

Macrophytes in the western Wadden Sea: monitoring, invasion, transplantations, dynamics and European policy

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Historic surveys of seagrass beds in the Dutch Wadden Sea were made in 1869, 1931 and 1972/1973. Annual quantitative analysis during 1995-2004 of seagrass-monitorings showed that the beds are highly dynamic. In the Balgzand area (western Wadden Sea), a dominance of eelgrass (*Zostera marina*) was recorded in the 1930s, followed by a dominance of dwarf eelgrass (*Zostera noltii*) in the 1970s. At present, the area is dominated by low densities of widgeon grass (*Ruppia maritima*), that has invaded the area in 2002 approximately. This sequence of macrophytes might be correlated to increasing soil level due to sedimentation (GIS-analysis of monitoring in 1930s, 1970s and 2000s), but a changed salinity regime may also have been of influence. Near the seaward edge of the *Ruppia* bed, reintroduced dwarf eelgrass (planted in 1993) and eelgrass (planted in 1999, 2003 and 2004) lead a vulnerable existence. The highly variable survival rates underline the importance of spreading of risks of reintroduction programmes, both in time and space. This spreading of risks is also a general population strategy of *Zostera*, and the resulting high population dynamics imply that a large buffer zone around the beds should be protected to allow for new colonisations. This is recommended to be included in EU directives.

Key words: Invasions, monitoring, policy, population dynamics, seagrass, trend analysis, water plants

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Introduction

Seagrass has played an important role in The Netherlands. Until the early twentieth century, hundreds of families earned a living from the collection and harvesting of the robust form of eelgrass that grew around Low Tide level (LT) or deeper. It was used as isolation and filling material, and until the eighteenth century also for dike construction. Due to its

former economic importance, historic maps of seagrass distribution from 1869 and 1931 are available (Oudemans et al. 1870, Reigersman et al. 1939). In the early 1930s, the robust form of eelgrass disappeared from the Wadden Sea. This was attributed to a seagrass disease, the closure of the Zuiderzee, two subsequent years of sunshine deficit or a combination of those three (Giesen et al. 1990a,b, den Hartog 1996). In the early 1970s, den Hartog & Polderman (1975) inventoried intertidal seagrass beds in the

Dutch Wadden Sea (dwarf eelgrass and the remaining flexible form of eelgrass). From 1995 onwards (and incidentally in previous years), seagrass beds are monitored on a yearly basis in the Dutch Wadden Sea by the Department of Public Works within the framework of the biological monitoring program (www.zeegras.nl). In 2002, widgeon grass colonised an area of more than 200 ha in the western Wadden Sea, in low densities (less than 1% cover). In this area, seagrasses got extinct in the mid 1970s. Since water clarity improved at the end of the 1980s, possibilities for reintroduction were investigated and transplantations were carried out in 1993, 1998, 2003 and 2004.

In this paper we will relate water plant distribution to location depth (tidal height) using maps from the 1920s onward, to gain insight in the depth distribution of the species eelgrass (two forms), dwarf eelgrass and widgeon grass in the western Dutch Wadden Sea (Balgzand). Secondly, we will analyse the dynamics of natural populations, and thirdly, we will summarise the transplantation results. This will lead to a number of policy recommendations.

Water plants at Balgzand 1931-2002

At Balgzand, in the westernmost part of the Wadden Sea, vegetation was mapped in 1931 by both Reigersman et al. (1939) by boat, and by Harmsen (1936) by foot, presumably also in 1931, or in 1932, the paper is not clear about this. In 1972, den Hartog and Polderman (1975) mapped the area, and in 2002 it was mapped again by van 't Veer, after the invasion of a new species in the area, widgeon grass (Figures 1 and 2).



Figure 1. Map showing the Dutch Wadden Sea with present locations of seagrass beds.

The maps were digitised by ArcMap 8.2. These distribution maps were related to tidal depth maps from the Ministry of Transport, Public Works and Water Management. These maps were made by sounding from a boat, and have an accuracy of ± 0.10 m. Tidal depth data were used for the period 1926-1934, 1971-1974, and 1997-2002. Tidal depth maps were converted from ASCII to grid using

ArcToolbox 8.2, and subsequently to feature by a spatial analyst (ArcMap 8.2). Grids were 20x20 m.

