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Aos meus Pais

À Memória da minha Avó,
Cândida Sofia

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Summary

Habitat loss is a major factor that can potentially affect the survival of wader populations (Aves: Charadrii) in many estuaries of Europe. A number of studies have been made on this subject in northern Europe, in the last two decades, but there is a lack of data regarding the southern half of the continent.

The main objective of this thesis was to investigate what could be the short to medium–term effects on waders of changes in the estuarine habitats as a consequence of two important factors directly or indirectly attributable to human action in the Mondego estuary (west Portugal): the loss of supra–tidal habitats (mainly salinas) and the increase of eutrophication in the mudflats, expressed as periodic 'blooms' of green macroalgae. The salinas are being abandoned and converted into fish-farms, which can not be used as feeding grounds by waders, at an alarming rate (5.2 ha.year^{-1}in the the period 1984–94). Of about 305 ha of salinas in the estuary, 35% have been deserted or drained, 23% have been converted into fish-farms and only 42% were still producing during the course of this study.

The Mondego estuary still hold a numerous and diversified wader assemblage, which was particularly important in winter and during the spring and autumn migratory passages. It includes one species (Avocet, *Recurvirostra avosetta*) of international importance, according to the criterion of the Ramsar Convention ($>$1% of the flyway population in winter) and at least 8 species of national importance ($>$1% of the national winter numbers).

Part of the birds that used the estuary throughout the year remained in the salinas both at low-tide (30%) and at high-tide (58%), and most of them feed there. The active salinas were clearly more used for feeding than the inactive ones, but these could still hold a number of birds for a long time (up to 20 years) after they have been abandoned, depending on the particular environmental conditions of each salina. An hypothetical destruction of the salinas would mean that the feeding would subsequently be restricted to the mudflats. According to the calculations developed in this study, such increase could reach up to 45% of the birds–feeding hours that presently occur in the mudflats. However, not all species were likely to be affected in the same way by the loss of feeding opportunities in case the salinas were lost. Little Stint, *Calidris minuta*, Redshank *Tringa totanus* and Common Sandpiper *Actitis hipoleucos* would be the species more affected by the loss of space, while Turnstone *Arenaria interpres* and Dunlin *Calidris alpina* would be those more affected by the loss of feeding time at high–water.

The predicted increase in the birds feeding–hours that would occur in the mudflats, if the salinas were to be destroyed, could be potentially detrimental for the survival of the estuarine populations of many species if the levels of inter-specific competition in the flats (which ultimately determine the carrying capacity of an area) were already high. The two tests used to assess the occurrence of inter-specific competition (feeding interference among birds and depletion of the prey resources) suggested that the phenomenon was probably too weak to be perceived in the Mondego, at least with the methodology used. Thus, at present, the mudflats

are probably still able to buffer the potential effects of the loss of space if the salinas were lost. However, the displaced birds would also lose feeding time, and this could not be recovered in the mudflats.

The increasing process of eutrophication in the area may represent a direct threat to the mudflats, while feeding grounds for waders, through changes in the composition of the invertebrate communities and in the abundances of the main prey species. The present study did not present any evidence that the algae repelled the birds. Indeed, some species (e.g. Dunlin) may even have benefited of a surplus of food provided by the mats when they first appear, in early spring, but not in autumn, after the algal crash. This suggests that, at present, the occurrence of the algae on the flats do not represent an immediate threat to the waders' feeding in the Mondego. Nevertheless, the situation can change in the long-term if the eutrophication process can not be reduced.

The main conclusion of this study is that, in spite of the heavy human pressure in the estuary, the survival of the wader populations seems not to be threatened in the short–term. However, in the medium–term, the destruction of the salinas could seriously affect the survival of some species, and the situation could still become worse if it was accompanied by a global change in the composition of the prey communities caused by an increase of the eutrophication process in the estuary.

Nevertheless, both these drastic alterations can still be prevented. Possible ways of achieving that in the salinas could be to find sustainable ways of keeping them actively producing, to manage inactive ones in order to make them suitable for waders, to carefully control the establishment of new fish-farms in old salinas, and to improve the design of the fish–ponds in order to provide waders with alternative feeding places to the destroyed salinas. As to the eutrophication process, any action to reduce it would have to be taken at a regional level, and would take more time. In any case, a long–term monitoring of the process at all levels of the trophic chain would be greatly welcome.

Resumo

A perda de habitat é uma das principais causas que podem afectar a sobrevivência das limícolas (Aves: Charadrii) em muitos estuários da Europa. Um razoável número de estudos tem sido feito sobre este tema no norte da Europa durante os últimos 20 anos, mas continua a existir uma lacuna no que se refere à metade sul do continente.

O principal objectivo deste estudo foi investigar quais as consequências para as limícolas, a médio e curto prazo, de determinadas alterações nos habitats estuarinos provocadas pela acção humana que, directa ou indirectamente têm tido lugar no estuário do Mondego: a perda de habitats supra–mareais (sobretudo salinas) e os *blooms* de macroalgas que ocorrem periodicamente nas vaseiras inter–mareais devido ao progressivo aumento do processo de eutrofização no estuário. As salinas estão a ser abandonadas e convertidas em áreas para aquacultura a um ritmo alarmante: 5.2 ha.ano^{-1}no período 1984–94. Dos cerca de 305 ha de salinas existentes no estuário, 35% encontravam–se abandonadas ou tinham sido aterradas, 23% tinham sido transformadas em tanques para aquacultura e apenas 42% se encontravam ainda activas no decurso deste estudo.

O estuário do Mondego ainda possui uma comunidade de limícolas bastante numerosa e diversificada, especialmente no inverno e durante as migrações de outono e de primavera. Esta comunidade inclui uma espécie (Alfaiate *Recurvirostra avosetta*) de importância internacional, de acordo com os critérios da Convenção de Ramsar ($>$1% do total de indivíduos que constituem a população da via de migração do Atlântico Oriental no inverno) e pelo menos 8 espécies de importância nacional ($>$1% da população nacional de uma espécie que inverne regularmente no país).

Parte dos indivíduos que passaram pelo estuário durante o ano, usaram as salinas quer na baixa–mar (30%), quer na preia–mar (58%), e a maioria fê–lo para se alimentar. As salinas activas foram nitidamente mais utilizadas como áreas de alimentação do que as inactivas, mas estas podem ainda ser utilizadas por aves a alimentar–se por períodos relativamente longos (até cerca de 20 anos) depois de terem sido abandonadas, dependendo das condições específicas de cada salina. Uma hipotética destruição de todas as salinas significaria, muito provavelmente, que a maioria das aves que actualmente usam este habitat para se alimentarem se concentraria sobretudo nas vaseiras inter–mareais do estuário.

De acordo com os cálculos desenvolvidos neste estudo, poder–se–ia assistir a um incremento até 45% do número de horas de alimentação.aves^{-1}que actualmente tem lugar nas vaseiras durante a baixa–mar. Contudo, as espécies não seriam todas afectadas da mesma forma pela perda de oportunidades de alimentação (que incluem a perda de espaço e tempo de alimentação). O Pilrito–pequeno *Calidris minuta*, o Perna–vermelha *Tringa totanus* e o Maçarico–das–rochas *Actitis hypoleucos* seriam as espécies mais afectadas pela perda de locais de alimentação, ao passo que parte das populações da Rola–do–mar *Arenaria interpres* e do Pilrito–comum *Calidris alpina* perderiam um considerável número de horas de alimentação du-

rante a preia–mar.

O incremento previsto do número de horas de alimentação.aves^{-1}, que ocorreria nas vaseiras se as salinas fossem destruídas, poderia ser potencialmente desvantajoso para a sobrevivência das populações de muitas espécies se os níveis de competição inter–específica na zona inter–mareal (que determina, em última análise, a 'capacidade de suporte' da área) fossem já elevados. Os dois testes utilizados neste estudo para avaliar os níveis de competição inter–específica (interferência entre aves em alimentação e uso dos recursos alimentares potencialmente disponíveis) sugeriram que o fenómeno era provavelmente pouco relevante neste estuário, pelo menos para poder ser detectado com a metodologia utilizada. Assim, é provável que as vaseiras possuam ainda uma boa capacidade–tampão para absorver eventuais indivíduos deslocados das salinas. Contudo, o problema da perda de horas de alimentação, que não poderiam ser recuperadas na zona inter–mareal, manter–se–ia para muitas aves.

O acelerar do processo de eutrofização no estuário pode constituir uma ameaça directa para as vaseiras, enquanto áreas de alimentação das limícolas, através de alterações na composição das comunidades de presas e na redução das respectivas abundâncias. O presente estudo não mostrou nenhuma evidência de que as algas afectassem de alguma forma as densidades de limícolas a alimentar–se nas zonas por elas colonizadas. Na realidade, algumas espécies (e.g. o Pilrito–comum) podem até ter beneficiado da presença das algas no início do processo de colonização, na primavera, embora não no outono, depois do 'crash' algal. Isto sugere que, de momento, a presença de algas no sedimento parece não representar uma ameaça para a alimentação das limícolas no estuário do Mondego. A situação pode alterar–se, contudo, a longo prazo, se a eutrofização do estuário não puder ser reduzida.

A principal conclusão que se pode tirar deste estudo é a de que, apesar da intensa pressão humana que se verifica actualmente no estuário do Mondego, a sobrevivência das populações de limícolas não parece estar em risco imediato. Contudo, a médio prazo, a destruição das salinas pode vir a afectar muitas espécies, e a situação pode piorar ainda mais se for acompanhada por uma deterioração das condições alimentares nas vaseiras inter–mareais em virtude do crescimento incontrolável das áreas periodicamente afectadas pelos 'blooms' algais, como consequência do processo de eutrofização acelerado que actualmente se verifica.

No entanto, qualquer destas drásticas alterações nos habitats de alimentação das limícolas no estuário do Mondego pode ainda ser prevenida. Algumas formas de conseguir isso, nas salinas, poderiam incluir a concessão de apoios para manter as salinas activas a produzir de uma forma economicamente sustentável, a gestão das salinas abandonadas, de modo a torná–las atractivas para as limícolas, e uma cuidadosa política de autorização de novos estabelecimentos de aquacultura no estuário, assim como alterações no *design* de novos tanques para piscicultura, de modo a proporcionar às limícolas áreas de alimentação alternativas às salinas destruídas no processo de construção das pisciculturas.

Quanto ao processo de eutrofização, uma eventual acção para o controlar fugiria do âmbito meramente local, necessitando do envolvimento, a vários níveis, de muitas entidades, públicas e privadas.

Entretanto, o estabelecimento de um programa de monitorização a longo prazo, abarcando todos os níveis da teia trófica, para acompanhar os efeitos do processo de eutrofização seria extremamente útil.

Contents

Chapter 1

General Introduction

1.1 The significance of estuaries for waders

Estuaries are known to be important areas for wading birds, particularly for the group of species generally designated as shorebirds or waders (Aves: Charadrii) (McLusky, 1989). According to the more recent estimates (Smit & Piersma, 1989), some 7.5 million waders winter along the Atlantic coasts of Europe and west Africa, which forms the East Atlantic flyway [1]. Most of these birds are found in estuaries or related coastal areas. In Europe, 38 of these sites support a wintering population of 2.6 million birds, 80% of the whole European Atlantic population (Smit & Piersma, 1989).

By far, the most important area for waders in Europe is the Dutch–German–Danish Wadden Sea. In winter, its large intertidal flats holds some 0.9 to 1.2 million birds (Smit & Piersma, 1989; Meltofte *et al.*, 1994). The other European sites considered by Smit & Piersma (1989) support considerably smaller populations. Still, numbers range from some 22 000 (Duddon estuary, north–western England) to 250 000 birds (the Delta, Netherlands), with an average of 72 000 per site (Smit & Piersma, 1989). The significance of these areas for waders is not confined, however, to the winter season, although this is the most important one. Numbers recorded during the migratory seasons can reach high values, too. For instance, Meltofte *et al.* (1994) estimated that the international Wadden Sea held 2.2 to 2.6 million birds from September to November (autumn migration), while some 1.3 to 2.2 million used the area between March and May (spring migration).

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[1] According to the definition of Campbell and Lack (1985, *in* Smit & Piersma, 1989), a ‘flyway’ is “a major route for birds on migration” and encompasses the annual migration routes followed by waders between their breeding and wintering areas, including the stopover sites between the two (Smit & Piersma, 1989). In Europe there are two recognized flyways. The most important, and intensively studied, is the East Atlantic flyway, which links arctic breeding grounds of eastern North America and western Siberia to the wintering areas ranging from Iceland and Scandinavia to the central part of the Mediterranean and West Africa as far as south as the Ivory Coast. The other is the Mediterranean flyway, which crosses the eastern Mediterranean, with wintering areas in Egypt and in the African coast between Ghana and Angola (Smit & Piersma, 1989)

Such an impressive demographic pressure of waders on relatively limited areas is due to the high values of secondary productivity that characterises the estuarine environments, which matches that of the most productive terrestrial ecosystem, the tropical forests (Whittaker, 1970). In fact, waders seem to take full advantage of the important food supply (mainly macroinvertebrates) available in estuaries. Goss-Custard (1980, 1984) present values for the winter reduction in the standing stock of prey densities in 16 English estuaries by several species of waders [2] of 25% to 45%. In a more recent review, Székely & Bamberger (1992) list 28 cases, ranging from 1% to 99%, for periods from 13-22 days up to 9 months. Other studies, however, failed to find a significant reduction in prey densities (e.g. Botton, 1984; Raffaelli & Milne, 1987).

The impact of predation by wading birds has also been highlighted in production studies, in which it is measured against the annual production of the prey and not just to their standing crop present at the start of one season. Baird *et al.* (1985) reviewed the available information, mostly in northern Europe, and found that 44% of the secundary production in the Tees estuary (northeastern England), 36% in the Ythan (Scotland), 17% in the intertidal area of the Dutch Wadden Sea and 6% in the Gravelingen estuary (Netherlands) was taken by birds. These authors also reported partial values for parts of estuaries, or for a limited number of wader and/or invertebrate species. For instance, in Lindisfarne estuary (northeastern England), 13% to 22% of the annual production of *Notomastus*, *Scoloplos* and *Arenicola* worms was removed by Ringed Plovers *Charadrius hiaticula* and Grey Plovers *Pluvialis squatarola* (Baird *et al.*, 1985).

In the African feeding grounds the impact seems to be quite similar – 17% in the Langebaan Lagoon (Puttick, 1980 *in* Baird *et al.*, 1985), 26% in the Berg River estuary (Kalejta, 1992), both in South Africa, and 14% in the Banc d'Arguin, Mauritanea (Wolff & Smit, 1990) – in spite of the higher production values recorded for the southern Hemisphere sites, as compared to their northern counterparts (Kalejta & Hockey, 1991; Hockey *et al.*, 1992).

Despite the observed variability in the proportion of the invertebrate production consumed by waders, these studies agree on the general significance of the birds to the energy flows in the estuarine ecosystems [3]. Thus, in summary, most wader species depend strongly on the abundant estuarine resources, both in winter and in the migratory periods, and may exploit them heavily.

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[2] Oystercatcher *Haematopus ostralegus*, Redshank *Tringa totanus*, Grey Plover *Pluvialis squatarola*, Curlew *Numenius arquata*, Bar–tailed Godwit *Limosa lapponica* and Knot *Calidris canutus*

[3] The values for some estuaries (Wadden Sea, Gravelingen Estuary, Ythan Estuary) include other non–wader carnivorous birds (gulls, terns, herons, grebes) (Baird *et al.*, 1985). Therefore, the strict contribution of waders to the energy transfer between trophic levels must be somewhat lower than that presented.

1.2 The human usage of estuaries and its consequences for waders

Estuaries are among the oldest and most human intensively–used coastal wetlands (McLusky, 1989; Prater, 1981). Because of the mani–fold ways of using estuarine resources (Smit *et al.*, 1987; Lambeck *et al.*, 1996), the impact of human activities may take several forms. In a recent review, Smit *et al.* (1987) described, largely in qualitative terms, the potential threats to the estuaries of the East Atlantic flyway, a summarised version of which can be found in Table 1.1. These authors considered that the most serious threat to the estuarine wetlands and, by extension, to their wader populations, was the irreversible loss of habitat, posed mostly by land–claims. This has been confirmed by an impressive number of studies in northern Europe (e.g. Goss-Custard, 1977a, 1979; Evans *et al.*, 1979; Prater, 1981; Smit *et al.*, 1987; Davidson *et al.*, 1991; Goss-Custard *et al.*, 1991; Schekkerman *et al.*, 1994; Meire *et al.*, 1994).

Table 1.1: Threats or potential threats to estuarine ecosystems along the East Atlantic flyway (adapted from Smit *et al.* (1987), and their potential impact on waders. Symbols as follows: 1) Type of effect: (I) irreversible; (L/M) long to medium–term recoverable; (S) short–term recoverable; 2) Geographical scale: (*) restricted to a few sites; (***) spread along the whole flyway; 3) Frequency, (+) rare; (++++) very frequent in most estuaries of the flyway 4) Effect on waders: (Y) confirmed direct effect; (y) probable direct effect, still to confirm; (Y?) confirmed side–way effect; (y?) probable side–way effect, still to confirm. *Direct effects* are defined as those that can, *per se*, conditionate the waders' fitness or survival hability, in opposition to the *side-way effects*, which act indirectly, sometimes potentiating direct effects (e.g. the sand extraction may leave long-term holes in the flats that cause the loss of feeding habitats)

Threats	Type of effect	Geographical scale	Frequency	Effect on waders
1. Habitat loss				
1.1 Land-claims	I	***	++(++)	Y
1.2 Sea-wall construction	I	***	++(++)	y?
1.3 Other constructions	I	**	+(+)	y?
2. Pollution effects				
2.1 Industrial, direct (heavy metals, chemicals)	L/M	***	++++	Y
2.2 Industrial, indirect (noise, termal pollution)	L/M	***	++++	y/y?
2.3 Sand & shell extraction and dredging	L/M	***	+(+++)	Y?
2.4 Agriculture, direct (aquaculture, pesticides)	L/M	**	++++	Y/Y?
2.5 Agriculture, indirect (eutrophication)	L/M	**	++++	Y?
2.6 Accidental (oil spills, broken pipelines)	L/M	*	+	Y
3. Human disturbance				
3.1 Fisheries	S	***	++++	y?
3.2 Bait-digging and cockle dredging	S	**	+++	y?
3.3 Aquatic sports	S	**	++++	Y
3.4 Shipping activities	S	**(*)	++++	y
3.5 Military activities	S	**(*)	+++	Y
3.6 Hunting	S	**(*)	++++	Y
3.7 Other (tidal walking, research, planes)	S	*(*)	++(+)	Y/Y?

More rare have been the studies dealing with the indirect effects on waders of the construction of sea defenses and other coastal protections (Davidson *et al.*, 1991). An elegant and classic example of this was the study of Goss-Custard & Moser (1988) who showed that the decrease in numbers of Dunlin *Calidris alpina* from 1977/78 in some British estuaries was associated with the spreading of *Spartina anglica*.

Although not so abundant, some work has been made regarding the long to medium–term effects of industrial and agricultural pollutants on the survival of the wader populations (references in Smit *et al.*, 1987; Lambeck *et al.*, 1996). The direct, deleterious effects of pesticides on the breeding parameters of waders, for instance, have been documented (Denker *et al.*, 1994; Denker & Buthe, 1995; Dietrich *et al.*, 1997) as well as the bioacumulation of organochlorine compounds and heavy metals in wintering (Lambeck *et al.*, 1991; Ferns & Matthews, 1994) and migratory shorebirds (Tatsukawa *et al.*, 1994).

An expanding field of interest within this area has been the potential effects of eutrophication at the higher trophic levels of estuarine chains, particularly birds (Soulsby *et al.*, 1982; Raffaelli *et al.*, 1989; Desprez *et al.*, 1994; Metzmacher & Reise, 1994, and references therein). The actual state of research suggests that beneficial short–term effects of eutrophication can occur due to a momentary surplus in the food available to waders (e.g. Van Impe, 1985; Metzmacher & Reise, 1994). Long–term effects, however, are likely to be detrimental, particularly if the weed mats become contiguous because, in these circumstances, all the refuges in the unweeded areas, from which macroinvertebrates can recolonise the affected areas in autumn and winter, would be lost (Raffaelli *et al.*, 1989).

Finally, the short–term effects of the diverse forms of human disturbance have only recently begun to be analyzed (Davidson & Rothwell, 1993; Lambeck *et al.*, 1996). It seems, though, that disturbance effects can play a more important role in the waders ability to survive, particularly in winter, than it was thought.

1.3 Present status of the Portuguese estuaries

In contrast with the situation in the north European estuaries, the influence of human activities in the more southern sites of the Atlantic coasts of Europe have only recently received some attention.

In Portugal, the contamination of the larger estuaries by pesticides and heavy metals is well known (e.g. Castro & Vale, 1995; Ferreira & Vale, 1995; Pereira *et al.*, 1995). However, only recently has a study analysed the bioaccumulation of heavy metals (cadmium and zinc) in chicks of a wader species, the Black–winged Stilt *Himantopus himantopus*, nesting in the salinas of the Sado estuary (Pimentel & Costa, 1996).

Eutrophication is also a widespread phenomenon, particularly in the smaller estuaries (Marques *et al.*, 1993a,b). Yet, no assessment has been made of the impact of eutrophication on estuarine waders, even though such studies were initiated for the lower levels of the trophic chain in the Mondego estuary (Martins, 1995; Marques *et al.*, 1997).

The effects of loss of habitat for estuarine breeding waders have recently been examined in relation to the destruction of the salinas, particularly with regard to the Black–winged Stilt (Rufino & Neves, 1992; Neves & Rufino, 1995). However, no attempts have yet been made to extend the analysis to the non–breeding waders, although the importance of the salinas for wintering and migratory birds has now been generally recognised (Rufino *et al.*, 1984; Luís, 1989; Batty, 1991, 1992).

Furthermore, no formal studies have been performed on the effect of human disturbance on waders in any of the Portuguese estuaries, except for Batty's preliminary study in the 'Ria' Formosa, Algarve (Batty, 1991). This too is an aspect of human impact on waders that needs attention.

1.4 Justification, objectives and structure of the present study

A major reason why so few studies on the potential damage of human action to estuarine waders have been performed lies, for most estuaries, on the lack of a suitable database on which these kind of studies must be based. Particularly important are (1) comprehensive mapping of the biological communities over the whole estuarine areas, (2) detailed quantitative data on abundance, densities and production of the main prey species, and (3) a detailed knowledge of the movements of the waders within the area, as well as their use of other supratidal habitats nearby (Moreira, 1995). These data are only possible to achieve in large estuaries with a considerable logistic staff and/or with enough time, two important constraints in most scientific studies.

In this sense, the estuary of the Mondego provides a good site to study the potential consequences of, at least, some human–induced changes. First, it is small in area (roughly 2 000 ha) and thus easy to survey, even with limited human and logistic facilities. Second, the most important intertidal area is located in a narrow, relatively quiet and easily accessible branch of the river. Third, the location of the main supratidal habitats for waders – mostly salinas – in a central island and in the left margin of the south arm, makes them easy to survey within a short time. Fourth, the wader populations are not too numerous, and are, therefore, easy to census. Fifth, a considerable amount of information on the intertidal prey communities in the whole estuary is already available. Last, but not least, the estuary is currently under a severe human pressure, these being harbour and industrial facilities, dredging activities, organic pollution from upstream the river and locally installed fish–farms, increasing eutrophication, and potential loss of habitat due to the destruction of the ancient salinas.

These reasons led to the examination of the possible consequences of some of the human activities in the estuary for the wader populations. Specifically, one tried to answer to the following question: **what may be the consequences for waders**

of the main anthropogenic changes that can be foreseen at the moment in the estuary, these being (1) the reduction in the number of active salinas and (2) the increase in eutrophication.

This study is subdivided into several independent but connected Chapters. After a brief description of the study area and a review of the present status of knowledge about its biology (Chapter 2), the data on the composition and seasonal dynamics of the wader assemblages in the estuary are detailed (Chapter 3), in order to highlight the significance of the Mondego from the ornithological point of view.

The following part (Chapters 4 to 6) deal with the potential effects arising from (1) the loss of habitat and (2) eutrophication. Specificaly, Chapter 4 assesses the importance of the salinas for waders as feeding grounds and the potential effects, in terms of the quantity of space and time that the birds will loose, were this habitat to disappear in the near future. Chapter 5 focuses on the extent to which birds in the intertidal areas already compete for food, in order to evaluate whether these feeding areas could accommodate all the birds displaced from the salinas, were these to be destroyed. Finally, Chapter 6, discusses the eutrophication of the estuarine waters and its possible influence on the feeding of waders in the estuary of Mondego.

The last part of the study (Chapter 7) summarizes and integrates the information presented, and attempt to discuss the implications of the findings for the whole ecosystem of the Mondego estuary.

Chapter 2

Study Area

2.1 Description of the study area

2.1.1 Location and general characterisation of the estuary

The Mondego estuary (40^o08' N, 8^o50' W) is the most important estuarine system of the Portuguese Atlantic coast between the Ria de Aveiro, in the north, and the Tagus and Sado estuaries in the south. The Mondego estuary is presently about 7 km long and 2–3 km wide, covering an area of approximately 1 072 ha with wetland habitats, although the tidal influence extends some 27 km to the east (D. N. Duarte, pers. comm.). Just before it enters the sea, the Mondego river is divided into two arms around an alluvian–formed island, the Morraceira. The two arms (north and south) join again before opening to the sea, in front of Figueira da Foz harbour (Figure 2.1). A small tributary, the Pranto river, opens directly into the downstream part of the south arm.

2.1.2 Climate and weather

The Mondego estuary is located in a warm–temperate zone, presenting characteristics of transition between the Atlantic and the Mediterranean climates (Borges, 1988). The winters are mild (15^oC –16^oC) and the summers not very hot (20^oC – 22^oC), the average annual temperature being 15^oC . Daily temperature ranges are of 7.4^oC in winter, 6.3^oC in summer and 6.7^oC annually (Proença, 1988). The rainfall is concentrated mainly in late autumn, winter and early spring, as in Mediterranean climates. The average annual precipitation for the area is 627.1 mm (Borges, 1988). For the period 1931–1960, the driest month was July, with an average rainfall of 5 $\mathrm{l.m}^{-2}$, and the wettest was December, with 800 $\mathrm{l.m}^{-2}$ (Jorge, 1991). According to Proença (1988), average values of insulation and humidity are high: 2 772 $\mathrm{hours.year}^{-1}$ and 79.4%, respectively. The predominant winds are those from the NW quadrant (NNW). The estuary receives some protection from the N and NW winds, due to the location of the Serra da Boa Viagem (see below). On the other hand, it is exposed to the winds from W and S quadrants. In summer

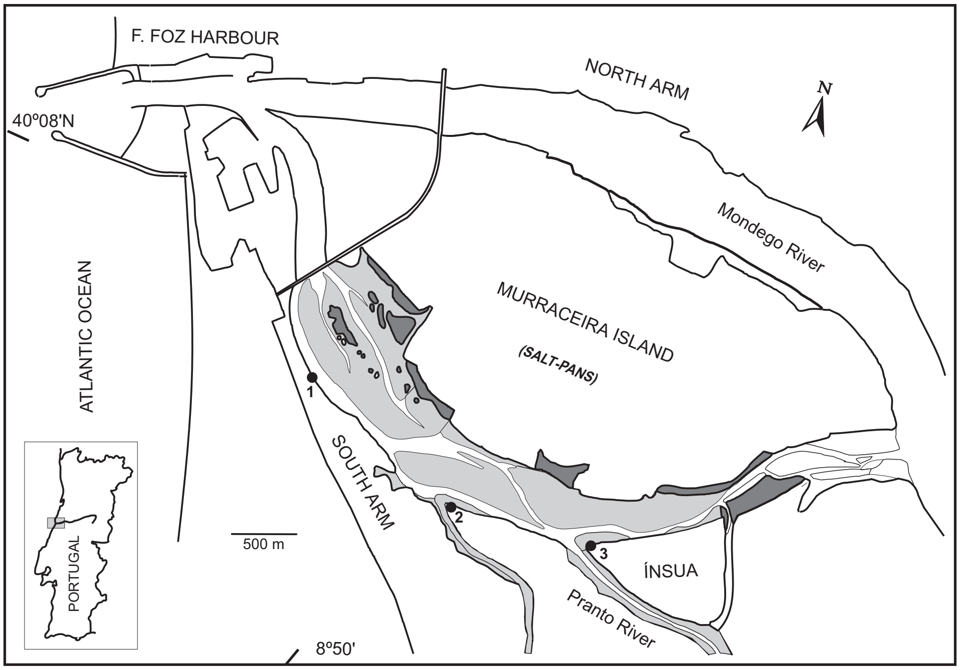

Figure 2.1: Map of the study area, showing the intertidal mudflats (light grey) and the saltmarshes (dark grey) areas of the south arm. Also shown are the positions of the three fixed stations used to census the waders in the south arm

there is a frequent occurrence of ‘nortadas’, strong winds from the north, that can last 3–5 days (Proença, 1988).

2.1.3 Geology

The whole area around the estuary resulted from a complex series of tectonic movements and sedimentation processes which took place in the mid/late Mesozoic (Jurassic–Cretacic periods), and led to the formation of the Serra da Boa Viagem (253 m above sea–level), some 10 km N of the estuary, which extends for 6–7 km in a general E–W direction (Proença, 1988). The basic aspect of the coast line has not changed over the last centuries. However, some alterations took place in the configuration of the area around the mouth of the estuary, due both to natural and human–induced high sedimentation processes (Gonçalves, 1991).The first documented proof of the existence of the Morraceira Island dates back to the 12th century, when King Afonso Henriques donated it to the Santa Cruz monastery of Coimbra (Proença, 1988). However, because of the strong tectonic activity that once occurred in the area, some authors (e.g. Ribeiro, 1869 *in* Proença, 1988) considered that the formation of the Morraceira Island is much older and was due not only to the accumulation of sediments in the formerly deeper primitive estuary, but also to the emersion movements of the coast.

Although the main geological formations in the estuarine environment are of sedimentary nature, mainly alluvium and sandstone (Ferreira, 1991), some differences can be found in the geologic composition of the soils along the margins of the two arms. In the north arm (and also in the Morraceira Island), the soils are mainly formed by Cretacic sandstone with Cenomanian–Turonian (Cretacic) calcareous spots. In Lavos (south arm), on the other hand, they are mostly constituted of clays with sandstone deposits (Ferreira, 1991).

2.1.4 Hydrography and bathimetry

The two arms are clearly different with regard to their hydrographic characteristics. The total submerged area [1] over an average high–water, amounts to 541 ha, of which 59.7% (323 ha) is in the north arm and 40.3% (220 ha) is in the south arm (Jorge, 1991). The freshwater flows mainly through the north arm, where the harbour facilities are located. The south arm, on the other hand, is heavily silted–up and presents many of the characteristics of a tidal lagoon (Marques *et al.*, 1993b). The average depth in the north arm is 8–10 m at high–tide, while in the south arm it is only 2–4 m. According to the tide–tables published by the Junta Autónoma do Porto da Figueira da Foz (JAPFF), the average tidal range in 1995 was 2.06 m. Average tidal heights were 0.97 m (low–water) and 3.03 m (high–water), while extreme values (spring tides) were, respectively, 0.23 m and 3.9 m (Anonymous, 1995).

2.1.5 Salinity and water temperature

Salinity values [2] differ between the two arms, reaching higher annual values in the north arm (range: 0.5‰–35‰, average: 21.5‰) than in the south arm (range: 0‰–25‰, average: 7.1‰) (Marques, 1989). This contrast is most strongly marked in winter; in summer, on the other hand, it seems that the difference is not so clear, with sites in both arms presenting similar values (Marques *et al.*, 1993b,a).

The water surface temperature fall in the range 12.3^{o}C –21.8^{o}C in the north arm, and 10.8^{o}C –21.4^{o}C in the south arm (minimum values in February and maximum values in July and August). A seasonal pattern similar to that previously described for salinity is also observed. In both cases, the differences found between the two arms can be attributed to different patterns in water circulation in the estuary, and also to depth differences from the mouth (Marques *et al.*, 1993b,a).

As to the salinas, there is virtually no information, at present. Some preliminary measurements of salinity and water temperature made on 29.10.93 in three active salinas fall in the range 18.7‰–37‰ and 18^{o}C –22^{o}C , respectively, depending on the type of pond within the salina (pers. observ.), but these values are likely

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[1] From the mouth of the estuary to the bifurcation of the two arms

[2] Taken in high–water spring tides (Marques, 1989)

to be substantially higher during the salt extraction period (see Batty, 1991, for comparison).

2.1.6 Sediments

A comprehensive description of the major sediment types in the estuary remains to be published. Most of the data presented here were kindly provided in a pre–publication form by D. N. Duarte.

The particular hydrodinamic conditions in each arm of the estuary are the main factor that affected the distribution of the sediments. In the more open and active north arm, the coarse–grained sediments predominate, either brought by the river, or dragged into the estuary by the tidal movements. Fine sediments are found only in 'sheltered' environments (e.g. within the harbour perimeter), although these are not so fine as the south arm's sediments (Pena dos Reis & Duarte, 1990). In the closed south arm, on the other hand, a more complex pattern of sediment distribution can be found. In those flats adjacent to the margins of the south arm, or of the Morraceira Island, a gradient of increasing coarser sediments (mud to muddy–sand) take place from the shore to the river channel along the whole reach of the south arm. In the more exposed flats between the channels, an upstream–downstream gradient is observed. The coarse sand sediments dominate the upstream central flats (although an area of softer sediments can be found in the middle section, around the *Spartina* marshes) and the upper part of the downstream central flats (Figure 2.2).

Moreover, the bottom of the channels change from sandy sediments to sandy–mud and finally muddy sediments in the innermost area.

2.2 Biological characterisation

2.2.1 Preliminary studies

Despite the recognised importance of the Mondego estuary in the regional context (Marques, 1989), almost no studies of its biological communities existed before 1984–1985. Concerning the benthic invertebrates, only the studies of Nobre, in 1938–1940, and Maren, in 1974, included information on some faunistic groups and, even so, only as part of a more global survey of the Portuguese coast (Marques *et al.*, 1984). A comprehensive and detailed survey was initiated only in 1983–1984 by the Zoology Department of the University of Coimbra. The aim of this work was to set up the basis for a better knowledge of the biological communities of the estuary, mainly focusing on the intertidal and subtidal benthic invertebrates.

Marques *et al.* (1984) presented the first preliminary analysis of the major intertidal macrobenthonic communities in the estuary. They located and described the main biotopes from both hard and soft sediments, as well as their associated fauna. According to this work, the most representative benthic species were the

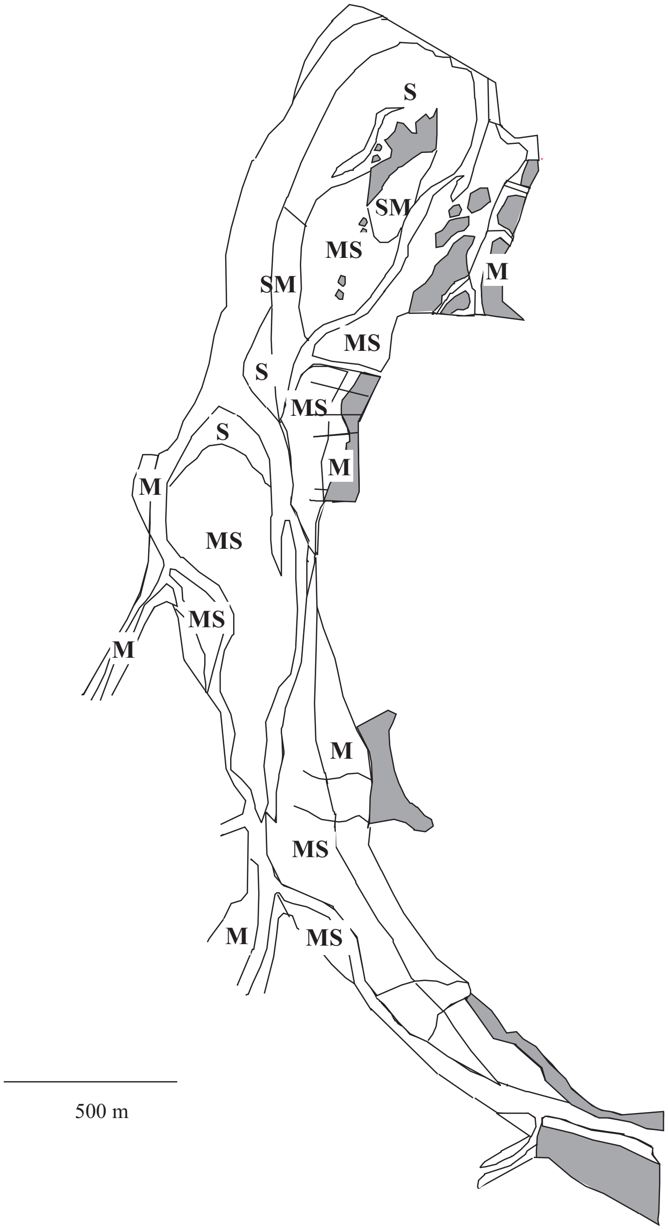

Figure 2.2: General aspect of the distribution of gross sediment classes in the south arm intertidal flats. M - mud; MS - muddy–sand; SM - sandy–mud; S – sand

polychaete *Nereis diveriscolor* and the bivalves *Scrobicularia plana* and *Cerastoderme edule*. The main epibenthic species were the crustaceans *Carcinus maenas*, *Crangon crangon*, *Paleamon longirostris* and *P. serratus* and the fish *Anguilla anguilla* and *Potamochistus minutus*.

As the sequel to this study, a considerable amount of work has been performed, embracing a variety of individual groups and/or communities. A summarised account of these studies is given below.

2.2.2 Studies on individual species and other taxonomic groups

Primary producers

General descriptions of the main communities of marine plants (either algae or vascular plants) have been given on several occasions (Marques, 1989; Marques *et al.*, 1984, 1993b), but until now, no detailed studies of particular species or groups have been attempted. A preliminary survey, initiated in 1995 detected a total of 10 species of vascular plants in the inner area of the estuary, five in the intertidal zone (*Zoostera noltii*, *Spartina maritima*, *Halimione portulanoides*, *Salicornia* sp. and *Arthrochnemum fruticosum*) and five in the supratidal habitats (*Limarium* sp., *Atriplex prostratum*, *Betta maritima*, *Sacrocornio perenis* and *S. fruticosum*) (M. J. Martins, pers. comm.).

Zooplancton

Gonçalves (1991) described the general structure and seasonal occurrence of the zooplanctonic community of the estuary, as well as several aspects of its ecology (larval emission, transport and retention, recruitment and vertical migration of the zooplanctonic species, distribution patterns and nictimeral cycles). He also studied in some detail the larval development of the crustacean *Rhitropanopnaeus arisii*, both in the field and under laboratory conditions.

Ichtyoplancton

Ribeiro (1991) made a preliminary description of the ichtyoplancton community of the estuary and studied several aspects of its ecology. He also made a detailed study of the ecology of the larval stages of the anchovy, *Engraulis encrasicolus*.

Polychaetes

The first study dealing with this group was performed by Gonçalves & Ribeiro (1987). These authors made a preliminary inventory of the polychaetes of the estuarine intertidal sediments, as well as of their spatial distribution. They concluded that *Nereis diversicolor*, *Stroblospio dekhuyzeni* and *Amage adspersa* were the most abundant species. Moreover, the first species was also distributed over the whole estuary. These authors also studied, in some detail, the life cycle of *N. diversicolor*, obtaining values of 1.6 to 3.6 for the P/B ratio of the adults, and maximum densities of 575.5 indivduals.m^{-2}.

Pardal *et al.* (1993) investigated the subtidal polychaete populations, both spatially and over the year. They found an impoverished fauna, regarding the number of species. The two arms had a clearly different faunal composition, with a large number of species and individuals in the south arm and an extreme faunal impoverishment in the north arm. The dominant species was *Amage adspersa*, followed by numerically less important populations of *Capitella capitata*, *Heteromastus filiformis*, *Polydora cyliata* and *Streblospio shrubii*, in the south arm, and *Nereis diversicolor* and *S. shrubii*, in the north arm. The highest total abundance was recorded in June, with 1 204 individuals.m^{-2}

Pardal (1995) studied the life cycle and population dynamics of the sedentary polychaete *Amage adspersa* in the subtidal zone.

Molluscs

A preliminary study on some aspects of the life cycle (growth and reproduction) and of the spatial distribution of *Scrobicularia plana* was undertaken by Guilhermino (1985), who found stable populations in two sampling stations in the south arm, but a decreasing one in a sampling station of the north arm. The highest density occurred in the south arm, with 460–470 individuals.m^{-2}.

Marques & Guilhermino (1988) made the inventory and studied the distribution of the intertidal molluscs. They assigned 22 species, the more important which were *Haminea hydatilis*, *Hydrobia ulvae*, *Littorina litorea*, *Cerastoderme edule*, *Mytilus galloprovincialis* and *Scrobicularia plana*.

The life cycle, population dynamics and production of *Hydrobia ulvae* in relation to the occurrence of macroalgal blooms were recently studied by Baptista (1997).

Crustaceans

This was the most intensively studied group. A preliminary investigation of the Peracarid crustaceans (Amphipods and Isopods) in the intertidal area was published by Marques *et al.* (1988). These authors recorded 21 species of Amphipods and 14 species of Isopods. *Cyathura carinata*, *Sphaeroma rugicauda*, *Echinogammarus marinus*, *Melitta palmata* and *Hyale stebbingi* were the most abundant species.

Marques (1989) described the life cycle of *Echinogammarus marinus* over two years. He found a maximum density of 1 781 individuals.m^{-2}, a net production of 6.36 g.m^{-2}.year and 8.81 g.m^{-2}.year in the first and the second years, respectively, and P/B ratios of 6.35 and 6.08. This author has also investigated the importance of the Amphipods in the macrobentihc communities of the Mondego estuary.

Marques *et al.* (1994) studied the population dynamics, life cycle and production of *Cyathura carinata* in two sampling stations of the south arm. They found maximum densities of about 1 000 individuals.m^{-2} and a P/B ratio of 1.65–2.03.

Fishes

Apart from some data on the biology and ecology of adult *Engraulis encrasicolus* presented by Ribeiro (1991), the only published study regarding the fish of the estuary was that of Jorge (1991). This author described the fish assemblages of the Mondego estuary, recording 48 genus and 67 species, distributed over 30 families. She also presented data on the growing period and spatial colonisation of the juveniles of the most important species of economic interest (*Dicentrarchus labrax, Sparus aurata, Scopthalmus rhombus, Diplodus vulgaris, D. sargus, Solea* spp. and *Platichthys flesus*).

Birds

Up to very recently, the only known data on the wading birds of the Mondego estuary were some winter census of waders coordinated by the CEMPA[3](Rufino, 1979, 1989, 1990; Rufino & Neves, 1986). Additionally, information on ringed waders deliberately shot or ‘found dead’ (most probably also shot) in this estuary was scattered through several ringing reports (Freire, 1969; Oliveira, 1974; Ferreira, 1979; Silva & Castro, 1991, 1992).

