

Restoration of Coastal Ecosystems



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M. Isermann & K. Kiehl

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Restoration of coastal ecosystems – an introduction

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During recent decades, coastal ecosystems have been increasingly affected by large-scale land-use change and by sea level rise due to global warming (Doody 2001, 2004). Both salt marshes and dunes have an important function in protecting coastal regions from flooding and erosion and provide habitats for plant and animal species with special adaptations for survival under harsh environmental conditions. For successful restoration, the dispersal of target species is less limiting in coastal ecosystems than in most inland ecosystems (e.g. Wolters et al. 2005a, Gallego Fernandez & García Novo 2007). However, restoration can be constrained by difficulties in restoring the natural dynamics of flooding, sedimentation, sand accumulation and erosion processes, and therefore the appropriate environmental conditions on which target species depend (Bakker et al. 2002, Grootjans et al. 2002, Wolters et al. 2005b).

Different processes threaten drift-lines, beach and dune ecosystems, for example the removal of sea-grass and seaweed, recreation activities, coastal protection measures, sea level rise, construction activities and pollution (Doody 2001). During the 20th century, land use changed and both shrub and woodlands, for example with *Hippophaë rhamnoides* and *Rosa rugosa*, spread rapidly and dominate at present large dune areas, forming dense, impenetrable thickets. Therefore, species richness and biodiversity, especially of open dune grasslands, declined (Isermann 2007, Isermann et al. 2007). Restoration efforts lead from dense shrubland to open species-rich dune grasslands (Provoost et al. 2004, Leten et al. 2005). Moreover, grazing (over- and undergrazing) of grey and brown dunes e.g. by cattle and rabbits, has still important effects on species composition as well as on habitat heterogeneity (Doody 2001). During the last decades, grasses such as *Carex arenaria*, *Deschampsia flexuosa* and *Ammophila arenaria* as well as mosses like the neophyte *Campylopus introflexus* encroached in grey and brown dunes mainly due to nutrient enrichment (e.g. Kooijman et al. 1998).

Natural processes and species richness of dune slacks were mainly affected by lowering of the water table due to drinking water production. Restoration measures, already established since the middle of the 20th century, included for example the re-introduction of traditional management techniques in dune slacks, such as mowing, grazing and sod removal, or construction of artificial dune slacks to compensate for biodiversity loss elsewhere (Grootjans et al. 2002).

Salt marshes and brackish coastal communities in Europe have been endangered due to large-scale embankments and intensive agricultural land use in the past (Dijkema et al. 1984, Stock et al. 1997, Bakker et al. 2002, Bakker & Piersma 2006). During the last decades, however, several projects on the restoration of coastal marshes by de-embankment and dike-breaching have been started at the North Sea (Crooks et al. 2002, Wolters et al. 2005b) and the Baltic Sea (Bernhardt & Koch 2003, Seiberling 2003) as well as in Southern Europe (e.g. Gallego Fernandez & Garcia Novo 2002 & 2007). Furthermore, European salt marshes were affected by land use changes during the last decades. At the Baltic Sea and in brackish marshes of the Wadden Sea area, the abandonment of traditional land use by mowing or grazing favours the spreading of *Phragmites australis* and other tall species, which can have a negative effect on low-growing halophyte species (Dijkema 1990, Jutila 2001, Esselink et al. 2002). Here, restoration efforts have to focus on land-use optimisation in relation to local environmental conditions.

In the Wadden Sea salt marshes, restoration targets have differed between countries over recent decades. In the Netherlands, nature conservation and restoration focussed on the maintenance and re-establishment of high plant species density (e.g. Bakker 1989, Bakker et al. 1997), whereas in the German National Parks the main target was to enhance natural dynamics both concerning abiotic processes as well as species interactions (Stock 1997, Stock et al. 1997, Bakker et al. 2005). In the Netherlands, re-introduction of cattle grazing in species-poor abandoned high salt marshes dominated by *Elymus athericus* led to an increase in plant species density but in the low marsh grazing and trampling had a negative effect on species density (Bakker 1989, Bakker & Piersma 2006). In the German Wadden Sea area, large-scale grazing reduction and abandonment was necessary to allow the spreading of typical salt marsh species, which had become rare after decades of very intensive grazing (Kiehl et al. 1996, Stock et al. 1997). Up to now, salt marsh restoration by grazing cessation and reduction in combination with neglecting the artificial drainage systems, showed mainly positive effects on species density and establishment of target species (Schröder et al. 2002, Kiehl et al. 2007).