In the 1930s, in the Wadden Sea, but also in the Thames estuary, the following zonation of *Zostera* species was encountered: in the highest (shallowest) zone dwarf eelgrass occurs, followed by a zone of the flexible form of eelgrass, an un-vegetated zone and a zone of the robust form of eelgrass (Wohlenberg 1935, Harmsen 1936, van Katwijk et al. 2000). In the Balgzand area, Reigersman et al. (1939) and Harmsen (1936) only mapped the eelgrass beds (Fig. 2a and b). Note the difference in areas mapped in 1931/2 by Reigersman et al. (1939) and by Harmsen (1936). The difference is probably due to the different aims and methods: Reigersman et al. had an economical interest, i.e., only in the robust type eelgrass growing around LT and deeper, and mapped the area from a boat; Harmsen (1936) had a botanical interest, and mapped the area by foot and omitted water covered areas (see Fig. 3).

In the 1970s both species of seagrass were mapped (Polderman & den Hartog 1975), dwarf eelgrass appeared to have been slightly dominant over eelgrass. In 2000, for the first time, a few widgeon grass patches had been discovered at Balgzand by Rob Dekker (personal communication), who frequents this area at least yearly since mid-1990's. In 2002 and 2004, 225 and 264 ha of widgeon grass were recorded, respectively (Groeneweg 2004a). Densities were less than 1%. The sequence of water plant species was correlated to tidal depth (Fig. 3).

During this period, the investigated area silted up due to sedimentation, resulting in decreased tidal depths (Table 1). The optimal depth ranges of the water plants in the Dutch Wadden Sea and particularly Balgzand are listed in Table 2, and visualised in Figure 3. Most of the seagrass beds mapped in the western Wadden Sea in 1931 were located subtidally with an optimum depth of around 1 m below MSL (Mean Sea Level) or 0.4 m below LT (Table 2 and 3).

This corresponds with recordings of Feekes (1936 in de Jonge & de Jong 1992). Ninety percent of the seagrass beds were located subtidally. This contrasts with the 44% that de Jonge & Ruiter (1996) calculated on the basis of nautical maps. Perhaps this difference is due to the unavailability of the detailed bathymetric maps at the time of de Jonge and Ruiter's study. Of interest is the higher optimum of the seagrass beds at Balgzand in comparison to the total Wadden Sea, MSL -0.7 versus -1.0 m, respectively (Table 2 and 3). When related to low tide level, the difference is less: LT -0.20 and -0.40 m, respectively. The zone with maximum cumulative wave dynamics roughly corresponds with these depths (van Katwijk & Hermus 2000). Further analysis of the maps in relation to exposure to waves and currents, and in relation to