A preliminary study on the relative abundance and distribution of the wading birds in the estuary was performed by Múrias & Ferrand de Almeida (1991), who counted 39 species and 10 families. The waders (families Charadriidae, Scolopacidae and Recurvirostridae) accounted for nearly 50% of the species represented.

Following this first approach, a series of studies on several aspects of the ecology (particularly the feeding ecology) and the behaviour of the estuarine waders were undertaken. Múrias *et al.* (1991) made a preliminary study of the seasonal and tidal pattern of use of some selected salinas in the Morraceira Island by the waders.

Rodrigues (1991) performed the first study on the behaviour of an estuarine species in this estuary: she analysed the feeding vigilance behaviour of the Dunlin *Calidris alpina* in the field under a variety of conditions (feeding place, flock density, flock composition).

Múrias (1993) studied the relationship between the density of waders and those of their prey in two selected sampling stations of the south arm, and made a preliminary description of the general feeding behaviour of the Dunlin and Grey Plover *Pluvialis squatarola*.

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[3] The CEMPA (Centro de Estudo das Migrações e Protecção das Aves – Center for Study of Migrations and Bird Protection) was founded in 1976 as the research branch of the Secretary of State for Environment. Today, it is a section of the Research and Ecological Studies Division of the National Conservancy Institute (ICN).

Cabral (1995) studied the effect of the algal increase on the distribution and on the behavioural response of some wader species (Dunlin, Grey Plover, Ringed Plover *Charadrius hiaticula*, and Kentish Plover *C. alexandrinus*) in three marked plots of the intertidal feeding grounds of the south arm.

Lopes *et al.* (1995) presented the first data on the diet of Dunlin and Grey plover. A preliminary study of the time–budgets of some low–water mudflat feeding species was also made by Lopes *et al.* (1996).

In a preliminary analysis, Neves *et al.* (1996) compared the reproductive success of the Kentish Plover and the Black–winged Stilt *Himantopus himantopus* in active and deserted salinas of the Morraceira Island.

A more detailed experimental study of the immediate consequences of eutrophicaton to the estuarine waders is currently in progress (R. Lopes, pers. comm.).

2.2.3 General community analysis

Based in part on the preliminary studies on specific groups or species of macrobenthic invertebrates referred above, and on the work of Marques (1989), a reference study of the intertidal macrobenthic communities was carried out by Marques *et al.* (1993b). These authors analysed the structural variations in the Mondego communities both in time (summer and winter) and in space (north and south arms). They identified 90 species of macrobenthic invertebrates distributed over 14 higher taxa. The south arm provided better conditions for the development of the typical estuarine species than did the north arm. The more favourable physico–chemical conditions in this arm, particularly the salinity, were the main factors that accounted for the faunistic differences between the two arms.

A similar analysis was performed on the composition of the subtidal community (Marques *et al.*, 1993a). A similar, although less diversified fauna than that of the intertidal sediments, occurred in the subtidal areas of the estuary. A total of 58 species, distributed through 11 higher taxa was found. As to the general aspects of the spatial distribution, and to the influence of environmental factors, they were essentially similar to those of the intertidal community.

2.2.4 Other studies

The recent tendency of the studies presently taking place in the Mondego estuary is to evaluate and try to predict the consequences of human action, namely the problem of increasing eutrophication, in the composition and in the structure of the biological communities.

The use of simulation procedures to achieve this goal is increasing. For instance, Martins (1995) studied, by simulation, the possible long–term effects of eutrophication on the biology and reproduction of *Cyathura carinata*. The consequences of the substitution of the *Zoostera nolti* meadows by green algae species (mainly *Enteromorpha* spp. and *Ulva* spp.), and its possible influence on the populations of some estuarine species, is also being studied (I. Martins, pers. comm.).

Ongoing studies deal with the attempts to devise a matemathical operational parameter to measure and to monitor the ‘health’ of the estuarine ecosystem in order to detect early the effects of the increasing eutrophication (Marques *et al.*, 1997). In parallel with these global studies, in–depth work on the general biology of the estuarine key–species (e.g. *Echinogammarus marinus* is progressing (P. Maranhão, pers. comm.).

2.3 The waders’ prey: potential feeding resources

2.3.1 The intertidal zone

The above mentioned studies on the invertebrate communities, particularly that of Marques *et al.* (1993b), provided a guidance to what could be expected in terms of the potential prey for waders in the Mondego estuary. Based on these studies, Múrias (1993) presented the first comprehensive data on the potential prey available for waders in two sampling plots of the south arm, which took into account the fact that only certain size–classes of prey are taken by the waders (Zwarts & Wanink, 1993).

According to Múrias (1993), the most abundant potential prey recorded in those two study plots were *Hydrobia ulvae* (more than 6 000 individuals.m^{-2}), *Scrobicularia plana* (828 individuals.m^{-2}), *Cyathura carinata* (731 individuals.m^{-2}) and 13 species of small polychaetes, largely dominated (76%) by *Amage adspersa* (6 614 individuals.m^{-2}) (Table 2.1).

If only the individuals of the appropriate size classes [4] were taken into account, the available densities decreased a lot (Table 2.1). Defined this way, the numbers may be directly compared with the data from Marques *et al.* (1993b), which refer to individuals larger than 1 mm long. In most cases, the average densities obtained by Múrias (1993) were higher than those found by Marques *et al.* (1993b). Surprisingly, the densities of *Nereis diversicolor*, a main prey of most wader species (see review of Goss-Custard *et al.*, 1991), were greatly reduced from 1986–87 to 1991–92. This seems to correspond to a real change and not simply to an effect of the spatial distribution of this species, as more recent surveys clearly indicate (J. C. Marques, pers. comm.; M. Pardal, unpublished). In contrast, the small polychaete *Amage adspersa* dramatically increased during this period (Table 2.1; M. Pardal, umpublished).

These and other changes observed in some species may be closely linked to the growth of extensive macroalgae mats that have periodically invaded the estuary since the mid 1980’s. The consequences of such alterations in the macrofauna to the waders are discussed in more detail in Chapter 6.

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[4] Considering only the individuals above 2 mm, according to the review made by Goss-Custard *et al.* (1991)

Table 2.1: Estimates of the main prey densities (ind.m^{-2}) in the estuary of Mondego in 1986–87 (adapted from Marques *et al.*, 1993a) and in 1991–92 (according to Múrias, 1993). All values are average densities for several periods of the year (two in the first case, four, in the second). (1) represents total densities and (2) densities of the size classes above 2 mm. The percentage of the densities of the higher classes (>2 mm) for 1991–92 is also shown.

	1986–87	1991–92 (1)	1991–92 (2)	%
Hidrobia ulvae	1420	6308	2955	47.0
Scrobicularia plana	193	828	261	32.0
Cyathura carinata	225	731	598	88.0
Other crustaceans	43	40	–	–
Carcinus maenas	14	9	–	–
Small polichaetes	185	6614	–[a]	–
Nereis diversicolor	776	23	10	43.0
Oligochaetes	<1	245	–[a]	–
Diptera larvae	6	–	–	–

[a] In most cases they were inferior to 2 mm, so their division into classes was considered unnecessary

2.3.2 The salinas

Knowledge of the benthic fauna that can be taken by waders in the salinas is still very poor in most Portuguese areas. In Algarve, where most studies have been performed, Rufino *et al.* (1984) found that most of the potential prey to waders in a group of salinas of the Ria Formosa, in winter, were Chironomidae (0–1 300 individuals.m^{-2} and Ephydridae (0–440 individuals.m^{-2}) larvae, and *Hydrobia ulvae*. Batty (1991), in his study of a large and traditional salina in the same area, confirmed the numerical importance of the Chironomidae in the autumn and winter months (reaching more than 5 000 individuals.m^{-2} in late winter). Conversely, the potential prey in the salina drastically decreased, by a factor of about ten, in the spring, summer and early autumn months (April to August), when they were dominated by a variety of other invertebrates, from larger Diptera larvae and pupae to crustaceans – *Artemia* – and beetles. A marked seasonal cycle on prey availability seems, therefore to take place in the salinas.

In the Mondego estuary, there is still no information available on the macrofauna of the salinas and adjacent channels. Only in late March of 1995 was an attempt made to collect some benthic invertebrates from a salina were many birds were seen feeding. Eight samples were taken with a corer (95 cm^{2}) up to 5 cm deep and transported to the laboratory, where they were washed, sieved through a 0.5 mm mesh, and stored in 4% neutralised formol. The collected organisms were later separated, preserved in 70% alcohol and identified and counted under a binocular microscope. The results are displayed in Table 2.2.

Apart from the Chironomidae, the taxa present belong all to coleopteran families. Nevertheless, the Chironomidae larvae and pupae accounted for 85% of the total densities present, with 1 037 and 133 individuals.m^{-2}, respectively. The

Table 2.2: Density (ind.m^{-2}) of invertebrate taxa present in a salina of the Morraceira Island, in late March of 1995. All taxa were identified, at most, to the family level.

Taxa	Density
Chironomidae larvae	1307
Chironomidae pupae	133
Dysticidae	114
Corixidae	5
Hygronemidae	20
Hygrobiidae	54
Hydrophylidae	20
Hydrobiidae	44

coleoptera of the family Dysticidae were the third most numerous group (114 individuals.m^{-2}, while all the other families presented densities below the 100 individuals.

These albeit very preliminary results, suggest that the dominant benthic fauna of the Morraceira's salinas must be similar to that of other estuaries, at least in late winter. This is, however, an aspect that needs to be more carefully examined.

At present, there is no quantitative information on the prey that lives in the water column, which are also consumed by waders (Batty, 1991). Occasional observations suggest that adult diptera may be specially abundant over the water surface in spring and summer, but no data has yet been collected that can show how important these, and other free-living species, can be to the waders' diets.

2.4 Major habitats for waders

Of the 1 600 ha of wetlands in the estuary, only 67% (1 072 ha) are really available to the waders (Table 2.3). The remaining areas are subtidal. At high–water, the percentage accessible is reduced to some 55%, corresponding to the supratidal habitats.

A little more than 53% of the potential feeding habitats (including both the intertidal and supratidal habitats) for waders is occupied by channels ('esteiros') and reservoirs ('viveiros') that feed the salinas with saltwater from the estuary (see Chapter 4 for a more detailed description). When these reservoirs are full, the waders are prevented from using them. However, many 'viveiros' are filled with water only during the salt extraction period, while others (specially those that feed abandoned salinas), are open to the tidal regime, thus providing more feeding opportunities to waders.

The salinas, *sensu strictu* (that is, the storage, preparation and crystallisation pans), are the second largest habitat present in the estuary. They occupy about 29% of the feeding supratidal area (and 23% of the total feeding habitats in the estuary) and are mainly located in the Morraceira Island and in the left margin of the south

Table 2.3: Main wader habitats and their approximate areas in the Mondego estuary (maximum area available in low–water spring tides). The area of salinas includes 21 ha of industrial salinas.

Habitats	Area (ha)	% of total area
Intertidal Habitats		
Mud and sandflats	134	12.5
Intertidal marshes	62	5.8
Supratidal Habitats		
Salinas	251	23.3
Aquaculture ponds	54	5.0
‘Viveiros’ and other	574	53.4
Total area	1072	100.00

arm [5]. Some salinas have been abandoned and later transformed into extensive or semi–intensive fish–farm ponds (Marques *et al.*, 1993b). They represent 5% of the potential available habitats, but their extent is tending to increase (see Chapter 4).

The main intertidal habitats, representing about 13% of the total area potentially available for waders’ feeding in the estuary (Table 2.3), consist of mud and muddy–sand flats, which cover most of the low–water area of the south arm. Actually, there are some sandflats in the north arm, which resulted mainly from the constant dredging and engineering works that have been performed to improve the harbour facilities. These flats were, until very recently, of little value for waders, although they were often used for feeding by herons and egrets (Ardeidae) and as low–water resting areas by gulls and shags (Phalacrocoracidae) (pers. observ.). However, the recent conclusion of the engineering works to regulate and make deeper the navigation channel of the north arm may have altered this situation (see below).

A small area of *Spartina* marsh (62 ha, about 6% of the total feeding area available) occurs in the south margin of the Morraceira Island (Table 2.3), specially in the upstream section. In addition, some brackish and freshwater marshes (mainly with *Typha* spp. and *Phragmites* spp.) can be found in long–term abandoned salinas, in the eastern part of the island. In the south–eastern part of the estuary, there are some hundreds of hectares of ricefields, a small part of the 15 000 ha present in the lower Mondego river valley. Although not belonging to the estuarine wetlands, this habitat can be used sporadically by some wader species (Chapter 3).

2.5 Human activities and disturbance

The Mondego estuary has been for a long time the easiest way of communication between the littoral and the inland areas of the central region of Portugal (Marques, 1989). Besides the busy trading harbour of the Figueira da Foz, which moved 400 000–700 000 tons of goods and 300–400 ships per year, during the 1983–1993

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[5] A number of abandoned or drained salinas located in the north arm, which are not used by the waders will not be considered here. A full description of this group can be found in Chapter 4.

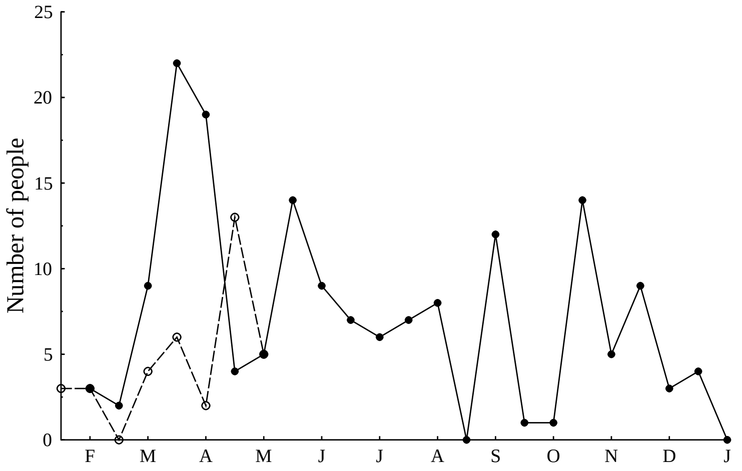

Figure 2.3: Seasonal changes in the number of people in the south arm mudflats in 1993–94 (open circles) and in 1994–95 (closed circles)

period (Anonymous, 1995), this estuary also supports a fishing port and an active shipyard.

The need to improve the harbour facilities has recently led to the onset of important engineering works and dredging activities in the north arm, which were the cause of important physical disturbance to the bottom (Marques *et al.*, 1993b). These activities may have increased the feeding area for waders in the north arm. They allowed a considerable area of sandflats to establish itself between the margin of the Morraceira Island and those of the new navigation channel. These flats, which are exposed at low–tide were rapidly colonised by algae and are now formed of much softer sediments (pers. observ.). The first waders (mainly Greenshanks *Tringa nebularia* and Black–tailed Godwits *Limosa limosa*) were observed there in August 1996 and their numbers, along with the number of species, have shown a tendency to increase since then, particularly in winter (R. Lopes, pers. comm.).

In addition to the harbour facilities, the estuary supports several industries, salinas and fish–farms. Moreover, it receives the nutrient and chemical discharges from agricultural fields of the lower Mondego river valley (Marques *et al.*, 1993a). These effluents are partially the cause of the macroalgae blooms (mainly of *Enteromorpha* spp. and *Ulva* spp.) that periodically occur in the mudflats (Marques *et al.*, 1993b, see Chapter 6).

Also important are the traditional activities of bivalve collecting and fishing. Although especially significant in the spring and early summer months, human disturbance in the mudflats in practically continuous throughout the year (Figure 2.3).

The effects of the human presence on the mudflat biological communities still need to be evaluated. However, as regards to the waders, they can be potentially detrimental. Recent studies (Davidson & Rothwell, 1993) have demonstrated that this kind of human disturbance may have implications for the survival of waders, particularly in cold weather, as they are forced into an unexpected and costly energetic expenditure, when they have to run–off or fly away from the disturbed areas. It is perhaps because of this that the major feeding areas for waders in the Mondego are located downstream, while most people concentrate in the upstream and, although slightly less, in the middle flats.

The intensive human usage of the estuary under all its forms is, of course, an important factor of disturbance of the estuarine communities, and is the reason why Marques *et al.* (1993b,a) consider that the estuary is currently under a severe environmental stress.

Chapter 3

The Wader Assemblage

3.1 Introduction

The first published data on the wader assemblage of the Mondego dates from the mid- seventies, and consists of exploratory winter counts performed by the CEMPA in January–February 1977 and 1978 (Rufino, 1979) to evaluate the major habitats for coastal waders in the country. These censuses did not reveal the presence of quantitatively important populations of waders and the area was, therefore, excluded from the regular counts in the following years (Anonymous, 1979, 1980, 1981, 1982; Oliveira, 1980). In 1986, the estuary was again surveyed, but less than 300 birds were counted (Rufino & Neves, 1986).

From 1987 onwards, a permanent team of the Zoology Department (University of Coimbra) organised the winter counting program in the area, on behalf of the CEMPA. The counts of 1987 and 1988 still showed relatively few birds to be present (89 and 154, respectively) (Rufino, 1990, and pers. observ.). However, in the following two years, with a better knowledge of the area, the numbers recorded increased significantly: 562 in 1989 and 754 in 1990 (Rufino, 1989, 1990).

No counts were performed in 1991, 1992 and 1993. However, a series of studies on several aspects of the estuarine wader assemblage were carried out (Múrias & Ferrand de Almeida, 1991; Múrias *et al.*, 1991; Rodrigues, 1991; Múrias, 1993; Cabral, 1995). These studies led to the conclusion that the previous winter censuses probably under-estimated the numbers really present. Moreover, they provided some data on the possible importance of the area outside the wintering season, especially during the migration periods. The better knowledge of the area and of the species acquired through these initial and partial studies provided a sound basis on which to carry on a detailed census program that could fill in the gaps.

The main objectives of this Chapter were (1) to study the species composition and its seasonal variation; (2) to assess the significance of the area for several species and (3) to place the Mondego estuary in the national and international perspective, and to provide data on the ornithological segment of the estuarine food-web that could be useful for future studies.

3.2 General Methodology

3.2.1 The census method: advantages and limits of its use

The most frequently used method to assess wader numbers is through direct counts of the birds on their feeding or roosting sites because, as Prater & Lloyd (1987) state "…they are large, readily identifiable without being collected and easy to see …". This method belongs to the family of *absolute methods*, as it allows, in principle, the assessment of the total number of birds present in the area to be made (Blondel, 1969). In large estuaries, specially, counts are performed two hours either side of high–water on spring tides, when birds are concentrated in a limited number of roosts (Prater, 1981; Prater & Lloyd, 1987) and movements between areas are at a minimum. In small estuaries, however, it is possible (and often advisable) to make the counts at low–water (Prater & Lloyd, 1987). In both cases, the method implies perfect knowledge of the area to be surveyed, in order to include all, or at least the most important, refuges and/or feeding areas (Prater, 1981).

In practice, however, it is seldom possible to count *all* the birds in an area even in the best conditions. When dealing with estimates from a single count, it is important to distinguish between *precision* and *accuracy* (or *reliability*). The first term refers to how close the estimate is to the number of birds really present. In practice, it will be very difficult to achieve (Bibby *et al.*, 1992), but it tends to improve with the increase in cover (e.g. by counting all the high–tide roosts). Accuracy, or reliability, on the other hand, deals with systematic departures from an average which cannot be improved by increasing the cover (Bibby *et al.*, 1992). Ideally, any count should be both precise and accurate. Errors are always present, however, and the main goal of anyone who initiates a census program should be to minimise the effect of such errors.

Prater (1981) and Prater & Lloyd (1987) identify four main factors that are likely to affect the accuracy of wader counts. Of these, they also identify two which are directly applicable to counts in a single estuary: (a) those that result from incomplete counts[1] and (b) those that derive from counting inaccuracy. The latter error is most likely to arise when direct counts of individual birds are not possible. The magnitude of the errors arising from both sources can be explored.

The margin of error due to *incomplete counts* can be substantially reduced through an increase in counting effort (usually resulting from a good knowledge of the area). An appraisal of the magnitude of error deriving from this source can be made by comparing high–water and low–water counts made in the same area and season. Although low–water counts may be as error–prone as high–water counts, the close relationship shown in many studies between high and low–water counts (e.g. Yates & Goss-Custard, 1991) does raise confidence that all roosting areas have been located: it would seem unlikely for exactly the same number of birds to be missed when birds are concentrated at roosts and when dispersed widely

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[1] Mostly due to the absence of birds that use, as resting and/or feeding areas, the inland habitats adjacent to the study area, and also to the omission of roost and/or feeding areas.

over the feeding areas.

The errors associated with *counting inaccuracy* are more difficult to overcome. In most counting procedures it is necessary to resort to estimates: the observer counts a small group of birds and, mentally, superimposes the group on to the rest of the flock (Blondel, 1969; Luís, 1989). Although at first sight this would seem to be inaccurate, the method is, nevertheless, surprisingly precise when the observers are trained (Prater, 1979, 1981).

The accuracy of the counts depends of several factors, including the species involved, the dimension and the density of the flocks, the topography, the weather conditions, and even the observer's experience (Prater, 1981). Generally, there is a tendency to overestimate small flocks (100–400 birds) and to underestimate large ones (1000–3000), the error margin varying between 10% and 25%, according to flock size and observer experience (Prater, 1979, 1981). In normal conditions with trained observers, the error is closer to the lower limit (Blondel, 1969; Prater, 1979, 1981; Prater & Lloyd, 1987). As a high degree of consistency has been found between counts made by the same observer and between counts made by different observers (Prater, 1979), the counts from different areas can usually be compared with confidence.

The test of counting accuracy, the second error factor, demands precise counts of the number of birds present which can be used in comparison with repeat estimates made using the some counting procedure. Although photographs of flocks of different sizes can be used to test the accuracy of an observer or inter-observer reliability, a photograph may not completely reflect the field situation, since factors such as habitat type, weather conditions, and the characteristics of dispersion, conspicuiness and behaviour of the species involved, can strongly influence the results (Prater, 1981). Therefore, it is highly advisable to repeat this kind of test in the field from time to time, as indeed is usual with professional teams (Prater, 1979).

3.2.2 Counting procedure

Field methods

According to several authors (in Batty, 1991), the accurate estimate of wader assemblage composition is best made on falling tides, just before the birds begin to spread throughout their feeding areas. Alternatively, birds can be counted in their refuge areas at high–water, which is the most widely used method (see above) (Prater, 1981; Prater & Lloyd, 1987).

However, the method used in this study was that of low–water counts because (1) another part of the study required the birds to be censused at this time (Chapters 5 and 6) and (2) accurate high–water counts were thought to be more difficult in this estuary, due to the particular nature and scattered distribution of the supratidal habitats which led to a scatter distribution of the birds themselves. Even so, birds may also have been missed in the low–water counts. Therefore, a series of high–water counts was made in the Morraceira's salinas on the same day that low–water counts for the purposes of comparison (see below).

The censuses were performed fortnigthly[2] from July 1993 (in the salinas) or October 1993 (in the mudflats) to May 1995. Whenever possible, low–water and high–water censuses were carried out in the absence of strong wind and/or rain or fog, since adverse weather conditions can strongly influence the results (Prater, 1981; Luís, 1989). The whole of the south arm mudflats was surveyed from three fixed stations located downstream (Chapters 5 and 6), while a transect was used to survey the supratidal habitats (salinas, 'viveiros', fish–farms and saltmarshes) in the Morraceira (Chapter 4)[3]. The position of the fixed stations was such that, in order to get there, it was necessary to cross most of the south arm supratidal habitats that were, in this way, also surveyed. However, these habitats proved to be seldom used, at least at low–water.

Initially, each census was completed in three days (two for the intertidal mudflats and one for the supratidal habitats). But with the expertise gained, from February 1994, it was possible to do the work in just to two days (one for each type of habitats), which greatly reduced the likelihood of double counting.

Counts were made by two persons using binoculars (10×50) and a telescope (30–90×60), within ±2 hours of low–water on spring tides. Birds were identified and counted, or estimated, and some additional information taken (details on Chapters 4 to 6). A total of 38 visits was made to the study areas.

Checking the reliability of the counts

Errors due to incomplete coverage In order to look for errors due to incomplete coverage, the total number of birds counted on the estuary at low–water was compared with the number counted at high–water roosts/feeding areas in the salinas of Morraceira. In most cases, fewer birds were found in the high–water censuses of the Morraceira than at low–water on the estuary by a proportion that was variable, but especially high in the first winter (Figure 3.1)[4].

This deficit in bird numbers at high–water was in contradiction with the general tendency found elsewhere (e.g. Prater, 1981; Yates & Goss-Custard, 1991) and could be due to (1) a deficient coverage of the main high–water roost/feeding areas, (2) inaccuracy in the high–water counts, due to the limitations and difficulties already mentioned, and (3) double-counting at low–water. Although none of the hypothesis can be completely eliminated, it is believed that most of the difference between high–water and low–water counts were indeed due to a less good coverage of the high–tide feeding areas in 1993–94.

In fact, from July 1994 onwards, after six new salinas had been added to the number visited in the Morraceira, most of the differences between high–water and

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[2] Except prior to February 1994, when only monthly counts were made.

[3] For a description of the study area, see Chapter 2

[4] The Avocet and the Grey plover were excluded, since they could all be counted at low–water. Conversely, the Black-winged Stilt and the Curlew Sandpiper were only found in the salinas and, therefore, were also excluded.

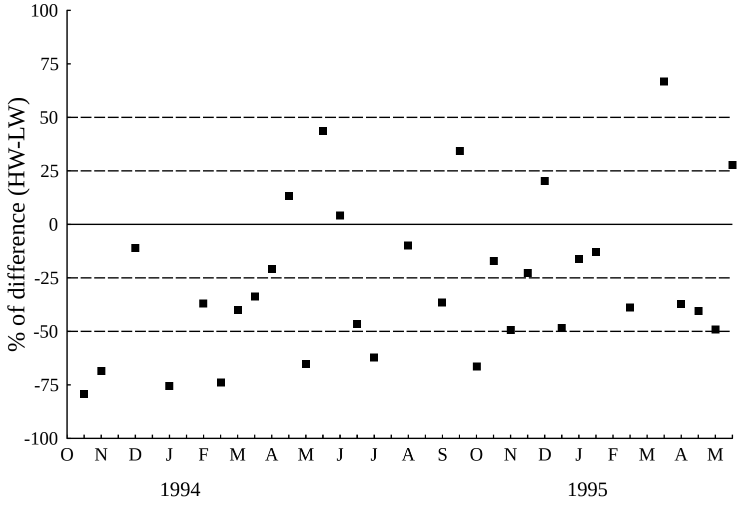

Figure 3.1: Percentage of difference between total low–water and high–water counts from October 1993 to May 1995.

low–water counts fell within the 50% limit of variation whereas, until then, they had largely exceeded this limit. Moreover, in the winter 1994–95 (November to February), a series of regular high–water counts was also performed in the supratidal habitats of the south arm. The average difference between high–water and low–water winter counts generally dropped in 1994–95 (–29.8%), as compared to the previous year (–53.2%), which further suggest that the uncounted areas may have also been used by waders in the winter of 1993-94. Overall, up to 77.8% of the counts differed by 50% or less, with only 22.2% being above this limit, which was considered a good correspondence (see Yates & Goss-Custard, 1991).

Errors in counting accuracy In general, it was possible in the mudflat areas to count the birds one by one, as they usually fed far apart. But on some occasions, particularly in winter, birds fed in large and compact flocks, and estimates had to be made instead. Compact flocks also occurred in the salinas at high–water. To verify the accuracy of the estimates, several trials were made in both sites with flocks of different sizes of Dunlin (small sized species) and Avocet (large sized species) (Table 3.1).

An overall, and acceptable, average difference of 14% was obtained, which is in agreement with the values of Prater (1981). The results of these trials also suggests that inaccuracies in high–water counts in salinas were not too large, and these counts could provide, in some cases, a better estimation of the numbers present than the low–water counts, particularly for the small sized-species (e.g. Dunlin).

Table 3.1: Comparison between estimates of the same Avocet and Dunlin flocks in the two main habitats. Flock sizes are the means of each set of estimates.

Species	Habitat	Date	Number of estimates	Mean flock size	Average difference from the mean
Dunlin	Salinas	7.1.94	3	395	12.6
Avocet	Mudflats	25.1.94	3	477	5.0
Dunlin	Mudflats	27.4.94	2	1986	10.1
Avocet	Mudflats	1.2.95	2	555	12.6
Avocet	Mudflats	17.3.95	4	160	18.5
Dunlin	Salinas	15.4.95	2	95	14.7
Average (SE)					**13.3 (5.3)**

To control for the inter-observer variability, regular comparisons of the results of simultaneous counts performed by the two observers were made.

3.2.3 Which counts to use?

Given the results presented in section 3.2.2, the highest counts obtained for each species in each sampling occasion were used in the analysis, on the grounds that the most serious errors arose from birds being missed rather than being not counted accurately, independently of the tidal state. They were considered to represent the minimum number of birds present in the estuary in a given census and suited well the purposes of this Chapter.

Data analysis

The individual censuses were used in all analyses. The few missing ones were interpolated by calculating the average of the two nearest censuses made before and after the missing data (Wolff & Smit, 1990). For the evaluation the seasonal variations in the birds' populations, the simple totals for each count were used. However, when refering to the total use of the area for the whole study period (i.e. from July 1993 to May 1995), total bird-days were thought to provide a better measurement. Bird-days were calulated as: average number of birds per month$\times$30 days$\times$23 months. The widely used index of Shannon-Wienner (Washington, 1984) was used to measure wader diversity. Factorial Analysis of Correspondence, or simply Correspondence Analysis (COA) (Legendre & Legendre, 1984), was used to study the affinities between the wader assemblages of the main portuguese estuaries.

Factorial analysis belongs to the group of ordination methods whose purpose is to simplify and condense a massive data matrice $X_{S,N}$, with S rows (species) and N columns (sampling units, SU), into a smaller matrice, p, with a minimum of information loss, in which there is one dimension or axis for each species where the SUs are positioned by orders of magnitude of the variable measured (e.g. abundance), into in a reduced space, preserving the relations between clusters.

This corresponds to a reduction of the *n*–dimentional space into a smaller *p*–dimentional space, where the relations between SUs can be more rapidly examined: this operation is called 'reduction of dimensionality' (Ludwig & Reynolds, 1988). SUs (and species) located on opposite sides of an axis are highly different; conversely, those located side by side along an axis are highly similar (Ludwig & Reynolds, 1988). If the SU ordination is based on the resemblance between species (i.e. rows) it is called R–mode ordination. Conversely, if it is based on SU resemblance (i.e by columns) it takes the name of Q–mode ordination (Legendre & Legendre, 1984).

The number of axes that is extracted is equal to the number of species (or SUs, depending of the mode selected for the analysis). However, more than three axes (or dimensions) are not able to be plotted on a graph and so inspected visually and are, therefore, difficult to interpret (Legendre & Legendre, 1984). Fortunately, the majority of the variability in most data sets is 'captured' in the first two or three axes, which greatly simplifies the analysis (Ludwig & Reynolds, 1988).

The correspondence analysis deals with sets of data described by only two qualitative characters (estuaries and species, in the present case), whose quantities are (1) homogeneous (in this case, the unit is the number of birds), (2) can be summed (e.g. number of birds of all species in an estuary), and (3) are amenable to being transformed into frequencies (e.g. proportion of birds of a species in a given estuary) (Ribeiro, 1991). The analysis is based on the χ^2distances of the weighted sums of the n species in the k SUs (estuaries, in this case) (Legendre & Legendre, 1984). The COA presents several advantages over other ordination methods: it does not need to fulfil the stringent conditions required for the use of parametric tests, and allows both the individual species and SUs to be simultaneously plotted, thus making the interpretation easier (Ludwig & Reynolds, 1988; Ribeiro, 1991).

The winter assemblages were chosen for this analysis, not only due to the lack of data for most estuaries for bird numbers outside the winter, but also because it is in winter that the Portuguese estuaries are of greatest importance for waders (Smit & Piersma, 1989). A matrix of 22 wader species$\times$8 sites (estuaries) was constructed, using the database of the winter national census, co–ordinated by the CEMPA. In the absence of more recent data, the numbers of waders for the period 1991 to 1993 (Rufino, 1991, 1992; Rufino & Costa, 1993) were averaged, for each species and site, except for the Ria de Aveiro, for which only two years data (1992 and 1993) were available. Averaging across years had the advantage of reducing inter-annual variability.

Ideally, data for the Mondego from the same period should have been used; however, no complete counts were available for this area prior to 1993–94. On the other hand, these data existed for the winters of 1993–94, 1994–95 and 1995–96, so an average across these three years was used. This allowed to compensate for any over-estimations due to the counting method used before February of 1994 (see above). In fact, it was found that the 1993–94 count in the Mondego overestimated the mean number of waders recorded for the winters of 1993–94 to 1995–96 by an average, across species, of 6.2%, although individual species differences could be as high as 44%. Moreover, six species were not present, as they were recorded only in later years. But as these species were present only in very low numbers (<15 birds), it was considered that their omission was not likely to affect the final results.

The following zones (from the north to the south of Portugal) were considered: the Minho estuary, on the northern portuguese border; the Ria de Aveiro, the Mondego, Tagus and Sado estuaries, along the western atlantic coast; the Ria de Alvor and the Ria Formosa, in western Algarve; and the saltmarsh of Castro Marim (Guadiana estuary) in the eastern Algarve. The calculations were performed with the program NTSYS–PC, version 1.7 software (Rohlf, 1988).

3.3 Results

3.3.1 The wader assemblage

Permanence time

The number of months in which a species is present in an estuary is called the 'permanence time' (Amat, 1984 *in* Batty, 1991). As Batty (1991) notes, permanence time refers to species, not to individual birds, which usually remain for shorter periods.

In the Mondego estuary, six species (Golden Plover *Pluvialis apricaria*, Snipe *Gallinago gallinago*, Lapwing *Vannelus vannelus*, Curlew Sandpiper *Calidris ferruginea*, Greenshank *Tringa nebularia*, and Sanderling *Calidris alba*) used the estuary for short periods (1 to 4 months). Seven species (Ruff *Phillomachus pugnax*, Turnstone *Arenaria interpres*, Avocet *Recurvirostra avosetta*, Bar-tailed Godwit *Limosa lapponica*, Curlew *Numenius arquata*, Knot *Calidris canutus* and Whimbrel *Numenius phaeopus*) were present for about half a year. Finaly, nine species (Black-winged Stilt *Himantopus himantopus*, Grey Plover *Pluvialis squatarola*, Black-tailed Godwit *Limosa limosa*, Little Stint *Calidris minuta*, Common Sandpiper *Actitis hipoleucos*, Redshank *Tringa totanus*, Ringed Plover *Charadrius hiaticula*, Kentish Plover *Charadrius alexandrinus* and Dunlin *Calidris alpina*) were present for most of the year (7 to 12 months)). (Figure 3.2).

Seasonal variations in total bird numbers, species numbers, diversity and species composition in the Mondego estuary

Total numbers, species numbers and diversity A total of 591 909 bird–days and 22 species were recorded in the Mondego during the study period. Across all species, more birds were present in 1993–94 than in the following year; however, the patterns of seasonal variation in total numbers were similar. The maximum values in individual counts occurred during the winter months (November–February).

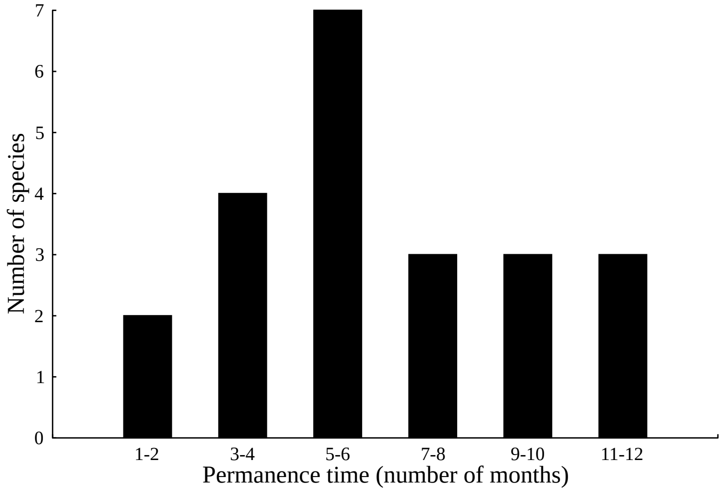

Figure 3.2: Permanence time of the species in the Mondego. Data refer to the number of months in which a species was recorded, independently of the year of occurrence.

A second peak ocurred in April–May (pre–nuptial migration) while a smaller peak was recorded from late July to September (post–nuptial migration). During the summer months (June, early July), few birds remained on the estuary (Figure 3.3). The number of species in the Mondego varied between 10–12 throughout the year (Figure 3.3).

The highest number of species was recorded during the spring migratory passages (March–May), while the lowest number was present during the summer months (June and early July).

In spite of the seasonal variations in numbers, the Mondego's wader assemblage was a diversified one, as was demonstrated by the genererally high values of diversity (Shannon's H) (Figure 3.4).

The diversity values decreased in late April–May, coinciding with the overwhelming numerical dominance of Dunlin during this period, and in June–July, where the nesting species (Kentish Plover and Black–winged Stilt) were virtually the only species present.

Species composition A particular group of species dominated the assemblage numerically throughout the study period. The winter months (November–February) were largely dominated by Dunlin and Avocet (Figure 3.5).

In late February–early March, the onset of migration resulted in the increase of the representativeness of another group of species (Black–winged Stilt, Ringed

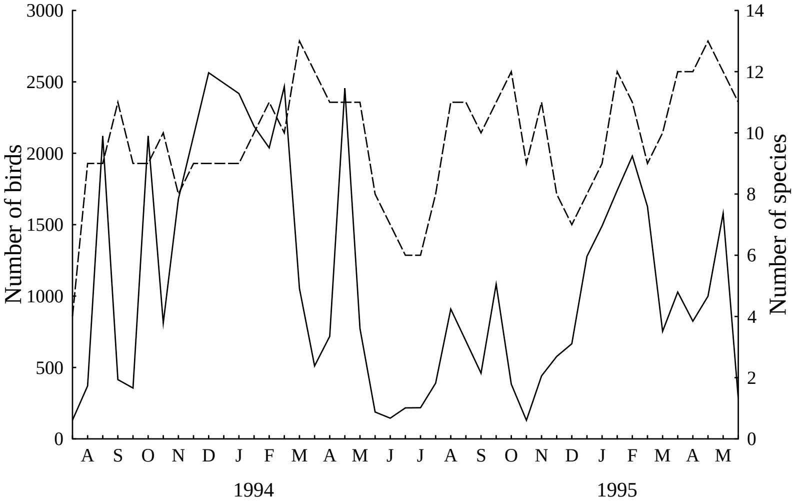

Figure 3.3: The number of birds of all species combined – heavy line – and the number of species – broken line – recorded each sampling occasion during the study period.

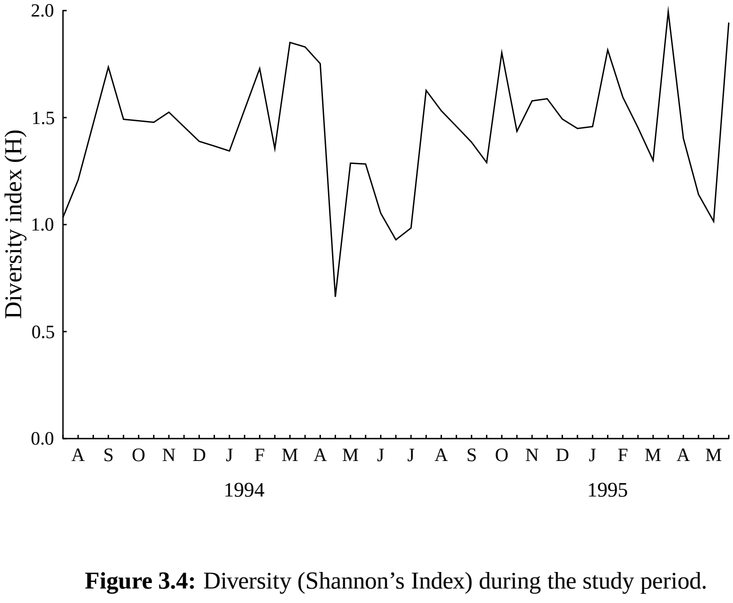

Figure 3.4: Diversity (Shannon's Index) during the study period.

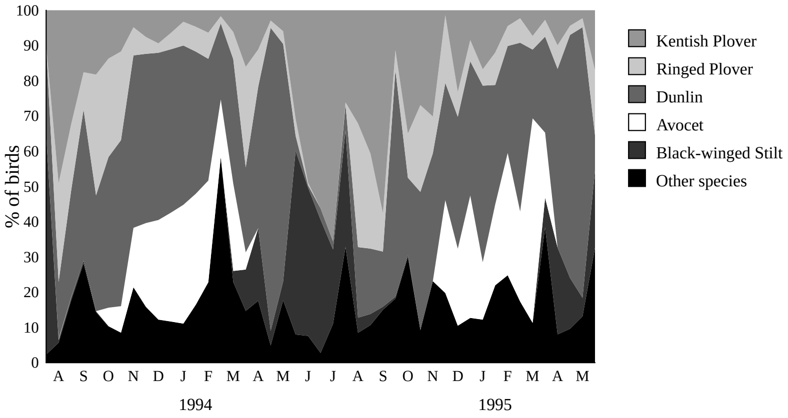

Figure 3.5: Species composition, in % of all the waders counted during each sampling occasion. Refer to Figure 3.3 to see total numbers

Plover). The spring passage (March–May) was again dominated, although briefly, by Dunlin, while Black–winged Stilt replaced the Avocet as the second most abundant species. In the summer months (June–July), the Black–winged Stilt and the Kentish Plover were the dominant species, as both nested in the area. The assemblage was more diversified in the autumn passage (August–October), although Dunlin and the two small plovers (Kentish and Ringed Plovers) were, again the most represented species.

Variations in the specific patterns of abundance Each species showed a characteristic pattern of occurrence throughout the year. The counts for July, August and September 1993 were the maximum numbers found in the salinas only, and so may underestimate the actual numbers. Nevertheless, they are presented as they are thought to be adequate to illustrate the general patterns of seasonal variation in species abundance. Data are presented in Figures 3.6 to 3.9

Kentish Plover. This species was present during the whole study period, with a total of 76 479 bird–days. The highest values in individual counts were reached in August and early September, during the passage of autumn migrants (and probably also of juveniles born in the estuary in this year) and in mid–winter (December and January). This species is one of the two that nest in the estuary, breeding in the salinas of the Morraceira and elsewhere (see Chapter 4). No attempt was made to count the number of nests. However, based on the total number of birds present in late March and April in the Morraceira, a minimum number of 30 to 40 pairs may have nested in the island in the 1994 breeding season.