In summary, these examples show that ecological restoration of beaches, dunes and salt marshes includes the maintenance and improvement of natural environmental dynamics and the restoration of biodiversity through the promotion of habitat-specific target species and by favouring environmental heterogeneity and landscape diversity. In practice, ecological restoration of coastal ecosystems has to comprise coastal conservation policies, technical aspects of restoration activities and habitat management as well as scientific research on the effectiveness of restoration and conservation measures (e.g. Bauer et al. 2004, Wolters et al. 2005, Zedler 2005).

The Fifth European Conference on Ecological Restoration of the Society for Ecological Restoration (SER) focused on “Land use changes in Europe as a challenge for restoration: ecological, economical and ethical dimensions” and was held from 21 to 25 August 2006 in Greifswald, Germany. During this conference, a special symposium on the restoration of coastal ecosystems was organised with sessions on saltmarshes and dunes. Six contributions from this symposium are published in this volume. They cover a broad field from restoration recommendations for mangroves in relation to their historic distribution (Kholeif 2007); vegetation studies on permanent plots for the evaluation of large-scale land-use changes in Wadden-Sea saltmarshes (Kiehl et al. 2007); the significance of ground beetles and spiders as indicators for the evaluation of salt-marsh and dune restoration (Desender et al. 2007, Maelfait et al. 2007); to studies on restoration of dune ecosystems for the promotion of target species (van Til & Kooijman 2007) and landscape perception (García-Novo et al. 2007).

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Palynology of mangrove sediments in the Hamata Area, Red Sea Coast, Egypt: vegetation and restoration overview

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Abstract

The present mangrove vegetation condition in the Abu Ghoson area of the Red Sea coast, Egypt can be recognised through field observations and palynological analysis of the intertidal flat and swamp environments, as well as through cores. In addition to illustrating the pollen distribution on the mangrove site, special attention is paid to *Avicennia marina* pollen distribution, which is the only mangrove pollen species in the investigated area. The maximum concentration of *Avicennia marina* is recorded ($> 9000 \text{ pcm}^2$) in the sediments which are taken in proximity to the aerial root (pneumatophore) of mangrove plants. Away from the area of mangrove roots and further seaward, it became rare to absent. This indicates that the pollen grains are inefficiently dispersed by the water current, which is considered the main tool of pollen dispersal away from the area of pollen production. The variation of mangrove pollen frequency in the core sediments can help to understand the vegetational changes associated with paleogeographic modification. In addition to poor pollen production, low catchment run-off and intense browsing have caused high stress conditions in mangrove communities. The threat to mangrove plants by extinction, hydrological changes, and physical stress from resorts around the mangroves would require assessment.

1 Introduction

Mangroves are the most important part of the coastal habitat, which occur in patches along the intertidal zone. Mangroves play a vital role in the health of the marine ecosystem, because they filter and block sediments from the unstable substrates of the wadi and desert, and preventing siltation of sea grass beds and coral reefs. One of the most important roles of mangroves is to increase oxygen into the marine environment. They also eliminate pollutants such as sewage waste, pesticide run-off, and toxins in waste dumped in the wetlands.

This paper gives a precise picture of the current situation of the mangrove vegetation carried out by palynological study and field observation of Abu Ghoson area on the Red Sea coast.

1.1 Mangrove reproductive strategies

Mangroves have a unique characteristic of spreading seedlings for forest continuation (Tomlinson 1986). Generally, the formation of new stands of mangroves, away from the area of origin, depends on the seedlings being carried by tidal action. Their ability to survive periods of immersion allows them to invade suitable coastal niches where they can establish and commence the development of a new formation.