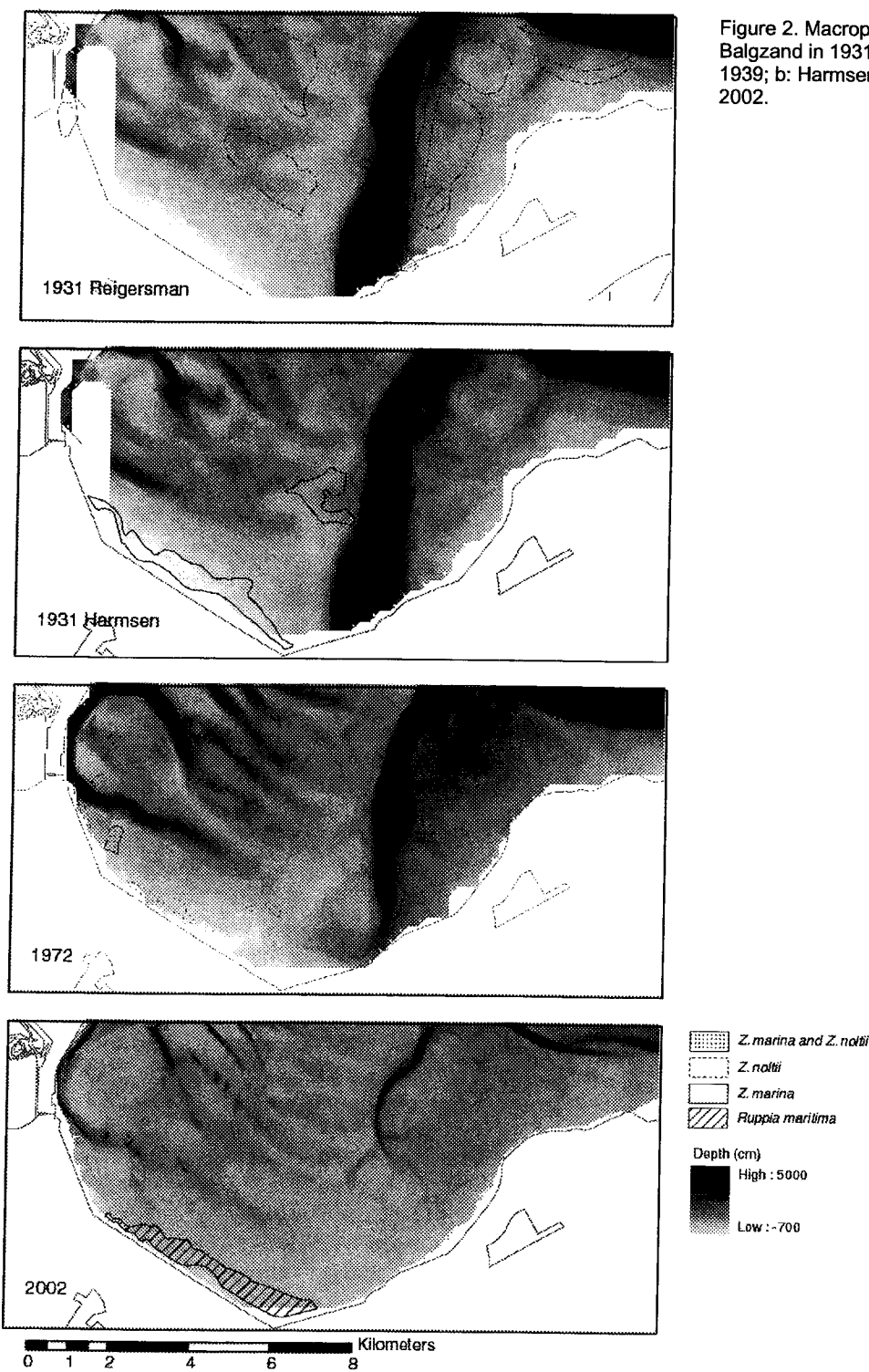


Figure 2. Macrophyte distribution at Balgzand in 1931 (a: Reigersman et al. 1939; b: Harmsen 1936); c: 1972 and d: 2002.

available substratum at each depth level, could offer explanations for the depth distribution of the seagrass beds in the 1930s.

The unvegetated zone that was found during the 1930s at several locations in the Wadden Sea, but also in the Thames estuary (Harmsen 1936, van Katwijk et al. 2000), appeared to have been located between MSL -0.25 and -0.4 m in the Balgzand area. This depth range of the un-vegetated zone as derived from the GIS analysis of seagrass and bathymetric maps, is consistent with field observa-

tions noted in literature, i.e. circa -0.20 below MSL and one or two decimetres above LT (van Goor 1920, Wohlenberg 1935, Harmsen 1936, Klok & Schalkers 1980, Boley 1988, van Katwijk & Hermus 2000). This consistency between the notes of eye-witness-scientists and the calculations performed in this study indicates that the data used and the analyses are sufficiently reliable, notwithstanding the inaccuracies in the sounding method and positioning.

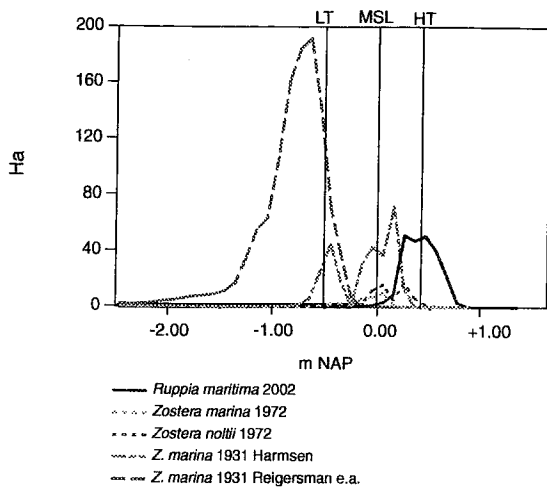


Figure 3. Depth distribution of water plants in Balgzand in 1931 mapped by Reigersman et al. (1939) and by Harmsen (1936), in 1972 mapped by den Hartog and Polderman 1975 and in 2002 by R. van 't Veer (unpubl.). LT: Low tide level, MSL: Mean Sea Level, HT: High tide level (Klok & Schalkers 1980).