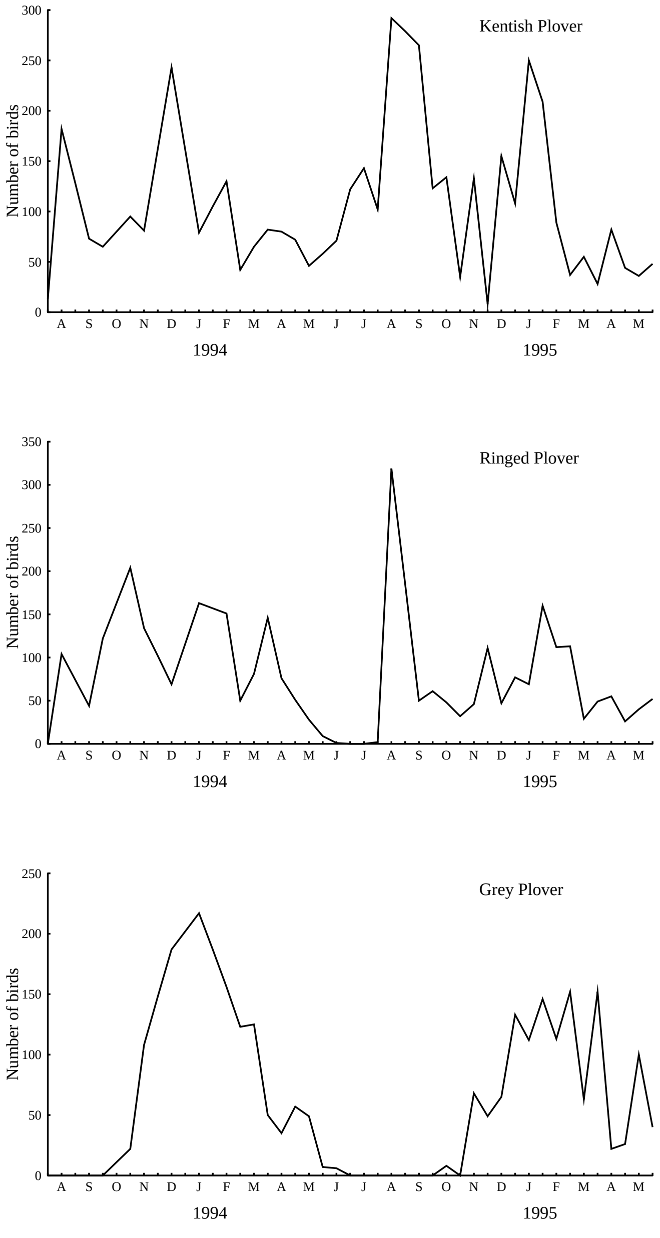

Figure 3.6: The number of the most abundant species that were recorded in each sampling occasion during the study periods: Kentish Plover, Ringed Plover and Grey Plover

Ringed Plover. This species was also present throughout most of the year, excluding late June and early July, with 45 528 bird–days. It occurred in the highest numbers during the autumn passage, although the peak values in individual counts recorded at that time of the year differed between years (204 birds in October 1993, and 319 in August 1994). In contrast, mid–winter numbers were small, although similar between the two years (163 and 160 in January of 1993–94 and 1994–95, respectively). Maximum numbers in spring differed between the two years (146 birds in April 94, and only 55 in April 95).

Grey Plover. This was a typical wintering species. Some 35 528 bird–days were recorded, mainly from October to May. The maximum numbers of birds present in individual counts were recorded in winter, but differed between the two years, with 217 in 1993–94 and 152 in 1994–95. However, the yearly pattern of occurrence was much more regular than that of most species, with higher numbers consistently being recorded in mid–winter, followed by a decrease in late winter and a smaller peak in late March to April/May, corresponding to the migratory passages. Maximum numbers recorded during this migratory phase varied between 125 (in 1994) and 152 (in 1995).

Curlew and Whimbrel. The two species of the genus *Numenius* were sometimes difficult to distinguish in the field. However, Curlew *N. arquata* was mainly a winter visitor, while Whimbrel *N. phaeopus* could be found intermittently from early spring to late autumn. The number of total bird–days were 468 for Curlew and 4 830 for Whimbrel. The peak number in individual counts for Curlew was 12 birds in late February of 1994, while Whimbrel reached its highest numbers of 35 birds in July 1994. Recent information seem to indicate that Whimbrel is present more continuously from April to October than is suggested by the counts, because at low–water Whimbrel also use the saltmarshes, where it can easily be missed.

Black-tailed Godwit. This species occurred mostly in January and February. Total bird–days for this species was 22 955, and the maximum winter peaks in individual counts were 1 285 in 1993–94 and 346 in 1994–95. A peak also occurred in March 94 (160 birds), probably associated with the passage of spring migrants. The autumn passage, from late July to October, was much longer than the spring passage, but the maximum number was considerably lower (63 birds). The 1994–95 numbers may be underestimated because, apart from a small group that remained in the salinas throughout the tidal cycle, this species moved into the rice–fields of the Lower Mondego river valley during the day, returning on the estuary mainly at dusk to roost.

Common Sandpiper. Although present for most of the year, except from late May to June, this species also occurred in the greatest numbers in winter. The peak for individual counts in both years were similar: 20 birds in early February 1994 and 18 birds in late January 1995. The first birds arrived in the estuary in late July, and the numbers increased until January. However, the numbers fluctuated a lot more in 1994–95 than in the previous year. A total of 2 908 bird–days was recorded.

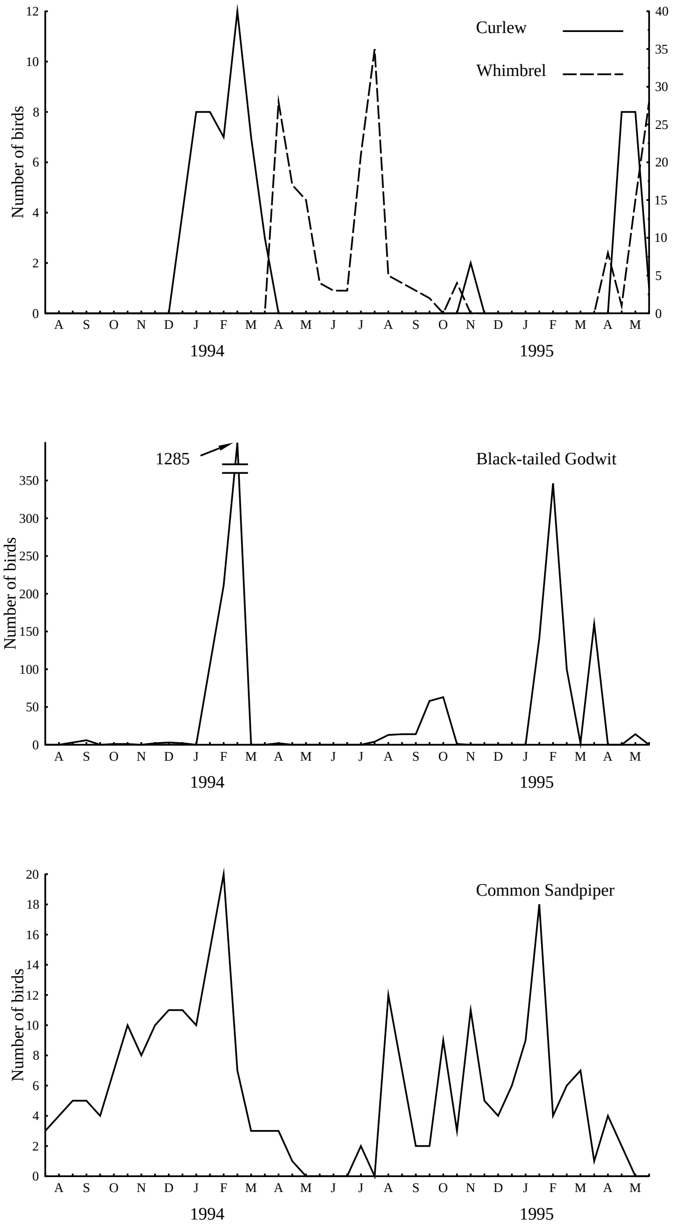

Figure 3.7: The number of the most abundant species that were recorded in each sampling occasion during the study periods: *Numenius* spp.: *N. arquata* (solid line) and *N. phaeopus* (broken line), Black–tailed Godwit and Common Sandpiper

Redshank. This species was present only during the migratory seasons, with a total of 9 056 bird–days. The maximum numbers in individual counts were recorded in July (77 birds) and September 1994 (68 birds), these numbers being very high compared with the autumn of the previous year. Spring migration numbers were similar in both years (late March, 34 birds in 1994, and 41 in 1995). However, recent information suggest that some individuals may remain on the estuary over the winter. Although breeding in this species has not yet been recorded in the Mondego, Redshanks breed in the more southern estuaries of the Tagus, Sado and in the eastern Algarve (Jardim, 1984; Rufino, 1989). There is, however, some recent evidence that one or two pairs may have nested in the Morraceira's salinas during the study period (pers. observ., J.P. Neves, pers. comm.).

Dunlin. This was the most abundant species in the estuary, with a total of 265 231 bird–days. A large wintering population was present from early November to late February, with more birds being counted in the first year (maximum of 1 228 birds, in January of 1994) than in the second (774, in January of 1995). However, the peak counts in each year were reached during the brief spring passages (2 112 in late April of 1994, and 1 211 in early May of 1995). In comparison, the autumn passage lasted for much longer – from late July to late September – with a small peak count in August (182) and a major one in September (698).

This apparently bimodal timing of the passage of migrating Dunlins may be due to different geographic populations migrating through the region at different times. Batty (1991, 1992), found that, in the Ria Formosa, the early migrants (late July) were adults belonging to the continental *schiinzi* race. These were then followed, from August to October, by the juveniles of this race and by both the adults and juveniles of the icelandic *schiinzi* race. The birds of the *alpina* race, which winter on the estuary, were present mainly in September and October. In view of these findings, it is possible that the July peak in the Mondego also corresponded to the passage of *schiinzi* birds, while the September peak was mostly formed by birds of the *alpina* and icelandic *schiinzi* races.

Little Stint. This small calidriid occurred mainly in the winter, with additional peaks in numbers occurring during the autumn and spring migration periods. Some 7 910 bird–days were recorded, and the highest peak in individual counts occurred in November (221 birds) and December of 1993 (109 birds). Since then, the maximum numbers have not exceeded 60. The spring migration occurred from late February to late April, although this pattern was not very clear in 1993-94. In contrast, the autumn passage, which apparently took place in August and September, was more evident during the first year of the study.

These annual differences in autumn passage numbers may be partly explained by difficulties in counting – particularly in the salinas – and to the possibility of confusion with similar species at a distance.

Ruff. A typical saltmarsh bird (Hayman *et al.*, 1991; Cramp & Simmons, 1983), the Ruff was observed during the migratory periods exclusively, in a total of 1 355 bird–days. Peak numbers in individual counts were in September of 1993 (21 birds) and late March of 1994 (17 birds).

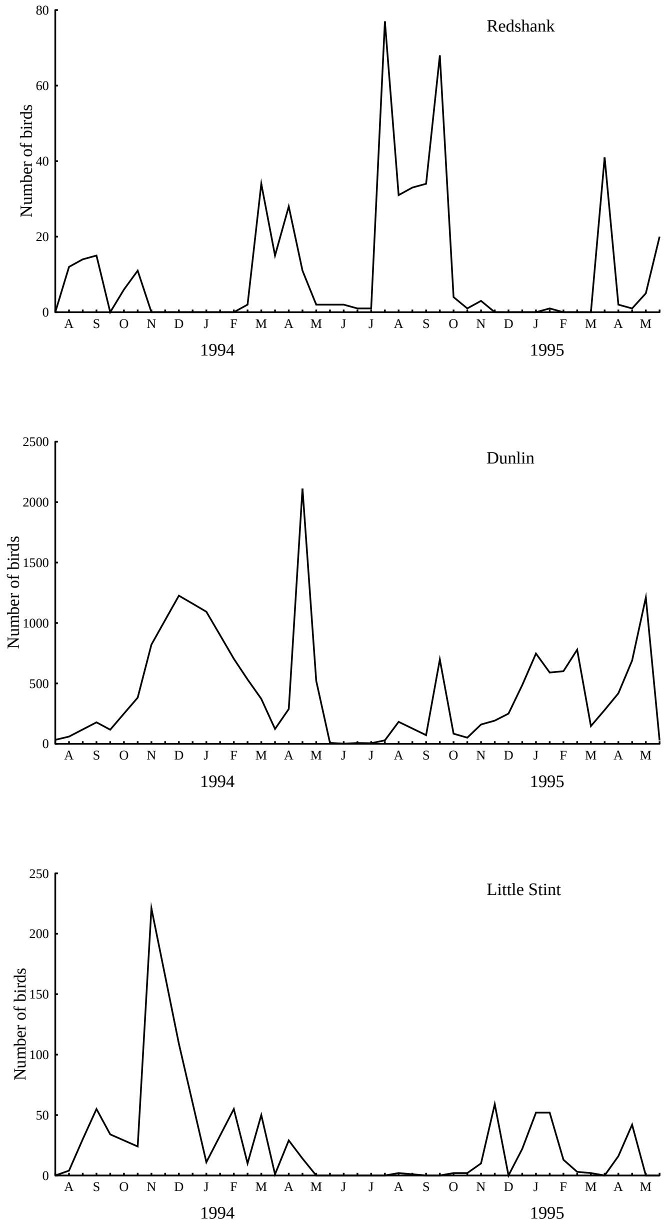

Figure 3.8: The number of the most abundant species that were recorded in each sampling occasion during the study periods: Redshank, Dunlin and Little Sandpiper

Avocet. With a total of 76 713 bird–days, the Avocet was the second most abundant species. It was found from late October to early March, with peak numbers in individual counts in mid–winter (815 birds in January 1994 and 687 in early February 1994). This is perhaps the only species that was counted very accurately, as all birds gathered in a single flock to feed at low–water on the south–eastern mudflats. As such, the changes in its abundance in the estuary can be confidently traced back to the late 1970's. In fact, from 1975 to 1978, Rufino (1979) found an average of 322 birds in the Mondego during January counts (with a maximum count of 700 in 1975). From 1988 onwards, regular January censuses were carried out, the following numbers of Avocets being counted: 270 in 1988/89 (unpublished data); 510 in 1989/90 (Rufino, 1989, 1990); 325 in 1991/92: (unpublished data); 815 in 1993-94 (Costa & Rufino, 1994, this study); 687 in 1994/95 (this study); and 968 in 1995/96 (unpublished data). The average of 596 birds counted from 1989/90 to 1995/96 represents, thus, an increase of 85% from the mid–seventies average.

Black-winged Stilt. This is the second species that breeds in the estuary. A total of 33 453 bird–days was recorded during the study period. The species is present from early March to August or September, the peak numbers in individual counts being reached in early April (148 in 1994, 203 in 1995), probably due to the presence of migrating birds. Most birds were found in the Morraceira Island, where the largest colony in Lower Mondego river valley occurs (Rufino & Neves, 1991, unpublished data). Based on counts of individual birds and assuming that two birds correspond to a breeding pair, the number of pairs in the island for the nesting seasons[5] of 1990 (Rufino & Neves, 1991), 1994, 1995 and 1996 was, respectively, 36, 49, 42 and 41. These data suggest that the numbers breeding have stabilised over the past few years.

Other species. Other species were present in small numbers and for very limited periods of time. Some used the area for brief periods, as a migratory stopover, in spring and/or autumn. For example, the *Turnstone* was recorded only in spring (April–May). The pattern was similar in both years, with maximum numbers of 18 (1994) and 16 birds (1995). Some 1294 bird–days were recorded.

The *Bar-tailed Godwit* occurred in both autumn (September–October) and spring (April–May), but the numbers present varied from year to year, either between seasons and in the same season. The peak number was reached in early May of 1994, with 42 birds, but the average number was 8 birds. Total bird–days reached 2 514.

The *Knot* was also present during the migratory seasons although, in 1993–94, some birds were seen in November and December. As with the previous species,

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[5] Rufino & Neves (1991), in their estimation of the national totals of breeding pairs for the Black-winged Stilt, used two counts at the beginning (April) and in the middle (early June) of the nesting season. As a more complete set of counts was available in this study, the number of breeding pairs was found by selecting the highest count made when the overall numbers stabilised. In this way it was avoided the introduction of biases due to the presence of migrating birds (in April) or the year's young birds (from July onwards), as the seasonal dynamics of the species may sligthly vary from year to year.

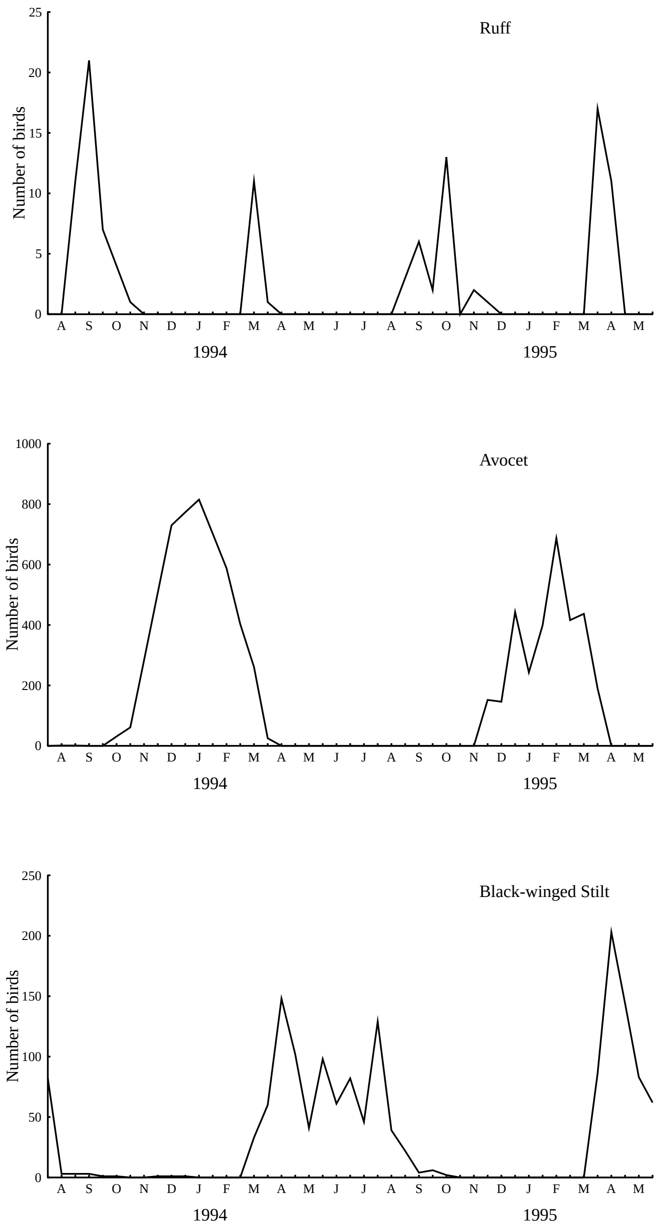

Figure 3.9: The number of the most abundant species that were recorded in each sampling occasion during the study periods: Ruff, Avocet and Black-winged Stilt

the numbers varied a lot. Two peaks in numbers occurred in late September of 1994 (42 birds) and in early May of 1995 (38 birds). Total bird–days for this species was 2 612.

The *Sanderling* was present from mid–winter (January or early February) to early spring (late March), with 1 479 bird–days. This species is mostly found on sandy beaches in winter (Smit & Piersma, 1989), but is frequently found in more inland waters and saline lakes during migration (Hayman *et al.*, 1991). So, the high numbers observed from mid–February to late March in the salinas probably reflects the passage of migrating birds from southern latitudes. The numbers found were similar in both years, although the peak counts were recorded in January of 1994 (22) and in late February 1995 (21 birds).

The *Curlew Sandpiper* was recorded continuously from the late summer to early autumn of 1994 (late July, August and early September). Only 35 birds were observed. This pattern is in agreement with what is known of the phenology of this species, which occurs in western Europe mostly during the autumn migration in scattered groups (Smit & Piersma, 1989; Hayman *et al.*, 1991), with only a small number wintering here (Encarnação, 1992; Perez-Hurtado & Hortas, 1991). The timing of migration found for this species was very similar to that observed by Rufino (1984) in the autumn of 1981 in the Tagus estuary. As with Little Stint, some confusion with the more abundant Dunlin may have occurred, which may have led to an underestimation of the numbers present.

Four species used the area intermittently, mainly in winter. They were mostly inland species that, for some reason, were driven to the estuary temporarily. The *Lapwing* and the *Snipe* are mainly ricefield users that occasionally occurred in the Morraceira's salinas in winter or early spring. No more than 9 birds from the first species and 15 from the second were recorded. Nevertheless, the Snipe may be more frequent in the supratidal habitats of the estuary than it appears, as the census method used is likely to have underestimated the numbers of this highly inconspicuous species.

The *Greenshank* can be found in both estuarine and inland habitats (Smit & Piersma, 1989; Hayman *et al.*, 1991). In the Mondego, it was only detected in the second year, from October to March, probably because it mainly used the ricefields and the adjacent south-eastern salinas of the Morraceira, which were visited only from August 1994 onwards. Total bird–days amounted to 912. The *Golden Plover*, on the other hand, is a typical inland species (Smit & Piersma, 1989; Hayman *et al.*, 1991), and its presence in early March of 1994 in the Morraceira's salinas (2 birds) must be considered accidental.

3.3.2 The Mondego's wader assemblage in the national context

Qualitative comparisons: permanence time of the species

The permanence time of the species found in the estuary of Mondego was compared with that of other Portuguese estuaries, for which data on the annual cycle of

Table 3.2: The permanence time of wader species in the Mondego and Tagus estuaries and in the Rias de Faro and Aveiro. Values are ranks, as follows: 1: 1–2 months; 2: 3–4 months; 3: 5–6 months; 4: 7–8 months; 5: 9–10 months; 6: 11–12 months. (?) means no data available. The comparison of ranks gives the qualitative ranking similarity between species according to the four sites in the sequence presented in the table. 1 means that the species concerned shares the ranking value with other estuaries; 0, means a ranking different from all other sites. Thus, 1101, for example, means that the ranking of the species concerned is the same in Aveiro, Mondego and Faro, but is different in Tagus. In case of double pairs (e.g. 1100), the 1 value is attributed to the Mondego and its co-sites. Values for the Mondego are in bold.

	R. Aveiro	**Mondego**	Tejo	R. Faro	Comparison of ranks
Kentish Plover	6	**6**	6	6	1**1**11
Ringed Plover	5	**5**	6	5	1**1**01
Lapwing	?	**2**	1	2	?**1**01
Snipe	?	**1**	2	1	?**1**01
Golden Plover	?	**1**	1	2	?**1**10
Grey Plover	6	**4**	6	4	0**1**01
Curlew	6	**3**	2	2	0**0**11
Whimbrel	4	**3**	1	4	1**0**01
Black-tailed Godwit	5	**5**	6	1	1**1**00
Bar–tailed Godwit	6	**3**	6	?	1**0**1?
Common Sandpiper	?	**5**	5	1	?**1**10
Redshank	6	**5**	6	6	1**0**11
Greenshank	?	**5**	6	2	?**0**00
Ruff	?	**3**	4	4	?**0**11
Knot	?	**3**	5	3	?**1**01
Dunlin	5	**6**	6	6	0**1**11
Curlew Sandpiper	?	**2**	4	2	?**1**01
Sanderling	?	**5**	4	2	?**0**00
Avocet	4	**3**	6	3	0**1**01
Black–winged Stilt	4	**6**	6	4	0**1**10

occurrence of the species were available: Ria de Aveiro (Luís, 1989), Ria de Faro (Encarnação, 1992) and Tagus estuary (Mendes *et al.*, 1996). The species were ranked according the number of months spent in each site (Table 3.2).

Some 60% of the species examined had a similar permanence time to the birds of, at least, one of the other estuaries, although no consistent latitudinal trend was evident. Among these species, and excluding those that were recorded in only some of the four sites, four species (Grey Plover, Black-tailed Godwit, Avocet and Black–winged Stilt) were present during the same number of months in the Mondego as they were on only one other estuary or 'ria' (36.4%). Two species (Ringed Plover and Dunlin) were present for the same period of time in two other estuaries or 'rias' apart from the Mondego (18.2%). Only one species (Kentish Plover) was present for the same period of time in all sites.

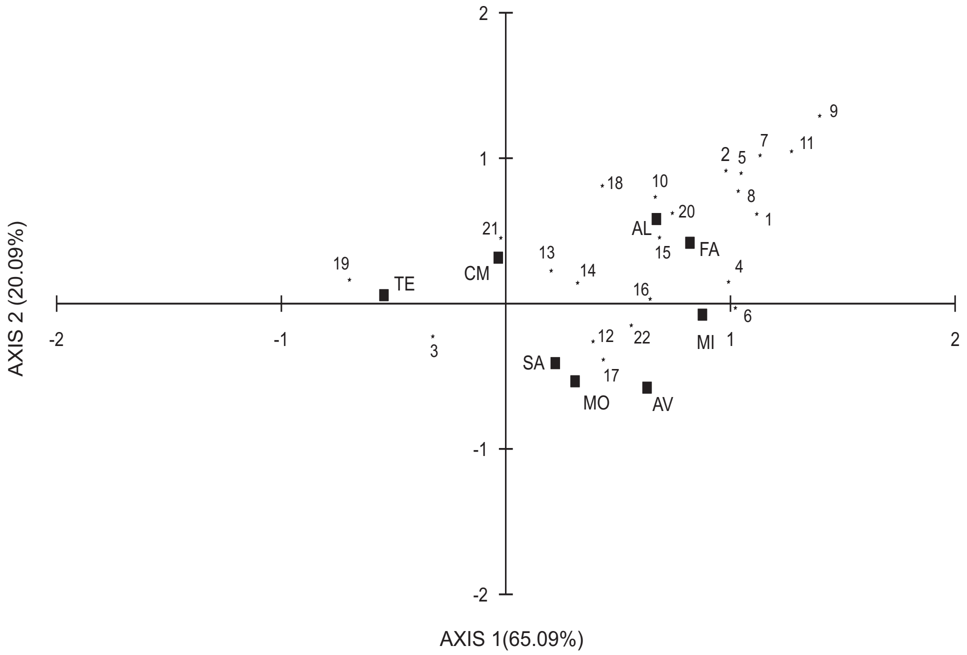

Figure 3.10: Analysis of the winter distribution of the waders amongst the most important portuguese estuaries, as determined by correspondence analysis to the matrix species×sites (estuaries). **TA** – Tagus estuary; **CM** – Castro Marim saltmarsh; **AL** – Ria de Alvor; **FA** – Ria Formosa; **SA** – Sado estuary; **MO** – Mondego estuary; **AV** – Ria de Aveiro; **MI** – Minho estuary. Numbers refer to the following species (the scientific names of the species not recorded in this study on the Mondego are given in brackets.): 1. Oystercatcher *Haematopus ostralegus*;. Black-winged Stilt; 3. Avocet; 4. Ringed Plover; 5. Kentish Plover; 6. Grey Plover; 7. Turnstone; 8. Dunlin; 9. Curlew Sandpiper; 10. Little Stint; 11. Knot; 12. Sanderling; 13. Redshank; 14. Spotted Redshank *Tringa erythropus*; 15. Greenshank; 16. Green Sandpiper *Tringa ochropus*; 17. Curlew; 18. Whimbrel; 19. Black-tailed Godwit; 20. Bar-tailed godwit; 21. Ruff; 22. Common Sandpiper.

Quantitative comparisons: the winter assemblage

The results from correspondence analysis showed the projection of both wintering wader species and the different estuaries and 'rias' considered along the Portuguese coast in the space of the first two axis of variability (Figure 3.10).

The Tagus estuary was clearly at the other extreme of axis I (which accounted for 65.1% of the total variability), compared to the other sites, being characterised by the presence of abundant Black-tailed Godwit. Along axis II (21% of the variability), the separation was especially marked between the Algarve zones (namely Ria de Alvor and Ria Formosa) and the estuaries of the Atlantic coast, excluding the Tagus. The species that characterised these southern Portuguese wetlands were Curlew Sandpiper, Turnstone, Knot, Black-winged Stilt, Kentish Plover, Dunlin, Oystercatcher Haematopus ostralegus, Little Stint, Greenshank and Bar-tailed Godwit. Conversely, the group of estuaries of the Atlantic coast, including the

Mondego, was not clearly distinguished by the presence of particular species, although the positions of Curlew, Sanderling and Common Sandpiper seemed to be more liked to these sites.

3.4 Discussion

The results of this study generally confirmed the idea that the importance of the Mondego estuary as an habitat for waders had been under-estimated. The average winter numbers[6] for the period 1994–96 (738) increased by 67.7% the figures given by Rufino (1979) for 1975–78 (440). This increase should be attributed more to a better coverage of the area than to a real increase in numbers. In fact, a complete count of the area, including all the habitats suitable for waders (although not the total area available for each habitat), was only achieved in 1994, the first year of the present study. The total number counted in the January census of 1994 was significantly higher than in previous years: 2 396 birds, most of them being Avocets (815) and Dunlins (1 093).

The numbers of the smaller species may have been overestimated, as the winter counts in 1994 were spread over three days and, in a small estuary such as the Mondego, this is likely to increase the probability of double counting. However, the high numbers that were recorded agreed well with the increased numbers recorded in the national winter census of 1994 relative to the previous year (Costa & Rufino, 1994). As Dunlin was one of the species that increased by the greatest amount nationally, the increased numbers recorded in the Mondego may be viewed with some confidence. Similar total numbers (1 877 and 2 079, respectively), were recorded in the Mondego in 1995 (this study) and 1996 (unpublished data), in spite of the adverse weather conditions of the 1996 count. Although these data cannot be compared with the national counts for these two years, as they are yet to be published, the inter–annual similarity in bird numbers in the Mondego estuary further suggest that the main feeding/roosting grounds have now been located and censused.

As expected, the winter wader assemblage of the Mondego is very similar to those of other estuaries along the western coast of Portugal, both qualitatively and quantitatively, but different from the Algarve sites. This is probably due to the geographic position of the Algarve coast in one of the crossroads of the European-African migratory routes – the Gibraltar Strait. Also, the Algarve is situated near the northern border of the wintering area for a number of species (e.g. Kentish Plover, Black–winged Stilt) (Smit & Piersma, 1989; Batty, 1992).

Regarding the numerical importance of the wintering populations of waders, the position occupied by the Mondego among the other Portuguese estuaries is modest. According to Farinha & Trindade (1994), the estuary of Mondego only accounted for less than 2% of the total number of wintering waders in the Por-

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[6] Calculated as the average of species' averages, in order to allow the comparison with the data from Rufino (1979)

Table 3.3: National and international importance of the Mondego estuary for selected wader species in winter. An estuary is considered to be of national importance (N) if it supports $>1\%$ of the total population of a species or a particular race wintering in a given country, and of international importance (I), if it supports $>1\%$ of the population of a species or a particular race wintering in a given flyway (the East Atlantic Flyway, in this case). Data for 1% tresholds taken from Farinha & Trindade (1994)

	1% of the national population		Average population of Mondego	Importance
	1986–1991	1986–1992	(1994–1996)	(1% threshold)
Black-winged Stilt	3	10	–	–
Avocet	185	186	727	N, I
Kentish Plover	25	60	141	N
Ringed Plover	29	36	129	N
Grey Plover	115	115	157	N
Knot	16	16	–	
Sanderling	2	4	18	N
Little Stint	3	15	26	N
Dunlin	502	571	861	N
Black-tailed Godwit	370	527	75	
Bar-tailed Godwit	41	41	13	
Curlew	21	33	8	
Redshank	60	60	2	
Common Sandpiper	3	3	17	N
Turnstone	4	4	–	–

tuguese wetlands for the period 1989–92. In fact, this proportion was still below 1% (0.5%), placing this estuary in the third position among the four smaller estuaries and rias of Portugal. However, the proportion of the national wintering populations of waders accounted for in the Mondego raised up to 1.5% in the period 1994-96, probably as a consequence of the better coverage achieved, placing the estuary in the second position within its group.

On the other hand, the data from the present study revealed for the period 1993–96 the presence of 8 species in nationally important numbers (that is, above 1% of the national population[7]), according to the values given by Farinha & Trindade (1994) (Table 3.3).

Also, most of these species are, in some way, protected by national (Red Data Book) or international (EU's Wild Birds Directive and Bern Convention) agreements (Table 3.4).

It is true that the 1% values refer to the national population estimates made over 1986–91 and 1986–92 periods, as no more recent estimates are available.

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[7] According to the Ramsar Convention, a given wetland may be considered 'internationally important to the waterbirds' if (a) it usually supports more than 20 000 birds, or (b) it supports a significant number of birds belonging to a group of species indicators of the value, productivity and diversity of the wetland, or (c) it usually supports at least 1% of the population of a species or sub–species of waterbird of a given biogeographic region (Farinha & Trindade, 1994). These criteria are also applied to one country's specific fauna.

Table 3.4: Wader species that regularly occur in the Mondego estuary, protected by national (portuguese Red Data Book, **RB**) or international (Birds Directive, **BD**; Bern Convention, **BC**) conventions

	BD	BC	RB
Black-winged Stilt	+	+	
Avocet	+	+	+
Kentish Plover		+	
Ringed Plover		+	
Sanderling		+	
Little Stint		+	
Dunlin		+	
Ruff	+		+

However, in most cases, the values obtained in this study lie well above these limits. Considering this, and also that the most recent published counts for Portugal (1993 and 1994) did not reveal any significant increase for the species concerned (Rufino & Costa, 1993; Costa & Rufino, 1994), the above conclusions are probably still valid. Moreover, this estuary acquires international importance for the Avocet, whose 1% threshold limit is between 670 and 700, according to the more recent estimates (Smit & Piersma, 1989; Farinha & Trindade, 1994). In fact, in the three last winters, the estuary retained this status in 1994 (815 birds) and 1996 (968 birds).

The winter counts have been adopted as the most accurate way of estimating the population sizes of wader species along the east atlantic flyway as, by then, the inter–site movements are minimal (Prater, 1981; Smit & Piersma, 1989). However, the phenologic cycles of most species also include migratory movements between the breeding and wintering areas, and many estuaries are also strongly (although briefly) used during these periods (Smit & Piersma, 1989).

The estuary of Mondego was shown to be important for a number of species during the spring and autumn migratory periods. Particularly significant was the number of Dunlins that used the area in spring, reaching values even higher than in winter. Also, the estuary supports two confirmed breeding species, the Kentish Plover and the Black–winged Stilt. At least in the second case, the area is the most important one for the breeding populations in the whole region of the Lower Mondego valley (Rufino & Neves, 1991, pers. observ.).

Overall, the Mondego estuary may be considered an important area for waders, reaching national and/or international standards in both a qualitative and a quantitative way.

Chapter 4

The use of the salinas by waders in the Mondego estuary

4.1 Introduction

Migrating waders (Aves: Charadrii) inhabiting or passing through an estuary face a set of problems in finding enough food for self–maintenance or for putting on fat to continue their migratory journey. In fact, since most wader species are unable to feed while swimming, they are confined to looking for food only during low–tide, when their main intertidal feeding sites are exposed for some 7.5 to 8.5 hours (Prater, 1981; Puttick, 1984). A series of biotic factors, such as high bird densities and fluctuations in prey populations, and abiotic factors, such as short day length, low temperatures and adverse weather conditions, acting in isolation or in combination, may further constrain the potential rate of food consumption (Evans & Dugan, 1984; Burger, 1984; Puttick, 1984).

As an adaptation to these limitations, waders have developed a series of alternative feeding behaviours, like foraging for longer periods and/or foraging at night, increasing their intake rate by foraging faster, and preying upon more rewarding size classes of their usual prey, or upon more rewarding prey species (see Puttick, 1984, for a review). Nevertheless, even employing these alternative feeding behaviours may, sometimes, be insufficient for them to fulfil their energetic demands (Davidson & Evans, 1986). Therefore, the use of supplementary man–made or man–modified supratidal habitats can be of great value when low–water feeding habitats are unavailable, or their use is restricted (Goss-Custard, 1969; Davidson & Evans, 1986).

The salinas are among the most important man–made habitats in many estuaries and shores of the southern European countries (Martínez–Vilalta, 1985; Britton & Johnson, 1987; Casini *et al.*, 1995; Rufino & Neves, 1992; Perez-Hurtado & Hortas, 1993b; Aymerich, 1995). Such habitats might provide alternative or supplementary feeding areas to the usual intertidal sites used by many wader species, although this depends on their management regime (Perez-Hurtado & Hortas, 1991;

Velásquez, 1992; Perez-Hurtado & Hortas, 1993a).

The term *alternative areas* is used here to refer to those non–tidal areas that are used over the low–water period, when the intertidal flats are exposed. The salinas provide then the most important alternative areas and are frequently used by birds at low tide, even though the mudflats of the estuary are fully available. The term *supplementary areas*, on the other hand, refers to areas used at high–tide by the birds that at low water fed on the intertidal flats of the estuary. When the flats are covered by the tide, birds can continue feeding by using areas above the water mark, such as the salinas. Thus, such areas provide feeding that is supplementary to the feeding done in the intertidal areas at low–tide.

In Portugal, the salt industry goes back to, at least, the 10th century and, until the 17th century, there were salinas in almost all estuaries and coastal lagoons (Rufino & Neves, 1992; Neves & Rufino, 1995). In the 16th century, salt production reached 300 000 tons/year (Oliveira, 1992) and, by the beginning of the 20th century, some 190 000 tons were still produced annually. However, from 1936 onwards, salt production entered a deep crisis. Despite the government's intervention in the early 1950's and a full–scale mechanisation in the most important areas in the late 1960's, more and more salinas were abandoned or, more recently, converted into more profitable activities, such as rice production or fish–farms (Rufino & Neves, 1992; Neves & Rufino, 1995).

These changes in land–use, namely the conversion of old salinas into fish–farms, are likely to have a direct and important effect on the wader populations which use the salinas, and need therefore to be carefully investigated.

In the Mondego estuary, there still remains an extensive area of unchanged traditional salinas. Nevertheless, since 1984, the area converted into fish–farms has increased considerably, as has the number of abandoned salinas. In spite of this, and of the recognised importance of the salinas to estuarine wader populations, there is still only a limited amount of information available on the use made of the salinas by waders in the Mondego estuary (Múrias & Ferrand de Almeida, 1991; Múrias *et al.*, 1991).

The objective of this Chapter is to assess how, and by which species, the Mondego salinas are used as feeding areas in order to provide information on the importance of these habitats for the estuarine populations of waders. Specifically, the following aspects were examined:

(a) the general use made of the Mondego's salinas by estuarine waders, by describing the birds' temporal, spatial and tidal distribution patterns;

(b) the effects of human management (or the lack of it) on the used made of the salinas by waders;

(c) the possible effect of the salinas destruction on the wader populations, through an evaluation of the present use made of this habitat by the species concerned, in terms of feeding space and of feeding time.

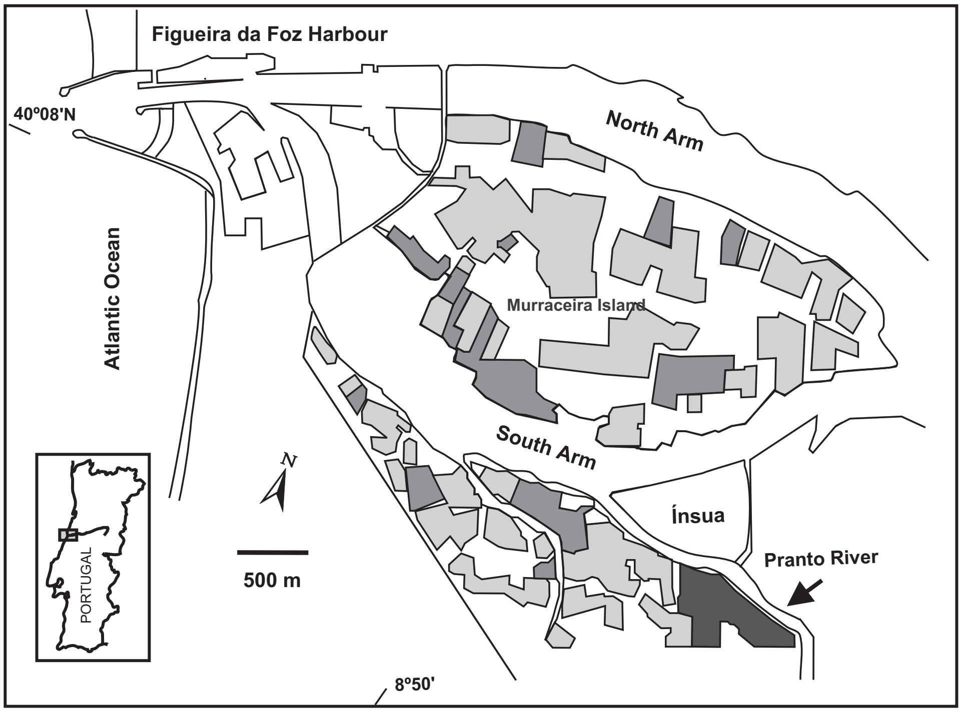

Figure 4.1: Location of the main area of artisanal salinas (light grey), fish–farms (dark grey) and industrial salinas (black) in the estuary of Mondego. The abandoned salinas in the north arm and in the Ínsua are not represented

4.2 Study Area and Methods

4.2.1 Study Area

The Mondego's salinas: location and characterisation

Location and history The Mondego's salinas extend for approximately 5 km from the mouth of the estuary and comprise an area of 305.1 ha. The salinas are distributed between the two arms of the estuary, the Vila Verde Group, in the north arm, and the Lavos Group, in the south arm, and in the Morraceira Island (Figure 4.1, Table 4.1).

Historically, the first mention of the salinas in the Mondego estuary dates back to the beginning of the 12th century. This first group of salinas was located near Figueira da Foz, in the right margin of the north arm (Vila Verde Group). Those of the Lavos Group are slightly more recent, while the Morraceira's salinas – the most important group today – were not built until the 16th century (Proença, 1988).

Table 4.1: The contribution of each group of salinas of all types in the estuary of Mondego to the total area of salinas. The number of salinas and cómoros (basic productive units of a salina – see text) in each group is also given (adapted from Lopes, 1955).

Salinas Group	N. of salinas	N. of cómoros	Area (in ha)	%
Vila Verde (North Arm)	5	15	12.8	4.2
Lavos (South Arm)	32	78	101.3	33.2
Morraceira (M. Island)	42	136	191.0	62.6
Total	79	229	305.1	100.0

The salinas: structural and functional characterisation The basic unit of a salina in the Mondego is the 'cómoro' or 'talhão'. Usually, a salina is divided into several 'cómoros', which are exploited by single saltworkers ('marnoteiros') (Figure 4.2).