Mangroves exhibit two relatively distinctive reproductive strategies: hydrochory and vivipary (Rabinowitz 1978, Tomlinson 1986). Hydrochory (dispersal by water) is a major means by which mangrove spreads seeds, fruit, or propagules. Tidal action can carry mangrove diaspores long distances from their point of origin. Vivipary means producing seeds that germinate while still attached to the parent plant for 4 to 6 months (McKee 1996). During germination, the seedling receives nutrients and water from the parent tree and it grows uninterrupted until, becoming too heavy to bear, it falls off into the trap of pneumatophores beneath the parent tree whilst others are dispersed by tides nearby or further away. If conditions are appropriate for growth, a new community is established. The seedling feeds

off the propagule until its roots develop sufficiently to anchor it in the mud and nutrients are obtained from the substrate.

1.2 Area of study

Abu Ghoson area (350 km²), lies on the Red Sea coast, south Marsa Aalm city (Fig. 1). It is a nature reserve created by Prime Ministerial Decree 143 of 2003. The coastline of the area is retreating inland towards the wadis. Mangrove plants are found in narrow spots along the shore and are extensive in the swamp area behind the coastline (Fig. 2). The mangroves range in length from 5-30 cm tall, except one tree (~ 4 m tall). The substrate is enriched by lime mud in the swamp area and in some areas along the coastline. In other areas, it becomes sandy (medium to coarse grain sand). The intertidal flat extends from the shoreline 60m off-shore and appears to be closed by the natural conglomerate bar at the end. In some localities, there are some small shrubs and remaining ares of low aerial roots.

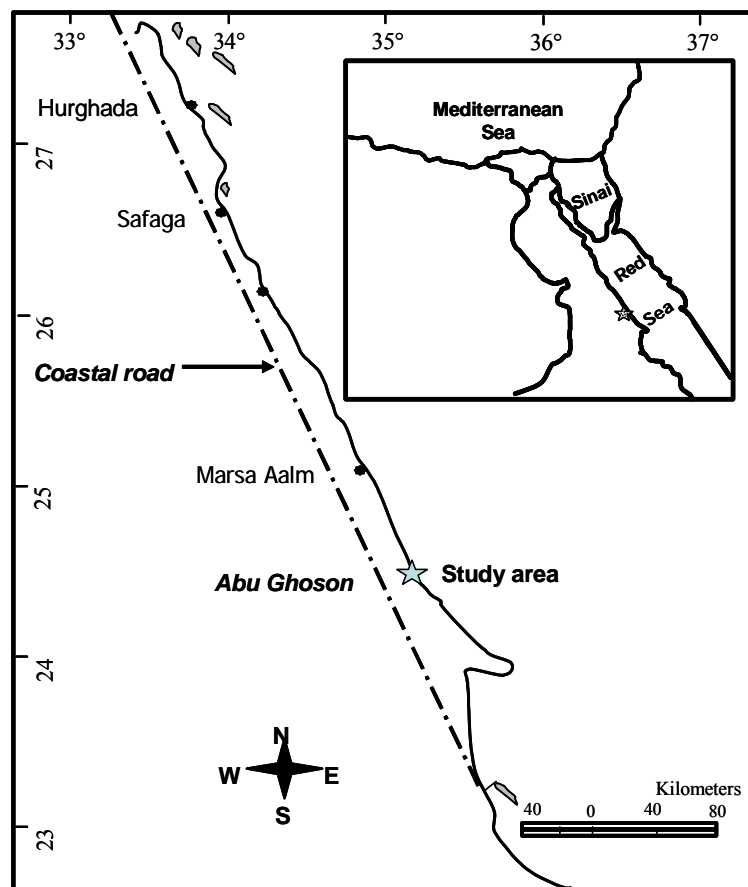


Figure 1: Location map of study area.

1.3 Material and Methods

Thirty-two sediment samples were collected from the swamp area, intertidal zone and from core 2 (14 cm deep) for palynological analysis. Sediments consisted mainly of lime mud and fine to medium grain sands. The intertidal transect covering the whole area, ran West-East starting from the shoreline to off-shore and was located between Latitude 24° 21' 42" and 24° 21' 29" N and Longitude 35° 18' 18" and 35 18' 23" E . The samples were collected (~50 m apart) by spoon and the readings of Hydro-lab and GPS were detected (Table 1).

