Chapter 9
Spartina

Friend or Foe?

9.1 Introduction

Spartina species form a group of salt-tolerant grasses which occur in several different parts of the world. Geographically centred along the east coast of North and South America, outliers occur on the west coast of North America, Europe and Tristan da Cunha (Table 19).

Table 19 Five of the fourteen or so **native** species of Cord-grass found in saltmarshes, their main areas of geographical distribution and tidal limits; MSL – mean sea level; MHW – mean high water mark

English name	Latin name	Natural distribution
Smooth Cord-grass	*S. alterniflora*	Eastern USA Maine – Texas from 0.7 m below MSL to c. MHW
Salt Meadow Cord-grass	*S. patens*	Eastern USA from MHW to c. 0.5 m above MHW
California Cord-grass	*S. foliosa*	Bodega Bay – Baja, California, USA
Dense-flowered Cord-grass	*S. densiflora*	Chile, South America near MHW, or just below it on open mud
Small Cord-grass	*S. maritima*	British Isles, Europe up to MHW spring tides

These species of grasses, have the ability to grow quickly and reproduce from seed, rhizomes or broken vegetative shoots. Once established they can colonise estuarine sand and mudflats quickly because of their ability to exploit areas outside the normal tidal range of the majority of perennial saltmarsh plants (Thompson 1991). The primary aim of this chapter is to consider the origins of the species, the reasons for its success and need for management.

9.1.1 The Nature of Colonisation

Most of the species (except the sterile hybrid *Spartina townsendii*) reproduce from seed. Plants flower quite late in the season, producing seed in the autumn. Colonisation by native species depends on the initial establishment of seedlings, which thereafter expand by vegetative means to form clones. *S. alterniflora* is one of the first colonisers of mudflats on the east coast of North America. In its native range, interspecific competition appears to limit its competitive ability at higher saltmarsh levels. Eventually, in the upper saltmarsh, the smaller *S. patens* becomes the dominant species (Bertness 1991; Figure 58).

Spartina maritima grows at a lower elevation than most other *Spartina* spp and in north-west Spain colonises from rhizome fragments. Thereafter, clones grow horizontally creating circular patches on bare mud (Sánchez et al. 2001). In the estuaries of Portugal and Spain it is quite robust (Figure 59).

In southern England, where the species reaches its northern limit, it was plentiful in the estuaries around the south and south east in the late 1800s, early 1900s (Marchant & Goodman 1969). It still survives in a few localities in south-east

Figure 58 *Spartina alterniflora* saltmarsh on the east coast of America, Chincoteague National Wildlife Refuge. Saltmeadow Cord-grass (*S. patens*) lies flat in the foreground of the photograph

Figure 59 *Spartina maritima* clones in the Faro Estuary, Portugal

England although it is less robust than the same species, further south. The expansion of the more robust hybrid, *Spartina anglica* and possible climate change are amongst the reasons cited for its demise (Rodwell ed. 2000). Other species, including the hybrids, show similar patterns of colonisation.

9.1.2 Hybridisation

In North America, particularly on the east coast of the USA, *Spartina* is a dominant component of the extensive saltmarshes occurring there. Two species are important, *Spartina alterniflora* (Figure 58) and the smaller *S. patens*. The first species, in particular, readily hybridises with other species, and this has had a significant impact on saltmarshes throughout the world.

The first recorded occurrence of hybridisation took place on the south coast of England in about 1816. The native *Spartina maritima* crossed with *S. alterniflora*, introduced by shipping from the USA, accidentally into Southampton Water (England). This hybridisation produced a sterile plant *S. X townsendii* (Goodman 1969; Goodman et al. 1969). Following a doubling of its chromosomes, a fertile species *S. anglica* appeared and expanded rapidly (Marchant 1967). This event has been widely studied and reported. Several publications summarise the biology, origins, spread and impact of the species (e.g. Doody 1984; Gray & Benham 1990; Adam 1990, pp. 78–87).

In south-west France there may have been a second hybridisation event, involving the same parent species, but with different parental (nuclear) genotypes (Baumel et al. 2003). This plant does not appear to have had the same history as *Spartina anglica* and remains as a small isolated clone.

Hybridisation also occurs in other parts of the world, in San Francisco Bay, the native *Spartina foliosa* hybridised with the introduced *S. alterniflora*, colonising the Bay's tidal mudflats and marshes to the detriment of the former species (Callaway & Josselyn 1992; Daehler & Strong 1997a). Also, in San Francisco Bay, Dense-flower Cord-grass (*S. densiflora*) has recently hybridised with *S. alterniflora* (Ayres & Lee 2004).

The tidal range for *S. alterniflora* is also wide and varies throughout the world. In the USA, this variation is attributed to differences in mean tidal range (MTR). At the same time, the *S. alterniflora* zone expands with increasing tidal amplitude (McKee & Patrick 1988). It has the potential to grow from the Mean Higher High Water (MHHW) to approximately 1 m below Mean Low Lower Water (MLLW) as seen in Willapa Bay, Washington (Sayce 1988). In San Francisco Bay, hybrids between the introduced *Spartina alterniflora* and the native California Cord-grass (*S. foliosa*) occur and grow at both lower and higher elevations than the native species. It is also more prolific than the hybrid (*S. anglica*) and the other non-native introduced species *S. densiflora* and *S. patens* present in the Bay (Ayres et al. 2004).

Hybridisation appears to give a competitive advantage over the original parents. In the UK, where the earliest recorded hybridisation between *Spartina alterniflora* and *S. maritima* took place, the resulting *Spartina anglica* has the ability to colonise almost any level in the tidal range. This included the 'absolute seaward limit of saltmarsh growth' to the landward limit of 'high water equinoctial tides'. It tolerates 'saline to brackish water conditions' (Ranwell 1972). At the lower tidal levels this new species was able to occupy a niche with limited competition and then only by scattered plants of annual *Salicornia* spp. (Gray et al. 1990).

In the USA, the *S. foliosa* X *S. alterniflora* hybrid in San Francisco Bay appears to be more robust than the native species, with superior seed set and siring abilities. This results in proliferation of hybrid clones capable of rapid expansion (Ayres et al. 2004). *S. densiflora*, a native of South America, is also tolerant of a wide range of conditions. It has not only successfully invaded the west coast of the USA, but also Spain and Morocco (Bortolus 2006). It is uncertain if a hybrid of this species, such as the one found in San Francisco, will create highly aggressive colonisers of tidal flats, similar to those of introduced *S. alterniflora*.

9.1.3 Pattern of Invasion

There are two key invasive species now: the native *Spartina alterniflora* and the hybrid *S. anglica*, included on the Global Invasive Species Database (see http://issg.org/database/welcome/). Their introduction, both accidentally and deliberately throughout the world, has lead to extensive colonisation of tidal mudflats. The nature of colonisation follows a similar pattern and is initially from seed or vegetative shoots. Towards their northern limit the number and viability of seeds depends on climatic factors, with the best seed set taking place in warmer years.