There are, on average, 3.2 'cómoros' per salina. Each of them is formed by three sets of pans of decreasing depth linked through a network of walls and ditches, allowing the water to circulate until the sodium salt precipitates. The water coming from the estuary first enters a *reservoir* ('viveiro'[1]) through a *sluice* ('greiro') and is stored there. The water depth in the viveiro can reach more than 1 m, although it usually does not exceed 50 to 80 cm (Lopes, 1955; Rufino & Neves, 1992; Neves & Rufino, 1995). The 'viveiro' opens into the *storage pans* ('vasa'), the first true, as well as the largest, compartments of the salina, whose function is to allow the precipitation of unwanted salts, mainly Fe salts and $CaCl_3$ (Rufino & Neves, 1992; Neves & Rufino, 1995). There are usually one or two storage pans per salina, and their depth usually varies between 20 and 30 cm, although in the Mondego's salinas it reaches only 6 cm (Lopes, 1955). The water flows next to the smaller *preparation pans* ('comedorias'), where most of the evaporation takes place. Usually they are grouped in several rows and have a depth descending from 5 cm to 4 cm. They open to the last group of pans, the *crystallisation pans* ('praias'), where the NaCl finally precipitates. These are the smallest ponds, and are also grouped in several rows, the depth ranging from 3 cm to 2 cm (Figure 4.2).

In contrast to the *mechanised salinas*[2], the Mondego salinas belong to a group generally called *artisanal* or *traditional salinas* where all the work is done by hand (Rufino & Neves, 1992; Neves & Rufino, 1995). The main structural difference between artisanal and mechanised salinas lies in their *total area*, this being less than 10 to 15 ha for the artisanal salinas but sometimes more than 100 ha for the mechanised ones. In the Mondego, the average area of a salina is 4.0±0.4 (SE) ha. Furthermore, the *proportion of the area occupied by the chrystallisors* is usually less than 10% of the total area in the artisanal salinas against 20% to 40% in mechanised salinas. Finally, there are *larger but fewer chrystallisors* in mechanised salinas, as compared to the artisanal ones (Neves & Rufino, 1995).

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[1] The nomenclature is that used in the Mondego's salinas

[2] In their turn, these are further subdivided into *semi–industrial salinas*, if part of the work is mechanized, and *industrial salinas*, where the labour is fully mechanized

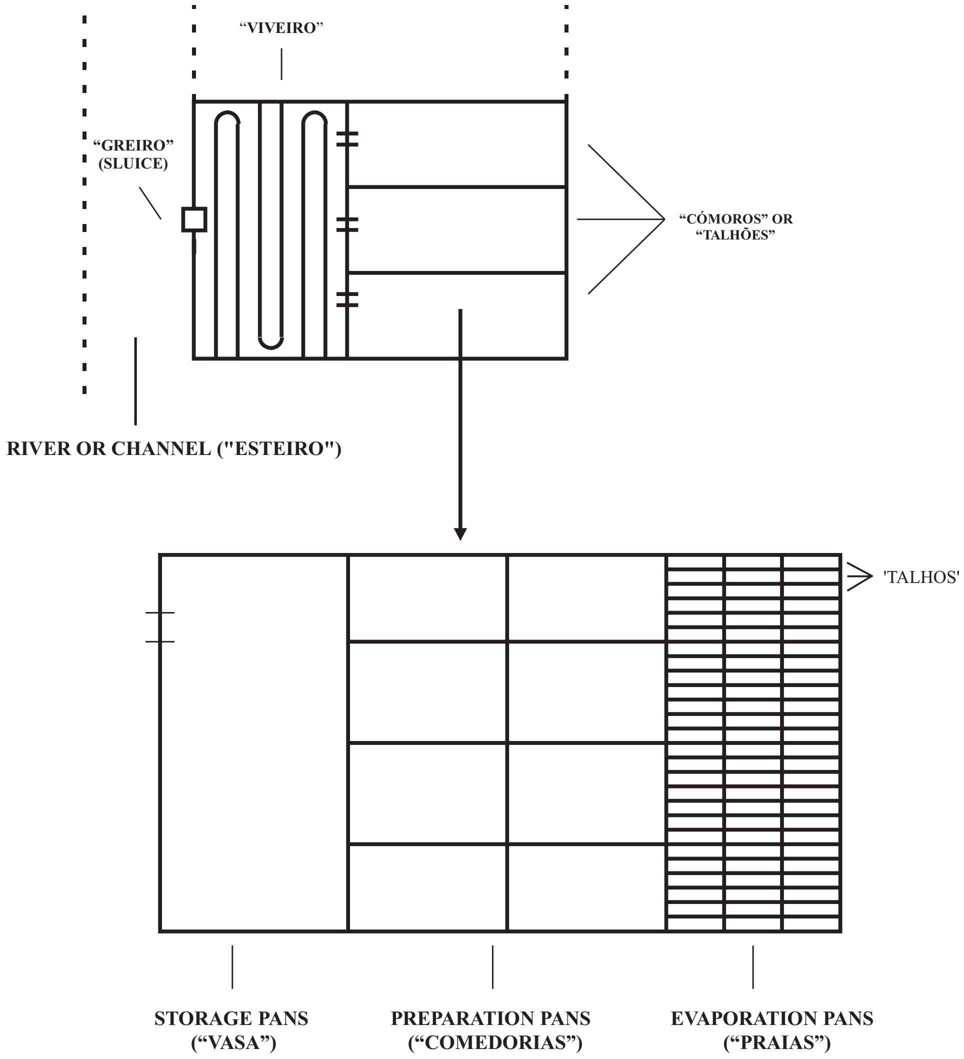

Figure 4.2: Plan of a salina, showing its general structure, with the 'viveiro' and the 'cómoros', the minimum area required for salt production (above), and a detailed description of a 'cómoro', with its storage, preparation and crystallization surfaces (below). The 'talhos' are the actual sites were the NaCl precipitates.

Although they are structurally very similar to the Aveiro salinas – which are also artisanal – the Mondego salinas show an important distinctive characteristic: because they were built inland, they lie above the tidal level (mean high water, spring tides). Therefore, they must be excavated in order to allow the water to enter. In contrast, in Aveiro, they are built at low levels and are separated from the sea by the construction of a protective wall (Proença, 1988; Gonçalves & Sobreiro, 1992).

This feature brings significant implications to the changes that occur in the salinas after they have been deserted. In Aveiro, the lack of maintenance of a deserted salina leads, in time, to the destruction of the protective walls and to the establishment of a confined tidal flat (A. Luís, pers. comm.). In the Mondego, on the other hand, where the salinas are not subjected to direct wear from the river, the walls are usually maintained intact and, because the sluices are, in general, kept closed, the ancient salinas become stagnant saltmarshes.

Annual cycle of the salt exploration The 'salt extraction season' usually lasts from May to September/October and is preceded by a preparatory phase of approximately one and a half month, when the ponds are cleaned and repaired in turn (Lopes, 1955; Rufino & Neves, 1992; Neves & Rufino, 1995). In artisanal salinas, the salt is collected four to five times per season, while in the larger mechanised salinas, a maximum of only two crops is possible (Neves & Rufino, 1995). In general, the harvested salt is not immediately stored; instead, it is accumulated in the salina, being then progressively removed and stored through the winter (Neves & Rufino, 1995). The Mondego salinas, where the crop is stored at the end of the season (Lopes, 1955), are the exception to this. During the salt season, the sluices are opened only twice: the first occasion is at the onset of the preparatory phase, to drain the accumulated rainwater from the winter, and after that, to fill the ponds in order to begin the evaporation process. At the end of the season, the salinas are flooded again and abandoned until the next spring.

Recent evolution of the salt industry in the Mondego

Situation of the Mondego salinas in 1993–95 During the study period, some 40% of the total area of the salinas was still occupied by active salinas, while 31% was occupied by inactive ones (34%, if the drained salinas are included in this number). About 2% were active industrial salinas, which differed from the artisanal salinas in their structure and methods of exploitation, while the fish–farms only accounted for 18% of the total area (Table 4.2). However, the importance of the fish–farms is slightly increased (to about 23%) if the number, instead of the area of the cómoros, is considered.

The distribution of each category of land–use (active salinas; inactive salinas; fish–farms) differed between the three groups of salinas in the Mondego, irrespective of the total area of each group (Table 4.2).

Table 4.2: Present–day land use (1993–95) of the former area of salinas in the three main groups of the Mondego estuary. Values are percentage of each category for each area. Original data as in Table 4.1.

	Morraceira[a]		Lavos		Vila Verde[a]		All estuary	
	Area	Cómoros	Area	Cómoros	Area	Cómoros	Area	Cómoros
Salinas								
Active	37.7	39.0	54.8	50.0	0.0	0.0	41.8	40.2
Inactive	43.5	39.0	4.4	14.1	46.0	46.7	31.0	31.0
Levelled	1.3	0.7	0.0	0.0	54.0	53.3	2.8	3.9
Salinas, rock–salt	0.0	0.0	20.6	5.1	0.0	0.0	6.8	1.7
fish–farms	28.2	21.3	20.2	30.8	0.0	0.0	17.6	23.1

[a] Excluding two salinas for whom was not possible obtain any data

Thus, in the Morraceira Island (the largest group), 43.5% of the area was occupied by inactive salinas, against 37.7% for active salinas and only 28.2% for fish–farms. In spite of these differences in area, the percentage of 'cómoros' in each category of salinas was nevertheless similar (39.0%).

In the Lavos (south arm) group, most of the salinas were still active (54.8% of the area and 50.0% of the 'cómoros'), but the fish–farms were important, especially in terms of the percentage of 'cómoros' occupied (30.8%). This was the only group which included an area of industrial (rock–salt) salinas. It had the typical characteristics of this type of salina (see above): a large area (20.6% of the total) but a small number of 'cómoros' (only 5%).

The remaining group of the estuary, the Vila Verde (north arm) group, was completely inactive during the study period. More than 50% of the salinas' area, and of the number of 'cómoros', had been drained, and the remaining areas had been deserted during the past decade.

The active salinas The crisis that affected the Portuguese salt industry also affected the Mondego salinas. According to Lopes (1955), 229 'cómoros' were exploited in 1954, and produced a total of 33 000 tons of salt. Almost forty years later, these figures have decreased to 6 200 tons produced by only 68 'cómoros' (Direcção–Geral dos Portos, pers. comm.), and the trend is for production still to decrease (Figure 4.3).

With the exception of the rock–salt industrial salinas referred above, mechanised extraction processes have not been introduced into the Mondego's salinas. The increase in industrial salinas usually imply the aggregation and complete reshaping of the ancient salinas. It is possible that the fragmented distribution of the salinas and the relatively small area of the ponds made the introduction of mechanised extraction processes rather costly and unprofitable.

The fish–farms As in other areas, the establishment of fish–farms in the estuary took place during the last decade, and tends still to grow. According to Alves &

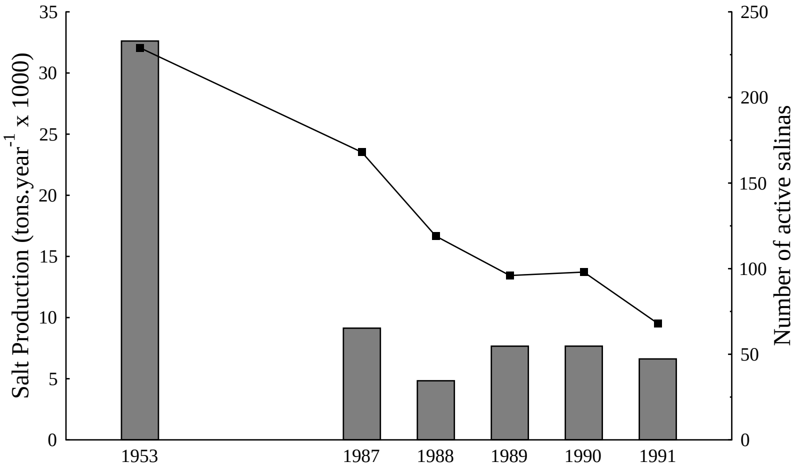

Figure 4.3: Change in the salt production (bars) in the estuary of Mondego from 1953 to 1991, as compared to the number of active salinas (line). The total number of 'cómoros' are shown. Data extracted from Lopes (1955) and DGP.

Marques (1995), some 22 fish–farms were already working in the estuary of Mondego in 1995, or had been authorised to do so. Of these, 11 were extensive mixed farms and 11 were intensive or semi–intensive monocultures, of which only 6 were active. From 1984 to the present, a total of 56.4 ha of salinas[3] were converted into fish–farms (Table 4.3, which represents approximately 20% of the total area available. The most significant increases occurred from 1984 to 1990, when 14 fish–farms, occupying 40.4 ha, where established.

In spite of the appreciable proportion of the total area that has been converted into fish–farms since 1984, the rate of transformation decreased from 1984–1990 to 1990–1994, whether measured in terms of the number of salinas (a reduction of 13%, from 2.3 salinas/year to 2.0 salinas/year, respectively), or in the area destroyed (a considerable reduction of 40%, from 6.7 ha/year to 4.0 ha/year).

This apparent disinvestment in fish–farming in the Mondego estuary from 1990 onwards is in opposition to a worldwide increase of this activity (Alves & Marques, 1995). The main reason for this seems to be the low profitability of the fish–farm industry at the local scale. Factors such as a heavy beaurocracy (it takes more than 3 years to build a production unit), the lack of a full–scale professional framework, the low diversification of the species used (mostly marine migratory fish) and the problems linked to the quality of water are limiting the rapid development

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[3] Excluding the area formerly occupied by 'viveiros' (see Chapter 2)

Table 4.3: Increase in fish–farm establishments in the Mondego estuary from 1984 until the present. The area of salinas converted into fish–farms in each period of time considered (1984–1990 and 1990–1994), as well as their respective percentage contributions to the total area of salinas, are also presented. All types of fish–farms that were established in old salinas were included.

	Before 1984	1984–1990	1990–1994
Number of fish–farms	–	14	8
Converted area/period (ha)	0.0	40.4	16.0
Converted area (%)	0.0	14.1	5.6
Cumulative converted area	0.0	40.4	56.4
Cumulative converted area (%)	0.0	14.1	19.7

of the fish–farming in the estuary (Alves & Marques, 1995). However, albeit at a slower rate, the conversion of salinas to fish–farms is still going on, and it would be expected to increase exponentially if the limiting factors were removed, or considerably reduced.

In the Mondego estuary, the fish–farms are rather narrow and deep, and have very steep margins, with each main pond being subdivided into several rectangular–shaped pans. This avoids their use by waders as feeding grounds, even when they have been emptied, which, in normal conditions, occurs approximately once a year. However, they are frequently used as high–water refuges.

The abandoned salinas Even if not immediately converted into fish–farms, many salinas were nevertheless deserted. At present, 93.7 ha, representing 43.3% of the total area of the unchanged salinas (i.e those not drained or transformed into fish–farms or ricefields) in the estuary have been deserted. In fact, the total area deserted in Morraceira Island decreased by some 28% over the last ten years (1984–1994), when compared to the previous 30–year period (25.5 ha and 35.3 ha, respectively). However, the *rate of desertion* increased about three–fold, from 1.2 ha deserted/period before 1984 to an average of 5.2 ha in 1984–1994 (Table 4.4).

At this rate of desertion (assuming that it will remain constant, as seems to have been the case in the past decade), the Morraceira's salinas will be completely abandoned within 14 years. Many of these deserted salinas will be converted, in the short to medium–term, into fish–farms.

In contrast to what happened in the Tagus and Sado estuaries, very few salinas in the Mondego were transformed into anything other than fish–farms. The exception was a group of the easternmost salinas on the right bank of the north arm (Vila Verde Group), described by Lopes in 1955, and still existing in 1984. These salinas were partly converted into agricultural fields and partly drained as part of the work carried out to regulate the banks in the outermost section of the Mondego estuary, which happened in the late 1980's.

After desertion, a salina goes through a series of successional steps which, if they are not interrupted by fish–farm conversion or reversed by reactivation of the salina, usually end up in a brackish lagoon or an area of salt–water marshland. The

Table 4.4: The abandonment of the salinas in the Morraceira Island up until 1994. The values for the first period may be slightly underestimated, since the salinas converted into fish–farms which were usually abandoned earlier, were not considered.

	Before 1984	1984–1990	1990–1994
Number of 'cómoros'	26	13	17
Area (ha)	35.3	29.9	21.1
Cumulative area	35.3	65.2	86.3
% of total salinas area[a]	12.4	12.2	9.2
Deserted ha/period[b]	1.2	5.0	5.3

[a] Calculated in each period by excluding the total area of fish–farms.

[b] The number of years is: pre–1984 – 30 years; 1984–90 – 6 years, 1990–96 – 6 years

fate of each individual salina, as well as the rate of transformation, is highly variable, depending on a combination of factors, such as the degree of communication between the different ponds and with the main water body, the soil permeability in the salina, and the colonisation pattern of allophytic plants (Neves & Rufino, 1995).

Therefore, excluding the long–term abandoned salinas – i.e. those abandoned for 20 years or more – it is possible to find a variety of habitats even among salinas deserted at the same time. Not all inactive salinas are lost, though. Sometimes, a salina is not used for 1 to 3 years, being left in 'poisio', according to the local terminology, and then is reactivated. Nevertheless, this is not the most common case and complete desertion is a widespreading phenomenon.

4.2.2 Methods

Field data collection

The counts Counts of waders were carried out in the salinas of the Morraceira Island, approximately 2 hours around dead low–water and dead high–water on spring tides. They were made monthly, from July 1993 to January 1994, and every two weeks, from February 1994 to May 1995. From July 1993 to June 1994, counts were also carried out monthly on neap tides in the week following the spring tide counts. But since no tidal or seasonal differences were found in the number of birds present in the salinas on neap and spring tides (two–way ANOVA, $p>0.05$), all counts were grouped for the analysis.

A car was used to move between salinas, following a fixed route, and all the birds seen were counted with 10×50 binoculars and a 30–90×50 telescope. Each count lasted about 2 hours, although it depended on the number of birds present. With few exceptions, the study area was surveyed in a single day or, at most, on two consecutive days, whether it was spring or neap tides.

The counts were carried out in such a way that the route chosen covered 24 of the 33 (72.7%) salinas still extant in the Morraceira island, as well as all the fish–farms and most of the 'viveiros'. In July 1994, six new salinas of more difficult access were included, allowing a better coverage (90.6%) of the total area: see Chapter 3 for a detailed analysis of the accuracy of the counts. Only three salinas, located in the remote northern part of the island, were not surveyed because previous observations showed that waders seldom used them.

In the south arm salinas and fish–farms, regular low–water counts were carried out from October 1993 onwards when the mudflat birds were being counted (see Chapter 3). Occasional high–water censuses were also performed in this area, particularly in winter and spring. No attempt was made to conduct counts in the few salinas of north arm, since no waders were ever found there.

Apart from identifying and counting the birds in each salina, the *number of birds feeding and not feeding* (resting, preening, standing) and the *type of salina* used (active or inactive) were recorded.

Assessing the macrohabitat variables: water depth and salinity Several authors (Britton & Johnson, 1987; Velásquez, 1992; Verkuil *et al.*, 1993) suggested that the water depth and salinity were two of the more important factors that affected the use of hyper–saline habitats by birds, either directly, by limiting prey accessibility (water depth) or indirectly, through changes in the composition and density of the prey populations (salinity). In this study, the two variables were assessed indirectly in 1993–94 (but not in the following year) by recording:

1. the *water depth at which each feeding bird was observed in the salina*, adapted from the scoring system used by Perez-Hurtado & Hortas (1991), based on the mean 'leg level' of the bird:

   W – internal wall of the salina
   0 – wet ground but no water
   1 – shallow water <5 cm (water <tibio–tarsal joint)
   2 – water level 5–11 cm (water = tibio–tarsal joint)
   3 – water level 11–20 cm (water >tibio–tarsal joint)

   For some species, the original data matrix contained many empty cells (i.e. with 0 values), which prevented the use of the χ^2test. Because of this, data were grouped and only three 'water levels' were used: (1) wall, (2) dry soil and shallow depth (0 and 1 levels) and higher depths (2 and 3 levels). This procedure allowed to increase the sample size.

2. the precise *location* within the salina where the focal bird stood, corresponding to an increasing level of salinity (Britton & Johnson, 1987):

   S – storage pans (lowest salinity)
   P – preparation pans (medium salinity)
   C – crystallisation pans (highest salinity)

Table 4.5: Identification, total area and state of management of the salinas considered in this study. See Appendix for more details concerning the specific areas and the number of cómoros in each management state.

N. of the salina	Name of the salina	Area (ha)	Actual state of management
1	Cavalo Branco	3.400	Partially active
2	Filipas	3.175	Active
3	Cruz	1.445	Active
4	Corredor do Padre	7.735	Partially active, includes an area of extensive fish–farms
5	C. do Cabaço	6.810	Inactive
6	Praias Grandes	8.000	Partially active
7	Morro	4.180	Partially active
8	Alhos 1	3.694	Partially active
9	Alhos 2	1.798	Partially active
10	Tapada Norte	6.230	Partially active
11	Tapada Sul	6.230	Active
12	Cerco	5.805	Partially active
13	Pestanas	5.145	Partially active
14	Corredor Novo	4.190	Inactive
15	Doutores 1	1.725	Active
16	Doutores 2	5.380	Inactive
17	Pontão	9.545	Inactive
18	Amante 1	1.585	Partially active
19	Amante 2	2.995	Inactive
20	Feras	7.800	Partially active
21	Donato 1	5.665	Partially active
22	Donato 2	4.835	Partially active
23	Ínsua	3.400	Inactive
24	Uxaria	9.630	Partially active
25	Venturas de Baixo	3.965	Partially active
26	Casa da Pedra	4.005	Partially active

Characterisation of the salinas The salinas in the Mondego estuary were mapped from aerial photographs, backed up by visits to establish their actual condition. Following the work of Lopes (1955), the number of 'cómoros' which were active, inactive, transformed into fish–farm, or simply drained, were assigned, by visual inspection, for each salina. The local salt–workers and fishermen were further contacted for approximate information on the time of desertion of the inactive 'cómoros' (or entire salinas), since no official records were available. The area of each salina was obtained from Lopes (1955) for the Morraceira's and the north arm's salinas. For the remaining salinas, a detailed map of the area was used to estimate areas. The name, location, total area and present state of management of the studied salinas in the Morraceira Island are presented in Table 4.5. A more detailed characterisation of the whole complex of salinas of the Mondego estuary is given in Appendix.

In order to calculate the average value of water level and vegetation cover in each salina and each season, the level of water and the vegetation cover were assessed in each visit to each salina, according to a previously established ranking scale. The two scales were as follows: For the *level of water*

0 – dry soil

1 – Low level (<5 cm)

2 – Medium level (5–10 cm)

3 – High level (10–15 cm)

4 – Flooded salina (>15 cm)

The water level was estimated visually from the banks of the salina, by comparison with the height of the main internal walls. This method is, of course, subject to some imprecision, particularly because the depth of the salina is not homogeneous, varying according to the type of pan. Thus, we selected, for each case, the value that seemed to occur in 50% or more of the pans. For the *vegetation cover*:

0 – No vegetation

1 – Sparse and low vegetation (<50 cm)

2 – Dense but low vegetation, covering less than 50% of the area

3 – Dense but low vegetation, covering more than 50% of the area

4 – Dense and tall vegetation (>50%), covering almost all the pans

As in the previous case, the estimates were made visually from the banks of the salina.

The distance of each salina to the nearest low–water areas was also measured, as the minimum distance (in meters) from the center of the salina to the edge of the intertidal mudflats. Additionally, human pressure was assessed by counting the number and thus density of people in all the salinas in each season. In practice, this variable was only relevant in autumn, when the salt was being collected from the pans and stored.

4.2.3 General data analysis

Counts were grouped by season and year. Although this necessarily involved the loss of some information, it was required in order to simplify the analysis, given the large number of species involved. Given the small number of counts available from each month, it also allowed within season variations to be explored. The number of counts varied from one (in summer 1993) to eight (in winter 1994–95).

The seasons considered were: (1) *summer, June and July*; (2) *autumn, August to October*; (3) *winter, November to February*; and (4) *spring, March to May*. Although this seasonal division was chosen for the Mondego (Múrias & Ferrand de Almeida, 1991), it is similar to that used by other authors (Batty, 1992). It roughly corresponds to the phenologic cycle of most wader species along the

Atlantic coasts of southern Europe (Luís, 1989; Batty, 1992; Encarnação, 1992). While this seasonal categorisation provided the main basis for the analysis, the data were sometimes combined in different ways (e.g., winter *versus* other seasons), or only part of the data set was used, according to the particular analysis being performed.

Following Komolgorov–Smirnov tests to test for normality, data were statistically analysed using ANOVA tests and simple regression tests. Categorical data were analysed with χ^2tests. When required, data were transformed; $arcsin(n)$ for percentage data and $log(n)$ or $log(n+1)$, for other data.

In order to study the multi–specific and simultaneous pattern of selection of the salinas by waders, a correspondence analysis was performed on the matrix *density of birds*$\times$*salinas* (see Chapter 3 for a full description of the method), excluding the salinas where no birds were ever recorded, as well as those with only one species and/or low densities ($<$0.1 birds.10 ha^{-1}). The analysis was carried out for the autumn and spring[4] of 1994–95, when most of the salinas of the Morraceira Island were surveyed. Only low–water counts were employed, to minimise the chances of including the non–feeding birds which moved to the salinas at high–water. The numbers of birds recorded in each salina at each count were averaged per season and converted into numbers.10 ha^{-1}of salinas, due to the generally small average densities ($<$0.1 birds.salina^{-1}) observed in individual salinas.

To investigate what could possibly determine the distribution of the feeding birds among the salinas, a preliminary analysis was made, relating a set of potentially important environmental factors (area of the salina, water depth, amount of vegetation, disturbance – density of people – only tested in autumn, and the nearest distance to the mudflats) to the density of the birds in the salinas, excluding, for each species, those salinas where no birds were recorded.

This was achieved through an indirect gradient analysis (Ludwig & Reynolds, 1988). The salinas coordinates obtained through the COA for the two first factorial axes (PC I and PC II) were correlated with the respective values for the environmental factors considered (Spearman's *r*). A significant correlation (p$<$0.05) indicates that the salinas (SUs) are positioned along the principal component axis (PC I and/or PC II) based on the overall variations in their species abundances, and these variations are significantly related to an underlying gradient of the factor(s) under study. Because the species coordinates are presented in the same scale than the SUs, they can be readily positioned along the gradient found for a given environmental factor.

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[4] The winter was excluded as, by then, most salinas were flooded, and the distribution of the birds among them could simply reflect the accessibility of the feeding areas, rather than any structural or human–induced factors

4.3 Results

4.3.1 Temporal and spatial patterns of use of the salinas by waders

Global use of the habitat

Morraceira Considering the study period as a whole, and thus excluding any seasonal or yearly variation, the 11 species commonly present in the Morraceira's salinas can be assembled into four groups (Table 4.6):

1. Species that were rarely found in the salinas (0% to 0.1% of the total numbers): Avocet

2. Species that used the salinas mostly at high–water (26% to 58%), but seldom at low–water: Grey Plover and Whimbrel,

3. Species that mostly used the salinas at high–water, but still occurred there in high numbers (12% to 20%) at low–water: the small species, like Dunlin, Kentish Plover and Ringed Plover.

4. Species that were present in roughly the same proportions at both high and low–water, usually with more than 50% of their total numbers being present: the true 'salinas species', Black-tailed Godwit, Black–winged Stilt, Common Sandpiper, Little Stint and Redshank.

Across all species combined, approximately 30%, on average, of the birds counted in the estuary used the Morraceira's salinas over low–water, while about 58% used them at high–water. Although, these figures illustrate the significance of the Morraceira's salinas as a habitat for waders in the Mondego, they do not indicate how the birds used it. In particular, the high proportions of birds in most species that were present at high–water suggest that the salinas could have been used mainly as a resting area when the mudflats are inaccessible, as is often the case with other supratidal habitats.

This possibility was explored by comparing the proportions of birds recorded in the salinas at high and low tides that were feeding when counted. Depending on the species, most (42%–70%) of the birds of the species that used the salinas extensively at both high and low–water were feeding (Table 4.7). Grey Plover and Whimbrel fed only irregularly in the salinas (less than 10% of the individuals present), and the Avocet seldom fed there, particularly at low–water (less than 5%).

Thus, with the exception of these 3 species, the results clearly indicated that (1) in all cases but the Kentish Plover, the proportion of birds feeding at both low–water and high–water was very high (more than 50%) and (2) no significant differences were found between high and low–water in the proportion that were feeding. It seems that, in the Mondego, the salinas were used mainly as feeding areas.

Table 4.6: Percentage (±SE) of the total number of birds counted in the whole study area at low–water (N) that used the Morraceira's salinas at low and high–water. The groups are described in the text. Values represent the average number of birds present per count in the estuary over the whole study period (maximum n=38 counts). Only the species with 5 birds per season and per year, or more, were considered.

	N±SE	n	low–water %±SE	high–water %±SE	Time of occurrence
Group 1					
Avocet	388.6±53.4	18	<0.1	0.1±0.0	Winter
Group 2					
Grey Plover	86.6±10.8	28	1.3±1.0	14.3±7.9	All year, excl. summer
Whimbrel	7.9±3.4	7	6.9±3.0	25.5±11.1	Late spring to autumn
Group 3					
Dunlin	451.6±74.4	37	20.0±3.3	60.0±1.8	All year
Kentish Plover	91.5±10.6	38	18.8±6.3	59.4±6.3	All year
Ringed Plover	66.7±11.3	34	12.1±1.7	57.7±3.6	Winter, autumn and spring
Group 4					
Bl.–tailed Godwit	143.4±85.7	16	30.8±20.9	52.2±14.3	Mainly winter
Black–winged Stilt	62.4±9.9	22	79.7±5.5	87.4±20.6	Spring, summer
Little Stint	25.3±11.1	21	52.9±8.3	77.4±11.7	Mainly winter
Redshank	12.4±3.7	26	53.3±1.3	69.0±17.5	As above
C. Sandpiper	5.5±0.9	29	65.5±11.1	64.0±16.3	All year, excl. summer

Table 4.7: Percentage (±SE) of total low–water birds of some selected species that fed in the salinas at both high and low–water. Original data as in Table 4.6.

	low–water	high–water
	% ±SE	% ±SE
Dunlin	70.1 ±10.1	71.2±3.8
Black-winged Stilt	68.9±3.7	70.0±1.1
Redshank	65.0 ±18.9	67.4±6.2
Ringed Plover	61.5±3.9	50.7 ±11.2
Little Stint	59.3±2.9	59.3±2.9
Kentish Plover	48.3±0.1	42.4±2.1
Whimbrel	5.0±3.5	3.6±3.6
Grey Plover	3.8±3.6	8.4±5.8
Avocet	0.5±0.5	3.7±3.7

Other sites Approximately 42% of the waders present in the estuary, were never found in the Morraceira's salinas. These birds were distributed among other feeding or roosting sites around the estuary and on the island itself (Table 4.8).

The most important of these sites were fish–farms, namely those of the left bank of the south arm. They were used only as roosting sites, in contrast to what has been found in other estuaries (Perez-Hurtado & Hortas, 1993a). The south arm salinas were also important sites, although they seem to have been used mostly in the migratory seasons or in winter when the Morraceira's salinas were particularly flooded, and thus unavailable to many species. Saltmarshes were only used on neap tides, when these sites were uncovered even at high–water. The ricefields were used for short and variable periods mainly in spring, when they were flooded at the beginning of the rice cropping cycle.

Seasonal differences

Low–water The proportions of the total number of birds of the more abundant species that were recorded in all habitats at low–water, whether feeding or not, that were present in the salinas were not significantly influenced by season and/or year effects (Table 4.9, left column). The only exception was Kentish Plover, which was present in the salinas in higher numbers in spring and summer, when the birds were attending their eggs and chicks, than in winter or autumn ($F_{3,28}$ =13.84, $p<0.00001$, Tukey test).

The proportion of the total number of birds recorded in the salinas at low–water that were feeding did not vary significantly between seasons in any of the species considered (Table 4.9, right column). Only the proportions of feeding Little Stints differed between the two years, with more birds feeding in 1993–94 than in the following year ($F_{1,14}$=5.84, $p<0.046$, Tukey test). A significant, albeit weak, interaction between season and year in the proportion of feeding birds was only found in the Redshank ($F_{1,9}$=5.29, $p<0.046$). This might have been due to the small proportion of birds that fed in the autumn of 1993, as compared to the spring of that year and to the autumn and spring of 1994. However, a multiple comparison test (Tukey test) did not detect any significant groups that might confirm this hypothesis.

High–water The proportion of all birds, whether feeding or not, that were in the salinas during high–water remained constant for most species between seasons and between years (Table 4.10, left column). The only exception was the Black–winged Stilt ($F_{2,15}$=6.07, $p<0.01$), for which the proportion of birds using the salinas at high–tide was higher in autumn than in summer and in spring (Tukey test). This is an unexpected result, as the numbers present in autumn (mainly in August and early September) were clearly less than during the spring migration and the early breeding seasons (see Chapter 3). In fact, it was observed that, when arriving on the estuary in late March/early April prior to establishing their breeding territories or resuming their migration, the birds usually aggregated in the large 'viveiros' to

Table 4.8: Sites other than the Morraceira's salinas that were used by waders at high–tide. Most of them were also used to some extent for feeding at low–water, particularly in winter and spring.

Site	When used and for what	How much	By which species
South arm salinas	Winter, all tides; feeding and roosting; spring/autumn, all tides, mostly roosting. In winter, used as alternative sites to the Morraceira's salinas	Large flocks (>200) gathered in a few salinas (winter); small flocks (<50) largely dispersed (spring/autumn)	Mainly Dunlin, but also *Charadrius* spp.
South arm fish–farms	Winter, all tides; roosting	Large flocks. A single, large fish–farm seem to be particulary used	Specially Grey Plover and Dunlin
South arm industrial salinas	In winter, when other sites become unavailable; feeding	Small to medium flocks	Dunlin and *Charadrius*; eventuallly other species
Morraceira's fish–farms	All seasons and tides, although somewhat irregularly; roosting	Large flocks. Rotative use of two or three ponds	Grey Plover, Dunlin, *Charadrius* spp, other species.
Morraceira's saltmarshes	Neap tides of all seasons; feeding and roosting	Large flocks in the southern area. A few birds in the western marshes	In S marshes, all species. In W marshes, Curlew, Whimbrel, *Limosa* spp.
Ricefields	Mainly in spring, although also in winter for some species; feeding, moslty	Small flocks.	Black–winged Stilt, Dunlin, *Limosa* spp. In winter, also Curlew, Lapwing and Snipe

Table 4.9: Two–way analysis of variance to test the effect of season (4 seasons – summer, autumn, winter and spring – but see note below) and year (1993/94 and 1994/95), on (1) the proportion of all birds in the estuary that used the salinas at low–water and (ii) the proportion of the birds recorded in the salinas at low–water that were feeding (arcsin transformed values). Number of counts as in Table 4.6 Significance levels as: ***p<0.001, **p<0.01, *p<0.05. The number of seasons was three for Ringed Plover (excluding summer), two for Black–winged Stilt (excluding winter and autumn) and two for Redshank (excluding summer and winter).

	All birds			Feeding		
	Season	Year	S ×Y	Season	Year	S ×Y
Kentish Plover	0.000***	0.102	0.578	0.457	0.987	0.443
Ringed Plover	0.353	0.935	0.603	0.495	0.161	0.237
Dunlin	0.793	0.477	0.304	0.259	0.208	0.776
Little Stint	0.952	0.630	0.148	0.280	0.030*	0.100
Redshank	0.738	0.618	0.637	0.287	0.836	0.047*
Black–winged Stilt	0.542	0.630	0.121	0.185	0.653	0.052

Table 4.10: Two–way analysis of variance to test the effect of season (4 seasons – summer, autumn, winter and spring – but see note below) and year (1993/94 and 1994/95), on (1) the proportion of all birds in the estuary that used the salinas at high–water and (ii) the proportion of the birds recorded in the salinas at high–water that were feeding (arcsin transformed values). Number of counts as in Table 4.6. Significance levels as: ***p<0.001, **p<0.01, *p<0.05. The number of seasons was three for Ringed Plover (excluding summer), two for Black–winged Stilt (excluding winter and autumn) and two for Redshank (excluding summer and winter) at both tidal states and three for Dunlin (excluding summer) at low–water.

	All birds			Feeding		
	Season	Year	S ×Y	Season	Year	S ×Y
Kentish Plover	0.072	0.362	0.459	0.296	0.810	0.423
Ringed Plover	0.521	0.882	0.150	0.369	0.398	0.038*
Dunlin	0.556	0.707	0.878	0.863	0.579	0.017*
Little Stint	0.857	0.844	0.216	0.068	0.818	0.136
Redshank	0.201	0.956	0.631	0.159	0.362	0.092
Black–winged Stilt	0.012*	0.381	0.080	0.000***	0.591	0.532

feed and roost, rather than in the salinas, which were occupied only later in the spring.

Seasonal changes in the proportions of the birds in the salinas that were feeding were only detected in Black-winged Stilt ($F_{2,14}$=14.98, $p<0.0003$) (Table 4.10, right column). A higher proportion feeding was found in the salinas in spring, compared to the other two seasons (Tukey test). Annual variations in the proportion feeding occurred in some seasons in both Ringed Plover ($F_{2,24}$=3.77, $p<0.037$) and Dunlin ($F_{3,25}$=4.13, $p<0.016$). While it was not possible to distinguish were the seasonal differences occurred in Ringed Plover (Tukey test), in the case of Dunlin the significant interaction between the two variables was entirely due to an increase in the proportion feeding in the autumn of 1994, as compared to the previous year.

With these few exceptions, the results suggest that, across all species, the proportion of the birds in the salinas that were feeding in any one season was largely independent of yearly differences in the population numbers.

Spatial use of the habitat

Species–specific patterns of occurrence in the salinas Waders may distribute themselves among salinas according to three basic patterns of dispersion: (1) random, (2) even or (3) clumped. Because these patterns can be described by well–known statistical frequency distributions that are characterised by specific mean–variance relationships (Poisson, negative binomial and positive binomial, respectively), they are amenable to certain statistical tests (Ludwig & Reynolds, 1988). In this study the simple variance–to–mean ratio was used to test the null hypothesis of a random distribution for all species in each specific season (or, in other words, to test the goodness–of–fit with a Poisson series) (Elliot, 1977; Ludwig & Reynolds, 1988). This index, also called Index of Dispersion (ID), is given by:

$$ID = s^2/x \tag{4.1}$$

where s^2 = sample variance, and x = sample mean.

If the sample is in agreement with a Poisson series, ID should be equal to 1.0. Significant departures from unity can be tested by a chi–square analysis (Elliot, 1977; Ludwig & Reynolds, 1988). For small samples ($n < 30$, where n = number of sample units), the χ^2distribution can be used, with $(n-1)$ degrees of freedom. The significance of the analysis (and the corresponding distribution type) was assessed visually by a graph relating the degrees of freedom (x–axis) to the χ^2values (y–axis) (see Elliot, 1977; Ludwig & Reynolds, 1988, for a detailed description of the method). All species occurring in autumn, winter and spring in all the salinas that were searched in the Morraceira Island ($n = 26$), in 1994–95 were used in the analysis.

The results (Table 4.11) suggested that, in winter and autumn, most species aggregated in certain salinas, while in spring they spread more evenly between salinas. It should be noted, however, that the apparent uniformity in the distributions of most species in spring may simply have arisen from the small numbers present (usually less than 0.1 birds.10 ha^{-1}).

Despite these general trends, some consistent patterns emerged. The Ringed Plover seemed to be distributed at random irrespective of the season. Conversely, the Dunlin always aggregated in a few salinas. In those species that were present mainly in autumn and winter, and whose distribution in spring could have been affected by their low densities, the Common Sandpiper also showed a random dispersion while, at the other extreme, the Little Stint and the Black–tailed Godwit always congregated in a limited number of salinas. The change in the distribution pattern in Kentish Plover and Black–winged Stilt from autumn and winter to spring may have been associated with the spreading of the birds through their breeding territories from April onwards. But with these exceptions, the results as a whole were very consistent and suggested that most species were not distributed at random among the available salinas within the Morraceira complex.

Table 4.11: Distribution of wader species among the Morraceira's salinas in autumn, winter and spring of 1994–95. The Index of Dispersion (I.D.) is given along with the χ^2value to assess its goodness–of–fit to a Poisson series (random distribution), as well as actual dispersion type of each species for a given season: C – clumped distribution; R – random distribution; Rg – regular distribution.

	Autumn			Winter			Spring		
	I.D.	χ^2	Distr.	I.D.	χ^2	Distr.	I.D.	χ^2	Distr.
Kentish Plover	2.31	57.73	C	6.94	173.43	C	0.46	11.58	Rg
Ringed Plover	1.39	34.81	R	1.01	25.29	R	0.67	62.75	R
Dunlin	208.32	5208.12	C	6.54	163.58	C	2.51	62.75	C
Little Stint	1.89	47.29	C	2.07	51.81	C	0.14	3.42	Rg
Curlew Sandpiper	2.75	68.77	C	–	–	–	–	–	–
Knot	24.71	617.65	C	–	–	–	0.15	3.65	Rg
Sanderling	–	–	–	3.62	90.49	C	0.26	6.60	Rg
Common Sandpiper	0.53	13.13	R	1.01	25.15	R	0.09	2.36	Rg
Whimbrel	1.79	44.67	C	–	–	–	0.08	1.91	Rg
Greenshank	1.31	32.86	C	4.93	123.32	C	–	–	–
Redshank	6.87	171.73	C	–	–	–	1.24	31.07	R
Ruff	3.22	80.55	C	–	–	–	0.81	20.15	Rg
Black–tailed Godwit	15.06	376.48	C	50.74	1268.50	C	0.36	9.03	Rg
Bar–tailed Godwit	3.53	88.20	C	–	–	–	0.08	1.91	Rg
Black-winged Stilt	8.70	217.51	C	–	–	–	0.47	11.66	Rg

Multi–specific analysis The distribution of the wader species throughout the salinas was found to be different in autumn and spring (Figure 4.4). In autumn, salinas 14 and 24, which were characterised by the three more 'terrestrial' species (Redshank, Ruff and Black–tailed Godwit), appeared separated from all the other salinas along the first axis of variability. On the other hand, Black–winged Stilt and Dunlin, and their associated salinas (18, in the first case, and 12, 23 and 7, in the second), were not included in either of the two main groups.

Along the second axis of variability, the Black–winged Stilt and Dunlin groups appeared separated from the rest. In spring, Black–winged Stilt and its associated salinas (5, 3 and 24) were separated from an indistinct group formed by the other salinas and species along axis I. In contrast, a clear gradient was found along axis II, with Kentish Plover on the positive side and Redshank, on the negative one. Nevertheless, two distinct sub–groups were apparent: one, formed by Kentish Plover, Little Stint and Common Sadpiper, with salinas 1, 6, 7, 9, 10, 14 and 18, and the other characterised by Ringed Plover, Dunlin and Redshank, with salinas 4, 8, 11, 12 and 22.