Table 1. Depth distribution of the coastal zone of Balgzand compared between the 1930s, 1970s and 2000s. In this GIS-analysis we selected grid cells that (a) had *Ruppia maritima* present in the 2000, and (b) had depth data available in the 1930s.

	area < 0.20 m MSL (ha)	area >= 0.20 m MSL (ha)
1930s	200	1
1970s	43	139
2000s	16	185

Table 2. Tidal depth optima of water plants in the western Wadden Sea, and in Balgzand in particular, on the basis of a GIS-analysis of water plant maps and tidal depth maps.

Macrophyte	Depth optimum in cm MSL	Depth optimum in cm LT
Eelgrass Waddenzee 1931, by boat ¹	-100	-40
Eelgrass Balgzand 1931, by boat ¹	-70	-20
Eelgrass Balgzand 1931 location A ²	-15 to +14	
Eelgrass Balgzand 1931 location B ²	-50	
Eelgrass Balgzand 1972 ³	-25 to -5	
Dwarf eelgrass Balgzand 1972 ³	-5 and +20	
Widgeon grass (van't Veer, this study)	-15 to +54	

¹Vegetation map of Reigersman et al. 1939

²Vegetation map of Harmsen 1936

³Vegetation map of Polderman & den Hartog 1975

Table 3. Average high tide (HT) and low tide (LT) level in the Balgzand area and in the seagrass beds in the Wadden Sea in 1931 (based on data of Klok & Schalkers 1980 and the seagrass bed map of Reigersman et al. 1939).

	m LT	m HT
Balgzand	-0.50	0.40
Wadden Sea seagrass beds 1931	-0.60	0.40

The settlement and expansion of widgeon grass in recent years may be explained by the decreased tidal depth following sedimentation (table 1); also in Chesapeake Bay and in the Baltic Sea, widgeon grass grows generally shallower than eelgrass (Orth & Moore 1988; Batiuk et al. 1992; Boström & Bonsdorff 2000, Moore et al. 2000). Obviously, the correlation is no indication for causality. The invasion of widgeon grass may indicate a lowered salinity, as this species has a lower salinity optimum than eelgrass (Verhoeven 1979, van Katwijk et al. 1999, Moore et al. 2000, La Peyre 2003). In the 1930's, in the Wadden Sea and Zuiderzee, the salinity range of eelgrass was 10-30 PSU (comparison seagrass maps of Oudemans et al. 1870, Reigersman 1939 with salinity data van der Hoeven 1982). At present, at the Balgzand the salinity drops frequently to 10-15 psu and occasionally as low as 5 psu, as appeared from a continuous monitoring program during 2005. The salinity drops were related to the discharges from Lake IJssel in combination with easterly winds (van Reen 2005). There are no indications that the discharge regime has changed during the last decades, though (van Reen 2005, www.waterbase.nl).

Dynamics in present natural populations in the Dutch Wadden Sea.

Since mid-1990s, four seagrass beds in the Dutch Wadden Sea have been monitored on a yearly basis (e.g. Groeneweg 2004b, Erfemeijer 2005, Fig. 1) by the Department of Public Works within the framework of the biological monitoring program. One of these beds, the eelgrass bed at Terschelling Harbour, had disappeared in 2003 (see also Fig. 4). A new bed has appeared in the Ems estuary, across a channel, 4 km west of the eelgrass population of 'Hond/Paap'. This area, called "Voolhok" is an area with high sedimentation rates. The area probably receives seed from the Hond/Paap beds since long, but only recently, the tidal depths have decreased sufficiently to provide a suitable habitat for germination and bed development. The bed was discovered in 2003 and was not present in 1999. Apart from the beds mentioned above, there are no significant seagrass occurrences in the Dutch Wadden Sea, except the small transplants of eelgrass and dwarf eelgrass at Balgzand, mentioned above.

