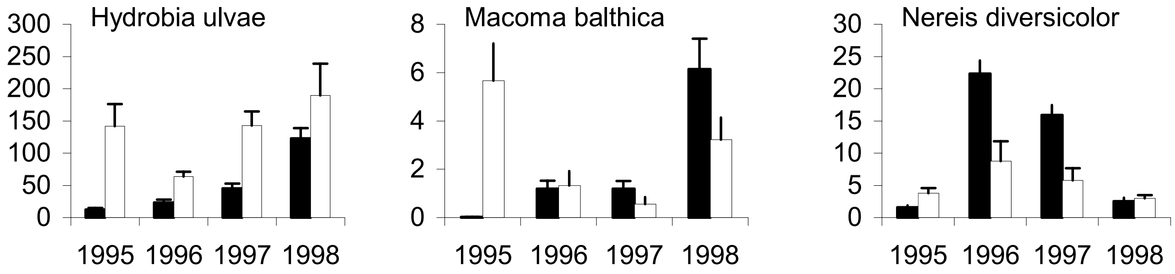


Figure A.6 Mean number of individuals per sample (± SE) *Hydrobia ulvae*, *Macoma balthica* and *Nereis diversicolor* in the Tollesbury retreat site (filled bars) and surrounding saltmarsh mud (white bars) between the retreat site creation in 1995 and 1998. A sample is a core of 10 cm diameter. Data from Reading *et al* (1999).

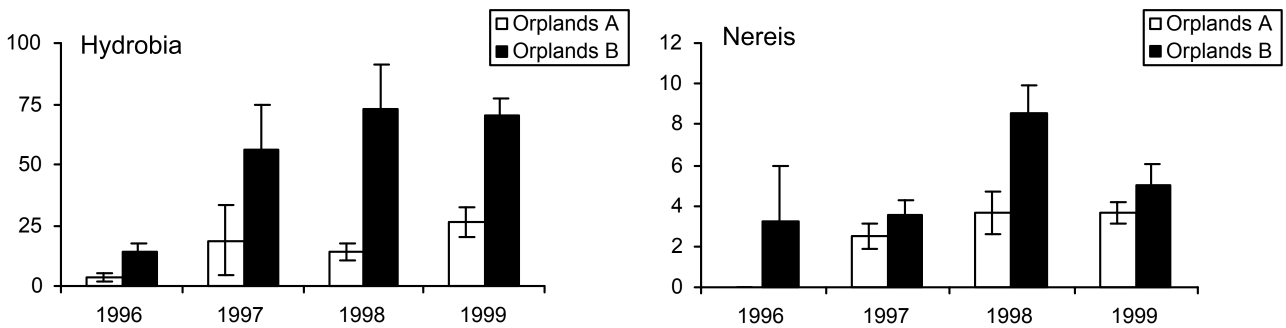


Figure A.7 Mean number of individuals per sample (+/- SE) of the two most common benthic invertebrates, *Hydrobia ulvae* and *Nereis diversicolor,* at the two Orplands managed retreat sites (data from Environment Agency Orplands monitoring report April to August 1999).