The gradient analysis did not reveal any significant relationships between the selected factors and the salinas location along both axes of variability in autumn (Table 4.12). In spring, only the distance to the nearest mudflat was negatively correlated with principal components in axis I (Table 4.12) that is, those species lying in the negative side of the axis (all but Black–winged Stilt) fed mostly in the salinas away from the mudflats, somewhat contrary to expectation. This relation was, however, barely significant ($p<0.05$) (Table 4.12).

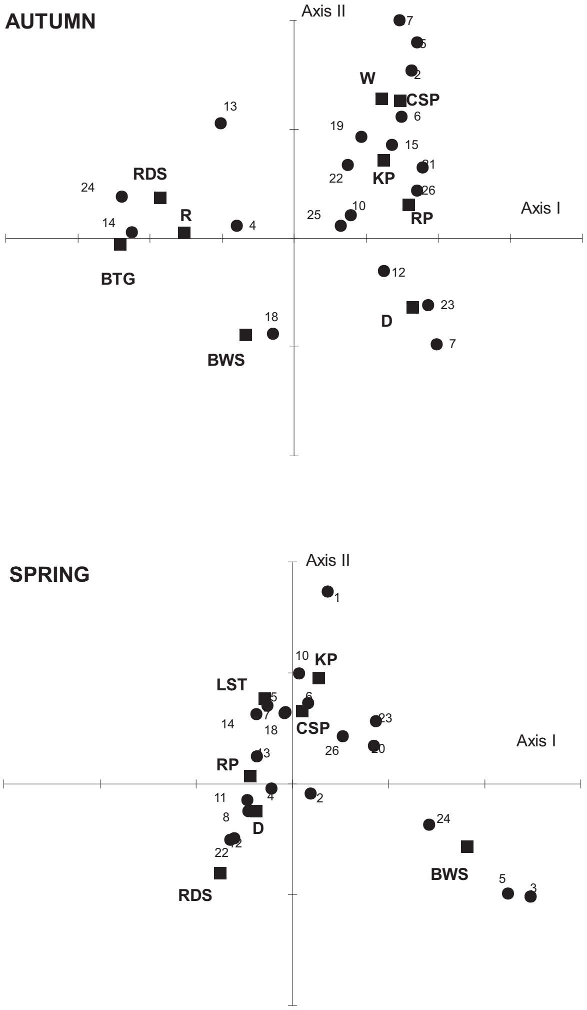

Figure 4.4: The first two axis of a correspondence analysis for autumn (above) and spring (below), of the matrix representing the occurrence of waders (birds.10 ha^{-1}) in the salinas (excluding, for each season, those salinas were no birds where recorded). These axis explain 59.2% of the total data variability in autumn (36.9% in the first axis and 22.3% in the second) and 64.9% in spring (41.4% and 23.5%, respectively). Numbers refer to the identification of the salinas, as in Table 4.5.Species abbreviations as follows: KP – Kentish Plover; RP – Ringed Plover; D – Dunlin; LST – Little Stint; CSP – Common Sandpiper; RDS – Redshank; BTG – Black–tailed Godwit; BWS – Black–winged Stilt

Table 4.12: Values of the Spearman correlation coefficients of environemental factors against salinas coordinates in the two first axis of the correspondence analysis (PC I and PC II) for autumn (n=20) and spring (n=21). * $p<0.05$.

	Autumn		Spring	
	PC I	PC II	PC I	PC II
Area	-0.37	+0.39	+0.11	-0.10
Water depth	-0.14	-0.30	-0.25	-0.37
Distance	-0.19	+0.13	-0.43*	+0.30
Density of people	-0.12	-0.20	-	-
Vegetation	+0.001	+0.28	-0.14	+0.31

4.3.2 Use of the salinas by feeding waders in relation to human management

This section investigates how the birds' use of some salinas' micro–habitats was affected by the management regime or the lack of it.

Active and inactive salinas differ in several important aspects (water level, density of vegetation, intensity of disturbance by man). However, active salinas are abandoned in late autumn, after the 'salt season' (section 4.2.1) and remain so until the next spring. Therefore, for the duration of the winter, the direct effects of management in the active salinas (particularly the level of disturbance and the control of the water level in the pans) are annulled, and both types of salinas apparently present similar characteristics.

Taking into account the seasonal differences mentioned above, the data were split, for each type of salinas (active and inactive), into two groups: winter period (November to February), and migratory periods[5] (March to May and August to October). The summer months were excluded, as by then the salinas were only used by the two breeding species, the Kentish Plover and the Black–winged Stilt (see Chapter 3). All analyses were performed with low–water data in order to avoid any possible influence of birds that used the salinas only to rest at high–water.

Active salinas vs. abandoned/inactive salinas: global patterns of use

Active salinas were more attractive to most of the species than inactive ones as, in all the cases, active salinas were used by more than 50% of the birds present (Table 4.13). Only the Black–winged Stilt showed no particular preference for any type of salina. All species that occurred in both seasons, with the exception of the Little Stint, used the active salinas significantly more in winter than in spring or autumn.

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[5] The two migratory seasons were grouped because of the small sample sizes for some months both in spring and in autumn, particularly in the inactive salinas. It is true that differences in some salinas's characteristics (e.g. water level, disturbance) can occur between the two seasons. However, between–season differences of spring and autumn are likely to be small when compared to the differences that both seasons present regarding the winter situation.

Table 4.13: Mean use (%) of the active salinas by selected species of waders in the spring to autumn and winter periods. Data for the two years were averaged by season (Sample sizes per season as follows: spring to autumn, n=14 (1993–94) and n=11 (1994–95); winter, n=8 and n=5, respectively. These values represent the maximum number of counts; individual species may have lower values.). Letter **a** denottes significant differences ($p<0.05$) in the proportion of waders (birds.10 ha^{-1}, arcsin–transformed) in active and abandoned salinas, for a given species and season (one–way ANOVA, data combined for the two years). Letter **b** shows where significant differences were found between seasons in a two–way ANOVA (testing for seasonal and annual differences), based on the proportion of birds in active salinas. See text for more details.

	March–October		November–February	
	Birds.10 ha^{-1}(SE)	% in active salinas	Birds.10 ha^{-1}(SE)	% in active salinas
Kentish Plover	1.4±0.1	58.6ab	2.6±0.8	95.4ab
Ringed Plover	1.4±0.2	59.6a	2.2±0.8	80.9a
Dunlin	8.0±4.9	58.1ab	20.7±7.9	89.7ab
Little Stint	4.8±1.2	62.6a	1.5[a]	100.0a
Redshank	0.6±0.5	62.6a	–	–
Black-winged Stilt	2.2±0.3	54.1	–	–

[a] Data for the first study year (1993–94), only.

Seasonal differences in microhabitat use in active and inactive salinas: pond type and water depth

A salina is a stratified habitat (see section 4.2.1), and each pan is characterised mainly by its water level and salinity which may provide different biological conditions and hence different microhabitats for waders. Different species are likely to use the pans in different ways according to their leg morphology (Perez-Hurtado & Hortas, 1991) and feeding ecology. The human management, or the lack of it, may force the waders to change the way in which they use the microhabitat because it changes the structural and physico–chemical conditions of a salina.

To examine this possibility, the response of waders to two aspects of the microhabitat – the water level and the type of pan (as an indication of the salinity) – were investigated in active and inactive salinas in both winter and during the migratory seasons of spring and autumn. Four of the more abundant species with different leg morphology were used for this analysis: the two visual–hunting plovers (Kentish and Ringed Plover) and the facultative sight–hunting Dunlin (short–legged species), and the facultative sight–hunting Black–winged Stilt (long–legged species). Only data from 1993–94 were used because no detailed records on the microhabitat use were taken in 1994–95. Data are depicted in Figures 4.5 and 4.6.

Kentish Plover. In the active salinas, Kentish Plover tended to feed more on the storage and preparation pans than on the chrystallisors in both seasons, although the difference was more pronounced in winter (χ^2=85.37, df=2, $p<0.001$) than during autumn and spring (χ^2=10.78, df=2, $p<0.0045$). In contrast, no differences in microhabitat use were observed in the inactive salinas at any time of the year (winter: χ^2=5.20, df=2, $p<0.07$; spring/autumn: χ^2=3.85, df=2, $p<0.15$).

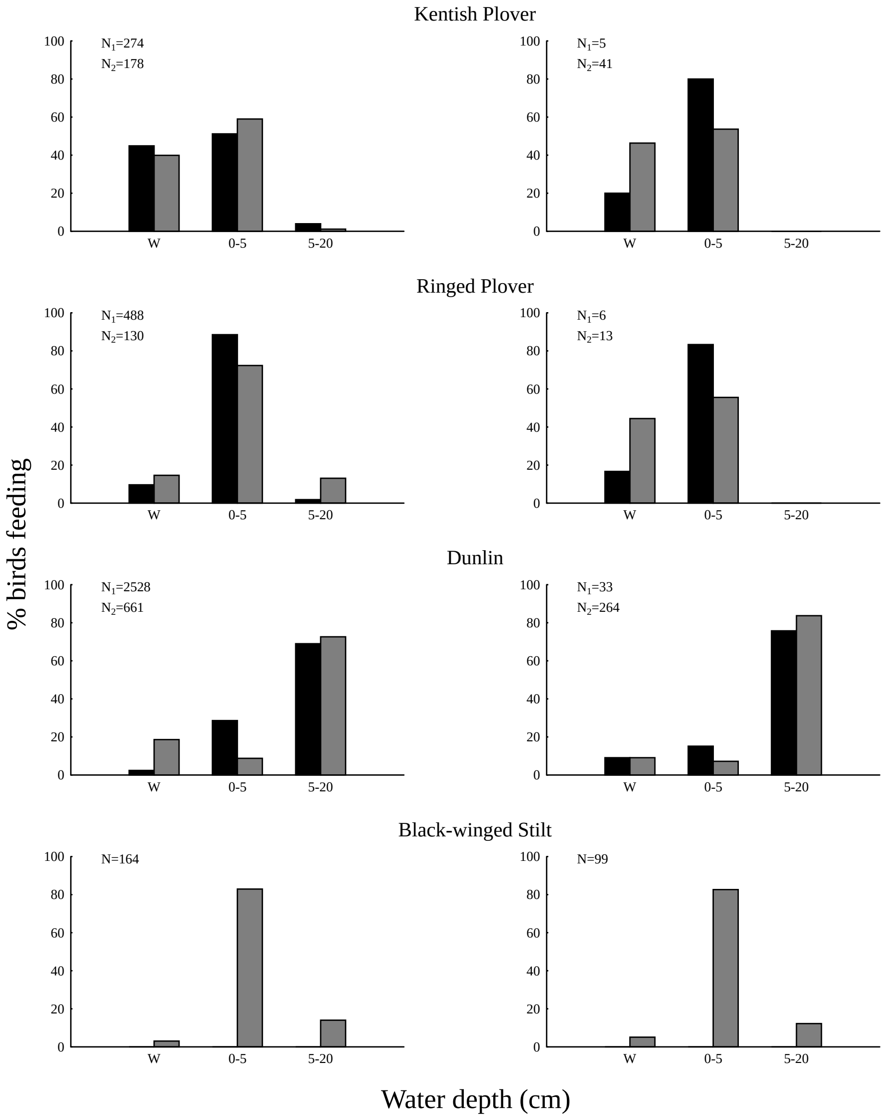

Figure 4.5: Water–depth selection by some wader species in active (left column) and inactive (right column) salinas, during the winter (N_1, black bars) and spring/autumn (N_2, grey bars) in the season 1993–94. W = internal wall of a salina.

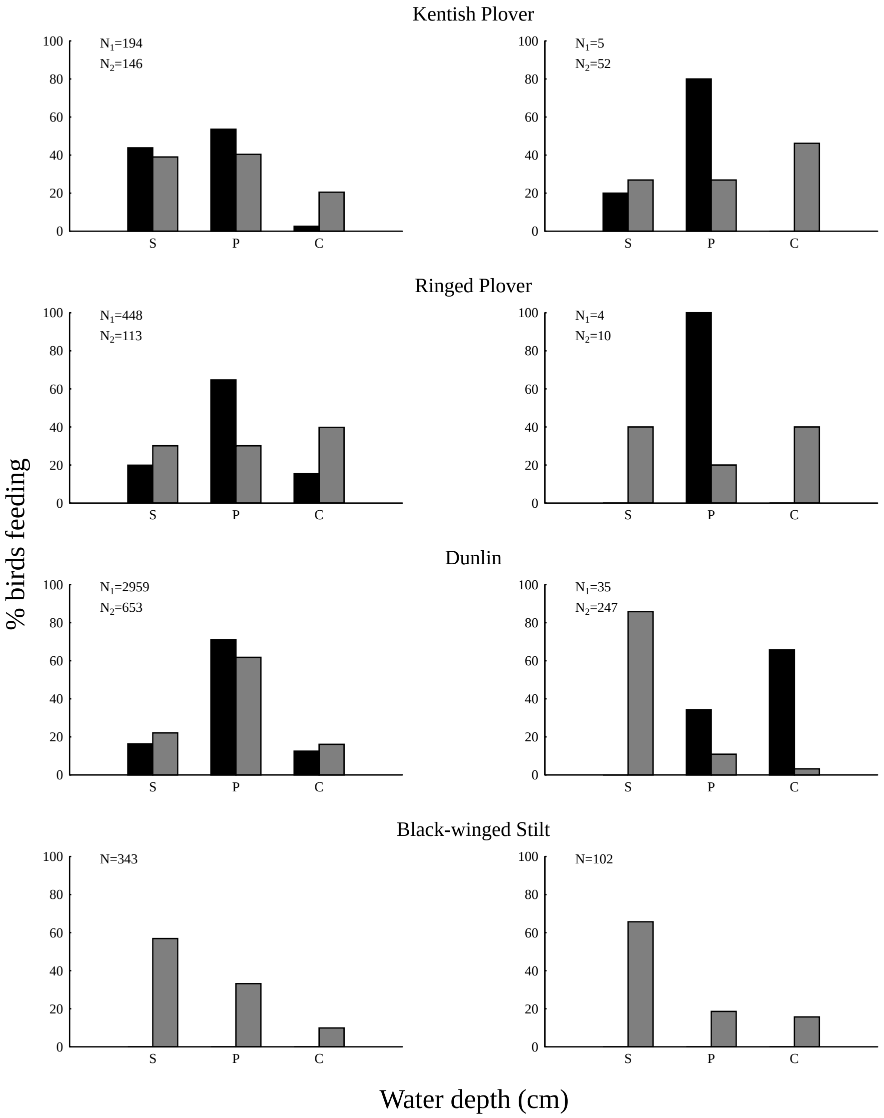

Figure 4.6: Pond type selection by some wader species in active (left column) and inactive (right column) salinas during the winter (N_1 and black bars) and spring/autumn (N_2 and grey bars) in the season 1993–94. S = storage pans; P = preparation pans; C = chrystallization pans

In both active and inactive salinas, Kentish Plover fed mainly in shallow water (0 to 5 cm) outside the wintering season (χ^2=92.84, df=2, $p<0.001$, and χ^2=30.47, df=2, $p<0.001$, respectively). In winter, it fed from the walls in the active salinas (χ^2=100.28, df=2, $p<0.001$). An apparent selection of the shallow water in the inactive salinas could not be statistically validated, perhaps due to the small sample size. In both cases, very few birds fed in deep water.

Ringed Plover. In the active salinas, Ringed Plover fed in the intermediate pans (the preparation pans) in winter (χ^2=200.09, df=2, $p<0.001$; no statistical validation was possible for the inactive salinas), but spread out over all the other microhabitats during spring/autumn, both in active (χ^2=2.14, df=2, $p<0.34$, ns) and in inactive salinas (χ^2=0.8, df=2, $p<0.57$). This species restricted its feeding to shallow water in active salinas (winter: χ^2=673.36, df=2, $p<0.001$; spring/autumn: χ^2=88.91, df=2, $p<0.001$) but did not show any depth preference in inactive salinas (spring–autumn: χ^2=1.00, df=2, $p<0.9$; no statistical validation possible for winter).

Dunlin. In the active salinas this species also fed mainly in the preparation pans throughout the year (winter: χ^2=1851.66, df=2, $p<0.001$; spring–autumn: χ^2=241.42, df=2, $p<0.001$). In inactive salinas, however, Dunlin showed clear seasonal variations: in spring/autumn the storage pans were used most (χ^2=308.51, df=2, $p<0.001$), while in winter the birds moved to the chrystallisors (χ^2=22.69, df=2, $p<0.001$). This species almost always fed in deep water, both in active salinas (winter: χ^2=1706.16, df=2, $p<0.001$; spring/autumn: χ^2=468.62, df=2, $p<0.001$) and in inactive salinas (winter: χ^2=26.91, df=2, $p<0.001$, spring/autumn: χ^2=301.66, df=2, $p<0.001$).

Black–winged Stilt. The only long–legged species studied, the Black–winged Stilt, was absent from the salinas in winter. During the breeding/migratory seasons, it consistently chose the storage pans as its main feeding area, both in active (χ^2=113.36, df=2, $p<0.001$) and inactive salinas (χ^2=48.18, df=2, $p<0.001$). However, somewhat contrary to expectation, it fed more in shallow waters than in deep waters both in active salinas (χ^2=184.48, df=2, $p<0.001$), and inactive salinas (χ^2=107.92, df=2, $p<0.001$).

In summary, most of the studied species were more selective with regard to the type of pond used in winter than in spring and autumn, this being particularly evident in the active salinas. The exception was the Dunlin, which showed a different pattern of use of the ponds in both seasons and in both types of salinas. In contrast, in all species, few differences in the feeding depth were found between the two types of salinas, although they could occur between seasons independently of the type of salina (e.g. in the two plovers).

Long–term use of inactive salinas

Even a long abandoned salina can still be used by waders. However, usage is likely to decline over time because of the lack of management, due partly to the development of an impenetrable layer of vegetation and partly to drastic alterations

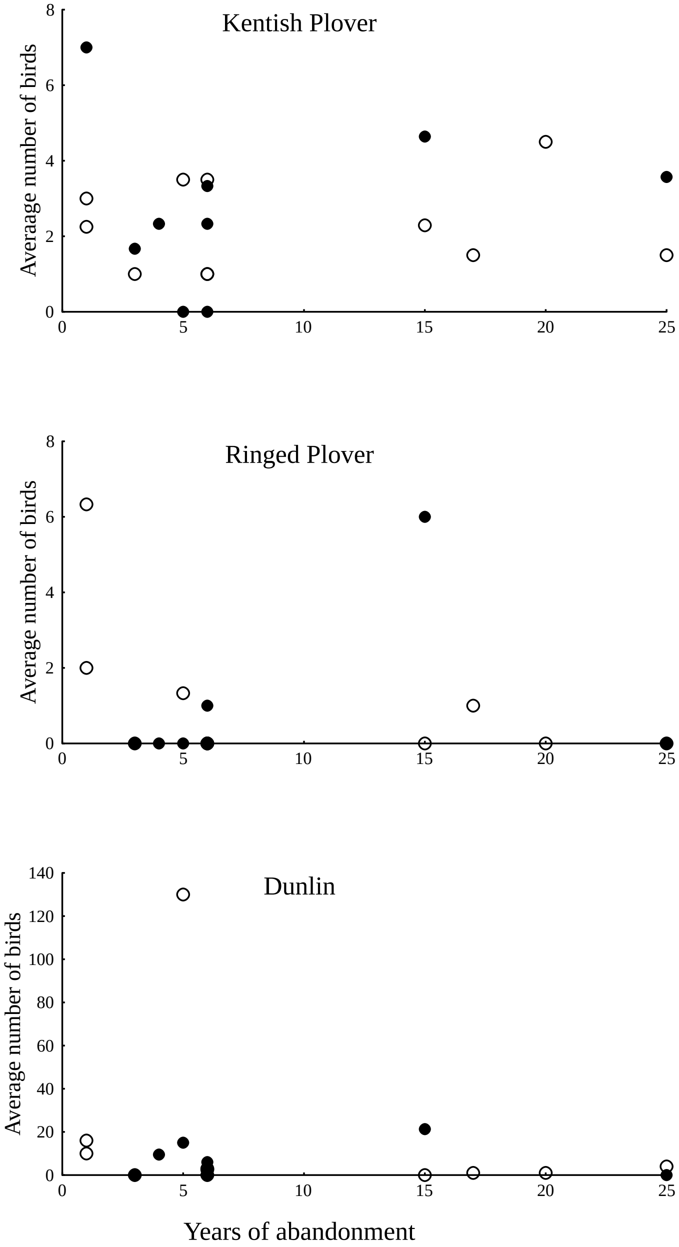

Figure 4.7: Relatinship between the number of birds of selected species feeding al low–water in abandoned salinas, and the time elapsed since these were abandoned. Data were plotted separately for 1993–94 (closed dots) and 1994–95 (open dots), and represent the average of spring and autumn means for each species. $p > 0.05$ in all cases.

in the water level. To establish the duration of the period over which the waders can use the deserted salinas, the average number of birds of selected species feeding at low–water in several abandoned salinas was related to the time elapsed since the salina had been deserted, as given by local salt–workers. Since not all the salinas were visited in the same year, the analysis was performed separately for each year.

The usage of deserted salinas by the species analysed was not correlated with the period of time elapsed since the salinas were deserted (Figure 4.7). This suggests that inactive salinas may provide adequate feeding conditions over a long period. Even a salina deserted for 25 years can support some – albeit few – birds. In fact, only the salinas that had been deserted for more than 40 to 50 years, and which by now had changed into brackish–water marshes, were totally ignored by waders.

The reason for the lack of a correlation between the time elapsed since the desertion of a salina and the number of birds that still use it, is probably the highly variable rate at which the ecology of deserted salinas change, as pointed out by Neves & Rufino (1995), and discussed above (section 4.2.2). It may be wise, however, to interpret the present results with some caution, primarily because the method for estimating the time elapsed since desertion was rather crude.

4.3.3 Consequences of the loss of the salinas

The previous sections have demonstrated the significance of the salinas as feeding areas for most wader species in the estuary, as well as how their use may be influenced by human management. This section deals with some crude predictions of how much feeding opportunity, defined in terms of feeding space and feeding time, would be lost to the waders if all the Morraceira's salinas were destroyed.

The data for the two years were analysed separately, in order to illustrate the year–to–year variability that would be likely to occur in the impact of such a loss of habitat. In each case, the average of autumn, winter and spring means of the feeding opportunities lost (see section 4.3.2) were used.

In case the effect of habitat loss varied between species of different size, the results were related to body mass. Body masses were taken from Cramp & Simmons (1983) selecting, for each season and species, the site(s) with the nearest latitude to the Mondego. Monthly data from southern Portugal were available for Dunlin, Kentish Plover and Redshank (Batty, 1991). This author reported, for Dunlin, an average mass of 48.2±4.9 (SD) g in April. This value was not statistically different (t=0.32, df=23, ns) from the average mass of a sample of Dunlins captured with mist–nets in the Mondego in April 1996 (47.7±4.5 g)(C. Pacheco, pers. comm.). It was therefore assumed that the body masses were likely to be similar, within a given month, between the Algarve and the Mondego and, consequently, the values given by Batty (1991) both for Dunlin and for the other two species were used in this study, as well.

Calculating the loss of feeding opportunities in the salinas

The basic assumption of the analysis was that the birds displaced from the Morraceira's salinas would not redistribute themselves over the salinas remaining in the south arm. In fact, it is possible that some birds would occupy the south arm's salinas.

There are, however, reasons to believe that these salinas could not entirely substitute for those of the Morraceira, mainly because of the difference in areas involved: the total area of the south arm's salinas represents only about 42% of that of the Morraceira's salinas (80.4 ha against 192.6 ha). Excluding the area occupied by fish–farms and drained salinas in both sites, the displaced birds – assuming that they would distribute randomly among the south arm's salinas – would have an area of suitable feeding equivalent to only 37% of that they would have lost in the Morraceira. However, on the few cases when the waders were observed in the south arm's salinas, they seemed to consistently use the same pans, even on occasions when they might have been expected to occupy all the potentially available feeding places in the salinas; for example, in severe weather conditions (e.g. winter gales), or when large numbers were present in the estuary (e.g. Dunlin, in spring). This suggests that the real feeding area available for waders in the south arm was probably even more restricted that the figure presented above and that the 'best' solution for most waders would be to redistribute themselves mainly over the intertidal flats of the estuary itself.

The importance of an area for feeding can be viewed in terms of both its spatial and temporal dimensions. Although each dimension was treated separately in this analysis, for the sake of clarity, biological realism demands that they should also be considered together. Here, the combined dimensions of feeding space and time available for feeding are termed *feeding opportunity*.

The spatial dimension. It is not easy to clearly assess the value of an area for feeding, as its quality will largely depend on the availability of food, and on the way this changes through autumn, winter and spring (Zwarts & Wanink, 1993). However, an approximate idea can be obtained by calculating the total amount of feeding done by waders in that area over a given period, such as a complete tidal cycle. The easiest and least ambiguous measure to use is the total number of bird–feeding hours during a complete tidal cycle, i.e., *the number of feeding birds*×*the number of hours available for feeding in a given habitat over a single tidal cycle*. This measure is easy to understand and can be readily used to measure the effect of the loss of space: if the area is destroyed, the birds will lose the number of bird–feeding hours that they previously had in that area. For simplicity it was assumed that the estuarine populations of most species could adopt only two strategies when feeding in the salinas: either they (1) remained in the salinas throughout the whole tidal cycle or (2) they used the salinas only at high–water, returning to the mudflats with the ebbing tide.

Therefore, for the birds that remained in the salinas throughout the tidal cycle (hereafter designated as *salinas birds*, or SB), the following formula was used:

$$SFH = TS_{LW} \times 12.5h \tag{4.2}$$

where

SFH – number of bird–feeding hours of the salinas population per tidal cycle

TS_{LW} – number of feeding birds counted in the salinas at low–water

and 12.5 h represents the average duration of a tidal cycle, from low–water to high–water (see Chapter 2).

For the *mudflat birds* (MB), a more complex calculation was needed in order to include both the hours spent feeding at low–water on the mudflats, and those spent feeding in the salinas at high–water. For this, the number of mudflat birds that joined salinas birds in the salinas at high–water had also to be estimated. Moreover, because some birds used the salinas at high–water only for resting, these also had to be taken into account in the calculations. It was assumed that all the feeding birds counted on the mudflats fed there for the duration of the emersion period of the intertidal flats, even if they actually made some pauses while feeding (e.g. Zwarts & Dirksen, 1990). The formula used was:

$$MFH = M_{LW} \times 8h + [(S_{HW} - S_{LW}) - (NFS_{HW} - NFS_{LW})] \times 4.5h \tag{4.3}$$

where

MFH – number of bird–feeding hours of the mudflat birds per tidal cycle

M_{LW} – number of feeding birds counted in the mudflats at low–water

S_{HW} – number of birds (feeding + non–feeding) counted in the salinas at high–water (salinas birds + mudflat birds that go to the salinas at high–water)

S_{LW} – number of birds (feeding + non–feeding) counted in the salinas at low–water

NFS_{HW} – number of non–feeding birds in the salinas at high–water (mudflat birds + salinas birds)

NFS_{LW} – number of non–feeding birds in the salinas at low–water (salinas birds)

with $SH_{HW} \geq S_{LW}$. The term $(S_{HW} - S_{LW})$ indicates the number of *mudflat birds* (feeding and non–feeding) that went to the salinas at high–water: it was thus assumed that the number of salinas birds feeding remained constant throughout the tidal cycle and was equal to the total number present at low–water (that is,

S_{LW}). The term ($NFS_{HW} - NFS_{LHW}$), on the other hand, is the number of *non–feeding mudflat birds* in the salinas at high–water. Consequently, $[(S_{HW} - S_{LW}) - (NFS_{HW} - NFS_{LW})]$ represents the number of *mudflat birds* that used the salinas for *feeding* at high–water. Occasionally, $S_{LW} \geq NFS_{HW}$. In this circumstance, the non–feeding birds were considered to be only mudflat birds, and a simplified equation was used:

$$MFH = M_{LW} \times 8h + [(S_{HW} - S_{LW}) - (NFS_{HW}] \times 4.5h \tag{4.4}$$

This situation, however, was rare. As in the previous case, 8 h corresponded to the maximum time available for feeding in the mudflats at low–water and 4.5 h was the maximum feeding time available at high–water in the salinas, before the flats were again exposed.

The temporal dimension. The removal of the salinas would remove all the feeding space presently used by the salinas birds, which would thus lose all the current feeding time as well. But the mudflat birds would not only loose feeding space, but also *feeding time* if all salinas were lost, because they would then be deprived of the extra–feeding time provided by the salinas at high–water.

A detailed assessment, for each species, of the time they fed over a whole tidal cycle was impossible, as it would have been too time–consuming. It would have required, for example, the proportion of the birds on both the mudflats and the salinas that were feeding to be measured as the tide receded and advanced as well as over dead low and high–water, and this would have to have been done throughout all seasons. However, a simplified estimation was obtained using the following formulae:

$$FT_{HW} = (FM \times 8h) + (FS \times 4.5h) \tag{4.5}$$

$$SFT_{LW} = (FT - 8h)/8h \times 100 \tag{4.6}$$

where

FT – total feeding time of the mudflat birds (in hours)

FM – proportion of the mudflat birds that were feeding on the mudflats at low–water

FS – proportion of the mudflat birds that were feeding in the salinas at high–water

SFT_{LW} – the shortfall in feeding time following the salinas loss as a percentage of maximum time that could be spent feeding on the mudflats after the salinas had been removed.

and 8 h and 4.5 h is the time available for feeding in the mudflats and in the salinas at high–water, respectively, as defined above. By applying Equation 4.5, the number of hours of feeding done by an average mudflat bird over both habitats per tidal cycle could be calculated. Equation 4.6 was then used to calculate the amount by which the present–day feeding time of the mudflat birds exceeded the maximum possible available feeding time in the mudflats (8 h) alone, as would be the case if the salinas were to be removed. This calculation is based, of course, on the assumption that birds could feed continuously for 8 h (but see Zwarts & Dirksen, 1990).

If $FT < 8$ h in a particular species, mudflat birds were assumed able to satisfy their energey requirements in the time that the mudflats were exposed. Values greater than 1 mean that, after the salinas had been lost, the birds would not be able to feed for as many hours as they do now over the whole 12.5 hours tidal cycle. Equation 4.6 thus provided a measure of the *percentage of the present–day feeding time that would be lost by the birds* if the Morraceira's salinas were destroyed.

Assumptions and limitations of the approach

The methodology made a number of assumptions that introduced some limitations on what could be done. It was necessary to assume that the number of feeding birds in the salinas remained constant through low and high–water. In fact, the values may have been overestimated, since the amount of feeding in many species changes through the tidal cycle, both in the intertidal areas (Katzemberg & Exo, 1994), and in supra–tidal areas, such as the salinas (Rufino *et al.*, 1984; Batty, 1991). Moreover, the use of 'average' values is an oversimplification, since the feeding requirements of individuals are likely to vary, depending on the sex, age, body condition, and also on the season (Pienkowski & Evans, 1985; Davidson & Evans, 1986; Goss-Custard *et al.*, 1996a,b,c).

The method for calculating the effect of the loss of salinas also assumed that two identifiable sub–populations – salinas and mudflat birds – did exist. Preliminary observations did indeed suggest that some birds relied on the salinas only as supplementary high–water feeding areas to their main feeding areas in intertidal mudflats (the mudflat birds), whereas others did remain in the salinas throughout the tidal cycle (salinas birds). The best way to find out would have been to individually mark some birds with colour rings, die–marks or radio transmitters, and to follow them for some weeks. This was not possible, due to time and logistic constraints. Nonetheless, the following evidence in favour of the distinction between salinas and mudflat birds hypothesis was obtained from field observations and bibliographic sources:

1. No birds were ever seen flying during low–water from the salinas to the adjacent intertidal mudflats. In fact, salinas birds showed a high degree of site–fidelity to their low–water feeding sites. Even when they were disturbed and forced to fly, they quickly returned after the disturbance ceased. If disturbance persisted for a long time, the flocks flew over the salina for some minutes before going away, but in many cases they returned later. Similarly, no flights from the mudflats to the salinas were recorded over the low–tide period, at least when most of the intertidal areas had emerged, 2 hours either

side of low–tide;

2. Large flocks ($>$1000 birds) of the smaller–sized species where rarely seen in a single salina at low–water. Therefore, when present, such flocks could be used as a marker to compare with the numbers present at high–water. On two occasions, in the winter 1993–94, large flocks of Dunlin consisting of approximately the same numbers were counted in the same day, in the same salinas, at both high and low–water;

3. In other salinas in southern Portugal and Spain, various degrees of fidelity throughout the tidal cycle to the salinas have been reported in winter (Rufino *et al.*, 1984; Perez-Hurtado & Hortas, 1993a) and spring (Evans, 1995). In the Portuguese areas, the fidelity seems to be quite high. Recently, some dye–marking experiments performed in the Ria de Aveiro, some 60 km north from the Mondego estuary, supported the conclusion that certain individuals remained in the salinas throughout low and high–water periods (A. Luís, pers. comm.).

These facts suggest that some degree of site–fidelity is common in the populations that use the salinas as feeding habitats, and thus give some strength to the assumption made. However, they do not show for how long the site fidelity is maintained by a single bird within a given season. In the present analysis, it was assumed that the number of birds using the salinas throughout the tidal cycle remained constant throughout a given season, and that they formed a distinct sub–population. This is likely to be the case during the migratory periods when the birds remain in the area for just some days (Evans, 1995, and pers. observ.) and, obviously, in the breeding season. In winter, however, the assumption that birds are faithful to the salinas is less well–founded, as the salinas are thought to provide less rewarding feeding areas than the mudflats (Batty, 1991). The possibility that they can move once in a while to the mudflats to feed, in order to avoid starvation is, therefore, very plausible (see, for instance, Davidson & Evans, 1986).

But in spite of these limitations, the present approach, which was adopted because of both human and logistic constraints, provide a first, albeit rough estimate of the reduction in feeding opportunities that would follow the complete removal of the salinas.

Loss of feeding opportunities

Loss of feeding space The complete disappearance of the Morraceira's salinas would have different impacts on the populations of the different species studied (Tables 4.14 and 4.15).

Independently of year, the loss of the alternative feeding habitat would have an important (more than 50%) impact in the three 'salinas species' – Redshank (74% reduction), Little Stint (72%) and Common Sandpiper (63%). The effect would be less severe for Kentish Plover (25%), Dunlin (24%), Turnstone (24%),

Table 4.14: The alternative (salinas birds feeding hours, SFH) and supplementary (mudflat birds feeding hours, MFH) feeding opportunities lost if the Morraceira's salinas were to be destroyed, expressed as a percentage of total bird–feeding hours: BFH. Values refer to 1993–94, averaged for autumn, winter and spring; therefore, n=3 except for Black–tailed Godwit and Whimbrel, for which n=2. The species are ranked according to the percentage of alternative feeding–hours that would be lost.

Rank Order	Species	Total feeding in the estuary (BFH±1SD)	% feeding–hours lost (SFH)	% feeding–hours lost (MFH)
1	Redshank	122.7±89.7	66.2	72.9
2	Little Stint	486.7±396.6	63.8	14.8
3	Common Sandpiper	96.8±48.6	44.4	11.2
4	Kentish Plover	597.9±304.7	24.9	11.1
5	Dunlin	5555.2±3323.5	23.2	10.4
6	Turnstone	93.4±64.1	24.0	26.0
7	Black–tailed Godwit	4100.5±5753.8	33.4	3.5
8	Ringed Plover	631.5±297.4	13.8	18.8
9	Grey Plover	510.9±393.5	7.0	0.0
10	Avocet	1644.0±2082.7	0.7	0.2
11	Whimbrel	63.5±51.6	0.0	0.0

Table 4.15: The alternative (salinas birds feeding hours, SFH) and supplementary (mudflat birds feeding hours, MFH) feeding opportunities lost if the Morraceira's salinas were to be destroyed, expressed as a percentage of total bird–feeding hours: BFH. Values refer to 1994–95, averaged for autumn, winter and spring; therefore, n=3 except for Black–tailed Godwit, Whimbrel and Turnstone, in 1993–94, and Avocet, in 1994–95, for which n=2. The species are ranked according to the percentage of alternative feeding–hours that would be lost.

Rank Order	Especies	Total feeding in the estuary (BFH±1 SD)	% feeding–hours lost (SFH)	% feeding–hours lost (MFH)
1	Redshank	277.1±246.7	73.7	0.9
2	Little Stint	181.3±137.7	72.2	22.2
3	Common Sandpiper	70.5±34.2	62.9	5.2
4	Black–tailed Godwit	721.8±460.0	59.7	25.3
5	Avocet	867.9±1209.6	50.0	0.0
6	Dunlin	4080.9±1533.9	24.2	22.9
7	Kentish Plover	739.3±373.1	19.3	21.8
8	Whimbrel	50.3±44.4	16.2	19.2
9	Turnstone	20.2±24.2	13.0	24.0
9	Ringed Plover	589.7±290.4	5.7	22.6
11	Grey Plover	387.8±366.8	0.0	21.9

Table 4.16: Total feeding time used at present by the mudflat birds and the percentage of it that would be lost were the Morraceira's salinas to be removed. Values are means of up to three seasonal (autumn, winter and spring) average values . In brackets, the number of seasons considered in each case. TFT=Total feeding time (hours±1 SD); % TL= % of time lost by removal of the salinas.

	1993–94			1994–95		
	TFT	n	% TL	TFT	n	% TL
Turnstone	12.4±5.1	2	55.0	11.3±0.0	1	41.3
Dunlin	10.8±0.7	3	35.0	11.2±1.5	3	40.0
Common Sandpiper	10.3±2.5	3	28.8	3.7±6.4	3	0.0
Kentish Plover	9.1±0.8	3	13.8	8.4±1.9	3	5.0
Ringed Plover	9.1±1.4	3	13.8	9.5±0.9	3	18.8
Grey Plover	8.8±3.2	3	10.0	9.3±2.3	3	16.3
Redshank	8.2±8.0	3	2.5	10.9±2.0	3	36.2
Whimbrel	7.8±0.0	1	0.0	11.8±0.8	2	47.5
Black–tailed Godwit	6.7±2.1	2	0.0	9.0±8.2	3	12.5
Avocet	6.4±2.2	3	0.0	4.5±0.0	1	0.0
Little Stint	4.8±6.1	3	0.0	3.9±6.7	3	0.0

and Ringed Plover (14%), and negligible for the Avocet (0.7%) and Whimbrel (0.0%)[6]. The species most affected by the loss of their high–water supplementary feeding would be the Redshank (73% reduction). Turnstone, Dunlin, Ringed Plover and Little Stint would be moderately affected, through reductions of 26%, 23%, 23% and 22%, respectively, while the other species would lose less than 20% of feeding. The least affected species would be the Avocet (0.2%). In the Grey Plover and Black–tailed Godwit, the impact of the loss of alternative and supplementary feeding would be more variable. Grey Plover would suffer a maximum of 7.0% reduction in its alternative feeding, but some 23% in its supplementary feeding. Black–tailed Godwit, on the other hand, would present reductions of, respectively, 60% and 25% in its feeding space.

Two main conclusions can, therefore, be drawn from the calculations using equations 4.2 and 4.3: (1) In general, the impact of the loss of feeding space in the salinas would be more severe for those parts of the populations that used the salinas at low–water as alternative habitats to the mudflats, and (2) yearly differences in the impact would occur, particularly in some species.

Loss of feeding time by the mudflat birds The time available for feeding throughout a complete tidal cycle for the mudflat populations of some species would also be affected by the loss of the salinas. Yearly differences in the predicted impact were less marked than those for the loss of space, except for the Common Sandpiper, the Redshank, the Whimbrel and the Black–tailed Godwit (Table 4.16).

The mudflat–feeding Turnstones and Dunlins would lose, depending on the year, up to 55% and 40%, respectively, of their present–day feeding time. Less

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[6] The value for the Avocet in 1994–95 is probably overestimated, as it refers to the presence of a single bird, in spring. The same is applicable to the Whimbrel.

Table 4.17: The birds feeding–hours in the Morraceira's salinas (bird–hours±1 SD) by the main species of waders (high–water and low–water combined) and, were the salinas to be destroyed, the predicted increase (in %) in the bird–feeding hours at the mudflats that would have to take place over low–water period for the present–day amount of bird–feeding hours to be maintained. Data details as in Tables 4.14 and 4.15.

	1993–94		1994–95	
	Total feeding in salinas	% of the total feeding hours	Total feeding in salinas	% of the total feeding hours
Little Stint	377.0±494.3	84.8	108.3±132.8	95.2
Redshank	34.3±8.2	72.9	104.8±148.6	59.1
Common Sandpiper	103.9±43.5	65.3	183.0±220.4	74.7
Kentish Plover	200.4±81.5	38.5	167.2±93.8	31.0
Dunlin	2009.5±1948.1	37.2	1707.7±715.7	47.9
Turnstone	20.2±35.2	37.0	18.4±26.1	55.6
Ringed Plover	599.0±670.9	28.2	181.9±102.1	25.6
Black–tailed Godwit	1005.7±1741.9	18.5	677.9±474.1	89.7
Grey Plover	14.3±24.8	8.0	4.5±4.5	1.0
Avocet	6.0±5.9	0.8	6.3±8.8	1.4
Whimbrel	0.0±0.0	0.0	8.3±7.2	16.2

dramatic losses would be faced by the Ringed Plover (13% to 19%), the Grey Plover (10% to 16%) and the Kentish Plover (5% to 14%). The Avocet and Little Stint would be least affected, as they require, on average, less feeding time than the 8 h available during the low–water period. Whimbrel, Redshank, Common Sandpiper and Black–tailed Godwit, would require up to 47%, 36%, 29% and 13% extra–feeding, but it is likely that these values could greatly change from year to year, as shown during the two years of study.

Extra–feeding pressure in the mudflats

An hypothetical destruction of the salinas would mean that feeding would subsequently be concentrated mainly on the mudflats. According to the calculations developed in this study, and depending on the species, this would imply an increase in bird–feeding hours on the mudflats of up to 55%, although the average increase would be of 35.6%±28.8%(SD) or 45.2%±33.1%, depending on the year. At the species level, there is a remarkable uniformity between years (Table 4.17).

The species that would contribute the most to the extra–feeding pressure on the mudfltas would be, of course, those that presently depend most on the salinas: the Little Stint (85% to 95% of extra–feeding bird–hours), Common Sandpiper (65% to 74%) and Redshank (59% to 73%). Several species (Dunlin, the two small plovers, Turnstone), would contribute with between 30% and 55% of the total extra bird–feeding hours across all species. The Black–tailed Godwit would have a variable contribution (19% to 90%), as well as the Whimbrel (0% to 16%), as a consequence of the more intermittent use made of the salinas by these species. The same applies to the Grey Plover and to the Avocet, whose potential contributions can probably be ignored (0% to 8%).

4.3.4 Is the loss of feeding opportunities a function of the species–specific metabolic requirements?

The above section showed that different species are likely to be differently affected by the loss of the salinas. Why is this so? One explanation might be that the impact is related to the different metabolic requirements of each species. Small birds with high metabolic needs per unit of biomass tend to feed for a greater proportion of the tidal cycle than waders with large body size (Davidson & Evans, 1986; Zwarts *et al.*, 1990). If this holds for the Mondego, it would be expected that smaller species would spend a proportionally larger part of their total feeding time in the salinas than the larger ones. Consequently, the small–sized mudflat species should be most affected by any loss of supratidal habitat that removes feeding time.