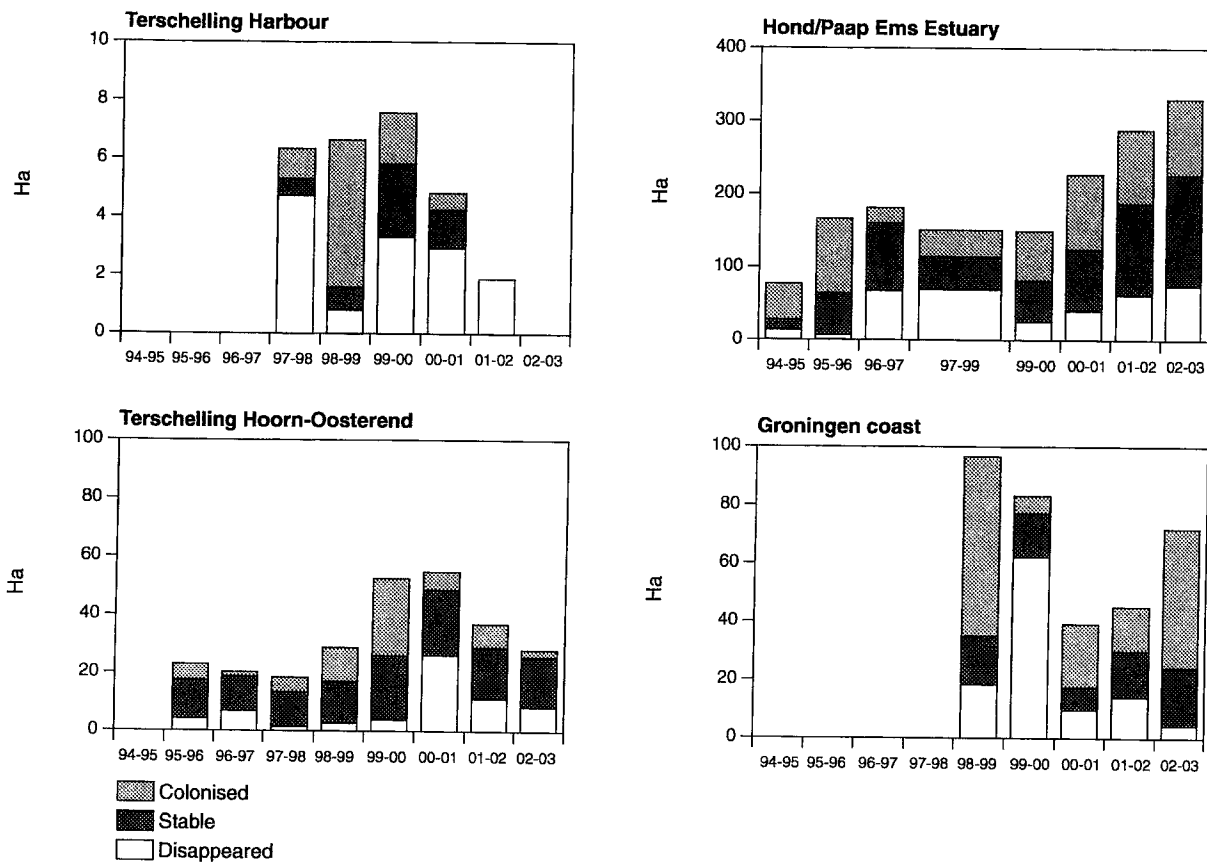


Figure 4. Dynamics of four seagrass beds in the Dutch Wadden Sea. Area of seagrass beds with 'colonised': area that is newly colonised in comparison to the preceding year, 'stable': area that was also vegetated in the preceding year, and 'disappeared': the area that was vegetated in the preceding year but not in the present year

The dynamics of the four seagrass beds in the Dutch Wadden Sea were analysed, using the monitoring data provided by the Ministry of Transport, Public Works and Water Management. Data of the following years were available and have been analysed: Terschelling Hoorn-Oosterend: 1995-2003; Groningen coast: 1998-2003; Terschelling harbour: 1998-2002; Hond/Paap in the Ems estuary: 1994-2003 with 1998 missing; we analysed 1999 data in comparison to 1997 data instead. Inaccuracies in monitoring can rise from the timing of the aerial and ground surveys, the spatial resolution of map data, the consistency in interpretation accuracy of aerial photographs and, for the category 0-5% cover that is hardly visible on aerial photographs, also the limited chance of detection in ground truthing (Frederiksen et al. 2004, Erfteijer 2005, D. de Jong, pers. comm.)