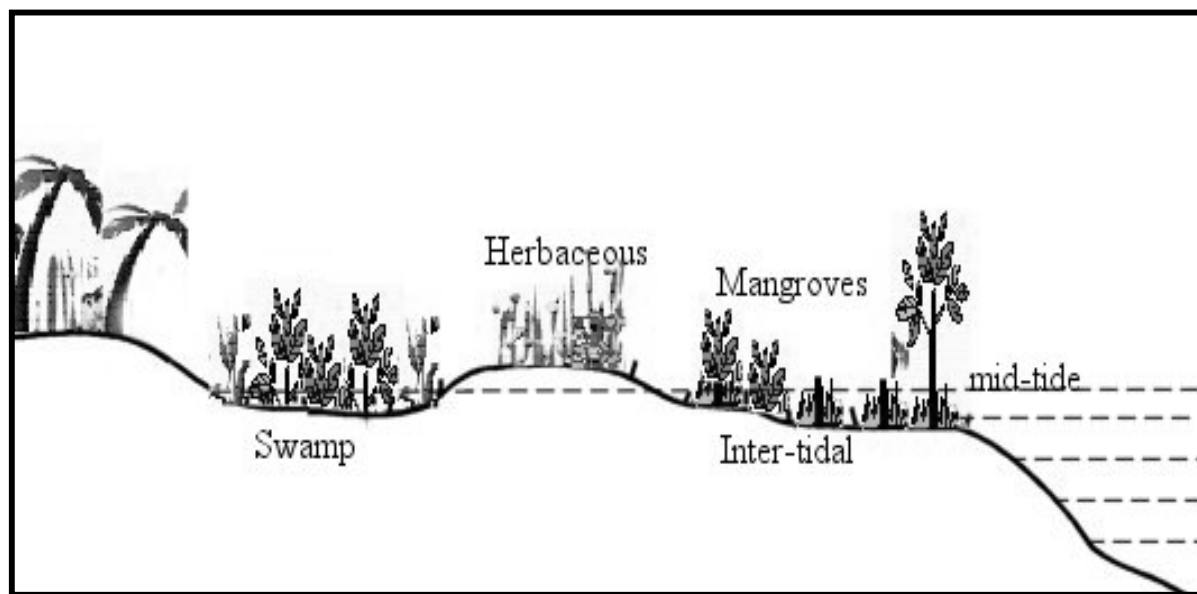


Figure 2: Simple sketch diagram showing the distribution of mangrove in the area of study.

The swamp transect collected from the area was located between Latitude 22° 21' 30" and 24° 21' 32" N and Longitude 35° 18' 12" and 35° 18' 13" E and covered the entire swamp spot. The upper three centimetres of sediments were nominated for palynological analysis. The cores samples were selected every 2 cm depth. All samples were treated chemically for pollen extraction following the standard palynological technique, which includes the use of HCl to remove carbonates and HF acid to remove silicates. The ultrasonic cleaner and 10 µm Nylon sieve were used to eliminate the finest particles.

The Lycopodium marker spore tablets were added in the first step of the technique to calculate the number of pollen in a sample using the following equation:

$$\text{Lycopodium spores added} / \text{Lycopodium spores counted} * \text{fossil pollen counted} = \text{no. of pollen per cubiccm}$$

Concentration was defined as pollen per cubic cm (millilitre). The calculation method of Stockmarr (1971, 1972), Maher (1981) and Bennet & Willis (2002) was applied to calculate the pollen concentration per cm². All sediments, palynological slides, and residue were depicted in the Geological lab at NIOF, Alexandria, Egypt.

Nearly 30 species belonging to 22 families were identified. Some of pollen genera could not be assigned to the species level. All pollen grains were counted and put in the tables (2-3), in addition the most abundant pollen families concentration were plotted (Fig. 3-5). Some photographs of some species were taken (Fig. 5-6).

Table 1: Abu Ghoson studied samples; T, S, DO refers to measurements of water temperature ($^{\circ}\text{C}$), Salinity (ppt), Dissolved Oxygen content (mg/l).