The salinas birds, on the other hand, are not so time–constrained, as they can, at present, feed for the entire duration of the tidal cycle in their preferred salinas habitats. For these species, the loss of feeding opportunities would be translated into the loss of alternative feeding space at low–water. Thus, if the small–sized salinas species were to be more affected by the loss of feeding opportunities than the larger ones, they would lose proportionally more alternative area than these species.

If the predictions regarding the mudflat birds and the salinas birds in relation to body mass were both met, then, the hypothesis that the salinas loss would most affect the small–sized species would be supported. This section tested the above predictions separately for each year. The time spent in the salinas by the mudflat birds at high–water was given by the percentage of hours that each species used over and above the time for which the mudflats were exposed (8 h), assuming that only the salinas were used for feeding at high–water. The loss of space, on the other hand, was measured as the percentage of bird–hours of feeding spent in the salinas at low–water in relation to the total feeding hours used at low–water in both the intertidal areas plus the salinas.

Both the percentage of extra time needed by the mudflat birds to feed over and above the 8 h that the intertidal areas were exposed, and the percentage of feeding–hours used by the salinas birds were correlated with body mass in 1993–94, but not in 1994–95 (Figure 4.8).

This suggests that the small–sized species would indeed be more affected by both the loss of time and space than the larger ones in some years.

4.4 Discussion

4.4.1 Use of the salinas by waders: conservation implications

The results of this study generally confirmed the findings published elsewhere (Rufino *et al.*, 1984; Velásquez & Hockey, 1991; Batty, 1991; Perez-Hurtado & Hortas, 1991, 1993a,b) regarding the importance of the salinas in providing high–water supplementary feeding for many species of waders, particularly the smaller

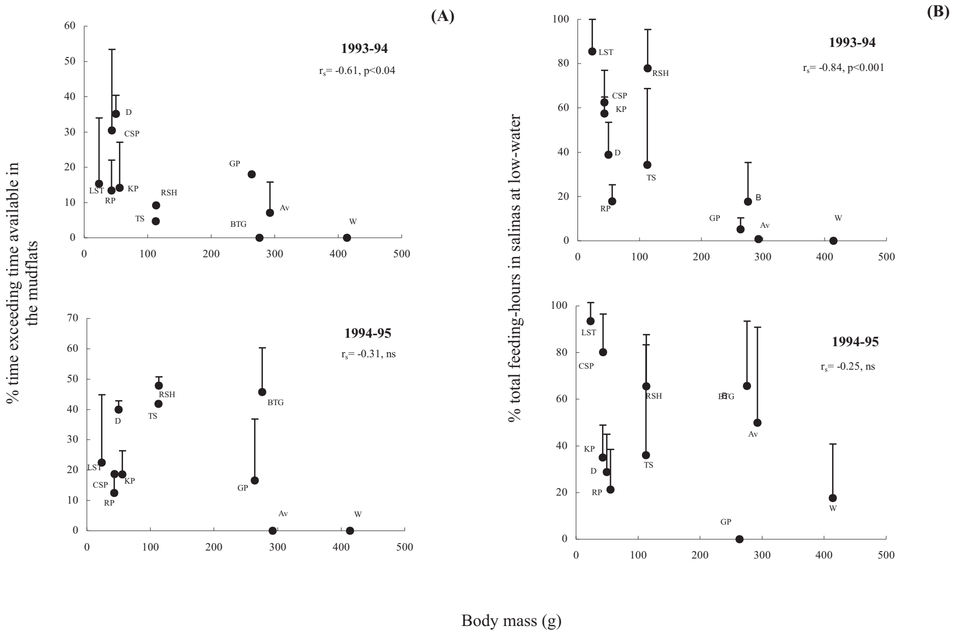

Figure 4.8: Relation between body mass and (A) percentage of time used for feeding in the salinas by mudflat birds, (B) percentage of bird–hours of feeding done in the salinas at low–water by the salinas birds. Data were plotted separately for 1993–94 and 1994–95. Values are average (±SE) of seasonal means in autumn, winter and spring. Coefficients of correlation (Spearman's r) are also given.

species, irrespective of their type of management. In this sense, the salinas may be regarded as south European supra–tidal feeding areas equivalent to the fields and peripheral wetlands of northern Europe (Goss-Custard, 1969; Davidson & Evans, 1986; Lambeck *et al.*, 1996).

However, contrasting with many northern supra–tidal habitats, the salinas are also frequently used as alternative low–water feeding areas, irrespective of the season. Values for the proportion of birds, relatively to the whole estuarine populations, that use the salinas at low–water range from 17%–19% (Rufino *et al.*, 1984; Velásquez & Hockey, 1990) to 34% (Perez-Hurtado & Hortas, 1993a), compared to the 8.3% recorded in the peripheral wetlands in England (Davidson & Evans, 1986). The Mondego salinas were not an exception to this pattern, with some 30% of the birds across all species using the habitat at low–water, 42.5% of which, on average, actively foraging there. The double role of the salinas as supplementary and alternative feeding areas in the estuary of Mondego confers to this habitat an added value, from the conservationist perspective, and raises the question of how detrimental for waders could be the transformation or desertion of the salinas.

The birds seemed to take advantage of the large number of salinas present, which presumably offered them a gradient of physico–chemical conditions (and therefore feeding resources) from which they could choose the most profitable, according to their energetic requirements. Due to the lack of permanent management (e.g. Velásquez, 1992), the feeding conditions in each salina are likely to vary over the year, which could explain why the birds changed the particular salinas they used on a seasonal basis. Both the importance of the salinas as all–tide feeding areas, and the fact that the birds changed their preferred salinas on a seasonal basis, have implications for the conservation of the habitat, as they imply that the integrity of the whole area of salinas should be maintained.

The abandonment of the salinas also contributes to the loss of feeding habitat if the waders are not able to use the inactive salinas. There are no published information on the relative use made of the active and inactive salinas by feeding birds, although there are indications that inactive salinas are less favoured feeding places (R. Rufino, pers. comm.).

In the Mondego, the active salinas were generally more used as feeding places than the inactive ones. Although more detailed data are needed, the preliminary results from the examination of the macrohabitat variables chosen (water depth and salinity) suggest that most species selected the pans of lower salinity (storage and preparation pans) for feeding or, when they used the higher salinity pans (the crystallizing pans), they did so in the abandoned salinas, where the salt concentration is likely to be much less than in the actively producing crystallizing pans. Possibly, the more constant and controlable level of water in the active salinas provide stable salinity conditions for the development of both the benthic and epibenthic prey (Velásquez, 1992) and is the main reason behind the observed pattern.

However, as suggested in this study, inactive salinas can still support waders for a variable period of time, depending on the particular characteristics of each salina. Also, a deserted salina is nearly always recoverable, while a salina transformed into a fish–farming area is completely lost for waders. Moreover, in spite of the severe losses sometimes caused by unexpected floods (Rufino & Neves, 1992), abandoned salinas are also used by breeding species (e.g., Kentish Plover, Black–winged Stilt). Thus, a main feature of the Mondego salinas is the diversity of macrohabitats that can be found even in the deserted salinas. The maintenance of this diversity is perhaps the key–factor that allow the waders' survival in this important supratidal habitat of the Mondego estuary.

This reasoning points towards the need to keep a nucleus of inactive salinas available alongside active ones in any future management plan for the area. Such salinas should be selected among those mostly used at present by the waders. Obviously, it is unrealistic to think of reactivating most of these salinas, when the current tendency is for the salt production to decline. However, as Neves & Rufino (1995) suggested, it would be possible to pay their owners to maintain some of them in a relatively clean and undamaged state. This would help to provide adequate feeding opportunities for waders, at least in the energetically more demanding periods of the yearly cycle, notably during the migratory seasons.

4.4.2 The effect of losing salinas

Considering the importance of the salinas in providing feeding opportunities for waders in the Mondego estuary, what could be the consequences for the wader populations if this habitat was destroyed? The preliminary calculations developed in this study should not be considered as predictions, in the strict sense of the term, because they are too crude and subjected to a number of many untested assumptions. They suggest, however, that the loss of the habitat would not have the same effect for all species, or species' sub–populations. There were some indications that the loss of feeding opportunities could proportionally affect more the small–sized species than the larger ones, but this tendency only held for 1993–94. How can these differences between years be explained? Seasonal differences in feeding opportunities could have been involved.

In fact, Zwarts *et al.* (1990) found that, in the Banc d'Arguin, many waders increased their feeding time in early spring, when preparing for their migratory departure, as compared to the winter. Furthermore, this increase was not the same across all species: it occurred mostly in the large species because the smaller ones were already feeding close to 100% of the time in winter. Because of this, no correlation was obtained between feeding time and body mass in early spring.

Although the Mondego data refer to yearly averages, they were based on seasonal means. Accordingly, it is possible to test whether a seasonal change, similar to that described by Zwarts *et al.* (1990), took place in this estuary. In fact, the mudflat birds of some larger species (Grey Plover, Black–tailed Godwit, Common Sandpiper) increased their feeding time in 1994–95, compared to the previous year, which could arise from significant seasonal differences in 1994–95, as suggested by the large SE for these species (Figure 4.8). On the other hand, the considerable increase in feeding time between years (from less than 10% to between 40%–50%) for some small species as Redshank and Turnstone was apparently not due to seasonal variations in the feeding time (small SE values). For these species, the average feeding time could have really changed, for some unknown reason, between the two years.

Methodological problems may also have been involved. The fact that only two counts on peak–tides were used to represent the whole tidal cycle have certainly led either to under–estimations or over–estimations of the real feeding time for some species, particularly for those that occurred in small numbers and only in some seasons. This could have been the case in mudflat–feeding Redshank and Turnstones, as these species used the mudflats intermittently during the migratorey periods.

However, the fact that a consistent, although not statistically significant, negative relation between body mass and the loss of feeding time (in the case of mudflat birds) or feeding space (in the salinas birds) was obtained in 1994–95, suggests that the idea that the small–sized species may be more affected by the loss of feeding opportunities than the larger ones, should not yet be completely discarded.

Despite this uncertainty, it is clear that the Mondego's salinas represent an invaluable feeding area for many species, either at low–water and at high–water. The need to protect this important man–made habitat in the Mondego estuary becomes even more urgent if it is recalled that the other estuarine supratidal habitats are not able to replace the Morraceira's salinas, either because the waders do not use them for feeding (the fish–farms), or because they occupy a much smaller area (the saltmarshes and the south arm salinas). Assuming that the smaller species are indeed more affected by a potential loss of this habitat, and since they form the bulk of the estuarine avifauna, the destruction of the Morraceira's salinas could imply either that (1) most birds would be driven to the mudflats, with a corresponding increase in the competition for feeding in these areas (see Chapter 5), or (2) they would simply leave the estuary. In either case, the ecological consequences for the whole estuarine ecosystem could be large.

The conclusions of this study contrast with the opinion from Smit *et al.* (1987), who claim that the construction of salinas inevitably cause the loss of feeding habitat, since they are usually built on intertidal mudflats or saltmarshes. This is probably true in the case of the large industrial Mediterranean salinas, such as those of the Camargue, south of France (Britton & Johnson, 1987). However, in Portugal (and in the Atlantic coast of Spain), the salinas are mainly built inland (Lopes, 1955; Gonçalves & Sobreiro, 1992), and even industrial salinas are usually built in areas formerly occupied by artisanal salinas. This not only avoids the loss of intertidal areas, but even increases the potential feeding areas available for waders.

Chapter 5

Competition for Feeding in the Intertidal Areas of the Mondego Estuary

5.1 Introduction

The loss of feeding area in an estuary may pose a problem for the waders that usually feed there. While it is true that birds may leave the area and move to another estuary, it might be expected that the first reaction of the birds affected by such an event is to redistribute themselves among the remaining feeding areas of the estuary (Goss-Custard, 1977a, 1979; Goss-Custard *et al.*, 1993, 1994; Dolman & Sutherland, 1994).

Waders usually aggregate in the areas where the densities of their potential prey are highest, thus allowing the birds to maximise their rate of energy intake and, therefore, their chances of survival (Brown & O'Connor, 1974; Goss-Custard, 1977a, 1979, 1984; Goss-Custard *et al.*, 1977a,b, 1991; O'Connor & Brown, 1977; Bryant, 1979; Evans *et al.*, 1979; Evans & Dugan, 1984; Piersma *et al.*, 1993b; Mercier & McNeill, 1994; Kalejta & Hockey, 1994). So, the immediate consequence of redistribution following habitat loss is an increase in the densities of birds feeding in those areas that remain.

As densities rise, competition between birds for resources will increase leading, in turn, to a reduction in intake rate. At a certain point as competition intensifies, the rates of emigration and/or mortality become density–dependent. Eventually, the point will be reached as densities increase at which, for each new individual that arrives, one will leave or die. Then, the *carrying capacity* of the area will have been reached and corresponds to the *maximum number of individuals that the area can support* (Goss-Custard, 1985).

In fact, the densities of birds in most estuaries are likely to be well below the maximum possible, although individual areas within them may already be at capacity (Goss-Custard, 1985, but see Meire *et al.*, 1994 and Goss–Custard *et al.*, 1996b). Up to now, only by simulation has it been possible to predict the carrying capacity of an entire estuary (Goss-Custard *et al.*, 1994, 1995a,b,c,d, 1996a,b). The ability to accurately predict when carrying capacity has been reached has a practical significance as it may help to guide conservation efforts when major developments are planned in estuarine areas. For this reason, an entire field of research has been stimulated in the past 20 years (Davidson & Pienkowski, 1987; Goss-Custard *et al.*, 1994, 1995a,b,c,d, 1996a,b).

As competition between individuals of the same species[1] is the factor that ultimately leads to carrying capacity being reached, it is important to study the mechanisms through which competition can operate. Moreover, in many cases, its detection may constitute the first, and often the only accessible, clue for determining whether an area is approaching carrying capacity.

According to Goss-Custard (1980), intra–specific competition may depress intake rate in two ways – interference and depletion. Interference may be defined as the depression of intake rate caused by the presence of other birds. The effect is reversible and operates in the short–term, as the density of available prey usually recovers quickly after the birds leave the area. Depletion, on the other hand, is the decrease in the food stocks due to predation. It takes more time to operate – usually some months in waders – but is irreversible, at least until the prey densities are re–established through growth and reproduction in the following spring and summer (Goss-Custard, 1980). While depletion is a function of predation pressure only, interference may arise through a variety of mechanisms, all of which lead to the reversible depression in the individual intake rates. Those already known include encounters over food, kleptoparasitism, disturbance in searching, exploitation of available prey, depression of prey availability and increasing use of poorer areas (Goss-Custard, 1980).

In practice, the existence of competition among foraging waders has been tested by investigating the occurrence of both interference (in at least some of its forms) and depletion (Goss-Custard, 1977a, 1979, 1980). Winter has been chosen as the appropriate season to study competition in the temperate latitudes. During this season, food resources are usually low due to the lack of recruitment and/or immigration in most prey species and to the low availability of most prey, as a direct consequence of the prevailing bad weather conditions (Evans & Dugan, 1984). Furthermore, it is in winter that the bird numbers reach their highest values in most estuaries (Smit & Piersma, 1989), and the birds' energetic requirements are greatest, due to increased thermoregulation costs (Evans, 1976; Piersma, 1996).

In the estuary of Mondego, any tendency for the increasing loss of the salinas over the last decade to be continued (see Chapter 4) may eventually cause birds that normally feed there to move to the intertidal mudflats. The effect that this would have on the estuarine populations of many species will depend on the levels

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[1] Inter–specific competition, although it certainly occurs, is likely to be of less significance in waders, as most species are ecologically segregated, largely preying upon different prey species or size classes (Baker & Baker, 1973; Baker, 1977).

of competition in the flats that would then occur. Accordingly, the objective of this Chapter was to test for the occurrence of intra–specific competition in the intertidal area of the Mondego estuary, through both interference and resource depletion, in order to make a preliminary assessment of the effects on wader populations of the loss of the supratidal feeding areas in the estuary.

5.2 Study Area and Methods

5.2.1 Study area

The study was undertaken on the 134 ha of intertidal mudflats in the south arm of the Mondego estuary. In spite of the small area, there was an appreciable variation in the type of sediments present. In general, the flats adjacent to the margins of the south arm of the Morraceira Island were made–up of finer sediments, while the central banks in the estuary consisted mainly of coarse–grained sediments (sand or muddy–sand, see Chapter 2) (D.N. Duarte, pers. comm.). In practice, it was possible to subdivide the estuary into three discrete areas (A1, A2 and A3, Figure 5.1), naturally delimited by channels and creeks. A dense *Spartina* spp. marsh was present in the margin of the island, while sparse meadows occurred downstream, in area A1.

Due to the small width of the south arm, the exposure time did not vary too much between and within the three zones: between 0.5 hour to 1 hour from A1 to A3, in an ‘average’ – 0.8 m – tide (J. Cabral, pers. comm.). Chapters 2 and 6 furnish more details on the prey populations and eutrophication of the estuary.

5.2.2 Methods

Interference

Selection of the method used With the exception of increased use of the poorer areas, all the interference mechanisms reviewed by Goss-Custard (1980) require the measurement of intake rates. Unfortunately, this was not possible in the present study as the species concerned mainly ate prey too small to be identified, or even to be seen being swallowed (Múrias, 1993; Cabral, 1995; Lopes *et al.*, 1996). Therefore, the chosen test for interference was based on the expected redistribution of birds over the feeding areas when the total number of birds on the estuary rapidly increased.

The hypothesis of interference competition predicts an increased use of the poorer ones, as total bird numbers – and thus interference competition – increases. As densities increase, the *total densities* of birds in both the preferred and non–preferred feeding areas may grow. However, if competition does indeed occur, the less competitive birds will be displaced from the more profitable feeding areas, redistributing themselves among the less used (and poorer) ones. The *proportion* of birds in the preferred areas, relatively to the total number present, will then

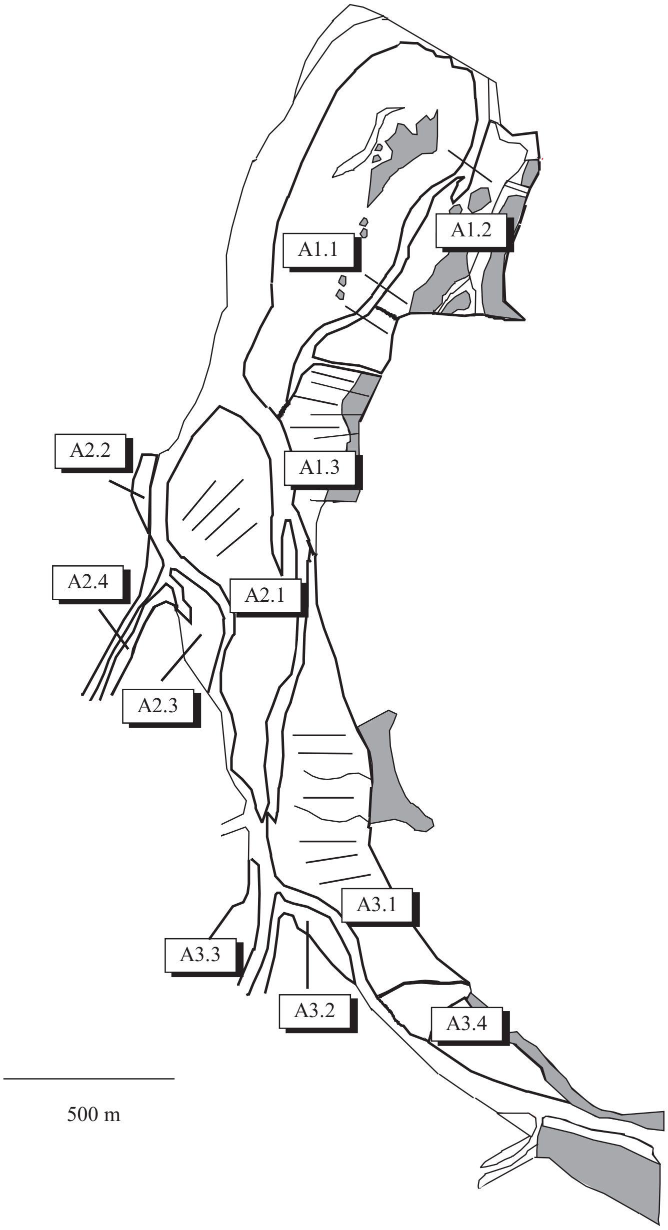

Figure 5.1: Map of the study area in the south arm of the estuary. The sub–areas considered are shown, as well as the transects used for sampling the macroinvertebrates

decrease while the proportion in the non–preferred ones will, correspondingly, increase.

Therefore, the occurrence of interference competition can be shown by plotting the change in densities and proportion of birds in both preferred and non–preferred areas against rapid changes in the total population size.

This method only requires counts of birds over short periods, during which total population size increased and prey depletion can be reasonably assumed to be minimal. Given the preliminary nature of the study, as well as the short time available, this option was considered to be adequate as a first attempt.

Field work Wader counts were performed monthly, from October 1993 to January 1994, and fortnightly, from February 1994 to May 1995, with 10×50 binoculars and a 30–90×50 telescope from three fixed positions around the mudflats (Figure 5.1). All counts were made within two hours of low–water on spring–tides because (1) all the main feeding areas were then exposed, allowing the birds to freely choose among them, and (2) very few movements of birds between mudflat feeding areas, or between mudflats and the salinas, a potential source of bias, were observed at this tidal phase (Chapter 3). All waders seen were identified, counted and plotted on a map of the mudflats, drawn from aerial photographs. Within two days, the numbers of birds present in the Morraceira's salinas (the most important supratidal area) were also counted (see Chapter 4, for more details).

It was assumed that all the birds present fed in the areas where they were observed, even if they were not feeding at the very moment they were counted. This is likely to be the case, because few birds usually rested on the mudflats at dead low–water. In contrast, at high–water in the salinas, only those mudflat birds that were feeding were considered because, many birds used the area for roosting at this period of the tidal cycle (see Chapter 4). The inclusion of the roosting birds in the analyses could thus have biased it.

Depletion

Difficulties in measuring food depletion At present, there are no absolutely satisfactory methods with which to measure the effect of waders on their food resources. Baird *et al.* (1985) reviewed the four main methods available. Two focus on temporal changes in the density of prey, and the other on daily food requirements of the birds.

They are (1) direct measurement of the changes in the food abundance before and after a period of supposedly intense predation by birds (e.g. winter); (2) use of exclosures, in conjunction with open control areas, to keep out avian predators and compare the changes in food abundance within each; (3) direct estimation of the food consumption of the birds, through measurements of the intake rates, the number of birds feeding in the area, and the proportion of time feeding, which are multiplied by each other and compared to the standing stocks of prey; and (4) calculation of the food consumption of birds by means of allometric equations relating basal metabolic rate (BMR) to body mass (e.g. Kersten & Piersma, 1987), multiplied by a factor of 3.5–5 to allow for the free–living energetic demands from thermoregulation and daily activities (feeding, flying, roosting etc.) (Pienkowski *et al.*, 1984; Baird *et al.*, 1985).

All methods have advantages, but also serious drawbacks (Goss-Custard, 1980; Baird *et al.*, 1985). For example, simply measuring the densities of the prey before and after a period of predation by waders, without using exclosures [method 1] may introduce bias due to the movements of the mobile prey in and out the area. Similarly, the use of BMR, in method 4, can be suspect because the multiplying factor vary to a certain extent with season and with latitude (Baird *et al.*, 1985; Zwarts *et al.*, 1990; Castro *et al.*, 1992). Therefore, the safer procedure is to combine two or more methods in order to get a number of independent estimates (Goss-Custard, 1977a).

Selection of the methods used Due to the difficulties of accurately estimating the intake rates of the species studied, no attempt was made to use method 3 in the present study. The second approach considered was to use exclosures. This method has been successfully used in several situations where the predation pressure was thought to be high due to the large number of birds present, either in intertidal (e.g. Goss-Custard, 1977a; Quammen, 1981; Botton, 1984; Wilson, 1991; Thrush *et al.*, 1994), or in non–tidal (Székely & Bamberger, 1992) sites.

However, as some authors (e.g. Goss-Custard, 1980; Baird *et al.*, 1985; Hall *et al.*, 1990; Thrush *et al.*, 1994) argue, exclosures may introduce some bias into the results through the action of potentially confounding factors. The most significant of these factors are: (1) the mobility of some prey, which may cause the difference in the changes in prey abundance in the exclosure and control areas to be reduced; (2) the alteration of the microhabitat inside the exclosures (sediment, water flow, growth of green algae), which may attract many invertebrates, and (3) the increase in predation pressure by epibenthic predators other than the birds (invertebrates, fish) that move into the exclosures to exploit the locally high densities of prey caused by the reduction in bird predation (Székely & Bamberger, 1992).

Moreover, as Kalejta (1993) demonstrated on a mudflat area of the Berg River estuary (South Africa) , the exclosure method may fail to detect the effects of quite heavy prey removal because of high variability between prey samples. The use of exclosures in the present case – where bird numbers were relatively low – would have required large number of samples in order to ensure a statistical validation of the results (D. Raffaelli, pers. comm.), a very time–consuming task, which was beyond the resources available. As a result, the exclosures method was abandoned. Instead, only the densities of some prey at selected points were measured before and after a period of (supposedly) intense predation.

In most estuaries the food resources decline during the winter, as no recruitment usually occurs during this period. Consequently, any increase in the number of birds in an area will lead to an increase in the rate of prey depletion and thus, perhaps, to a decrease in the individuals' rate of intake, so that the competition between them will increase (Goss-Custard, 1977a, 1979, 1980, 1984). In general, for sedentary or essentially immobile prey, the percentage depletion of prey is likely to be higher where the initial stocks of prey are high than when they are low, because the predators congregate where prey are abundant, i.e. the predation rate is density–dependent (Goss-Custard, 1977b). Since waders also usually aggregate where the densities of their prey are highest (e.g. Goss-Custard, 1970), the decreases in their prey will be greatest in such areas. To assess prey depletion in such circumstances, overwinter loss must be related to initial density in each site.

But as there are predators of the macrobenthos other than waders – such as fishes – it is advisable to check whether waders are likely to be the major cause of any spatially density–dependent prey loss over the winter. For this, method 4 was also used. This method is based on the comparison of calculated estimates of the bird populations' total energy requirements and the total energy of the main prey available in the feeding areas at the beginning of the winter. Assuming that no or little growth or recruitment of the prey occurs in winter (McLusky, 1989), the proportion of the standing stocks of prey depleted by overwintering waders may be assessed.

Although this method may under–estimate the real food intake due to, e.g., difficulties in measuring nocturnal feeding (Goss-Custard, 1984) it may, nevertheless, provide a useful and independent clue as to the magnitude of the depletion due to waders. It also provides a useful additional test of the significance of predation in the event that the overwinter prey loss proves to be density–independent.

Field and laboratory work In October of 1994, a series of transects were marked out in each of the three main feeding areas. The number of transects was a compromise between the need to cover the most significant parts of each area for waders while keeping sediment and exposure time as homogenous as possible, and the logistic and human resources available. Six transects were placed in area A1, four in area A2 and six in area A3 (Figure 5.1). Each transect contained an average of five, regularly spaced sampling points, except in area A2 were, due to the characteristics of the terrain, the average number of sampling points per transect was 8. Points were marked with bamboo stakes. A total of 90 sampling points, 30 per area, was used.

In late November and in late March, one sample was taken at each sampling point in each transect, with a small core of 95 cm^2, up to 5 cm deep. This was considered to be the maximum depth to which Dunlin, the numerically most important wader could forage (Kelsey & Hassall, 1989; Mouritsen & Jensen, 1992; Zwarts & Wanink, 1993)[2]. Because many mobile prey (e.g. large polychaetes such as *Nereis*

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[2] As to the other important species, the three plovers (Kentish, Ringed and Grey Plover) capture the prey at the surface (Pienkowski, 1981, 1982, 1983), the Avocet (only numerically important in winter) scythes the mud with the bill at roughly the same depth (Moreira, 1994c), and the long–billed species (the two Godwits *Limosa* spp., the two *Numenius* spp , which usually forage at higher depths, were not numerically important.

diversicolor) can burrow deeper than 5 cm, the sampling method may have introduced a potential bias in the results, unless the small–sized individuals taken by Dunlin spend all this time near the sediment surface, as may be the case. However, as a precaution, the analysis was limited to the gastropod *Hydrobia ulvae*. Besides being one of the most abundant and consumed prey of most species (Chapter 2), *Hydrobia* is usually found in the top 2 cm of sediment, with most individuals lying in the first 2 mm (Mouritsen & Jensen, 1992).

The samples were taken to the laboratory, washed and sieved through a 0.5 mm mesh and stored in 4% neutralized formol. The collected organisms were later separated, preserved in 70% alcohol and identified and counted under a binocular microscope.

The sieving procedure used was able to catch most of the animals present but the smallest individuals (i.e., in the present case, less than 0.73 mm). According to Marques (1989) more than 90% of all the benthic individuals of most species were caugth in this way. As such, it was considered that the proportion of individuals not caught was negligible and unable to strongly influence the results.

Simultaneous but independent data on invertebrates densities were collected fortnightly in three representative areas of the estuary by Pardal and co–workers (M. Pardal, unpubl. information) between January 1993 and June 1994. On each occasion, six to ten replicates were taken with a core (141 cm^2) to a depth of 20 cm. The samples were treated according to the method described above, except that no measurement of individuals were made. This provided a useful supplementary source of data.

5.2.3 General data analysis

Interference

Counts A total of 30 counts on the mudflats and in the salinas was available from the period of late July–May in 1993–94 and 1994–95. However, due to the lack of autumn and some early winter counts in 1993–94, only data of the second year were used. This provided a total of 20 counts, from late July 1994 to May 1995. Some additional counts made on consecutive days in September and October of 1994 were also considered.

Selection of preferred areas Goss-Custard *et al.* (1982) operationally defined preferred areas in two ways. They are those areas which (1) are occupied first, as bird numbers increase from late summer with the arrival of the wintering birds, and/or which (2) attract high densities of birds at all times. In the Mondego, numbers fluctuated a lot between late July and December–January, corresponding to brief passage of migratory birds in August–September, followed by a general decrease in late September–early October and by the main winter build–up phase, from late October to early January (see Chapter 3).

Perhaps because of these variations, and of the relatively small number of counts for each of the above periods (3–5 counts), the occupation of the feeding areas proved to be rather erratic. Moreover, the identification of discrete feeding areas proved to be quite difficult as the wader species considered preyed upon widely distributed prey (e.g. *Hydrobia* or small polichaetes) and so ranged widely themselves. Therefore, preferred areas could not be well described, in the present circumstances, by the first operational definition of Goss-Custard *et al.* (1982).

However, some areas were found to be, on average, more used by the birds than others. This was in agreement with the second operational definition of preferred areas Goss-Custard *et al.* (1982). In view of this, preferred areas were defined, in this study, as *those areas that, in average, attracted higher densities of birds over the entire study period.*.

Redistribution over the feeding areas In order to study redistribution over the feeding areas as the population increased, the three main areas of the estuary (A1, A2 and A3) were subdivided into several sub–areas on the basis of natural features, such as creeks (Figure 5.1). Whenever possible, each sub–area consisted of an homogeneous sediment type. Sediment homogeneity was easy to obtain for most areas, apart from the uppermost central flats (area A1.1, see Figure 5.1) where there was a mixture of sandy and muddy–sand flats. However, further subdivision of this area on the basis of sediment type did not reveal any tendency of the birds to distribute according to a given sediment type, partly because of the imprecisions in the location of both the flocks and the sediments inherent to the mapping method (see below). Therefore, only one large area of mixed sediments was considered, this being area A1.1. A total of 11 sub–areas (3 in area A1 and 4 in each of the other main areas) was thereby obtained. The size of the areas varied between 27.1 ha and 0.7 ha (average 10.02±11.8 (SD) ha).

The maps of the bird flocks recorded in each visit were later superimposed on the map of the sub–areas, allowing the allocation of each flock (or part of it) to a given sub–area to be made. Whenever a flock spread over several sub–areas, the number of birds composing it was assigned to each sub–area in proportion to the respective area occupied by the flock.

All the analyses were restricted to the three species most likely to be affected by interference due to their high abundance and widespread distribution, both in space and in time: the Kentish Plover, the Ringed Plover and the Dunlin. More details on the methods used will be given when appropriate in the Results.

Depletion

Overwinter depletion The mean densities (individuals.m^{-2}) of *Hydrobia ulvae* of the 30 samples in each area were used as the measure of prey abundance in each of the three areas (A1, A2 and A3) in early winter (December), before the main arrival of the wintering birds, and in early spring (March), after most wintering birds had gone, but before the arrival of the spring migrants. When necessary, data were divided into two groups, roughly corresponding to the small and presumably less profitable size classes – <1.5 mm – and the larger and presumably more profitable size classes – >1.5 mm – of this prey (see Goss-Custard *et al.*, 1991, for a review of this subject).

Food requirements of birds For measuring the food requirements of birds, the main prey species known to be taken by waders were selected (see above) and data on their biomass densities in early winter (November) of 1993–94, were supplied by M. Pardal (Pardal *et al.*, unpublished). The prey species and groups used were the gastropod *Hydrobia ulvae,* the polychaete *Nereis diversicolor* and all the combined species of crustaceans and polychaetes (other than *Nereis*). In all cases, it was assumed that all the size classes were taken by waders. While this was probably true for the small polychaetes (Goss-Custard *et al.*, 1991), this assumption may have over–estimated the energy available from the other prey species/groups, as only the larger size classes are usually consumed by the birds (Pienkowski, 1982; Goss-Custard, 1984; Piersma, 1986; Goss-Custard *et al.*, 1991; Velásquez & Navarro, 1993; Moreira, 1994a,b). Nevertheless, these values can be used as a measure of the total gross energy available to waders in the mudflats (Evans *et al.*, 1979)

The total gross energy present (TGE) on a given date for the whole area of mudflats (in kJ) was calculated as:

$$TGE = B \times A \times 5.25 \times 4.19 \tag{5.1}$$

where B is the average biomass density (g AFDW.m^{-2}) of the prey species, according to the data obtained from Pardal et al (unpublished), and A is the total area of the mudflats (134 ha). The value of 5.25 represents the average caloric content (kcal.g AFDW^{-1}) of a sample of invertebrate groups (Goss-Custard, 1979), and the value 4.19 is the equivalent energetic value (in kJ) of 1 kcal.

Only the gross energy requirements of the birds were calculated, as these determine prey loss. The birds' gross energy requirements were calculated for the total bird–days (BD) present during the period considered (November to March of 1993–94), calculated for each species as follows:

$$BD = N \times 30 \tag{5.2}$$

where N is the sum of the average monthly number of birds present, calculated from the mean of all the counts in each month, and 30 represents the average number of days in a month.

The daily gross energy requirements, or DEE, were calculated for each species by two methods:

(1) as a multiple of the Basal Metabolic Rate[3] (BMR), the traditional approach

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[3] Defined as the amount of energy consumed by a resting, non–absortive bird in a thermoneutral environment (Kersten & Piersma, 1987; Pienkowski *et al.*, 1984; Piersma, 1996).

(Pienkowski *et al.*, 1984) where:

$$DEE = n \times BMR, \tag{5.3}$$

in which $BMR = 437 \times BM^{0.729}$ (Kersten & Piersma, 1987), where BM is the body mass and n is a factor that represents the extra energy expended on thermoregulation costs and the normal daily activities (feeding, roosting, preening, flying), estimated from measured daily energy budgets for several species (Pienkowski *et al.*, 1984; Kersten & Piersma, 1987). In the present study, n=2.5 following the value provisionally used by Moreira (1995) for the Portuguese Tagus estuary, assuming that the birds assimilated 85% of the energy ingested (Kersten & Piersma, 1987).

(2) through the equation derived by Nagy (1987, in Turpie & Hockey, 1993), in his revision of published studies using the labelled water (D_2O) method, the Field Metabolic Rate (FMR):

$$FMR = 10.9 \times BM^{0.604} \tag{5.4}$$

where FMR represents the gross daily energy expenditure of a bird involved in 'normal' activities, as above. This value was corrected for assimilation efficiency in a similar way to method 1 by dividing FMR by 0.85. The values of BMR, DEE and FMR are expressed in kJ.bird.day^{-1}.

Both methods have been widely applied, but Nagy's method has the advantage of avoiding the need to use the multiplicative factor required by the traditional method of calculating the DEE. This is particularly important in this case, because no published estimates of the multiplicative factor of BMR exist for any south temperate European estuary[4]. As the energy expenditure usually decreases with latitude (Klaassen *et al.*, 1990; Castro *et al.*, 1992), the use of the multiplicative values derived to the north temperate estuaries is likely to introduce a bias when uncritically employed (Batty, 1991).

The total gross biomass consumed (TGB), in kJ.bird.period^{-1}, for each species was then calculated as:

$$TGB = BD \times grossDEE \tag{5.5}$$

or

$$TGB = BD \times grossFMR \tag{5.6}$$

in case (1) and case (2), respectively, where BD is the number of bird–days, calculated as above.

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[4] In fact, Batty (1991) has calculated these factors for three species (Dunlin, Kentish Plover and Redshank) in Ria Formosa, Algarve (south Portugal). However, the climatic conditions in Algarve are unique in the country, and the results, therefore, should not be extrapolated to the more northern estuaries and rias of Portugal, including the Mondego.

5.3 Results

5.3.1 Interference

Redistribution of birds on the feeding areas

In order to test for the redistribution of the birds over the preferred and non–preferred feeding areas as total numbers increased during the autumn/winter build–up phase, the numbers found in each count, from late July 1994 to January 1995 in all preferred areas were added and compared to the total numbers present on all the mudflats for the same count. The same method was applied to the non–preferred areas, excluding those where no birds were ever recorded. The proportion of birds present in each type of area, in relation to total numbers was then calculated and plotted along with numbers in the same graph. Because of the lack of independence between the variables in both axes, no statistical analysis could be performed.

The number in both the preferred and in the non–preferred areas generally increased as the numbers in the estuary increased (Figure 5.2). However, the plots of the proportions gave no indication that the proportion of birds in the preferred areas declined as total numbers increased. The present results, thus, did not present any clear evidence that the birds' redistribution as total numbers increased occurred in a manner consistent with the interference hypothesis.

Alternative approaches to testing for interference

The failure to demonstrate the presence of interference competition through changes in their redistribution over the feeding areas as total densities increased, may indicate either (1) that *the Mondego estuary has unusually low bird densities and is thus free from interference competition*, or (2) that *competition exists but it is weak and/or difficult to demonstrate*, or simply (3) that *all areas are equally preferred*. These hypothesis were tested in turn.

Hypothesis (1). Prater (1981) showed that the mid–winter peak numbers of waders in British estuaries were positively correlated with the size of the estuaries. It is possible that bird densities on the Mondego are so low that the area is free of interference and, indeed, other forms of competition. On the assumption that the general relationship described by Prater (1981) is valid outside Britain, and that the mid–winter numbers are the best approximation to the maximum numbers that occur on a given estuary (Prater, 1981; Smit & Piersma, 1989), the possibility that bird numbers (and hence, densities) were unusually low in the Mondego was tested.

The mid–winter numbers of Kentish Plover, Ringed Plover and Dunlin on the intertidal areas of the Mondego estuary were compared with those of other British, Dutch, French and Portuguese estuaries[5]. In order to avoid any possible biases due to the inclusion of unsuitable feeding areas in the calculations, a possibility that is

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[5] In the case of Kentish Plover, only the Iberian and north and west African sites were used, due to the more southern distribution of the species in winter.

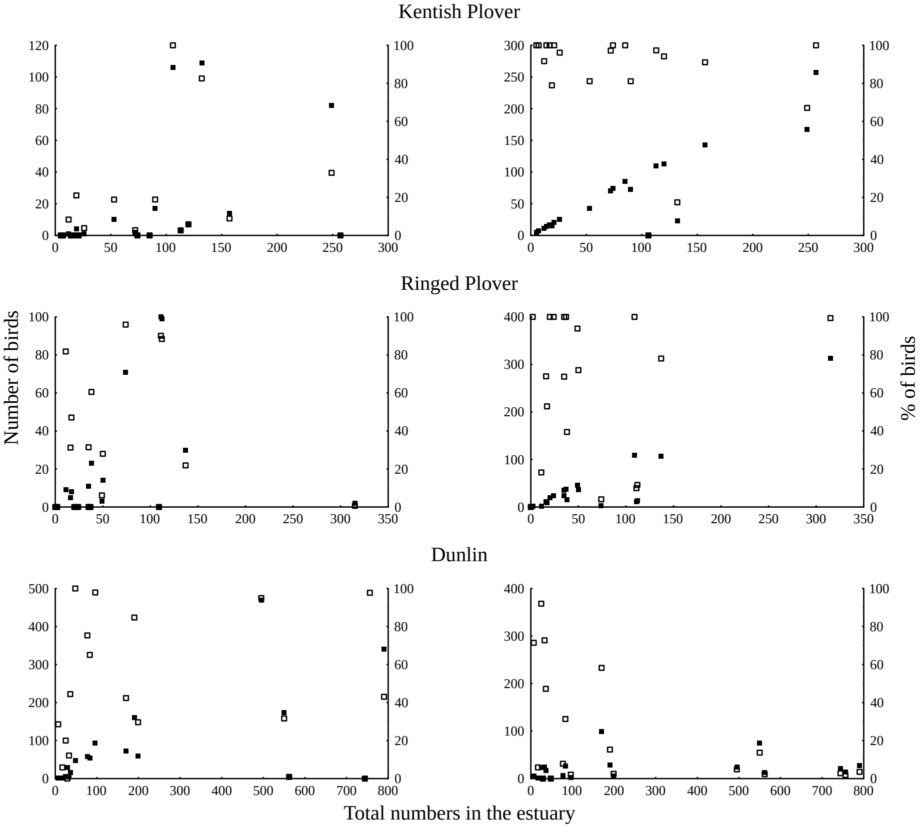

Figure 5.2: Number (closed squares) and percentage (open squares) of birds occurring in a preferred (left column) and no–preferred (right column) feeding area as a function of the total number of birds in the mudflats for three selected species of waders (from top to bottom, Kentish Plover, Ringed Plover and Dunlin).

more likely to occur with the increasing size of the estuaries (Prater, 1981), only the intertidal areas of medium–size (100–2 000 ha) estuaries were used[6].

These areas are thought to represent the main feeding sites on which interference competition usually occurs. Data for British estuaries were taken from the counts of the Wetland Bird Survey for 1993–94, published by the British Trust for Ornithology, and from Smit & Piersma (1989). Data for the other European and North African estuaries were extracted from Smit & Piersma (1989), and for the Portuguese areas, from Rufino (1991, 1992), Rufino & Costa (1993), and Costa & Rufino (1994). All data were log–transformed.