Using GIS-analysis, we calculated the differences in surface area between two subsequent years, and made a distinction between newly colonised areas (areas that were not covered by seagrass in the preceding year), stable areas (that had seagrass cover in both years), and areas where seagrass had disappeared (local extinction; Fig. 4). The average percentage 'colonised', versus percentage 'stable', was

relatively constant per population (no large standard errors of the mean, Fig. 5).

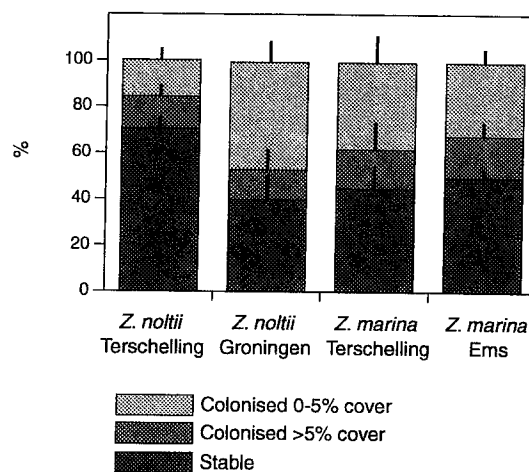


Figure 5. Percentage of the seagrass area that was newly colonised versus the area that was already covered by seagrass in the preceding year in the four seagrass localities present in the Dutch Wadden Sea. Newly colonised areas are subdivided in <5% and >5% seagrass cover.

In three out of four populations the newly colonised area was between 50 and 60% of the total population. The 'colonised' areas mainly have a cover of less than 5%. Only the dwarf eelgrass population of

Terschelling near Hoorn and Oosterend was less dynamic: at average only 30% of a seagrass area in a given year was (re-) colonised and the percentage 'stable' area (i.e. that was also seagrass-covered in the preceding year) was 70% at average. As this population shows no net losses or increases (Erfte-meijer 2005), the yearly extinction in the covered area is also about 30%, which can also be seen in Fig. 4. The low extinction percentage may have been due to the compact, stabile sediments at this location (drown salt marshes) in combination with the predominantly perennial strategy of dwarf eelgrass. The low (re-) colonisation percentage indicates that the number of suitable areas in the vicinity is limited.

The high (re-) colonisation area in most seagrass beds of 50-60% yearly, means that if only the existing beds are protected, one would potentially lose 50-60% of the population in the subsequent year. The risk is particularly high in winter and spring, when the new colonisations are not yet visible. Secondly, in summer when new monitoring results are not yet present and available, the risk of overlooking a new colonisation is present when the beds are sparsely covered, i.e. <5%. At average the area with risk of overlooking was calculated to be 30-50% of the seagrass covered area (except for the Hoorn/Oosterend dwarf eelgrass population: 15%; Fig. 5). Then, in summer, a new monitoring is performed and seagrass area (= area of protection) is adjusted to the new situation. When monitoring is not performed on a yearly basis, the risk of losses increases further. This follows from the situation that the protected area will be partly un-vegetated, the unprotected area will be partly vegetated as a consequence of the yearly bed dynamics.

Notably, comparison of historic maps of 1869 and 1931 reveals that also the subtidal, perennial form of seagrass showed large dynamics: 75% of the seagrass beds present in 1931 were a new colonisation compared to 1869, whereas 25% had remained at, or re-vegetated the same location during these 72 years. 55% of the vegetation present in 1869 had disappeared in 1931 (the seeming contradictions in these percentages are due to an increase in the total seagrass cover during the period 1869 and 1931).