In some locations, they only spread slowly by vegetative growth following the first phase of establishment. This may last for many years with little or no lateral expansion. For the first 50 years the population of *Spartina alterniflora*, accidentally introduced to Willapa Bay in 1894 and only identified in the 1940s when it flowered, changed little. However, between 1945 and 1988, the species established itself throughout the bay (Sayce 1988). This rapid expansion appears to have occurred in favourable (warmer) years, with the production of fertile seed.

Other non-native *Spartina* species establish themselves in similar ways. *Spartina densiflora* in California invaded the upper levels of tidal flats mainly by vegetative spread, following seedling establishment (Kittelson & Boyd 1997).

Rapid invasion does not happen in all cases. In California, *Spartina anglica* had not spread beyond its original 1970s introduction site in 30 years. *Spartina densiflora* has spread to cover only 5 ha at 3 sites in the Central Bay over the same time period. *Spartina patens* had similarly only expanded from 2 plants in 1970 to 42 plants at one site in Suisun Bay (Ayres et al. 2004).

In other situations, expansion and contraction followed by reestablishment can take place. *Spartina townsendii* colonised the tidal flats on the eastern shore of Skallingen, Denmark resulting in a 'large area of coherent vegetation' from 1954 to 1964. 'Die back' occurred in the area of coherent vegetation from 1976 and 1988 only to be recolonised by 1995. To seaward, a series of circular patches stretched into the lower tidal area, the limits of which remained more or less stable throughout (Vinther et al. 2001).

The eroding saltmarshes of south-east England also have local examples where accretion takes place in sheltered locations such as the Blackwater Estuary. An example of this (Figure 60) also shows the typical process of colonisation, whereby clumps of *Spartina anglica* arise by vegetative growth, following seedling establishment.

9.1.4 Rates of Sedimentation

Worldwide vertical sedimentation rates of 20–80 mm per annum occur. Rates of between 100 and 120 mm per annum for *Spartina* were recorded for Bridgwater Bay, Somerset, in south-west England (Ranwell 1964). In exceptional circumstances, over short periods and in rapidly accreting saltmarsh, these can be as high as 200 mm per annum (Ranwell 1967). These compare with a range of 2 and 10 mm generally for middle and upper saltmarshes in Europe and eastern America (Ranwell 1964), in line with the 2.5 and 4.7 mm per year on a long established *Spartina alterniflora* saltmarsh, on the east coast of America (Flessa et al. 1977). Very similar accretion rates of between 2.4 and 4.8 mm per annum occurred in the American Pacific Northwest (Thom 1992).

Figure 60 Coalescing clumps of *Spartina anglica* in the Blackwater Estuary, Essex, England

Sediment rates vary considerably depending on tidal range, sediment availability, and rate of compaction. Time of year is also important with rates being generally highest in the autumn (Ranwell 1964). Even at the lower end of the scale, *Spartina anglica* saltmarsh accretes at a rate that is some 10 times that of a typical 'natural' saltmarsh.

9.2 World Domination

Once the sterile *Spartina townsendii* had doubled its chromosomes and become fertile, the new species *S. anglica* showed a rapid expansion along the south coast of England. The plant first attracted attention when Lord Montagu of Beaulieu referred to the advance of *Spartina townsendii* (rice grass) over the tidal flat until it covered several thousand acres, in his evidence to a Royal Commission on Coastal Erosion in 1911 (Carey & Oliver 1918). Its ability to stabilise mud flats, led to its promotion for erosion control and as a land reclaiming agent (Oliver 1925). Plantings took place extensively in the UK (Goodman et al. 1959), with Poole Harbour on the south coast of England being the main source of material. Estimates of the overall area of *Spartina anglica* by 1967 amounted to 12,205 ha in 86 sites (Hubbard & Stebbings 1967).

Active promotion of the properties of the plant resulted in the export of considerable quantities of both seeds and plants from Poole Harbour (Ranwell

1967) to many parts of the world. In addition to material from Poole Harbour, a Mr J Bryce sent other material from Essex. For example, in 1927/28, he exported seed to New Zealand and much later (between 1947 and 1955) plants (Partridge 1987). The result of this enthusiastic promotion resulted in a considerable increase in the world resources of this plant.

9.2.1 World Resources

Although not all the material successfully established (some seed was sterile) and conditions were not always suitable, by the early 1960s, the area of *Spartina townsendii* (s.l.) recorded worldwide was between 21,000 and 27,700 ha (Ranwell 1967). In addition to the large area in Great Britain, this included substantial areas in other European countries as well as smaller areas in Australia, New Zealand and the USA (Table 20).

Table 20 Dates refer to the first appearance (or known introduction). Area estimates are of *Spartina* with ground cover of >50% and are very approximate (derived from Ranwell 1967)

Country	Date of origin	Area (ha)
Ireland	1925	200–400
Denmark	1931	500
Germany	1927	400–800
Netherlands	1924	4000–5800
France	1906	4000–8000
Australia	1930	10–20
Tasmania	1927	20–40
New Zealand	1913	20–40
USA	1960	<1

There were some early misgivings about the likely long-term effects of this rapid colonisation, 'whether the result (of *Spartina* establishment) will in the end be beneficial, or to the contrary will depend greatly on local conditions' (Stapf 1908). However, the general perception in these early days was that it is a great asset for sea defence, land reclamation and to a lesser extent animal fodder.

9.2.2 Spread in China, Australia and New Zealand

There are no native species of *Spartina* in China, Australia or New Zealand. Two species of *Spartina* (*S. anglica* and *S. alterniflora*) spread rapidly following their introduction to China in 1963 and 1979 respectively (Chung 1990; Bixing & Philips 2006). By 1985, the area of *S. anglica* reached 36,000 ha in 18 counties (Chung 1990). In North Jiangsu, *Spartina alterniflora*, covered some 410 km out of a coastal length of 954 km, with a maximum width over 4 km (Zhang et al. 2004). In China as a whole, the area increased from 260 ha in 1985 to 112,000 ha in 2002 (An et al. 2004). A review of the impact of the species over a 30-year period suggested that this colonisation was positive for partial control of siltation, fodder, nesting and feeding grounds for migratory birds, increased estuary productivity and in papermaking. *S. alterniflora* extracts provide additives for soda water, beer, milk, wine, tea and bathing lotions and have been trialled for their medicinal effects (Chung 1990; Chung 1993; Qin et al. 1997).

In Australia *Spartina anglica* infestations are found in the southern States of Tasmania and Victoria. In Tasmania, its introduction early in the nineteenth century was for the potential benefits in sea defence. It had invaded seven regions of Tasmania's coastal zone, occupying nearly 600 ha by 1997. Two sites, the River Tamar with 415 ha and the Rubicon estuary with 135 ha, represent a very small percentage of its potential habitat (Hedge 2002). In Victoria the estimated area of *S. anglica* was 186 ha (Hedge et al. 1997).