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[6] Except in the case of Kentish Plover were the lack of sites forced to include the Banc d'Arguin (180 000 ha) in the analysis

Positive correlations between bird numbers and the estuary size were obtained for Kentish Plover and Dunlin, as expected (Figure 5.3), although not for Ringed Plover.

This allowed the numbers of the Mondego's population of Kentish Plover and Dunlin to be compared with the average to be expected for an estuary of its size (regression line). In both species, the Mondego estuary fell close to the regression line, suggesting that the numbers of the two species in Mondego were in the range expected for an estuary of its size.

But although bird numbers (and thus, probably, densities) of many species in the Mondego were quite typical for the flyway, food may have been so unusually abundant that little competition occurred there. This was tested by comparing the abundance of food in the Mondego with that in other estuaries over the East Atlantic Flyway.

Food abundance is likely to vary with the area of the estuary (Prater, 1981) but also with latitude. Although this seems not to be an absolute rule (see Piersma *et al.*, 1993a), some evidence of a north–south latitudinal gradient of increasing food abundance[7] does exist (Kalejta & Hockey, 1991; Hockey *et al.*, 1992), and it has been attributed to the more favourably higher temperatures in the more southern sites.

Therefore, food abundance (in grams of ash–free dry weight.m^{-2}) in the Mondego and other European and African estuaries were regressed, in a multiple regression analysis, against estuary size and latitude in relation to the equator, over the a range of 57^{o} N to 34^{o} S. Data on estuary size, latitude and total invertebrate biomass on European and African estuaries of the East Atlantic Flyway were obtained from Kalejta & Hockey (1991), Hockey *et al.* (1992) and Piersma *et al.* (1993a). For the Mondego, data given in Múrias (1993) for two representative areas of the estuary were used, excluding *Scrobicularia plana*, as this species was not an important prey for the wader species considered (Lopes *et al.*, 1995).

There was no relationship between either estuary size (partial r=-0.2, ns) or latitude (partial r=-0.18, ns) with food abundance, when both dependent variables were analysed in a multiple regression analysis (r^2=0.03, $F_{2,8}$=0.11, ns, n=11) (Figure 5.4)

It is possible that these results may have arise in part from differences in field methodologies and sampling periods, as illustrated in Table 1 of Piersma *et al.* (1993a). However, a visual inspection of data plotted in Figure 5.4 shows that the Mondego was consistently located among the main cluster of points, whether these were plotted against estuary size or latitude. This suggests that the food supplies in this estuary are not different from other European sites. As both bird numbers and food abundance on the Mondego seem to be quite typical, the hypothesis that the Mondego estuary is a competition–free area seems to be quite improbable.

Hypothesis (2). As bird numbers and also probably food abundance were quite typical in the Mondego, competition may have existed but it may have been too

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[7] Taking food abundance as equal to invertebrate production (Hockey *et al.*, 1992).

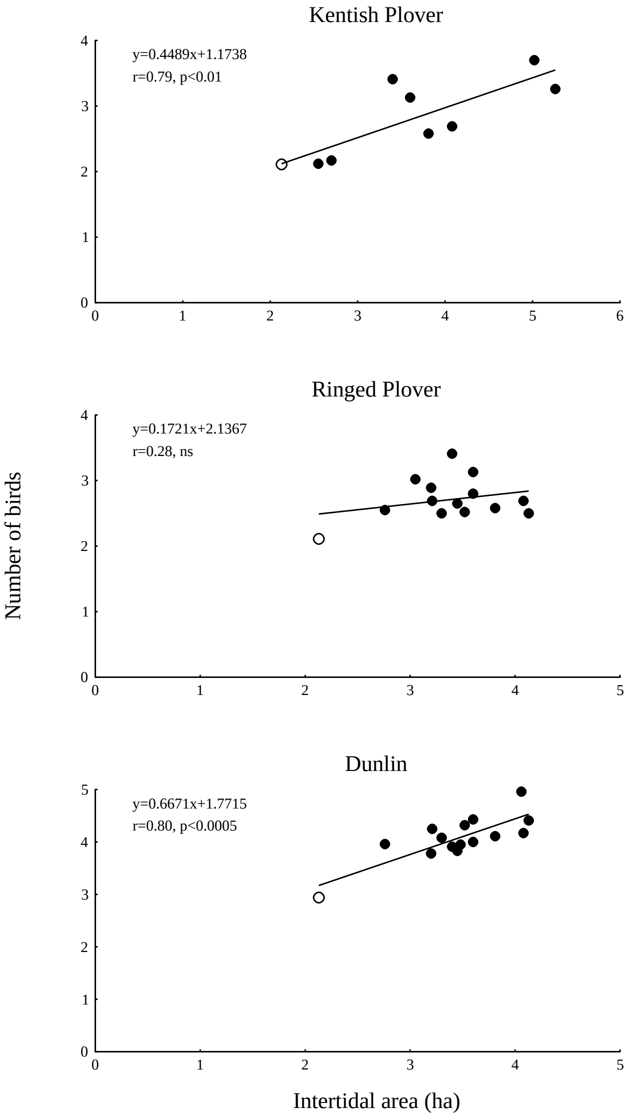


Figure 5.3: Relationshiop between the number of birds and estuarine intertidal area for three species of waders. Estuaries up to 5 000 ha were selected in England, Portugal, Netherlands and along the west African coast up to Guinea–Bissau, depending on the species. Open dot indicates the Mondego. All data are log–transformed (log n+1). See text for details.

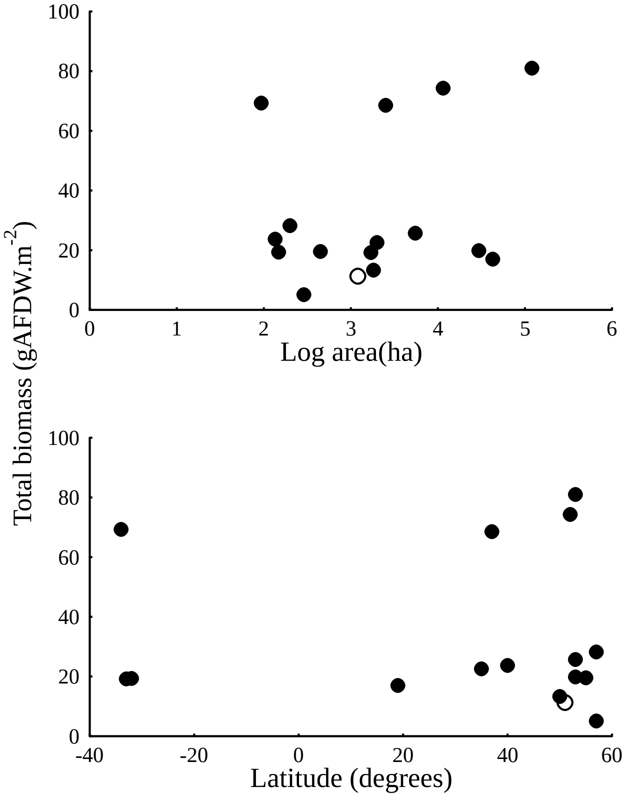


Figure 5.4: Relationship between total prey biomass (g AFDW.m^{-2}) and estuarine area (above) and latitude (below). Data are for european (German, Dutch, British and Portuguese) as well as for African (Moroccan, Mauritanean and South African) estuaries. In both cases the correlations were not statistically significant (r^2=0.03, $F_{2,8}$=0.11, ns, n=11; estuary size: partial r=-0.2, ns; latitude: partial r=-0.18, ns). Open dot indicates the Mondego position.

weak to be demonstrated by the traditional method of assessing the sequential occupation of the feeding areas. If so, an alternative analysis was needed to test for competition in this area. This was done by studying the change in the use made of the alternative feeding and supplementary areas (the salinas) as the numbers of birds in the estuary increased.

The competition hypothesis gives the prediction that, as the overall number of birds increases in the estuary, an increasing proportion would be forced, by intensifying competition, to use salinas, either at low–water or high–water, or both.. This was tested with data from both low–tide and high–tide periods; in the later case, only the surplus of birds that moved to the salinas as the tide flowed (see Chapter 4) was considered. Only counts from spring and autumn (March to May and August to October, respectively) were used, as counts in the salinas in the winter may have been influenced by the restriction in the available feeding areas in salinas due to flooding in many of them. To allow for any differences between years, a multiple regression analysis was performed, including the year of study (1994 and 1995) as a 0/1 dummy variable. There was no evidence that the proportion of birds occurring in the salinas at low–water increased as the numbers in the estuary as a whole increased (Figure 5.5).

No trend was apparent for the proportion of birds in the salinas to vary in relation either to total numbers or to the year, in Ringed Plover (r^2=-0.06, $F_{2,14}$=0.56, ns) and Dunlin (r^2=-0.07, $F_{2,14}$=0.48, ns). Indeed, contrary to expectation, the proportion of Kentish Plover in the salinas decreased significantly (r^2=-0.51, $F_{2,14}$=7.26, $p<0.007$) as the total numbers present in the whole area increased (partial r=-0.43, $p<0.04$), although independent annual differences may also be involved (partial r=-0.48, $p<0.03$).

This may suggest that, instead of competition on the intertidal feeding grounds determining the numbers feeding in the salinas, the reverse applied. That is, increased competition in the salinas as numbers increased may have caused more birds to move to the estuary at low–water. However, the trend may not be related to competition on the feeding areas, but rather to the occupation of the breeding grounds in the salinas, as the high percentages of occurrence in this habitat were recorded in spring counts, independently of the year considered. To test this, the analysis was repeated for Kentish Plover, using the autumn counts for the 1994–95 season, as only one autumn count was available for the 1993–94 year. Again, a strong negative association was found (r=-0.82, $p<0.002$, n=7), suggesting that the negative association previously observed may be indeed associated to some form of competition for the salinas at low–water in Kentish Plover.

There was, therefore, no evidence to suggest that an increase in numbers in the estuary directly influenced the numbers that use the salinas at low–water. However, increased numbers may have had some effect in those birds that, although feeding in the mudflats at low–water, used the salinas at high–water. In fact, if these birds were to be affected by competition on the mudflats, they would be expected to use more the salinas at high water, in order to achieve their energy requirements. If so, numbers in the salinas at high–tide should increase as the overall numbers on the

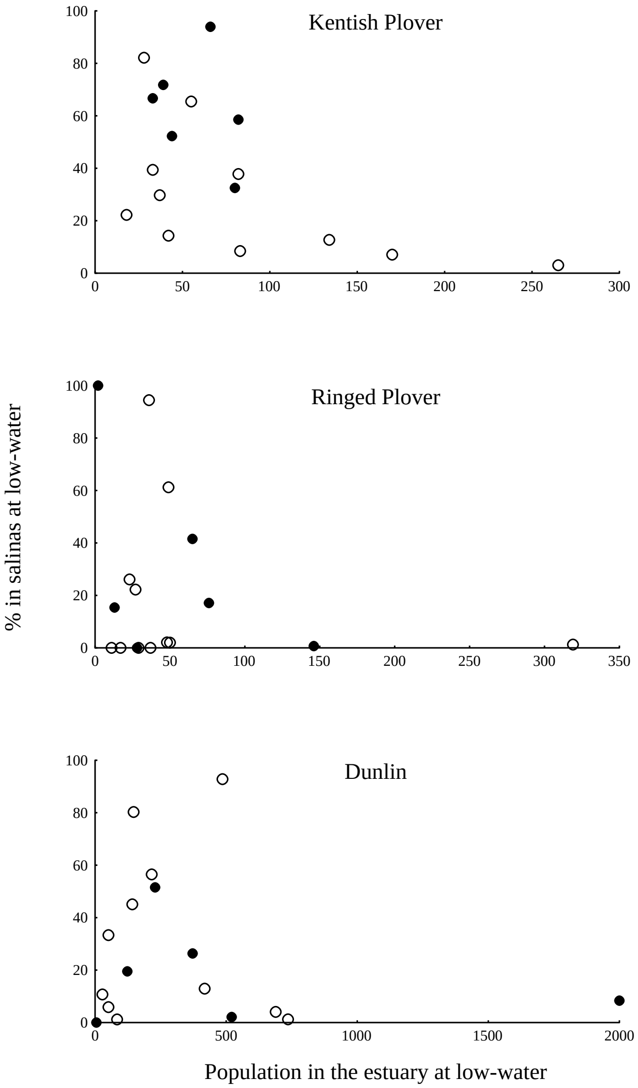


Figure 5.5: Relationship between the percentage of birds of selected species in the salinas at low–water and the total population of each species present in the estuary and salinas combined. Data are for the periods of March to May and August to September of 1994 (closed dots) and 1995 (open dots). A multiple regression on percentege of birds and year was performed. Results are statistically significant for Kentish Plover (r^2=-0.51, $F_{2,14}$=7.26, $p<0.007$; total numbers: partial r=-0.43, $p<0.04$; year: partial r=-0.48, $p<0.03$), but not for the other two species (Ringed Plover, r^2=-0.06, $F_{2,14}$=0.56, ns; Dunlin, r^2=-0.07, $F_{2,14}$=0.48, ns).

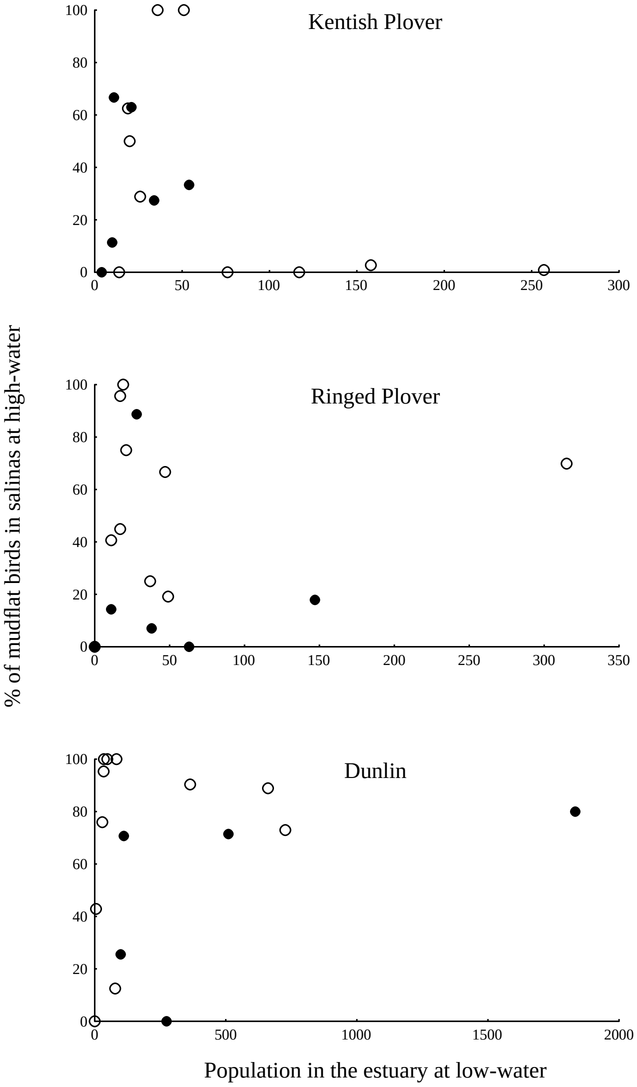


Figure 5.6: Relationship between the percentage of birds of selected species in the salinas at high–water and the total population of each speceis present in the estuary and salinas combined. Data are for the periods of March to May and August to September of 1994 (closed dots) and 1995 (open dots). A multiple regression on percentage of birds and year was performed. Results are not statistically significant for any species (Kentish Plover, r^2=0.05, $F_{2,13}$=1.42, ns; Ringed Plover, r^2=0.09, $F_{2,13}$=1.73, ns; Dunlin, r^2=0.09, $F_{2,13}$=1.73, ns).

Table 5.1: Relationship between the pecking rate of selected wader species and the density of birds around the focal bird. *$p < 0.05$.

Species	Foraging method	r_S	n	Method of measuring densities	Source
Kentish Plover	Visual	+0.24*	144	Direct measurement[a]	Cabral, 1995
Ringed Plover	”	-0.09	149	”	”
Grey Plover	”	-0.001	133	”	”
Dunlin	Visual/tactile	-0.12	135	Ranking of densities[b]	Múrias, 1993

[a] Birds in 1 ha plots.

[b] Rank order as follows: (1) – <5 birds around the focal birds; (2) – 5–10 birds; (3) – >10 birds.

estuary at low–tide increased. However, for none of the three species considered was there any obvious tendency for the percentage of low–water birds that fed in the salinas at high–water to increase as the numbers in the estuary at low–water increased (Figure 5.6).

These results are consistent with the hypothesis that the interference competition, if it existed, was too weak to be detected.

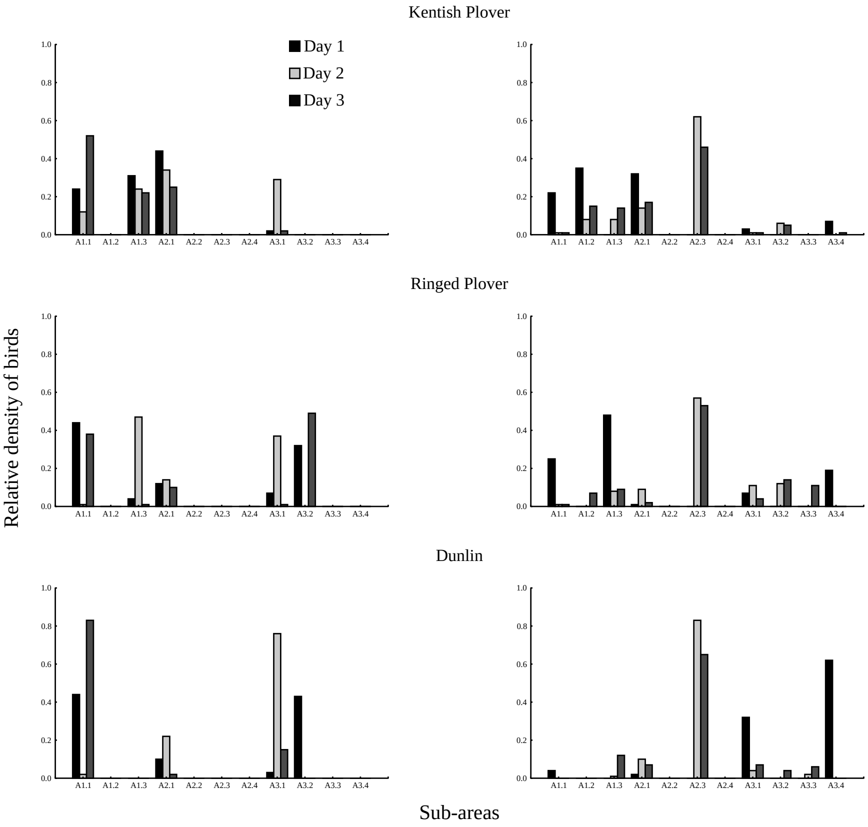
Hypothesis (3). Another possibility is that either all areas were non–preferred or they were all preferred, and the birds moved between them on a daily basis, thus making difficult to get a clear pattern of the usage. To test this, day–to–day variations in the use of the feeding areas were investigated for Kentish and Ringed Plover and Dunlin, using a serie of three consecutive counts in September (21, 22 and 23) and in October (7, 8 and 9) of 1994. All areas were considered, and relativized densities were used in each count.

Significant day–to–day changes in the relative densities of birds using the different sub–areas were observed in September and particularly in October for Ringed Plover and Dunlin, but less so for Kentish Plover (Figure 5.7).

As a whole, there was no clear evidence that the birds consistently chose the same areas in the short–term (i.e. over the three–days period), or in the medium–term (i.e. in the 15–days period between the September and October counts) although some areas were more often used than others. The present results, thus, reject the hypothesis that all areas were consistently equivalent, but they do not clearly show any tendency for some areas to be preferred to others, either.

Behaviour changes with competitor density

The above analysis suggested that competition levels in the estuary were low. Direct confirmation of this would be provided by studying the intake rates of waders as densities increased, but this was not possible. However, previous studies (Múrias, 1993; Cabral, 1995) measured some other parameters of feeding in selected species, over a range of densities. These data enabled the possibility that foraging behaviour was affected by bird density to be tested. A summary of the results is given on Table 5.1.



Kentish Plover

Ringed Plover

Dunlin

Figure 5.7: Daily change in the use of the sub–areas in the mudflats over three consecutive days in September (left column) and October (right column) by Kentish Plover (top, N_1=257, N_2=93 and N_3=150 in September, and N_1=120, N_2=91 and N_3=137 in October), Ringed Plover (center, N_1=32, N_2=5 and N_3=14 in September, and N_1=23, N_2=6 and N_3=4 in October) and Dunlin (bottom, N_1=33, N_2=23 and N_3=31 in September, and N_1=83, N_2=139 and N_3=230 in October). Values are relativised densities.

Densities were calculated in two different ways. Cabral (1995) measured true densities, as all observations and subsequent bird counts were performed in marked 1 ha plots. The study by Múrias (1993), on the other hand, only recorded relative densities, as this author counted the number of birds within a fixed distance (about 1 m, measured in bird–lengths) of the focal bird. Although the methods used to estimate densities were not the same for all species, the main conclusion that emerges from these studies is that the increase in densities seemed not to affect any of the foraging activities measured in any species and this conclusion was unaffected by the hunting method used by each species[8]

Conclusion

Overall, the tests of the interference hypothesis suggested that the competition through interference in the feeding areas of the mudflats either did not occur or was too weak to be demonstrated.

5.3.2 Depletion

Overwinter reduction in prey abundance

Overwinter reduction was found to be spatially density–dependent in the smaller size–classes ($<$1.5 mm) but, quite unexpectedly, not in the larger size–classes ($>$1.5 mm), which showed the exactly opposite trend to that expected (Figure 5.8). When both classes were combined, the density–independent nature of the overwinter change in prey abundance was even more clear.

The possibility, remained, however, that the growth of the smaller individuals into the larger size–classes replaced the losses due to wader predation, thereby preventing the detection of the expected density–dependent reduction in the prey densities of the adult (and hence profitable) *Hydrobia*. In fact, the overwinter reduction in the small–sized classes suggests that this could have occurred.

In order to test this hypothesis, the densities of all size–classes were compared over the two periods considered (i.e. early winter and late winter/early spring), along with the total densities. If the food had decreased through the winter, a significant decrease should be noted in the small–sized class (due to natural mortality and growth into larger individuals), in the larger–sized class (through mortality and predation) and, consequently, in all size classes combined, since no recruitment of *Hydrobia* took place in winter in the estuary (Baptista, 1997).

The results (Figure 5.9) do not support this hypothesis. In fact, they showed an increase in the large–sized classes by approximately the same amount that the small–sized animals decreased, while the total densities remained stable. This suggest either that (1) depletion by waders and other predators was slight or (2) new

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[8] Those species that use sight–hunting techniques (the plovers) are expected to be more vulnerable to the effects of interference, both due to the presence of the birds themselves, and due to the indirect effects in the prey behaviour (decrease in prey availability) (Goss-Custard, 1970).

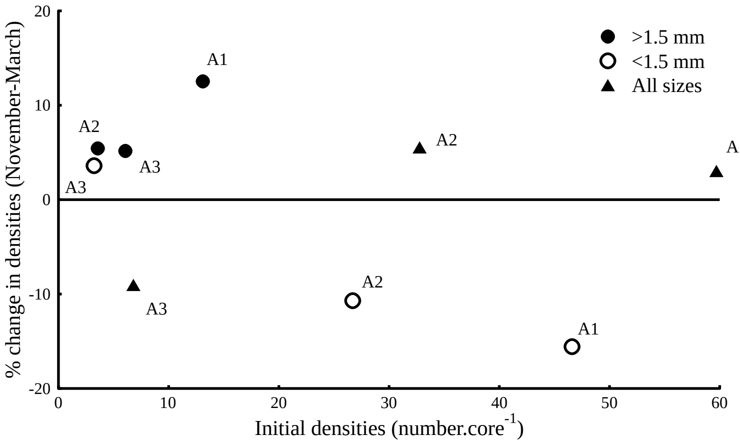

Figure 5.8: Percentage change in the densities of *Hydrobia ulvae* from December to March in relation to the initial densties present in early December, in three areas of mudflats of the Mondego estuary. Data are plotted separately for the class sizes >1.5 mm (closed dots)), which are predated by the birds, and for the class sizes <1.5 mm (open dots), which are not used as food resources, and for the two classes combined (closed triangles). Labels refer to the three areas (see text).

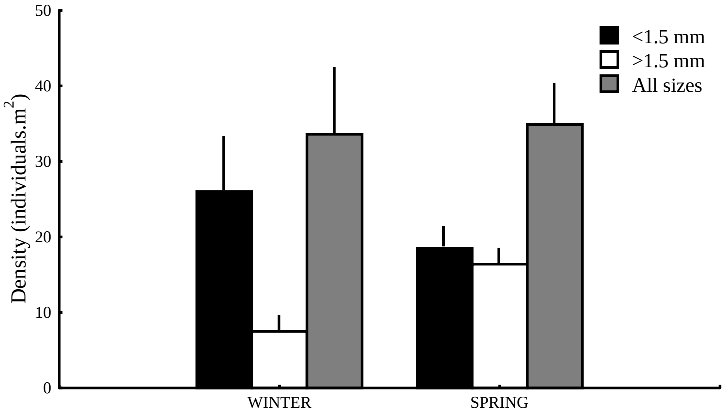

Figure 5.9: Average densities (±SE) of *Hydrobia ulvae* in each of the two size classes considered independently and all sizes grouped, in early winter and early spring.

Table 5.2: Total estimated gross biomass consumption (TGB) of the wader species present in the winter of 1993–94 in the Mondego estuary. TGB (1) is calculated as 2.5 x BMR (DEE), and TGB (2) is calculated from bird time budgets, as described by Nagy's formulae (Nagy, 1987). An assimilation value of 85% was used in both cases to account for non–metabolised energy (Kersten & Piersma, 1987). See text for details.

Species	Body mass (kg)	DEE ($kJ.day^{-1}$)	FMR ($kJ.day^{-1}$)	Bird–days	TGB (1) (10^6kJ)	TGB (2) (10^6kJ)
Kentish Plover	0.05	105	98	15930	1.67	1.57
Ringed Plover	0.05	105	98	18960	1.98	1.87
Grey Plover	0.19	277	220	22260	6.16	4.91
Numenius spp.[a]	0.56	609	423	690	0.42	0.20
Black–tailed Godwit	0.29	377	285	19410	7.31	5.52
Comm. Sandpiper	0.05	105	98	960	0.10	0.09
Knot	0.14	221	183	840	0.19	0.15
Dunlin	0.05	105	98	91200	9.54	8.97
Little Stint	0.03	72	72	1650	0.12	0.12
All species					**52.57**	**52.90**

[a] *Numenius arquata* and *N. phaeopus*. The values for these especies were averaged in all cases.

prey moved into the estuary from elsewhere, or (3) small individuals grew into large animals, replacing those lost. But in all cases, the net effect of birds' predation seems to have been low.

Energy requirements of the birds

This conclusion was reinforced by the analysis which compared the birds' energy requirements through the winter of 1993–94, with the energy values of the standing stocks of the main prey species present in November of 1993 (Tables 5.2 and 5.3).

Depending on the method used to calculate TGB (see section 5.2.3), the birds removed only some 4%–5% of the gross energy available. These low values were, of course, due to the overwhelming contribution of *Hydrobia ulvae* to the total energy available. If this species was excluded, the predation pressure in the remaining groups was very high (78% to 96%, according to the method used). Nevertheless, given the importance of *Hydrobia ulvae* in the diet of, at least, the most abundant species, it is reasonable to conclude that food was not heavily depleted over the winter.

5.4 Discussion

Many studies have attempted to assess the consequences of habitat loss for waders in their wintering or staging sites, but they have been more concerned with the role of food depletion than with that of interference competition (e.g. O'Connor & Brown, 1977; Evans *et al.*, 1979; Meire *et al.*, 1994). However, as several authors showed (Goss-Custard, 1977a, 1979, 1980, 1985, 1993; Goss-Custard *et al.*, 1994, 1995a,b,c,d, 1996a,b), the overall effects of competition in the feeding areas arises

Table 5.3: Energy potentially available to waders in November 1993 (TGE) and the percentage taken by the birds over the winter, given the estimated birds' consumption (TGB). (1) and (2) represent the estimates from the two methods of calculating TGB (see Table 5.2). See text for details on the calculation of the total energy.

Taxa	Densities (indiv.m^{-1})	Average dry weigth of soft parts (g)	Total biomass g AFDW) in (Nov. 1993)	Total energy (10^6kJ) in Nov. 1993	% consumed by waders by March 1994	
					(1)	(2)
Hydrobia ulvae	49991	0.00075	50.2×106	1105		
Polichaetes	3868	0.00015	0.8×106	17		
Crustaceans	1982	0.00024	0.6×106	14		
Nereis diversicolor	27	0.03000	1.1×106	24		
All taxa				**1160**	**4.7**	**3.7**

primarily from a close interrelationship between the two independent feed–back processes of interference and food depletion. In the Mondego estuary, an attempt to study both processes was made in this study.

The occurrence of redistribution could not be detected by the traditional method (Goss-Custard *et al.*, 1982), nor by alternative tests using bird numbers in the supratidal habitats. If interference did occur, its effects must have been operating only at low level, and so probably not affecting the intake rates of the birds. This is further suggested by the lack of relationship between pecking rates and increasing bird densities, measured in the Mondego estuary in some charadrids, for which pecking rates roughly correspond to the intake rates (Pienkowski, 1982). Indeed, in one case, (Kentish Plover), the pecking rate even increased with increasing densities (Cabral, 1995).

The overwinter reduction in the main food supply of *Hydrobia* was negligible. In fact, the abundance of the larger (>1.5 mm) size classes actually increased, while that of the smaller ones decreased almost the same amount. This could be attributable mainly to the effects of (1) the growth of the smaller animals into the larger size classes and (2) the low total energy requirements of the birds, as calculated for this estuary. According to these calculations, the birds would remove, through the winter, only 4%–5% of the total gross energy present in November, a value well below the typical percentage given by Goss-Custard (1984) for north temperate estuaries (22% to 45%).

The possibility that the estimate of the waders' impact on their prey was too low must be considered, though, as the invertebrate sampling carried out by M. Pardal and co–workers was designed specifically to cover weeded areas, or areas that would be covered by algae at some time in the year, where some invertebrate species reach very high densities. Some of the sampling areas used in their study, therefore, may not have been representative of the whole estuary. As an example, the densities of *Hydrobia* in the *Zoostera* meadows in November of 1993 reached some 80 000 individuals.m^{-2}, compared with the 30 000 to 44 000 recorded in the less weeded areas (Pardal *et al.*, unpublished). Indeed, if the area of *Zoost-*

era was excluded from the calculations, the total estimate of the energy present would decrease by 24.3% (from 1120 to 937 10^6 kJ). But even if this allowance for algae–covered areas is made, the impact of waders would be increased only 2%, compared to the original values. The estimated impact of the waders on their prey thus remains very low.

Without *Hydrobia ulvae*, the potential predation pressure on the remaining taxa present (total small polichaetes, plus *Nereis diversicolor* and total crustaceans) would increase to 78% to 96%, which would lie within the range of impact described by several other authors (Evans *et al.*, 1979; Baird *et al.*, 1985; Székely & Bamberger, 1992). Rates of prey depletion of this magnitude can translate into a depression of intake rate in the order of 5% to 25%, or more (Goss-Custard, 1984; Székely & Bamberger, 1992). Clearly, this could be detrimental to the waders that fed in the Mondego's mudflats. This underlines the importance of the abundant food supply provided by *H. ulvae*, which is the main prey of most species in the Mondego (Lopes *et al.*, 1995).

In summary, the present data indicate that competition through interference and depletion on the mudflats of the Mondego estuary must have been weak during the study period. In consequence, the mudflats were probably able to accommodate some hundreds of birds more, before food competition would intensify to the point at which some birds would die or leave the estuary. This suggests that, were the salinas to be lost, most birds would still be able to remain on the estuary simply by moving themselves to the mudflats, thereby buffering the loss of the salinas.

However, this conclusion must be viewed with some caution. The salinas are used either at low–water, as alternative feeding areas to the mudflats, or at high–water, as supplementary feeding areas (Chapter 4). High levels of competition on the mudflats would only affect the loss of space/feeding supply for the birds. However, not all the birds were able to meet their requirements only at low–water (Chapter 4). For whatever reason, and apparently not because of competition, some birds need to supplement their feeding in the salinas at high–water. These birds would be in trouble if the salinas were lost. Therefore, low levels of competition on the mudflats do not necessarily mean that this feeding area would be able to substitute all the feeding currently made on the salinas by both salinas birds and mudflat birds.

Furthermore, the loss of the salinas would also be detrimental to the breeding populations of at least two species. In fact, they are the exclusive or, at least, the most important breeding areas for Kentish Plovers and Black–winged Stilts (Rufino & Neves, 1991, see also Chapters 3 and 4). The complete loss of the salinas could bring about the disappearance from the area of the whole population, or part of it, of these species.

Finally, even if at the present the effect of competition seem to be low, it is not known by how much it would increase if all the salinas birds were displaced onto the mudflats. This would depend not only of the number of birds present, but also of the seasonal fluctuations in the populations of the potential prey and, perhaps more importantly, of their availability to the birds. Only long–term field studies combined with simulation analysis of the habitat loss could predict whether the complete loss of the salinas would reduce competition (see, e.g. Grant, 1981; Goss-Custard *et al.*, 1994, 1995a,b,c,d, 1996a,b).

Chapter 6

Short–term Effects of Intertidal Macroalgal Blooms on the Macrohabitat Selection of Waders

6.1 Introduction

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The intake rate of wading birds (Charadrii), can be affected by variations in prey abundance and accessibility (Goss-Custard, 1984; Zwarts & Wanink, 1993). The presence of thick layers of algae on the sediments due to eutrophication affect both of these aspects of the food supply of these birds over both the short–term and long–term (Raffaelli *et al.*, 1989). Short–term direct effects on prey abundance can be either positive or negative, depending on the species involved (Raffaelli *et al.*, 1989; Everett, 1994). Similarly, accessibility can be either increased, because the vertical migration of many species is impaired (Kalejta & Hockey, 1991), or decreased, because the mats can provide a refuge against predators on the surface (Everett, 1994). Indirect positive effects can occur through a general increase in the productivity of adjacent unweeded areas (Raffaelli *et al.*, 1991) and through an increase on sediment penetrability under the mats due to wetter substrates (Kalejta & Hockey, 1991).

Waders might respond to these changes in a variety of ways. In the short term, they could (1) only select for feeding either the unweeded or covered areas according to which is the more profitable, or (2) use alternative feeding habitats if their normal areas are rendered unusable by weed cover, or (3) change their foraging behaviour, depending on the foraging strategy they use. In the latter case, it might be expected the tactile–hunting scolopacids to be more flexible than the sight–hunting plovers, which rely on prey movement at the sediment surface to detect their food (Pienkowski, 1980). Here, the response of the birds will also depend on the way in which the presence of algae modifies the normal behaviour of the prey.

In the estuary of Mondego, where eutrophication has been monitored over the last eight years (Marques *et al.*, 1984, 1993b), both areas covered by macroalgae and areas free from algae can be found throughout the year. The situation thus provides a very good opportunity to study the short–term response of shorebirds to eutrophication. The aim of this Chapter was to discuss the way in which the presence of algae in the sediments affected selection of feeding habitats, as well as the feeding behaviour of waders. In view of the exploratory nature of the study, it was chosen in the later case to look only for gross differences in observable prey taken and in the rates of at which birds foraged. The adopted philosophy was that, if such simple behaviour data did not help to interpret any effects of algae on bird numbers, it would be possible to go on and to examine more subtle influences of algae on bird behaviour at a later stage.

6.2 Study area and methods

6.2.1 Study area

Fieldwork was performed on the 134 ha of mudflats of the south arm of the Mondego estuary, as well as in 21 salinas in the adjacent Murraceira island,from October to May, in 1993–94, and from August to May, in 1994–95. These represented 57% of the total number of pans and 52.2% of the total area they occupy. The south arm and these salinas provided the main low–tide feeding areas for the wading birds in the estuary (Múrias & Ferrand de Almeida, 1991). See previous Chapters, for a more detailed description of the area.

In recent years, periodic blooms of macroalgae (mainly *Enteromorpha* spp. and *Ulva* spp.) have been occurring in the Mondego, covering large areas of the flats. Usually, they last from late March to September/October (J.C. Marques, pers. comm.). Apparently, the mats are not restricted to a particular level, although they colonize muddy substrates more easily than the sandy ones.

6.2.2 Methods

Wader counts and behaviour

Monthly and, from February 1994 onwards, fortnightly[1] wader counts were made using 10×50 binoculars and a 30–90×50 telescope from three vantage points adjacent to the mudflats (see Chapter 2) and from the banks of a sample of salinas. Feeding birds were counted and, along with these of the algal mats, their positions were plotted on a map, drawn from published aerial photos. Therefore, it was possible to assign each bird as to whether or not it was on sediments that were covered by algae. Censuses were made within 2 hours of low–water on spring tides, as the flats were exposed to their maximum extent at this time so the birds were able to

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[1] Except in August of 1995 in which only a single count could be made

select amongst all feeding places. The algae, and most waders at this stage of the tide, were distributed without obvious regard to the shore level, apart from the Grey Plover, which tended to favour the upper levels.

Data on the feeding behaviour of a visually–feeding species, the Grey Plover, and a sight–feeding species, the Dunlin, were obtained in 1991–92 (Múrias, 1993), using standard techniques (e.g. Goss-Custard, 1969; Pienkowski, 1982, 1983). Only pacing rate (number of paces.min^{-1}) – a gross measure of the searching effort – and pecking rate (number of pecks.30 sec^{-1}) were measured. As already noted in previous Chapters, it was not possible to measure the success rate because most prey taken were too small to be seen as they were being ingested. This was confirmed by the analysis of fecal pellets obtained from Dunlin (n=170) and Grey Plover (n=200) in February and March of 1995. In both species the gastropod *Hydrobia ulvae* (with 32% and 25% of the total percentage of individuals per pellet, respectively) and unidentified amphipods (34%, in both cases) were the most utilised species (Lopes *et al.*, 1995). However, in the visually–hunting Grey Plover, it is likely that the pecking rate corresponds roughly to the actual sucess rate, since Pienkowski (1982, 1983) calculated that the charadriids were sucessfull in 93–98% of their capture attempts.

Determination of the algae biomass

The proportion of the sediments covered by algae may not estimate accurately the amount present because the density of the matts is not homogeneous. A much more appropriate measure of algae abundance is its biomass (grams of dry weight.m^{-2}). Data on algae biomass were available for areas A1 and A2 (see below) but not for area A3 (M. Pardal, pers. comm.). An indirect method was therefore used to calculate the average algae biomass present in that area for each count. This was done in three steps. First, the proportion weeded was mapped for each area. Next, the average amount of algae biomass present in areas A1 and A2 was calculated based on a data set of weekly samples of 10 to 20 cores taken in each area since January 1993, and selecting those corresponding to the census periods of the present study. Finally, a regression equation of algal biomass as a function of the proportion of area that was weeded was obtained from these data ($y = -0.972 + 58.76x$, $p<0.01$, n=11) and used to predict the biomass in area A3 in each period. The same equation was used to calculate the biomass values for the 1994–95 year.

This method is rather imprecise, mainly due to inaccuracies in the mapping work, but it was the only one available in the present circumstances, in view of time and logistic constraints. Nevertheless, it was considered sufficient for the purpose of this study. The original data on percentages of algae cover in the flats was used just to illustrate between–year differences, and the relationship between the percentage of cover and biomass.

6.2.3 Data Analysis

For the purpose of the present study, the whole area was subdivided into three sub–areas on the basis of the main water channels, named from the mouth of the estuary, A1 (66.8 h), A2 (30.93 ha) and A3 (36.12 ha) (see Chapter 5). In the event, these proved to correspond, effectively, with distinct feeding areas. For most analyses, the counts made in each area on a given census were treated separately. In order to calculate the waders' preferences, if any, for weeded or algae–free sediments, the average numbers of birds in weeded and unweeded areas were calculated. A total of 84 counts were obtained for the whole study period, distributed as follows: 33 counts (3 areas$\times$11 occasions) in 1993–94 (October–May) and 51 counts (3 areas$\times$17 occasions) in 1994–95 (August–May). Not all these data were used simultaneously, however, depending on the particular kind of analysis to be performed. Details will be given as necessary in the text.

The species that fed regularly on the mudflats differed in their average densities and sesonal peak of occurrence (see Chapter 3). In a preliminary analysis of the whole wader species assemblage, the number of all species present was combined, regardless of their typical densities. Clearly, the results were dominated by the most numerous species, which tended to be the small–sized birds. In order to compensate for the species differences in body size, species numbers were transformed to biomass values by multiplying the species' numerical densities by their body mass in December, obtained from Cramp & Simmons (1983). The analysis of single species were restricted to those present in most of the counts ($n>8$, in each year) at sufficient high densities (>0.5 birds.ha^{-1}). These were the three plover species, the Dunlin, the Curlew and the Avocet. However, as the Curlew did not met the above conditions in the second year, it was excluded from the analysis. On the other hand, although the Avocet fulfilled the conditions for inclusion, it avoided the weeded areas completely, so there was no point in including it.

Since the absolute abundance of both the birds and the algae varied between months, both variables were standardized to a common scale for most analyses. The procedure used was that of Goss-Custard (1977b) and consisted of expressing each individual value for a given month as a proportion of the total for the three areas combined. Unfortunately, because of the lack of independence in data treated this way, probability values cannot be determined (Goss-Custard, 1977b). To assess the probable importance of the correlations, the empirical classification of Guilford (in Martin & Bateson, 1993) was followed, and all cases where $r>0.4$ were considered as a strong association.

The null hypotheses were that (1) the birds would distribute themselves indifferently between the weeded and unweeded areas, and (2) for the two species studied, the feeding and searching rates, as revealed by the pecking/probing rates and by pacing rates, would not differ between the two kinds of areas. In the latter case, such a result would either imply that the birds did not change their behaviour or that adaptations occur at a more subtle and unobservable level. In particular, birds may have changed the types of prey captured or the sizes taken which, in view of the small sizes of those consumed could not be detected.

Statistical analysis included the non–parametric Mann–Whitney U–test, the Kruskall–Wallis ANOVA by ranks and the Spearman correlation coefficient. The multiple regression analysis was preceded by Kolmogorov–Smirnov tests to assess the normality of data (Zar, 1984).