Transplantations of eelgrass and dwarf eelgrass at Balgzand, 1993-2004

As part of a large reintroduction programme (van Katwijk et al. 2000, van Katwijk 2003, Bos et al. 2005, Bos & van Katwijk subm.), seagrass transplantations have been carried out in 1993, 1998, 2003 and 2004. In 1993 both eelgrass and dwarf eelgrass were introduced. Dwarf eelgrass transplantations (methods: 1x1 m, 100 plants, planting date May 19th 1993, planting depths between MSL -0.4 and + 0.3 m; survival in 1994 reported between MSL -0.1 and + 0.15 m, Hermus 1995) appeared to have been successful,

as they have been and are still spreading since (Fig. 6), though all colonisations are 1-2 m in diameter,

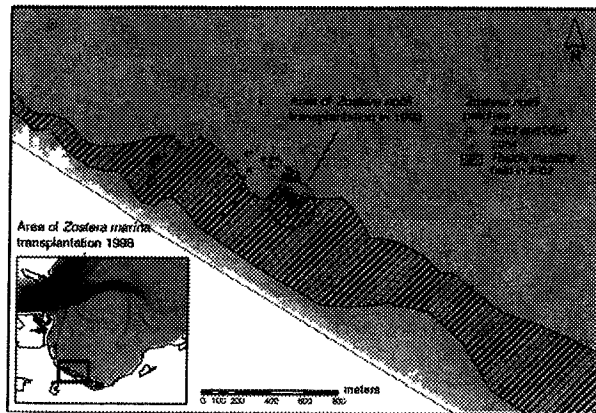


Figure 6. Dwarf eelgrass in 2002 and 2004, transplanted in 1993 at the encircled location; Inserted: location of eelgrass transplantation in 1998.

and do not form beds, yet. In 1993, eelgrass transplantations were successful only during the first growing season and did not survive through the winter, neither by surviving plants, nor through recruitment by seed (van Katwijk & Hermus 2000). Transplantations at the same location in 2003 gave similar results (Bos & van Katwijk 2005). In December 1998, seed bearing shoots (circa 5 kg wet weight) were deposited at another location within the Balgzand area (Fig. 6, details of the area see van Katwijk & Wijgergangs 2004). This transplantation was successful. However, the numbers of plants are highly variable, making this transplantation vulnerable to extinction (Table 4).

Table 4. Development of a transplantation of eelgrass plants at Balgzand, resulting from a donation of seed bearing shoots in December 1998.

year	number of plants	ha
1999	Ca. 100	
2000	Ca. 300	
2001	Ca. 200	
2002	13	
2003	Ca. 800	5.3
2004	50	

Conclusions and recommendations

From monitoring data covering 75 years, we could relate the sedimentation at Balgzand (decreased tidal depth) with a sequence of water plant species dominance in the area (eelgrass dominance, followed by dwarf eelgrass dominance, followed by widgeon grass dominance, Table 1 and 2, Fig. 3).

This temporal relationship corresponds to known spatial relationships (zonation): also in Chesapeake Bay and in the Baltic Sea (Orth & Moore 1988; Batiuk et al. 1992; Boström & Bonsdorff 2000, Moore et al., 2000). *Ruppia* grows shallower than *Zostera*, and dwarf eelgrass is known to occur shallower than eelgrass in the rest of the Wadden Sea, but also in the UK and Brittany, France (Harmsen 1936, Wohlenberg 1935, den Hartog & Polderman 1975, personal observation), with exception of an eelgrass zone with a wintergreen perennial strategy, that occasionally can be encountered in a zone above the dwarf eelgrass zone (van Katwijk et al. 1998). This consistency between the temporal relationship and the zonation patterns suggests that this relationship may be causal. In other words, the invasion of widgeon grass may have been caused by the sedimentation in the Balgzand area. However, based on the depth profiles, one would have expected the invasion sooner: already in the 1970s, the area had silted up considerably (Table 1).