In New Zealand, expansion began in 1913 with the introduction of *Spartina townsendii*, which by 1952, in at least one site, the New River Estuary, Invercargill, had expanded to 40 ha (Partridge 1987). The maximum rate of spread of vegetation was 5.3 m per annum (Lee and Partridge 1983). Following the further introduction of *S. anglica*, by 1973, 90 ha of saltmarsh meadow and 250 ha, of scattered clumps were present, covering 15% of the mudflat (Hubbard & Partridge 1981).

9.2.3 USA, Washington State and San Francisco Bay

In the USA, there are three principal native species, two distributed on the east coast (*Spartina alterniflora* and *S. patens*) and on the south Pacific coast (*S. foliosa*). There are four introduced species:

1. *Spartina alterniflora* dominant, mainly Willapa Bay;
2. *Spartina anglica* widespread, dominant in Puget Sound;
3. *Spartina patens* small populations;
4. *Spartina densiflora* small populations first discovered in 2001 (Figure 61).

Initially non-native *Spartina townsendii* covered less than 1 ha (Ranwell 1967). Today the non-native invaders occur on the west coast where there are four principal sites (Figure 61). In Puget Sound, Washington State, *Spartina anglica* was introduced deliberately for shore stabilisation and as potential feed for cattle in the 1960s. Since then, in 36 years the plant has successfully invaded 73 sites, affecting 3,311 ha of marine intertidal habitat (Hacker et al. 2001). Also in Washington State *Spartina alterniflora*, accidentally introduced to Willapa Bay in the 1880s, had by the 1950s, expanded to cover about 10% of tidal flats. It spread

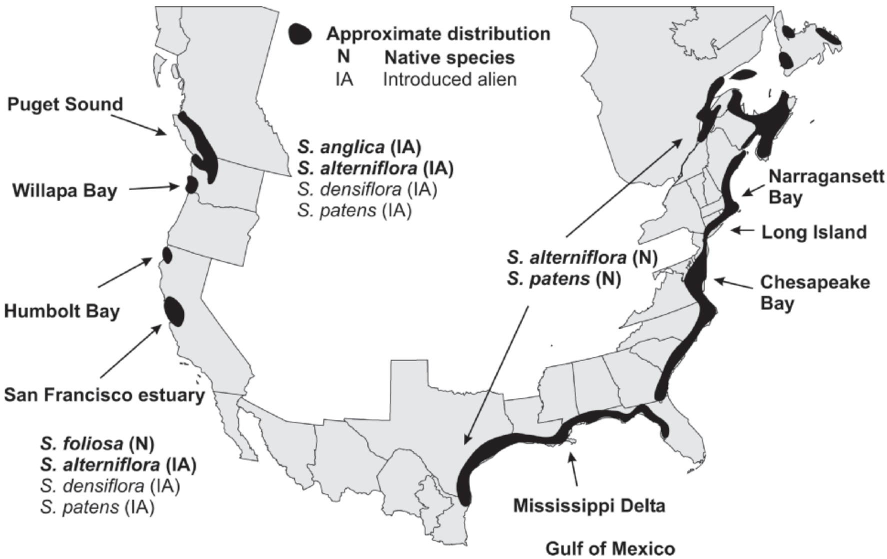

Figure 61 Approximate distribution of the main native and introduced *Spartina spp*. in North America. **N** – native to area; **IA** – introduced alien

laterally at a rate of 0.79 m per annum (Feist & Simenstad 2000). Given the uniform slope of the intertidal flats of Willapa Bay, the potential area for colonisation is 66% or approximately 12,600 ha of the intertidal area (Sayce 1988). Between 1995 and 2000 the total area of non-native *Spartina anglica* and *S. alterniflora* in Washington State was 8,093 ha (Hedge et al. 2003).

For more information, see the Willapa Bay web site dedicated to providing information on *Spartina* spp. spread and control (see http://friendsofwillaparefuge.org/spartina.htm). The Washington State Department of Agriculture, Noxious Weed Control Board web site provides more general information on *Spartina* spp. (see http://www.nwcb.wa.gov/). It includes useful summaries of the species of *Spartina* found in the state.

In San Francisco Bay, hybrids of *Spartina alterniflora* and *S. foliosa* are the most numerous exotic species and spread rapidly (Strong & Ayres 2005). Three other non-native species are also present *S. anglica*, *S. patens* and *S. densiflora*, which together occupy approximately 5 ha. The total coverage of hybrids and other exotic species was 195 ha, slightly less than 1% of the area of tidal flats, although the potential area for colonisation was much greater (Ayres et al. 2004).

9.3 Changing Perceptions

The success of introduced *Spartina anglica* and *S. alterniflora*, hailed as a boon for sea defence, land claim and other human uses soon became questionable. The first suggestion that this rapid expansion might not always be beneficial came when the spread of *Spartina townsendii* (Oliver 1925) coincided with a loss of populations of *Zoster marina* in England in the 1920s and 1930s. Although there was no direct evidence of 'cause and effect', there are a number of accounts, referred to by Adam (1990) of *Spartina anglica* replacing native *Zostera* spp. It seems likely that *Spartina anglica* was the lucky recipient of the available niche, created by the loss of *Zostera marina* through a 'wasting' decease (Davison & Hughes 1998), rather than the agent of its demise. Experience from the UK suggested that, from a nature conservation perspective at least, the overall conclusion is that '*Spartina* provides a threat in estuaries of high wildlife interest, both to bird populations and to natural saltmarsh succession' (Doody 1984). However, it is the implications for bird populations feeding on exposed tidal flats, which was the principal cause of concern in the UK.

9.3.1 Impacts on Bird Populations in the UK

Concern about the rapid expansion of *Spartina anglica* and its effect on wintering waterfowl was one of the driving forces for a *Spartina* meeting held in Liverpool in 1982 (Doody 1984). This publication included papers on the origins, history and spread of the hybrid species *Spartina anglica*. The meeting discussed the implications of the spread for waders in the Dyfi Estuary in Wales (Davis & Moss 1984) and on invertebrates and shorebird populations at Lindisfarne National Nature Reserve, Northumberland, north-east England (Millard & Evans 1984). It was clear from both studies that the spread of *Spartina* reduces the area of upper open intertidal flats and with it the most profitable feeding zone for wading birds.

Lower down the shoreline *Spartina* may also have restricted the areas of *Zostera*, an important food for the Brent Goose and Wigeon. A more detailed study of the situation at Lindisfarne showed that, of all the factors potentially affecting wintering Brent Geese and Wigeon, the loss of the upper shore to *Spartina anglica* was more significant than sea-level rise and intermediate loss of *Zostera* spp. (Percival et al. 1998). These concerns resulted in several attempts to control the species at this site (Corkhill 1984; Frid et al. 1999). Further examination of these concerns, as they affected the wading bird, Dunlin in British estuaries, showed a negative correlation with *Spartina* expansion. However, the evidence that there was a causal link was not conclusive (Goss-Custard & Moser 1988).