6.3 Results

6.3.1 Temporal and spatial variations in the abundance of algae

The quantity of algae present in the three areas varied during the study period[2] from barely none to about 70 g DW.m^{-2}, in 1993–94, and 50 g DW.m^{-2} in 1994–95. Average monthly biomass values were similar for both years (24.3±22.4 (SD) g DW.m^{-2}, in 1993–94; 23.1±14.0 g DW.m^{-2}, in 1994–95; Mann–Withney U–test, U=74.0, ns; n_1=11, n_2=16). The maximum percentage of algae cover was reached in 1994–95, with 82.1% of the total area being covered in late April, whereas in 1993–94, coverage never exceeded 36.7% (late May). However, average monthly values did not differ significantly between the two years (16.0%±12.4% of total area in 1993–94 and 35.5%±27.4%, in 1994–95; Mann–Withney U–test, U=91.0, ns; n_1=11, n_2=16) (Figure 6.1a). The total biomass present was low in mid and late–winter months (January and February), although higher values were found in 1994–95 (range: 6.94 to 12.7 g DW.m^{-2}) than in 1993–94 (range: 1.58 to 5.11 g DW.m^{-2}). However, a rapid increase occurred, in both years, from late March onwards (range: 31.2 to 62.3 g DW.m^{-2}, in 1993–94; range: 10.2 to 49.5 g DW.m^{-2}, in 1994–95) (Figure 6.1b).

Apart from the total quantitative differences that have been mentioned so far, there were also obvious temporal differences in the total amount of algae present during the spring months. In 1993–94, there was a continuous increase in the algae biomass until late May, where the highest value was reached. In the following year, on the other hand, the peak biomass value was reached earlier, in late April, and an appreciable decrease in the amount present (of about 36%) occurred in May. In general, higher biomass values were recorded in late autum and winter in 1994–95 than in 1993–94, while the reverse applied in spring.

Spatial variations were also found in the pattern of occurrence of the algae in the two years of study. In 1993–94, marked seasonal fluctuations in the biomass of the algae occurred near the mouth of the estuary (area A1), while the upstream section (area A3) showed a much more regular pattern. In contrast, the middle section (area A2) was colonized by the algae only later in the season, with a sudden burst in April (Figure 6.2a). Conversely, in 1994–95, variations in the biomass of algae among the three areas were much less pronounced and occurred mainly in autumn and early winter (August–November) (Figure 6.2b).

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[2] Excluding the months of August and September of 1994, in order to allow proper comparisons between the two years

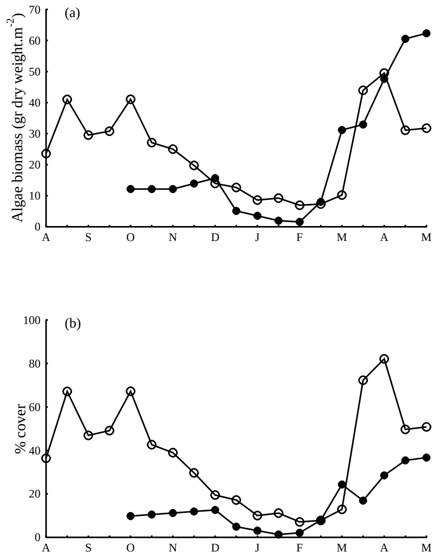

Figure 6.1: Seasonal varition in (a) the biomass and (b) the percentage cover of algae on the sediments of the Mondego estuary, in 1993–94 (closed circles) and in 1994–95 (open circles)

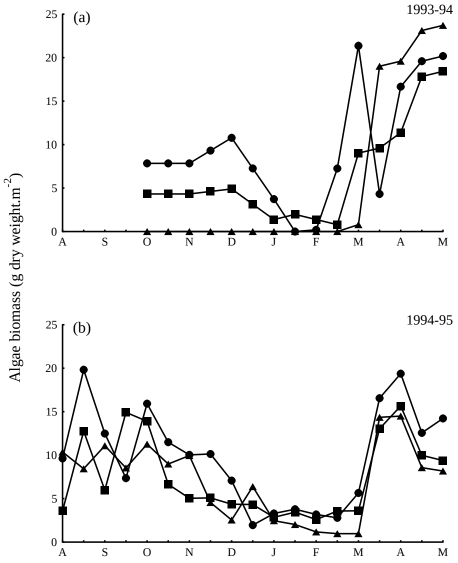

Figure 6.2: Seasonal varition in the biomass of algae in the three sub–areas considered, in 1993–94 (a) and in 1994–95 (b). Symbols: circles – area A1; triangles – area A2; squares – area A3

6.3.2 Distribution of the feeding waders on the mudflats in relation to the presence of algae: spatial and temporal analysis

Absolute avoidance or attractiveness of the algae by feeding birds

In order to test the hypothesis that the birds were either attracted or repelled absolutely by the algae, a test was conducted for two extreme situations: (1) when the average algae biomass was low ($<$5 g DW.m^{-2} *per* area, in winter and early spring months – January and February, in 1993–94 and late January to early March, in 1994–95) and (2) when the algae was abundant ($>$10 g DW.m^{-2} – late March to May, in both years). The average numbers of feeding birds in both weeded and unweeded areas were calculated for both cases and compared (Table 6.1).

In general, more birds fed on unweeded areas when the algal cover was scarce, the difference being statistically significant in all cases but one (the Grey Plover, in 1993–94). On the other hand, when the amount of algae was abundant, most species distributed themselves indifferently between algae–free and weeded areas, and no statistically significant differences were found between the two, with the exception of Ringed Plover, which was found to use the weeded areas more in 1994–95. Clearly, waders were not deterred completely from places where algae were abundant.

Temporal and spatial variations in the use of weeded and unweeded areas by birds

However, even though the algae did not deter the birds completely, they may have reduced bird densities This possibility was tested both in space and in time. Data on bird density and algae abundance were available for each individual census made in each month. A minimum value for the algae abundance of 5 g DW.m^{-2} in at least one of the three areas was imposed as the condition necessary to include the month in the analysis. Therefore, counts from January and February, in 1993–94, and from late January to early March, in 1994–95, were excluded.

Overall, there was no association between relativized bird abundance and relativized algae abundance, whether the species were considered together or individually (Figure 6.3, Table 6.2). Indeed, two of the ten associations had a positive sign, both in 1993–94 and in 1994–95. However, visual inspection of the data (Figure 6.3), suggested an association for some periods of the year, so a separate analysis was performed for each of these periods (autumn, winter and spring).

Strong (r$>$0.4) and negative associations, as predicted predicted by the hypothesis that algae reduced bird densities were obtained in all cases, except for Dunlin, only in 1993–94 (Table 6.2). In all other seasons and across both two years, a mixture of positive and negative associations were obtained, with four positive (Kentish Plover, in spring 1993–94 and winter 1994–95, Ringed Plover in winter 1994–95, and Dunlin, in spring 1994–95), and one negative (Grey Plover, in spring 1994–95) strong associations. Across all species and years, only Grey Plover consistently showed a negative association with algae, although significantly so in only

Table 6.1: Bird feeding numbers (average±1 SE) in weeded and unweeded areas when algae was scarce (December/January to early March) and when it was abundant (late March to May) in 1993–94 and in 1994–95. n, number of counts, U, Mannn–Whitney U–test; * $p<0.05$; ** $p<0.01$; *** $p<0.001$

	Scarce (<5 g DW.m^{-2})				Abundant (>10 g DW.m^{-2})			
	n	No algae	Algae	U	n	No algae	Algae	U
1993–94								
Kentish Plover	7	15.3±4.9	0.3±0.3	49.0*	10	4.5±1.1	4.0±2.5	69.5
Ringed Plover	5	37.4±17.4	5.8±5.8	22.0*	10	8.7±3.9	1.9±0.9	75.0
Grey Plover	10	42.1±9.0	13.8±5.6	71.5	10	8.1±4.8	12.1±5.7	67.5
Dunlin	8	207.1±75.4	14.1±5.9	54.0*	11	223.1±162.1	14.0±7.3	70.0
1994–95								
Kentish Plover	15	35.3±6.3	3.3±2.0	12.0***	21	21.3±6.4	12.4±4.3	167.0
Ringed Plover	13	35.9±6.0	1.8±0.9	9.0***	21	2.5±1.2	9.0±1.8	82.5**
Grey Plover	19	122.7±20.5	9.3±3.2	68.0**	12	9.6±3.4	12.7±2.4	46.0
Dunlin	17	188.6±31.4	55.1±25.2	53.0**	23	28.8±24.8	77.5±24.8	180.0

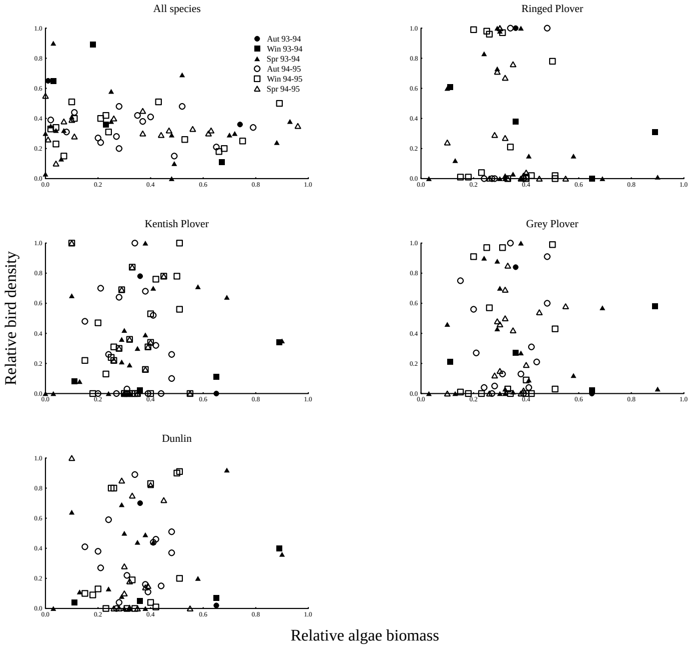

Figure 6.3: Relationship between the biomass densities of all species or relativised bird specific densities and the amount of algae present on the flats. Data are for all the sub–areas combined. Seasonal values are shown separately for each year: 1993–94 (closed symbols) and 1994–95 (open symbols). See Table 6.2.

one case. Overall, these results indicate that there was not a consistent effect of algae on the bird densities.

The results raised the possibility that the strong associations were spurious, and perhaps, due to another factor (e.g. sediment) which independently affected both the abundance of birds and the abundance of the algae. To test this, associations between bird density and algal abundance over time were looked for. If birds were repelled by the algae, the densities in one area should have decreased over time as the abundance of algae in that area increased, taking changes in the algae cover in the other two areas into account. Such a negative association would suggest that algae deterred the birds, since any confounding factor would hopefully have not changed.

A temporal analysis was, therefore, carried out in two steps. First, general trends in the total bird numbers over the whole estuary were looked for. Second, changes in bird densities between the three areas, were examined both for all species combined and for the most abundant species individually. As bird densities in the whole study area varied with time, those months in which numbers were more–or–less stable (that is, when no emigration or immigration was taking place)

Table 6.2: Values of the Spearman correlation coefficient of algae biomass against the total biomass densities of all the species combined and the numerical densities of individual species during the study period of 1993 to 1995, over: (1) the whole period (excluding January and February counts in 1993–94 and late January, february and early March counts in 1994–95), (2) the autumn months (only in 1994–95: August, September and October), (3) the winter months (November and December and, in 1994–95, also early January) and (4) the spring months (March–late May in 1993–94, and April–May, in 1994–95). The number of counts for each period is given in bracketts. "Significant" values for the individual areas (r>0.4, see text for explanation) are shown in bold typing.

	1993–94				1994–95			
	All year (n = 27)	Autumn[a]	Winter[b] (n = 6)	Spring (n = 18)	All year (n = 46)	Autumn (n = 15)	Winter (n = 16)	Spring (n = 15)
All species	-0.24	–	**-0.43**	-0.07	-0.10	-0.07	-0.03	-0.004
Kentish Plover	+0.18	–	**-0.46**	**+0.55**	-0.10	-0.11	**+0.53**	-0.11
Ringed Plover	+0.05	–	**-0.41**	+0.31	+0.12	+0.06	**+0.40**	-0.06
Grey Plover	-0.24	–	**-0.43**	-0.30	-0.14	-0.32	-0.26	**-0.42**
Dunlin	-0.11	–	-0.09	-0.03	+0.11	+0.13	+0.07	**+0.43**

[a] Only two observations for October of 1993 were available, therefore this periods was left out of the analysis

[b] The Area A2 was not considered in the counts, since no algae were present in the months sampled

were selected for the analysis. These were the winter and early spring counts from November to March, in both years. For the whole–estuary analysis, raw data counts were used, thus enabling statistical validation of the results. In the other cases, relativised data were used, so probability values could not be established formally. Because of this impossibility of establishing formal probability values in the case of the individual areas, the results of the whole–estuary analysis and those of the individual areas on all species were not directly comparable. However, if similar trends were observed in the whole–estuary and in the individual areas' analyses , this would have strongly supported the hypothesis that the algae indeed affected the densities of all species combined.

Total bird density did not correlate with the changes in the algal biomass over the whole estuary in 1993–94, but showed the expected significant negative association in 1994–95 (Table 6.3). In the individual areas, strong and contradictory associations of bird densities with algae biomass were found in areas A2 (negative association) and A3 (positive association) in 1993–94, while no strong associations were found in 1994–95. The results for the individual species were similarly variable and contradictory. The sign of the association differed between areas and/or years in all species except Ringed Plover. Across all species and areas combined, there were five positive associations and seven negative associations in 1993–94, and six positive associations and six negative associations in 1994–95, but high values of correlation – above 0.7, see Martin and Bateson (1994) – were only found in two occasions, both in 1993– 94: Kentish Plover in A1 (+0.74) and Ringed Plover in A2 (-0.92) . Only Ringed Plover exhibited the same trends in both years, but the strong associations were recorded in different areas (A1 in 1993–94 and A3 in 1994–95).

In view of the small data sets, particularly in 1993–94, some of these correlations may have arisen by chance, of course. But even without formal statistical testing, there is little to suggest that the birds changed their feeding areas in response to temporal variations in the abundance of algae.

6.3.3 Use of alternative habitats: the salinas

Another way to test whether the algae deterred shorebirds was to see if birds left the mudflats altogether to feed in the alternative areas provided by the salinas when algae was particularly abundant. This was done by relating the proportion of the whole population that was feeding over low–tide in the salinas to the total algae biomass on the flats. The total population was all the birds in the entire study area, the estuary and the salinas combined. The analysis was again performed for both the whole estuarine population of waders and for the most abundant species individually. As the size of the study area did not change, actual bird numbers were used, both for the total population biomass of all species combined and for the individual species. Only data from the winter months (November–February) were used.

There were no clear or statistically significant trends both for the whole popu-

Table 6.3: Values of the Spearman correlation coefficient of algae biomass against total bird biomass densities for the whole estuary, for the separate areas (A1, A2 and A3, see Chapter 5) and for the relative densities of individual species in each area during the winter (November to February) and early spring (early March) months of 1993–94 and 1994–95 years.n=7 in 1993–94 and n=10 in 1994–95. Formal statistical tests were performed for the whole estuary situation(* $p<0.05$). Otherwise, ‘significant’ values for the individual areas ($r>0.4$, see text for explanation) are in bold typing.

	1993–94				1994–95			
	All estuary	Area A1	Area A2	Area A3	All estuary	Area A1	Area A2	Area A3
All species	-0.53	+0.29	**-0.89**	**+0.46**	-0.65*	+0.05	+0.38	+0.00
Kentish Plover	–	**+0.74**	-0.29	+0.07	–	-0.06	**-0.57**	-0.39
Ringed Plover	–	+0.14	-0.14	**-0.92**	–	**+0.40**	-0.07	-0.23
Grey Plover	–	-0.26	-0.37	-0.25	–	**-0.49**	+0.03	+0.06
Dunlin	–	**+0.67**	**-0.84**	+0.32	–	**+0.46**	**+0.47**	+0.06

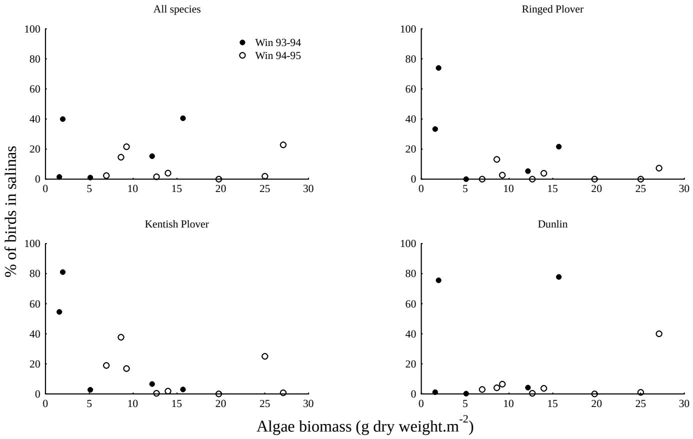

Figure 6.4: Relationship between the percentage of birds present in the salinas at low–water and the algae biomass on the intertidal sediments. Symbols as in Figure 6.3. See Table 6.4 for the statistical analysis

Table 6.4: Values of the Spearman correlation coefficient of algae biomass with the proporton of birds feeding in salinas in the winter seasons (November–February) of 1993–94 and 1994–95. * $p<0.05$, ** $p<0.01$

	Winter 1993–94 (n = 5)	Winter 1994–95 (n = 8)
All species	+0.50	-0.24
Kentish Plover	-0.60	-0.43
Ringed Plover	-0.50	0.00
Dunlin	+0.50	-0.24

lation and for individual species that the proportion of birds in the salinas tended to be high when the overall algae biomass was high (Figure 6.4 and Table 6.4). On the contrary, with the exception of the Kentish Plover, all plots suggested precisely the general opposite tendency, although seasonal and yearly variations could be observed.

There was the possibility, however, that any effect of algae abundance may have been masked by variations in competitive pressure associated with changes in total bird numbers (Chapter 5). The possible joint effects of algae and total bird numbers were therefore tested by a multiple regression analysis, controlling for the year of study and the season. This allowed to cover both the period (winter) where bird numbers were more stable, but the algae biomass was less abundant, and the migratory periods, where high variation in bird numbers took place, but the amounts of algae reached their highest values. As some birds fed exclusively in one of the habitats (see Chapter 4), the individual analyses were restricted to those

Table 6.5: Multiple regression analysis of the effect of algal biomass and the total number of birds of each species in the estuary (controlling for the year of study and the season), on the proportion of the birds feeding in the salina. Values of the coefficient for each variable are shown, as well as their sign, r and F values. n=31, * $p<0.05$, ** $p<0.01$

	Algae biomass	Total numbers	Year	Season[a]	Constant	r	$F_{(4,26)}$
All species	+0.07	-0.29	-0.15	-0.05	+35.96	0.31	0.70
Kentish Plover	+0.06	-0.05	-0.14	+0.70**	-11.26	0.72	6.03**
Ringed Plover	-0.33	-0.35	-0.16	+0.45*	+25.56	0.51	2.31
Dunlin	-0.33	-0.30	-0.13	-0.01	+60.10	0.30	0.61

[a] Measured as the number of days since the beginning of the autumn (October, in 1993–94, and August, in 1994–95)

species that used both the salinas and the mudflats at low–water.

Again, there was little to suggest that shorebirds as a whole, or individual species, used the salinas more when algae were abundant in the mudflats (Table 6.5). More Kentish Plovers were found in the salinas as the season advanced from autumn to spring, but this is likely to be linked to the onset of the reproductive period in this species, which breeds exclusively in the salinas (see Chapters 3 and 4).

6.3.4 Foraging behaviour

Only small differences were observed in the foraging parameters (pecking and pace rates) studied, in both the tactile–feeding Dunlin and the visual–feeding Grey Plover (Figure 6.5a and 6.5b, respectively). In fact, Dunlin showed a significant tendency to increase its pecking rate in areas covered by algae (45.19±2.91 *versus* 35.52±2.39 pecks.30 sec.$^{-1}$, n= 46 and n=68 respectively; Kruskal–Wallis ANOVA by ranks, χ^2= 6.72, $p<0.01$), whereas the pacing rate did not change (114.19±5.11 and 102.57±4.2 paces.min^{-1}, n= 46 and n =68; χ^2= 2.24, ns). On the other hand, Grey Plover maintained the same pecking rate in both covered and uncovered areas (4.42±0.38 and 4.14±0.33 pecks.30 sec^{-1}, n=52 and n=67; χ^2=0.01, ns), but walked significantly more quickly on uncovered sediments (53..38±3.56 *versus* 28.31±3.14 paces.min^{-1}, n = 52 and n = 67; χ^2= 24.87, $p<0.001$).

6.4 Discussion

The possible effect of the presence of algae on the feeding behaviour of wading birds has only just begun to be studied (see, e.g. Raffaelli *et al.*, 1989, for a review), and there is, therefore, a lack of data on the subject. But despite this, two major conclusions are already emerging. First, most of any effects seem to operate in the long–term. Second, the response of the birds is strongly species–specific, depend-

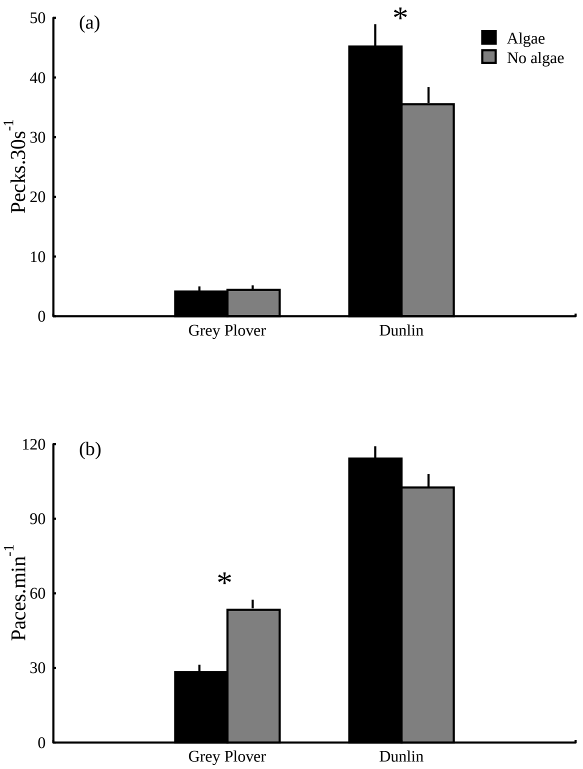

Figure 6.5: Pecking rate (a) and pace rate (b) of Grey Plover (GP) and Dunlin (D) in weeded and unweeded areas. Significant differences are signaled with an asterisk. see text for more details

ing on the particular prey and on the feeding behaviour of the species concerned (Raffaelli *et al.*, 1989; Desprez *et al.*, 1994). Viewed in this light, the results of the present short–term study are not surprising. Generally, the amount of algae present during the study period seemed not to deter the birds, and the species responded differently. This conclusion was reinforced by the absence of any evidence that the waders left the mudflats to feed on the saltpans when algae was abundant.

In view of these results, it seems more likely that an alternative factor actually influenced the spatial distribution of the birds, the sediment type being a strong possibility. Indeed, algae can trap fine particles which would otherwise be re–suspended and redeposited elsewhere, thus leading to a generally finer substrate type (Everett, 1994). This, combined with the anoxia observed beneath the mats, could lead to the development of a totally new, and sometimes richer, macrofaunal community (Everett, 1994) which, in turn, might attract at least some bird species, as Metzmacher & Reise (1994) have experimentally demonstrated. There is some indication that this could have happened with Dunlin in the Mondego. In spring, when the abundance of invertebrates was generally high in the mats (M. Pardal, pers. comm.), there was a strong positive association between Dunlin density and algae biomass (r=+0.43). Conversely, in autumn, after a reduction in prey abundance through predation or, as the algae decayed in late summer, through emigration or death by suffocation (Everett, 1994; Martins, 1995), had taken place, no association between Dunlin density and algae biomass was apparent (r=+0.13). Even though the data were not amenable to statistical testing, the considerable difference between the two coefficients of correlation is suggestive. The hypothesis that the birds distributed themselves indifferently regarding the position of the mats, or that they might even have been attracted to them in some cases, is reinforced by the absence of any evidence that the waders left the mudflats to feed on the salinas when algae were generally abundant (Table 6.1).

Perhaps the birds were able to continue feeding on weeded sediments by changing their foraging behaviour. Although the tactile–feeding Dunlin was more flexible in its response to the presence of algae than the sight–feeding plover, if this did happen in the Mondego, it must have been achieved through quite subtle changes in feeding behaviour. In neither of the two species studied were striking differences found in the feeding behaviours in weeded and algae–free areas. Although the prey could not be identified as they were taken, no large items were seen to be swallowed in either algae–free areas or in the weeded areas. In fact, in the Mondego, all wader species seemed to consume small–sized prey, the majority probable being small polychaetes and *Hydrobia ulvae*. These prey are extremely abundant (see Marques *et al.*, 1993b) in the weeded areas (M. Pardal, pers. comm.).

Overall, the results suggest that the presence of algae on the sediment does not strongly influence the feeding distribution of the birds in the Mondego estuary. However, for three main reasons this cannot be a definitive conclusion:

1. The average percentage cover of the flats by weed during the study period did not exceed 40% and values obtained at any one time were frequently

lower than 30%, whereas Soulsby *et al.* (1982), found that changes in the infauna related to the presence of the mats only occur when 25%–75% of the area is covered by algae. Assuming, as these authors did, that the change occurs when 50% of the area is covered, it is evident that a considerable area of the mudflats may still be occupied by algae before the numbers of avian predators exploiting the infaunal species may seriously decline;

2. Too few data for each period of the year were available to draw statistically strong inferences, and the need to use relativised values prevented a full test of the statistical significance of the associations revealed by the analysis.

3. The short duration of the study. As showed previously (Raffaelli *et al.*, 1989; Desprez *et al.*, 1994), changes in both the numbers of birds and in their feeding behaviour in response to an increase of algae, occurs in the medium to long–term, thus proceeding in parallel with the slow changes in the densities and structure of the prey populations.

Only more intensive and extensive studies carried out in the longer–term can more critically evaluate the effect that algae has on the feeding of the waders in the Mondego estuary.

Chapter 7

General Discussion

Habitat loss is probably the most intensively examined of the factors that are known to potentially affect the survival of estuarine wader populations (Goss-Custard *et al.*, 1996a,b). Most studied cases have focused on the consequences for waders of direct loss of intertidal low–water feeding areas through land reclamation for industrial, agricultural or water storage purposes (Davidson *et al.*, 1991), and of indirect losses due to the submersion of previously accessible feeding areas following the construction of tidal power or storm–surge barriers, or sea–walls (Meire *et al.*, 1994; Lambeck *et al.*, 1996). In contrast, there have been few studies dealing with the consequences of the loss of supratidal feeding areas, probably because in north European estuaries, these are not heavily used, or critically endangered, although their importance is recognised (Davidson & Evans, 1986; Hotker, 1994).

The situation may be different in the south European Atlantic estuaries. Supratidal habitats, such as the salinas, usually occupy large areas within the estuaries and 'rias', and seem to be intensively used throughout the tidal cycle, by a lot of species (Rufino *et al.*, 1984; Perez-Hurtado & Hortas, 1991, 1993b). These man–made wetlands are currently more threatened in the southern European estuaries than the natural intertidal areas, mainly due to their abandonment and/or transformation into fish–farms or ricefields (Rufino & Neves, 1992; Neves & Rufino, 1995; Perez-Hurtado & Hortas, 1993a).

The loss of the supratidal habitats raises a major problem for estuarine waders, which differs from that faced by waders when they lose intertidal habitats. When birds are able to meet most of their daily energy requirements in the intertidal areas at low–water, and are displaced from them, they will mostly lose feeding space. They can always (and do) try to re–establish themselves in the remaining intertidal areas (Meire *et al.*, 1994; Lambeck *et al.*, 1996). Whether or not they can recover equally good feeding opportunities as the ones that were lost, and if this redistribution will affect the carrying capacity of the area[1], and thus the survival or the permanency of all birds in it, is another question.

However, birds that rely heavily on the supratidal habitats to fulfil their ener-

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[1] Defined in the *sensu* of Goss-Custard (1985), see Chapter 5

getic needs at high–water, when intertidal habitats are submerged, will also lose feeding time if supratidal habitats disappear. On many occasions, particularly in the more energetically–demanding periods of the year, these birds will not be able to recover this feeding time by extending their foraging at low–water through an increase in the foraging time and/or in the intake rate (Davidson & Evans, 1986; Goss-Custard *et al.*, 1996c). Their only chance, if they are to remain on the area is, therefore, to look for alternative supratidal sites.

The results of this study suggest that, in the estuary of Mondego, many birds could be in trouble if all the salinas were to be destroyed, due to time loss. The present data suggest that the effects of competition on the mudflats, either through interference or through resource depletion, are currently weak (Chapter 5). Therefore, a number of the birds that use the salinas as alternative feeding areas to the mudflats could eventually re–establish themselves in the intertidal flats at low–water, buffering the loss of low–tide feeding space. However, even these birds seemed to intensively use the salinas for feeding at high–water (Chapter 4). So, the need for finding supplementary supratidal feeding areas could involve both the mudflat–feeders and the salinas–feeders.

The problem arises because no other supratidal habitat in the estuary is likely to offer the good feeding conditions that birds can presently find in the salinas. The ricefields are only usable at certain times of the year, otherwise being too flooded (in winter) or too dry (in summer). Furthermore, it is unlikely that some bare–sediment specialists, such as the small plovers (Kentish and Ringed Plovers) would use this habitat at all. Saltmarshes are also commonly used supplementary feeding sites for mudflat feeders, both in Europe and in South Africa (Davidson & Evans, 1986; Velásquez & Hockey, 1990, 1991). However, in the Mondego, they are mainly formed by *Spartina* sp., whose dense stands are known to drive out many small species, such as Dunlin (Goss-Custard & Moser, 1988). Only the larger species (Grey Plover, *Numenius* spp., godwits) were occasionally seen using this habitat in the Mondego, but these would probably be the least affected species by the loss of the salinas (Chapter 4).

A less suitable but still usable alternative supratidal site to the salinas could be provided by the fish–farms. They have the advantage of being encircled by wire fences, thus providing quiet and relatively safe places from attack by most terrestrial predators[2]. In fact, waders use them now as roosting sites. However, the ponds are usually too deep for waders, even for the long–legged species, and in practice they are used only when they are periodically emptied for cleaning (Perez-Hurtado & Hortas, 1993a,b). In the Mondego, even this periodic usage seems to be infrequent (at least it was never observed during the present study), probably due to the depth and to the narrow dimensions of the ponds.

Loss of intertidal habitat could also occur in the estuary, through the increas-

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[2] The predation effects of birds of prey on adult or juvenile waders seem to be relatively unimportant in this estuary, at least judging from the absence of any observations of raptor attacks on feeding or resting waders

ingly extensive 'blooms' of green macroalgae that occur seasonally due to eutrophication (Marques *et al.*, 1993b,a). The presence of contiguous and extensive algae mats would be expect to decrease the abundance of many prey species (see, e.g. Everett, 1994), and the subsequent recolonisation after the algal crash, from the small islets of unweeded areas that would remain, would be insufficient to replace the losses (Raffaelli *et al.*, 1989, 1991). Waders could then be unable to find enough food to meet their daily requirements in the previously weeded areas.

In fact, although the effects of the eutrophication at the low levels of the trophic chain can be assessed in a matter of 2–3 years (e.g. Soulsby *et al.*, 1982; Desprez *et al.*, 1994; Everett, 1994), its consequences for waders may take more time to be established. Subtle changes in the diets of some wader species usually appear when the populations of their main prey are being affected, long before any change in numbers begin to be noted (Desprez *et al.*, 1994). Only in a later stage of the process do waders respond by changing their feeding areas. In the Mondego, no indication was obtained that the waders changed of feeding areas due to the presence of algae (Chapter 6). On the contrary, although circumstantial, there was evidence that some species (e.g Dunlin) could even be attracted to the mats during, or soon after, the algal 'bloom', in early spring.

Apart from some methodological reasons and the time–scale used, detailed in Chapter 6, another factor may account for the lack of relationship between bird numbers and algae in the Mondego. The most consumed prey of waders in this estuary seem to be mobile sediment–water interface feeders, such as some errant polychaetes and *Hydrobia ulvae* (Lopes *et al.*, 1995), which may be favoured, at least in the earlier stages of the season, by the growth of algae (Soulsby *et al.*, 1982; Everett, 1994), thus providing enhanced, even though seasonally–limited, food resources.

In summary, this study suggest that the main threat to estuarine waders in the Mondego at present is the ongoing destruction of the supratidal habitats. It seems that there is some buffering capacity on the mudflats to receive a number of displaced birds from the salinas. However, the lack of sufficient area on the supratidal habitats themselves that would enable birds to recover the feeding time that would be lost with the salinas, could be detrimental to the populations of many species. In the long–term, the effects of the loss of supratidal habitat could be further aggravated if the eutrophication continues to increase at the present rate, thereby perhaps also reducing the available feeding space for waders at low–water.

From a conservation point of view, there is always the possibility of creating artificial supratidal habitats to replace those lost (Davidson & Evans, 1986, 1987; Hotker, 1994). There are, however, some limitations on the creation of artificial supratidal wetlands, these being (1) the large areas that are required to allow the settlement of all the displaced birds; (2) the need to provide similar habitats to those destroyed, particularly in terms of their sediment types and invertebrate faunas, in order to attract the same species that were displaced; and (3) the need to begin the work some years (2–3) in advance of the destruction of the primary habitat, due to the time required to find an appropriate place, prepare the area and allow the settlement and growth of the invertebrate prey (Davidson & Evans, 1987).

But even if these constraints can be overtaken, there is no absolute assurance that waders will accept the new sites. Hotker (1994) showed that at least two of three artificially–created supratidal habitats in the German and Danish part of the Wadden Sea, that were constructed to compensate for the loss of reclaimed intertidal habitats, did contribute to the increase the number of bird species and densities in the area, but they did not fully compensate the losses due to land claims.

It seems that prevention is still the best way to avoid the more deleterious effects of habitat loss. In the Mondego, there may still be time to reverse the present trend. Many salinas have been abandoned, but not yet transformed. As this study showed (Chapter 4), deserted salinas are not completely unattractive to waders, although they are far less used than the active ones. An effective and relatively inexpensive way of recovering these salinas for waders would be to pay their owners to keep them clean and to maintain a permanently controlled water level. This would avoid the salinas conversion or drainage. An alternative or, even better, complementary solution would be to impose strict rules on the construction of new fish–farms in the estuary, by improving the design of the pond walls, in order to create areas of shallow water, as it was suggested by Rehfisch (1994) for man–made brackish lagoons in England. This could allow even the smaller waders to use the ponds, although some care should be taken to prevent the access of piscivourous birds (e.g. herons).

Regarding eutrophication, any local intervention (e.g. by imposing some form of treatment of the urban, agriculture and fish–farm discharges to the estuary) would not be enough. It would be also necessary to control the urban and agricultural discharges along the whole lower river valley, in which the majority of ricefields and other extensively irrigated lands are located. This is a very difficult task, however, as it requires the involvement of many different official and private organisations.

Besides the obvious need to preserve the estuarine biodiversity and the health of the whole ecosystem, of which waders are an important component, there is another important reason why the quality of the habitat for waders should be maintained or even enhanced in this estuary. Small estuaries like the Mondego, with relatively low number of waders, as compared to the major estuaries of the East Atlantic Flyway (Chapter 3), may act as ‘emergency’ sites for some migrating or wintering birds. Emergency sites are areas where, in normal conditions, few birds land, but where, under adverse weather, they stage in great numbers (Piersma, 1987 in Smit & Piersma, 1989). This may prevent many birds from starving, avoiding the high mortality rates which, otherwise, would probably occur. Moreover, the real number of birds of all species that use the estuary of Mondego may have been underestimated, particularly during the migratory periods. As Kersten and Smit (1984) and Kersten *et al* (unpubl.) (in Smit & Piersma, 1989) showed for a small Moroccan estuary (Sidi Moussa), the spring migration peak of 7000 birds underestimated by 3 times the real number of birds that crossed the area over a two–month period, as inverstigated by an intensive counting (3–5 days counts) and colour–

marking program. If this is also the case in the Mondego, its perceived importance for waders would naturally increase.

The present study was intended to be a preliminary study of the important issue of habitat loss in the environmentally stressed estuary of Mondego. In the course of this work, many new questions were raised. As a guideline for future work on the subject, some suggestions are made:

1. Detailed studies on the feeding ecology of the most important species (including diet, time–budgets and measurement of intake rates) are very badly needed, as they constitute the basis for assessing the relative importance of the main feeding areas. Such studies should be performed both in the intertidal areas and in the salinas and, preferentially, be accompanied by detailed measurements of the food supply.

2. The patterns of between–habitats movements in the estuary, particularly those related to the tidal cycle, should be more fully investigated, both in winter and during the migratory seasons, possibly by means of colour–marked or radio–tagged birds. This would help to determine how reliable is one of the main assumptions of this study, viz. the existence of two distinct sub–populations that preferentially use either the mudflats or the salinas. Such a study could also provide more complete data on the use of the other supratidal habitats, and on the relative importance of all habitats during specific periods of the year, and/or weather conditions.

3. Detailed studies of the factors that determine the selection of active versus inactive salinas by feeding waders should be conducted, if possible using experimental approaches. Such studies could be very helpful in providing basic information for future management plans of the area. Experiments to attract waders could be conducted in the existing fish–farms (e.g. by gradually lowering the water level when the ponds are being cleaned, or by extending the period where these ponds are more accessible to waders), in cooperation with the respective owners. Testing new designs for fish–farms that would allow waders to feed there, would be invaluable, and could be achieved in the preparatory and initial stages of the construction of new ponds.

4. A refinement of the methodology to study the competition effects on the mudflats would be greatly welcome. In particular, it should be possible to measure intake rates of at least some species and use this approach, in combination with the one developed in this study. It would also be desirable (and possibly easier) to extend the study to the salinas.

5. A long–term study of the effects of algal growth on wader numbers should be conducted by monitoring both parameters for some years. However, detailed studies on how algae could affect bird diets could be performed over shorter periods and should be also conducted, both empirically and experimentally.

6. The significance of the estuary for migratory populations of some selected species (e.g. Dunlin) should be more fully examined. This could be evaluated by an intensive counting and marking program. Data on the total number of waders that stage on the estuary in each period, population structure and habitat use could be obtained in this way.

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Appendix

Table 0.1: Present status of the salinas in the Mondego estuary. The salinas surveyed in more detail in this study (Chapter 4) are marked with an asterisk. The identification of some salinas may differ from that presented in Table 4.5. In such occasions, the names of the salinas as they appear in Table 4.5 are given in brackets.

Group	Salina	Area (ha)	N. of cómoros	Actual status of management
Morraceira Island	Adeiro Velho	2.572	1	Inactive
		1.725	6	Fish-farm
	Alhos*	1.790	1	Active
		1.725	1	Inactive
	Amante*	2.995	2	Inactive
		1.585	3	Active
	Aveiró	1.280	1	Active
		1.770	1	Extensive fish-farm
	Casa da Pedra*	1.030	1	Inactive
		2.975	1	Active
	Cavalo Branco*	1.076	2	Active
		2.560	1	Drained
		0.800	1	Extensive fish-farm
		2.230	1	Active
	Cerco*	3.045	4	Active
		1.861	2	Inactive
	Corredor do Cabaço*	6.810	1	Inactive
	Corredor Novo*	4.190	1	Inactive
	Corredor do Padre*	1.660	2	Active
		6.830	7	Inactive
		1.775	2	Extensive fish–farm
	Corredor do Reveles	1.830	1	Active
		1.515	2	Inactive
	Corredor do Sol	5.880	3	Active
	Correias	3.045	2	Fish-farm
	Cruz*	1.445	1	Active
	Esteiro de Aveiró (Filipas)*	3.175	1	Active
	Donato*	4.230	2	Active
		2.830	2	Inactive
	(Ínsua)	3.400	3	Inactive

Group	Salina	Area (ha)	N. of cómoros	Actual status of management
	D. Maria	4.715	3	Inactive
		1.730	1	Active
	Doutores*	1.725	1	Active
		5.380	2	Inactive
	Feras*	5.205	4	Active
		2.595	1	Inactive
	(Donato II)	3.070	3	Active
		1.765	1	Inactive
	Ínsua	3.250	4	Extensive fish-farm
	Isca	3.815	3	Extensive fish-farm
	Joaquim da Fonte*	5.110	1	Inactive
	Moleiras	3.635	3	Fish-farm
	Mondeguinho	3.150	1	Fish-farm
	Morro*	1.485	2	Active
		1.515	2	Inactive
		6.005	5	Inactive
	Norte	2.640	2	Active
		1.665	1	Inactive
	Praias Grandes*	2.205	1	Active
	Pestanas*	2.640	3	Active
		2.505	3	Inactive
	Pinheiros	5.865	3	Fish-farm
	Pontão*	9.545	8	Inactive
	S. Julião	1.715	1	Fish-farm
	Tapada Norte*	3.110	3	Active
		3.165	2	Inactive
	Tapada Sul*	4.440	3	Active
	Ucharia*	3.255	2	Active
		6.105	2	Inactive
	Venturas de Baixo*	2.800	2	Active
		1.165	1	Inactive
	Venturas de Cima	1.040	1	Active
		2.175	3	Inactive
		2.785	2	Fish-farm
Lavos (South Arm)	Boca da Veia	0.937	2	Inactive
		0.449	1	Extensive fish-farm
	Torrão	0.599	1	Active
	Torrão, Negra, Estacas	5.927	5	Fish-farm
	Vale da Vinha, Estacas, Armazéns	4.790	3	Fish-farm
	Vale da Vinha	0.837	2	Active
		0.074	2	Inactive
	Freiras	0.262	0.5	Fish-farm
		0.166	0.25	Active
		0.581	0.25	Inactive

Group	Salina	Area (ha)	N. of cómoros	Actual status of management
	Jorge	0.531	1	Active
		1.062	2	Inactive
	Toscano, Tabuões, Corredor Velho	20.891	4	Active
	Negrão	4.578	6	Active
	Caldeira	1.312	2	Active
	Cobra	2.338	1	Active
	Quebradita	0.975	1	Active
	Quebrada	1.925	1	Active
	Morgada	1.438	2	Active
	Armazéns	1.325	2	Active
		0.837	3	Fishfarm
	Pedrosa	0.074	1	Inactive
	Corredio	0.462	1	Active
	Terça	0.975	1	Active
	Vermelha	6.955	5	Active
	Cavada	3.327	0.66	Active
		1.225	0.33	Fish-farm
	Morro Comprido	2.451	3	Fish-farm
	Eiras Largas	0.887	2	Inactive
		0.612	2	Active
	Craveiras de Baixo	3.063	2.66	Active
		0.692	0.33	Inactive
	Craveiras	3.302	0.5	Fish-farm
	Marachões	1.200	0.5	Active
	Quadros	0.599	1	Fish-farm
	Areia	0.324	1	Active
	Craveiras de Cima	1.663	2	Active
	Noventa Talhos	0.187	1	Inactive
		0.612	1	Extensive fish-farm
	Ínsua d’el Rei	-	2	Inactive
Vila Verde (North Arm)	Salmanha	-	1	Drained
	Salmanha	-	1	Inactive
	Herdeiros	2.435	3	Drained
	Gramatal	6.895	6	Inactive
	Ladeiras	3.450	6	Drained

This document was written in LaTeX using MikTeX in Windows 95 and teTeX in Linux (Slackware96). Graphs were designed using GLE 3.3h and CorelDraw 6.0.