Additionally, other explanations for the observed sequence of macrophyte cover over 75 years are possible. For example, it is known that widgeon grass has a much lower salinity optimum than *Zostera* species (e.g. Verhoeven 1979, Moore et al. 2000). Low salinities have been measured at the Balgzand area (van Reen 2005), offering a possible explanation of the presence of *Ruppia maritima*. However, also in this case an invasion would have been expected earlier as the discharges have not increased or decreased during the last 30 years (www.waterbase.nl). Distributional impairments could have caused the delayed establishment: seeds can travel by birds (Figuerola & Green 2002, Figuerola et al. 2002), but only occasionally this may result in a successful establishment (Clausen et al. 2002). Another means of transportation would be detached shoots bearing seeds (Cho & Poirrier 2005), which can probably travel over large distances, as was found for eelgrass (Harwell & Orth 2002, Reusch 2002, Erftemeijer et al. *subm.*). Once established, that plants can expand rapidly (Silberhorn et al. 1996, Cho & Poirrier 2005). One may also tentatively speculate that the low general environmental quality during the 1970s and 1980s (high levels of turbidity, eutrophication, heavy metals, toxicants, Marijnissen et al. 2001, de Jonge & de Jong 2002, van Beusekom & de Jonge 2002), may have prevented an earlier establishment of *Ruppia*.

Our GIS-study shows that seagrass bed dynamics are high. Between 1869 and 1931, only 30% of the subtidal beds of the robust type of eelgrass had remained at the same location, whereas 70% had 'moved'. Additionally, the population had expanded with a 25% increase in surface area (note that these are net values: we do not know the dynamics in the intermediate years). Present day seagrass beds, composed of the

flexible type of eelgrass and dwarf eelgrass, show yearly shifts of 50-60%. Below, we will elaborate the implications of these dynamics to protection measures.

The importance of regulating shellfish fisheries to effectively protect seagrass beds was shown by de Jonge & de Jong (1992), van Katwijk (2003) and Essink et al. (2003). At the Groningen coast, Essink and co-workers recently recorded the loss of a substantial part of the dwarf eelgrass population following shellfisheries activities. They recommend a buffer zone of 400 m for the Groningen coast. Our study of the observed year-to-year dynamics of the Dutch Wadden Sea seagrass populations indeed urge the need for protection of a larger area than only the present seagrass bed. By doing the latter, one may potentially lose more than 50% of the bed each year (Fig. 5). We recommend that the buffer zone should be established on the basis of the area surrounding the beds that can be considered as suitable for seagrass. The latter can be accomplished by using the habitat suitability model (de Jong et al. 2005, Bos et al. 2005), and on-site expert judgement. Additionally, long-term trend analyses of the bed dynamics (e.g. Erftemeijer 2005, www.zeegras.nl) could help to establish the potentially suitable areas surrounding seagrass beds.

To allow for a larger scale expansion of existing seagrass populations, protection of high potential seagrass areas remote of existing beds is recommended. The recent establishment of a bed more than 4 km remote from an existing bed proves that seagrass is capable to establish outside the direct margins of existing beds. To assign high potential areas, the habitat suitability model for seagrass in the Dutch Wadden Sea can be used (de Jong et al. 2005, Bos et al. 2005). Within the areas appointed by the model, a further refinement should be made at the site, to account for local circumstances. Both the establishment of large buffer zones surrounding seagrass beds and the establishment of protected areas at high potential areas are recommended to be incorporated in EU-regulations and in Wadden Sea management plans.

From the results of transplantations at Balgzand in 1993 (dwarf eelgrass) and 1998 (eelgrass), and from the GIS-analyses in this study, we can conclude that both seagrass transplantations and beds are highly dynamic. A dynamic population strategy is obviously the best strategy in a dynamic environment such as the Wadden Sea. From this, we recommend that transplantation programmes should spread risks in space and time, which is basically what natural populations do as well.

Another recommendation that rises from the observed dynamics is to monitor the abiotic variables in the seagrass beds. Correlations between seagrass dynamics and these environmental variables will provide invaluable insight in the habitat require-

ments of these threatened species. Variables of interest are, for example depth, salinity and nutrient concentrations during a tidal cycle and over a season (van Katwijk et al. 2000). Additionally, to get insight in the causes of local disappearances that occur in natural beds (Fig. 5) as well as in transplantations (Table 2, Bos et al. 2004, Bos & van Katwijk 2005), continuous visual monitoring is recommended. In monitoring the transplantations, very sudden disappearances during the growing season were detected. When did these plants disappear exactly? After an abundant visit of foraging birds, or after a period of a particular combination of wind speed and direction? Etcetera. Continuous visual monitoring, e.g. using webcams, is a technique that is presently coming into reach to efficiently provide this information.

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