9.3.2 Impacts on Amenity Beaches, North-West England

Changes associated with *Spartina* also have the potential to impact on recreational use. The Cheshire shore of the Dee Estuary, UK, provides an illustration of the extent of change and implications for recreational interests. Here, the marsh front advanced along the shore of the estuary, partly because of the growth and establishment of *Spartina anglica* (Taylor & Burrows 1968). The conversion of the sandy beach to a saltmarsh resulted in the loss of amenity beaches along the shore. This resulted in a request from the local authority to investigate the biology and control of the species (Taylor & Burrows 1968). By the mid-1990s the saltmarsh was 1.2 km wide in front of Parkgate; a sandy beach covered by most tides in 1939 (Figure 40; Pye 1996).

A similar problem arose on the south shore of the Ribble Estuary, further north along the coast of Merseyside. Here, Sefton Municipal Borough Council sought to control the expansion of *Spartina anglica* onto amenity beaches using the herbicide 'dalapon' (Robinson 1984; Truscott 1984).

9.3.3 Problems in the USA

At the height of the invasion of estuaries in Washington State, USA, dense swards of single species replaced natural vegetation. In the process, they destroyed important migratory shorebird and waterfowl habitat, increased the threat of flooding and severely affected the state's shellfish industry (Murphy 2005). These and other possible impacts (Table 21) led to increasing concerns about the invasion of *Spartina alterniflora* in the estuaries of the State.

Table 21 Potential affects of *Spartina alterniflora* spread in Washington State (adapted from Callaway & Josselyn 1992)

Possible impact	Cause
Competitive replacement of native plants	Higher seed production & germination; higher vegetative production
Effects of sedimentation	Greater stem densities, larger & more rigid stems
Changes in available detritus	Differences in quantity & quality of detritus
Decreased bottom-dwelling algae production	Lower light levels beneath *Spartina* canopy
Increased wrack deposition & disturbance to upper marsh	Greater stem production & subsequent deposition in high marsh
Changes in habitats for native wetland animals	Greater stem densities
Changes in bottom-dwelling invertebrate populations	Higher root densities & lower intertidal distribution
Loss of shorebird & wading bird foraging areas	Lower intertidal distribution

Similar concerns exist in San Francisco Bay, where the plant which was only introduced in 1970, has spread rapidly right around the bay. Whilst the area of invasion represents only 1% of the tidal flats, the potential for further colonisation is problematic, not least for the survival of the native *S. foliosa* (Ayres et al. 2004). Some of the impacts of invasive *Spartina* highlighted for San Francisco Bay, USA, are:

- Loss of biodiversity as a result of the competition with native flora, including *S. foliosa* and *Salicornia virginica;*
- Hybridisation with native *S. foliosa;*
- Loss of mudflat and channel habitat;
- Loss of foraging and nesting habitat for numerous shorebirds, where the potential for loss is high (Stralberg et al. 2004) and waterfowl, including the endangered California clapper rail;

- Change in macro invertebrates (Neira et al. 2005);
- Clogging flood channels;
- Increasing rate of sedimentation and marsh elevation.

The San Francisco Estuary Invasive *Spartina* Project web site, (see http://spartina.org/index.htm) provides information that is more detailed.

9.3.4 Studies Elsewhere

In Victoria, Australia, concern about the spread of *Spartina anglica* began in 1991 when the Department of Natural Resources and Environment initiated a study to:

- Map the spread of the species;
- Raise the profile of the issue through workshops;
- Assess methods of control (Williamson 1995).

In February 1995, a *Spartina* Control Project for the area, containing approximately 98% of Victoria's estimated 186 ha of *Spartina* infestations, was initiated (Hedge & Kriwoken 1997).

New Zealand experienced much lower sediment accretion rates for *Spartina anglica* than elsewhere. 12 mm per annum in dense swards on a muddy substrate, to as little as 3 mm per annum on sandy substrates, with turbulent water in one estuary. The potential for loss of biological value still led to concern and attempts at control (Lee & Partridge 1983). The ecological, social and economic costs associated with its continued spread in Tasmania have resulted in the development of a management programme supporting eradication and control (Kriwoken & Hedge 2000).

In China, the apparent benefits attributed to *Spartina anglica*, resulted in the introduction, in 1979, of *S. alterniflora*. However, this species, brought in to check erosion, had spread to such an extent that it is 'choking estuaries, and crowding out native species such as the bulrush, *Scirpus mariqueter* together with its rich diversity of bird species (Chen et al. 2004) and reducing feed and habitat for fish and migratory birds'. The recent increase in trade with the USA has led to greater concern for the impact of alien species generally in China (Yan et al. 2000). Amongst these species is *S. alterniflora* listed by the Chinese Government on a 'Black List' of species, which should not be imported into the country (Normile 2004).

9.4 Methods of Control

In the early days, any thought that the control of *Spartina* spp. would become a major issue would have been an anathema to many people. This is especially true for those concerned with coastal erosion protection or land claim. However, it is clear from the examples described above that the hybrid *Spartina anglica* and American species, notably *S. alterniflora*, when outside their native range, can be aggressive invaders.

This has resulted in a change in the perception of the value of the plant from being a ‘friend’ to a ‘foe’, where habitat loss and degradation are key issues. As a result, the ‘route to the restoration’ involves reversing the encroachment of the species onto open intertidal sand and mud flats. Some of the methods are considered next.

9.4.1 Herbicides

A common method of control is spraying with herbicides. Marketed under a number of different commercial names, they can represent the most effective means of control. However, depending on their toxicity to humans, effects on the environment generally, and on non-target species, they are closely regulated in most countries. Making specific recommendations is thus not possible. What follows is a brief review of some of the methods and their efficacy from published material around the world.

In Great Britain, attempts to control the species included the use of a variety of herbicides at Lindisfarne National Nature Reserve, Northumberland, northern England (Corkhill 1984). This included Dalapon (sodium dichloropropionate), also used to control *Spartina anglica* on amenity beaches (Truscott 1984). The treatments met with varying degrees of success. It was particularly effective on the amenity beach, where it achieved 99% kill after three applications. Although 90–100% kill occurred in some trial areas after 5 years most of the *Spartina* continued to grow and expand (Corkhill 1984). There was evidence of a return of some of the bird populations using the site. However, there was no clear indication of the long-term efficacy of the treatment (Evans 1984).

The herbicide Roundup PRO, based on glyphosate, was much less effective at Lindisfarne National Nature Reserve, achieving only a 50% kill in initial applications (Corkhill 1984). Reviews that are more recent suggest that in Washington State, USA, where Dalapon is no longer used, the only alternative, the Rodeo formulation of glyphosate, showed very variable effectiveness (Hedge et al. 2003). The annex (p. 123), gives a summary of the properties of Rodeo (Department of the Interior, US Fish and Wildlife Service, Willapa National Wildlife Refuge 1997). Alternatives that may prove more effective include Imazapyr (Patten 2002).

Elsewhere, such as Australia, other herbicides also proved successful. Other than for small infestations, the only herbicide which proved to be effective was Fusilade (active constituent: 212 g/L fluazifop-P present as butyl ester (FPB)) providing up to 100% kill, with ‘acceptable’ environmental side effects (Hedge & Kriwoken 1997). In Tasmania, the same chemical was the only suitable and effective herbicide to control ‘rice grass’ (Hedge 2002). In New Zealand, chemical treatment was by far the most effective. Here, treatment is mostly with the herbicide Gallant (Haloxyfop), either Dalapon/Weedazol or Roundup (Shaw & Gosling 1997).

In China, the recent change in approach to *Spartina* spp. invasion has resulted in work to find a weed-killer suitable for treatment. Studies identified an herbicide called micaojing, which killed all of the above ground parts of *Spartina* app. within

30 days. It is reported that the below ground parts are also susceptible, being killed within 60 days. Micaojing appears to be harmless to wild animals including clams, tuna and prawns and disappears from the environment in 30 days (Jian et al. 2005). To date there is no common agreement on the best herbicide. Much depends on the physical conditions at an individual site.

9.4.2 Physical/Mechanical Control

A detailed Environmental Assessment (EA) for the control of *Spartina alterniflora* on Willapa National Wildlife Refuge, Washington State from 1996, considered a variety of physical/mechanical methods. These included commercial harvesting, trampling/crushing, excavation, scraping, ploughing/rotovating, dewatering/draining, flooding/inundating, burning, steaming, covering, use of laser beams and freezing roots. Of those investigated, hand pulling or 'pushing' plants into the mud worked, but only on young seedlings. Even then, it is important to remove both above and below ground parts of the plant. Cutting, mowing alone, or burning needed several treatments and only eliminated infestations at high cost (Department of the Interior, US Fish and Wildlife Service, Willapa National Wildlife Refuge 1997).

At Lindisfarne National Nature Reserve, chemical control all but ceased. Instead, in the 1990s mechanical trials were undertaken using a machine, which 'buried' the plants. Burying using a rotoburying machine at Lindisfarne killed over 95% *Spartina anglica* after two years (Denny & Anderson 1999). Use of a light-weight tracked vehicle driven repeatedly over *S. anglica* resulted in a stem density approximately half that for untreated plots (Frid et al. 1999).

In Victoria, Australia, methods included slashing, burning, sluicing, digging, smothering and herbicides. Except for small newly formed infestations only herbicides proved to be effective (Hedge & Kriwoken 1997). Smothering techniques also proved less than successful, being very labour-intensive, suitable for small areas only and susceptible to damage by storms and from vandalism (Hedge 2002).

9.4.3 Grazing

Grazing clearly has an effect on *Spartina* swards, as it does on saltmarsh more generally (Section 7.3). Grazing, more than mowing or cutting, is likely to reduce seed set and hence expansion. However, it is unlikely to eliminate the saltmarsh from the tidal flats and does not appear to have been used as a control mechanism.

9.4.4 Biological Control

Greenhouse experiments found that *Spartina alterniflora* clones became stressed or killed by moderate populations of *Prokelisia marginata*, a Homopteran leafhopper common to the home range of *S. alterniflora* (Daehler & Strong, 1997b). A greenhouse population of *S. anglica* introduced to Puget Sound in Washington was also vulnerable to high populations of planthoppers from California (Wu et al. 1999). Thus, early results suggest the most effective biological control so far appears to be from *Prokelisia marginata*. This feeds on *Spartina* fluids, by piercing the leaf (Hedge et al. 2003).

9.4.5 Summary of Control Measures

Throughout the world, there have been many attempts to control *Spartina*. These are mostly costly and/or ineffective (Table 22).

Table 22 A summaries of control measures, taken from various sources. See, for example, Hammond & Cooper (2002); the San Francisco Invasive *Spartina* Project, which provides a wealth of information as well as links to many other sites (see http://www.spartina.org/index.htm); Hedge (2003) and Roberts & Pullin (2006)

Method	Effectiveness	Advantages & disadvantages
Herbicides (Dalapon, Glyphosate)	Can be effective although Dalapon is difficult to obtain and Glyphosate and other herbicides not fully trialled	Requires continual treatment. Relatively expensive
Digging	Partially and on a small scale (mainly seedlings)	Labour-intensive and costly on a large scale
Dyking and inundation	Partially effective in preventing spread	Costly and damaging to other saltmarsh communities
Bulldozing (removal of surface)	Ineffective	Potential damage to mud surface
Rotovating & harrowing	Counterproductive	Greater propagation from broken rhizomes
Burying (ploughing & rotoburying)	Effective if plants are covered. Effective for up to four years	Difficulties of access
Crushing	Partially effective	Requires repeat treatment, vehicular access difficulties
Burning	Ineffective	Impractical
Grazing	Prevents seedling production and hence can restrict spread. Cost-effective	Increases shoot density, no reduction in clumps
Mowing	Prevents seedling production, and hence can restrict spread. Can be labour-intensive	Can increase shoot density, no reduction in clumps. Requires continuing treatment
Covering (black plastic)	Partly effective on a small scale	Difficult to keep plastic in place
Biological control	Can be effective on an individual site basis. Avoids use of chemicals	Involves introduction of alien species. Experimental

9.5 *Spartina* spp. Friend or Foe?

The several species of *Spartina* pose an interesting dilemma for the conservationist and coastal manager alike. Native species of *Spartina* in their own environment provide a significant contribution to the functioning of the coastal ecosystem in which they occur. They also form an important component of many restoration schemes, especially in the USA (Section 6.2.5). However, aggressive hybrids and species outside their normal range can cause significant environment problems, as has been outlined in this chapter.

It is generally accepted that native *Spartina* spp., which form part of a natural succession, will have values associated with accreting State 3 or State 2 saltmarsh. It is also true that, promoting saltmarsh accretion through the introduction of *Spartina anglica* and other *Spartina* spp. is very successful. This has created a backlash in many parts of the world with the resulting ‘demonisation’ of the plant and attempts to eradicate it (Section 9.4).

However, factors such as the apparently natural ‘die back’ and a reappraisal of the role of *Spartina anglica* in the ‘natural’ succession have raised questions over eradication as a form of restoration, especially in the UK (Lacambra et al. 2004). The rest of this chapter looks at the pros and cons of those species of *Spartina* currently perceived as being a threat to one or more environmental values.

9.5.1 Control – Concerns and Costs

Overall, *Spartina* control has proved to be a costly and complex process (Hedge et al. 2003). It also appears that many of the methods employed are inefficient. Herbicide treatment remains the most effective control mechanism. However, even here ‘approved’ chemicals are not always completely successful and may require several treatments for near 100% eradication. Reinvasion is always possible and it seems likely in most areas that total eradication is not possible, except where small-scale local invasions are involved. Many of the apparently most effective herbicides (used in Tasmania, New Zealand and China, Section 9.4.1), are largely untested, elsewhere. Regulatory approval, in the face of concerns about toxicity to other non-target species and possibility of persistence in the environment, will continue to make widespread acceptance of their use difficult.

In Washington State, some of the most intensive and expensive control programmes have been undertaken. Despite five years of treatment between 1995 and 2000, which covered an average of 15% of the *Spartina* infestation, the total area increased considerably (Hedge et al. 2003). Following an increasing effort for three years up to 2005, there were still 2,550 solid hectares in Willapa Bay in 2004 and 223 solid hectares in Puget Sound. Nearly 80% of the former area and 95% of the latter were treated in 2005. The cost of control is impressive, with a budget of more than $1.5 million allocated for 2006 (Murphy 2005). Details of the programme are contained in annual reports from 1998 to 2006, (see http://agr.wa.gov/PlantsInsects/Weeds/Spartina/default.htm).

9.5.2 'Natural Die Back'

It is perhaps not surprising, given the often rapid and extensive colonisation of *Spartina* described above, that control measures take place. 'Die back' is a phenomenon often used to describe *Spartina* spp. plants exhibiting reduced growth, which can result in death of individual plants and ultimately the loss of large swaths of *Spartina*. Soon after the rapid expansion of *Spartina townsendii* s.l., in the early 1900s, slowed in the 1950s, 'die back' appeared on the south coast of England.

Studies of the process identified two distinct forms 'edge die back' and 'die back' in and around 'pans' in the centre of the saltmarsh (Tubbs 1984). Losses at the edge of the saltmarsh were attributed to wave attack, whilst those in the centre seemed to be associated with water logging and soft-rotting of the apex of the rhizome (Goodman 1960). The process of expansion and retreat, described for the south coast of England, represents a typical pattern of change (e.g. Goodman et al. 1959; Gray & Pearson 1984). Langstone Harbour, in the Solent Estuary provides an illustration of the scale of the change. Erosion and slumping followed the expansion of *Spartina* in the first half of the twentieth Century, such that its area was considerably reduced by 1980 (Figure 62).

A similar pattern of change occurred subsequently in South Wales, and along much of the east coast including North Norfolk and in northern France and south-west Netherlands (Gray et al. 1997; Gray & Raybould 1997). 'Die back' also appears to occur naturally in inland areas of native *Spartina alterniflora* saltmarshes in Louisiana, representing a significant area of loss (Mendelssohn & McKee 1988). The Mississippi River Delta suffered a major and rapid loss in 2000.

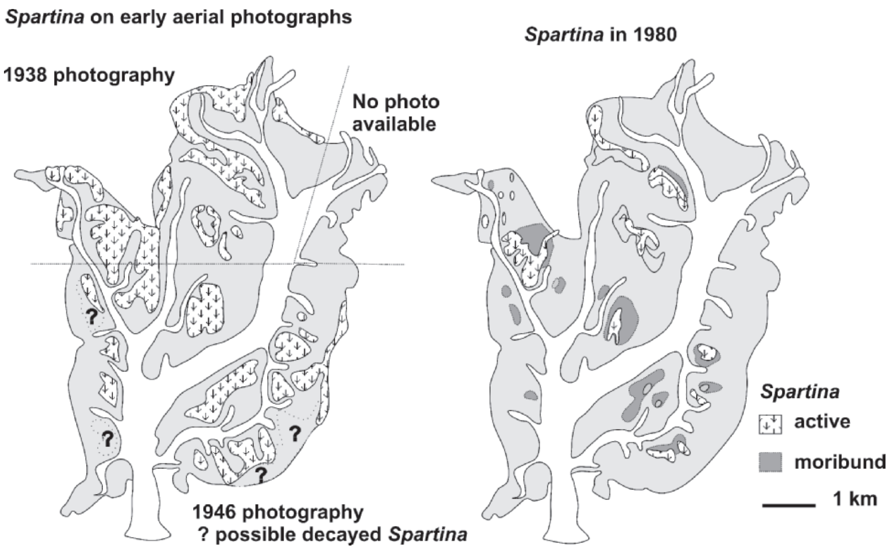

Figure 62 Change in the area of *Spartina* in Langstone Harbour, redrawn from Haynes (1984)

However, these areas recovered relatively quickly (McKee et al. 2004). Large-scale 'die back', affecting approximately 800 ha of saltmarsh, occurred in Georgia, although again this was not long lasting (Ogburn & Alber 2006).

The invasion of *Spartina anglica* in China seems to have followed a similar pattern. After its introduction in 1963, it had spread to 36,000 ha by 1985, in 18 counties. However, by 2002 there were only a few small and scattered colonies in three counties, the majority having suffered extensive 'die back'. By contrast the spread of *S. alterniflora* continued. Following its introduction in 1979, it had spread to 260 ha by 1985, reaching 112,000 ha by 2002 (An et al. 2004).

The production of phytotoxins under anaerobic soil conditions created by poor drainage, or even to rising relative sea levels is one of the most frequently suggested causes. Water logging seems to be a key factor in increasing soil reduction and sulphide concentrations (McKee et al. 2004). The development of 'salt pans' in Australia is attributed to water logging of stands of *Spartina townsendii* (s.l.) (Boston 1983). Although there is no specific mention of 'die back' this could be a similar process. Whatever the mechanism, it is possible to view the expansion and subsequent retreat as a natural process, whereby a new species occupies a previously unoccupied niche and has paved the way for its own destruction (Gray et al. 1991).

The Langstone Harbour example also illustrates the nature of the change. As saltmarsh is lost, dense growths of algae cover the mudflats, resulting from an increase in eutrophication, partly brought about by decaying *Spartina* (Figure 63).

Figure 63 Growth of algal mats in Langstone Harbour, following the loss of *Spartina*, photograph taken in 1980. *Spartina* remains present as isolated clumps in the middle distance

Once ‘die back’ occurs there is little evidence of reinvasion, suggesting that conditions remain unsuitable for some time. A nature conservation assessment of Langstone Harbour in January 2007 concluded that in approximately 50% of the intertidal area, the condition of the site in three of the main units was ‘unfavourable’, due to the continued erosion of saltmarsh. Information from the Natural England web site, Site of Special Scientific Interest Assessment Report, January 2007 (see http://www.english-nature.org.uk/Special/sssi/sssi_details.cfm?sssi_id=1001182).

Holes Bay, Poole Harbour, on the south coast of England, provides an indication of the timing of these changes. *Spartina* arrived in 1899 and expanded relatively rapidly to produce swards occupying 208 ha, more than 60% of the intertidal mudflats by 1924. By 1972 it had retreated to less than half the area, by 1994 to less than a third (63 ha) (Gray & Raybould 1997).

9.5.3 Changing Patterns of Invasion – Great Britain

The pattern of change described for individual sites appears to extend to a wider geographical area though with distinct regional differences. The estimated 12,205 ha of *Spartina anglica* in 86 sites in England and Wales in the 1960s (Hubbard & Stebbings 1967) had apparently fallen to 6,950 ha by the end of the decade (Way 1990, quoted in Lacambra et al. 2004). By the 1990s, approximately 10,000 ha of *Spartina anglica* represented nearly 25% of the total saltmarsh in Great Britain (Gray et al. 1997). A review of the status of *Spartina* in Great Britain in the late 1980s helps to explain this apparent reversal of fortunes. Taking the area of *Spartina* in Hubbard and Stebbings (1967) as a starting point, it is possible to compare the changes taking place in different geographical areas (Table 23).

These figures are not directly comparable but support the view that an early rapid phase of expansion and retraction took place in the south. This reflects the stages in the growth, establishment and recession in different geographical areas. Despite an increase in the total number of sites on the south coast, there has been an overall reduction of 11% in the area of *Spartina* saltmarsh. This reduction is even more obvious on the east coast. The increase on the west coast is equally clear.

Table 23 Areas of *Spartina anglica* as given for three geographical coastal areas in Great Britain. A literature search and limited survey provide the basis for the comparison (Charman 1990)

	South	East	West	Total
Hubbard & Stebbings (1967)				
Area of *Spartina* (ha)	3,326	6,568	2,312	12,205
Number of sites	24	27	35	86
Updated figures (Charman 1990)				
Area of *Spartina* (ha)	2,951	3,655	3,248	9,854
Number of sites	29	27	55	111
New sites / old sites	(+6 −1)	(+1 −1)	(+20)	(+27 −2)
Change	−11%	−44%	+40%	−19%

It seems that the rapid growth and subsequent 'die back' on the south coast has occurred at a later date on the east coast. Expansion continues on the north-west coast of England (see Section 9.5.4).

Analysis of data from two other sites in England helps provide further appreciation of these changes and the management response. The main expansion of *Spartina anglica* in Bridgwater Bay, Somerset, occurred between 1947 and 1971. There appears to have been a contraction from 1971 to 1982 and again by 1994, though at a slower rate. A survey in 1999 showed the boundary to be similar to that for 1994. This natural decline took place without any intervention to control the colonisation. At Lindisfarne, National Nature Reserve, Northumberland, the expansion of *S. anglica* was later still. Although present in the 1960s, its slow expansion did not cause alarm. However, after 1964 with the building up of the causeway road to Holy Island, a more rapid expansion took place. Due to the threat to wintering waterfowl, it was decided to control the plant by hand-pulling and digging in the 1970s, and later by the use of chemicals (Corkhill 1984). Despite these control measures, *Spartina* continues to spread (Lacambra et al. 2004).

9.5.4 Spartina in North-West England, a Case of Succession

A visit by the author to the island of South Walney in August 1981 proved to be of some interest in relation to the '*Spartina* story' in the UK. The extensive tidal flats east of the island were in the first phase of a remarkable transition. At the time, the area formed part of an extensive Morecambe Bay Site of Special Scientific Interest. It was also identified as a part of a Nature Conservation Review site (Ratcliffe 1977), which formed the basis for the selection of sites designated under the European Union, Habitats and Species Directive. The principal reasons for selection include sand and mudflats exposed at low tide, together with their feeding wintering waterfowl population (Table 24). The site is also important for its colonising *Salicornia europaea* and Atlantic saltmarshes.

The history of *Spartina anglica* expansion in the Morecambe Bay estuary is typical of the situation in the rest of Great Britain and many other parts of the world. It appears to have arrived naturally in the 1940s, from which time its population remained more or less stable. By the end of the 1960s, there were a few new sites but there had been only slow spread. By 1982, the species had established in a few localities, although most populations occurred as isolated clumps (Whiteside 1987). Outside the survey area, in the outer reaches of Morecambe Bay *Spartina* clumps were clearly visible off the eastern shore of South Walney Island (Figure 64) in August 1981 (Figure 65).

In 1985, these isolated patches had coalesced to form continuous swards (Figure 66). A survey of the whole shore in the lee of the island, at about the same time, estimated saltmarsh as covering 240 ha (Burd 1989). Partly because of this and other changes in *Spartina* noted on the UK coastline, a symposium took place in Liverpool in November 1982, to review the status of *Spartina* (Doody 1984).

Table 24 Principal wintering waterfowl using the tidal flats of Morecambe Bay Special Protection Area (designated under the European Union Birds Directive) as a percentage of the world, North West Europe (NW) and East Atlantic Flyway (EAF)

Migratory >1% International Biogeographical Population	
Pink-footed Goose	4.1% world
Shelduck	2.3% NW
Pintail	3.8% NW
Oystercatcher	6.0% EAF
Ringed Plover	3.0% EAF
Grey Plover	1.1% EAF
Knot	8.3% EAF
Sanderling	3.0% EAF
Dunlin	4.3% EAF
Bar-tailed Godwit	1.8% EAF
Curlew	3.6% EAF
Redshank	4.3% EAF
Turnstone	2.5% EAF

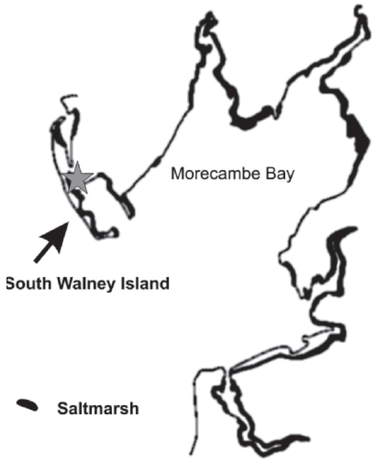

Figure 64 The approximate location of the photographs taken from South Walney Island and shown in Figures 65–67 is indicated by a star

A further visit in 2005 showed how the upper levels of *Spartina* appeared to have begun to give way to a high-level Atlantic saltmarsh community, one of the features for which the site was designated under the European Habitats Directive (Figure 67). In this community, *Spartina* is much less dense and other species such as *Limonium vulgare*, *Triglochin maritima*, *Juncus maritimus* and *Atriplex portulacoides* appear in the sward. The presence of low-level cattle grazing (Figure 66) may have helped the successional process.

Figure 65 August 1981 circular patches of *Spartina anglica* stretching onto the tidal mudflats of the Piel Channel Flats, Morecambe Bay, from South Walney Island

Figure 66 August 1985 *Spartina anglica* meadow (compare with Figure 65)

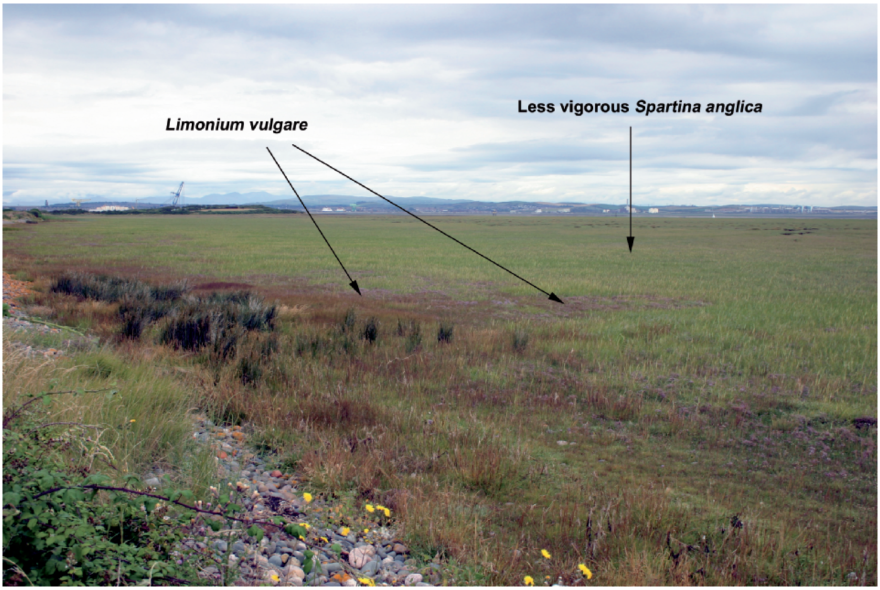

Figure 67 July 2005 similar view to Figures 65 and 66. There is abundant *Limonium vulgare* and other species characteristic of upper saltmarshes in the foreground

9.6 Conclusion

Promotion of the hybrid *Spartina anglica* as a land-reclaiming agent was one of several positive values attached to the species. The resulting introduction to estuaries around the world of this species, and *S. alterniflora*, has been remarkably successful. However, in recent years, a consensus has grown that suggests the positive economic benefits associated with the invasion of *Spartina* spp. no longer outweigh the largely negative environmental consequences, especially those associated with nature conservation. Even in China, despite its commercial use, recent reports indicate a change in attitude away from promoting its spread to one of control. Consequently, in many parts of the world, *Spartina* hybrids or native species outside their normal geographic range are subject to strategies designed to control or eradicate them.

9.6.1 Spartina anglica – A Natural Component of Saltmarshes in the UK and Ireland?

Given the concern around the world about the spread of *Spartina anglica*, it is perhaps surprising that the species is now considered an endemic 'native' in the

Atlas of the British & Irish Flora (Preston et al. 2002). This recent reclassification reflects a developing view that the species has been around long enough to become a 'natural' component of saltmarsh vegetation.

In an attempt to inform management policy on nature reserves and other protected areas, English Nature (Natural England) commissioned a review of the species. The detailed report includes a review of the situation generally, as well as providing a comparison between two different National Nature Reserves (NNRs) in England both with mudflats, important for wintering waterfowl. At Bridgwater Bay, NNR, despite rapid expansion of *Spartina anglica* on to the mudflats, there has been no control. By contrast, at Lindisfarne NNR, control has been taking place since 1970. At the former site, *Spartina* has stabilised, covering an area less than its maximum spread. At the latter site, it continues to expand (Lacambra et al. 2004). The limitation on expansion at Bridgwater Bay may result from the effects of wave action (Morley 1973).

These two different approaches show the importance of determining management on a site-by-site basis. The situation on the tidal flats of South Walney also reflects a differing view on the threat posed by rapid expansion of the species. In this case, a recent review of the nature conservation status of the saltmarsh and mudflat by Natural England considered 'condition' of the tidal saltmarshes at South Walney as 'favourable'. This was despite a recorded doubling of the area of *Spartina* saltmarsh in recent years. There have been no major attempts to control the species in this area despite the high value of the site for wintering waterfowl, many of which use the tidal sand and mud flats to feed on.

In Ireland, there is no indication that it is considered a problem, indeed where it occurs it is thought to enhance biodiversity (Curtis & Sheehy Skeffington 1998). By contrast, in Northern Ireland *Spartina anglica* is a major issue in several estuaries, notably Strangford Lough. Originally introduced in the 1940s, it spread to the point where it became a threat to wintering waterfowl populations. In the 1960s, an attempt to eradicate it using herbicides, failed (Hammond & Cooper 2002).

9.6.2 Friend or Foe

The question as to whether *Spartina* spp. is friend or foe, depends on a number of issues. In its natural geographical areas, it is valued for the protection it affords to both the hinterland and for species living in and around it. It provides many of the other values associated with saltmarsh in terms of productivity, acting as a pollution sink, etc. (Section 4.3.2) and nature conservation (Section 4.5). In the early days, whether as a hybrid or a native species outside its natural range, its values for coast protection and as an aid to land reclamation outweighed the environmental disbenefits. With the recognition of the environmental and nature conservation problems, such as the loss of intertidal feeding for wintering waterfowl, the view of the plant has changed. The '*Spartina* phenomenon' had become a problem requiring control or eradication (Doody 1990).

However, despite this it seems that control may not always be appropriate. The incidence of ‘natural’ change, notably ‘die back’, suggests that for *Spartina anglica*, leaving things alone may be all that is needed. The cost, the ineffective nature of some methods of control and the need for repeat treatments, also call it into question. Natural succession, in the UK at least, appears to be leading to the development of habitats of high nature conservation value in their own right at some sites. Although leaving things alone may result in the expansion of areas of saltmarsh at the expense of tidal flats, the overall nature conservation value appears to survive.

Thus, the loss of tidal flats to *Spartina* invasion will almost certainly illicit a first response similar to that adopted worldwide. Given the speed and scale of invasion and the known effects on many features of nature conservation and economic interests this is entirely understandable. However, this initial response requires modification in order to avoid costly and ineffective management.

In the Odiel Estuary, *S. densiflora* invasion may have helped create the large expanse of saltmarsh at this site. Today it dominates some 18% of the saltmarsh community (Mateos Naranjo et al. 2006) but elsewhere it is scattered throughout the vegetation (Figure 68). The saltmarshes form a significant component of this important site, which is a Natural Park, Ramsar Site and Special Protection Area designated under the European Union, Bird Directive.

There is no information on its original expansion at this site. The species probably first appeared following its accidental introduction from South America to the Gulf of Cádiz in the sixteenth century (Castillo et al. 2000). In the estuaries in north-west England, the recent invasion of *Spartina anglica* has not resulted in a clamour for its destruction, at least not on ornithological grounds. Although *S. densiflora* may represent a threat to saltmarshes and mudflats elsewhere in southern Europe, it is difficult to see what can or should be done to curtail its expansion. The Odiel Estuary is sufficiently large to support a rich bird fauna, including international important wintering waterfowl. Thus the appearance of this alien may have changed the nature of the estuary without destroying individual components including its birdlife.

The situation for some of the other alien species, especially *Spartina alterniflora* is less clear. The speed of invasion, its hybridisation with native species in the USA, the knock-on effects for nature conservation and impact on commercial interests may point to the need for control. However, for many of the sites, invasion is relatively recent. Given time, the situation in the UK suggests that invading *Spartina anglica*, at least, can become an acceptable component of the saltmarsh habitat. By allowing time, natural processes such as ‘die back’ or succession can lead to ‘incorporation’ of the species into the ‘natural’ habitat.

Figure 68 *Spartina densiflora* in a matrix of *Sarcocornia perennis with Atriplex portulacoides*, Odiel Estuary, Spain