Sample no.	Water Depth [cm]	Latitude N	Longitude E	Hydrolab reading			
				T	S	PH	DO
Intertidal Transect							
1	30	24° 21' 34	35° 18' 10	-	-	-	-
2	50	24° 21' 36	35° 18' 11	22.4	41.4	8.6	6.6
3	70	24° 21' 40	35° 18' 12	23.0	41.3	8.5	7.4
4	20	24° 21' 29	35° 18' 23	22.4	41.3	8.5	6.6
5	40	24° 21' 36	35° 18' 19	22.4	41.4	8.6	6.6
6	50	24° 21' 39	35° 18' 19	22.0	41.2	8.5	7.3
7	80	24° 21' 32	35° 18' 22	21.6	41.3	8.5	6.4
8	120	24° 21' 32	35° 18' 22	23.2	41.0	8.6	7.7
9	140	24° 21' 32	35° 18' 21	23.2	41.2	8.6	7.3
Swamp transect							
10	0	24° 21' 30	35° 18' 19	23.4	44.9	8.6	5.2
11	5	24° 21' 30	35° 18' 19	22.5	44.5	8.6	5.5
12	8	24° 21' 30	35° 18' 19	22.4	45.0	8.4	5.6
13	10	24° 21' 30	35° 18' 18	23.0	44.8	8.4	6.0
14	15	24° 21' 29	35° 18' 18	26.9	45.9	8.4	6.7
15	20	24° 21' 32	35° 18' 13	26.0	45.9	8.4	6.5
16	25	24° 21' 32	35° 18' 13	26.2	45.3	8.7	5.8
17	30	24° 21' 32	35° 18' 13	25.9	45.6	8.7	5.3
18	30	24° 21' 32	35° 18' 12	25.6	45.7	8.5	5.4
19	35	24° 21' 32	35° 18' 12	26.0	45.8	8.4	5.1
20	45	24° 21' 32	35° 18' 12	26.3	45.9	8.5	5.4
Core 2		Location					
1	Depth [cm]	24° 21' 32					
2	0-2	35° 18' 12					
3	2-4						
4	4-6						
5	6-8						
6	8-10						
7	10-12						
	12-14						

2 Results

2.1 Mangrove pollen

Avicennia marina was the only mangrove pollen species in the study area. It showed a maximal value near the root of mangrove plants with concentration 9292 per cm^2 (first sample of intertidal transect) and with 6194 per cm^2 at the vicinity of mangrove trees (swamp sample 10). On the other hand, it decreased seaward in the intertidal transect to minimum value 495 per cm^2 (sample 8). All core 1 samples from the intertidal zone were barren of any pollen grains except some Poaceae and microforaminiferal linings. On the other hand, the mangrove species in core 2 sediments, had a value ranging from 4570 to 3700 per cm^2 in the lower part (depths from 10 to 14 cm) but were poorly represented in the samples 1 and 2 (0 to 4 cm depth).

2.2 Other terrestrial pollen

Other pollen families associated with mangrove pollen are mostly herbaceous pollen. These pollen grains are often associated with mangrove ecosystem in Wadi El-Gemal (Kholeif & Khalil 2006). The first, most dominant families came after *Avicennia* pollen in the studied sediments. These were Chenopodiaceae\Amaranthaceae with the maximal value >7600 per cm^2 in the transect sample 2 and

swamp sediments sample 20. It declined to 2425 per cm² at 8 cm depth (sample 4) in the studied core (Table 3). The second best representative families in all productive samples were Poaceae and Asteraceae. Poaceae was well represented in the all swamp samples with the concentration a proportion of 7850 to 588 per cm². The concentration of Poaceae in the examined sediments was inconsistent with the concentration in Wadi El-Gemal mangrove sediments, as it had the highest percent and concentration in the mangrove sediments (KHOLEIF & KHALIL 2006). Asteraceae had a maximal value in the first sample of studied transect (5013 per cm²) and least value (120 per cm²) in the upper core depth (2 cm deep; sample 1).

Palmae came in the third position. It was the main pollen tree family in the Abu Ghoson area. It was well represented in all productive samples and exclusively in the swamp sediments with concentrations ranging from 3562 to 336 per cm². This may have been due to the near zone of Palmae tree from swamp area in the mangrove site.

Fabaceae, Meliaceae, Solanaceae, Polypodiaceae, Plantagnaceae, and Brassicaceae were the next most representative families in the studied samples with significant concentration. The other recorded families were represented by low concentration (Tables 2-3).

3 Discussion and Conclusion

The pollen assemblages contained many herbaceous and terrestrial pollen types beside the *Avicennia marina*, among which were Chenopodiaceae\Amaranthaceae, Asteraceae, Poaceae, Fabaceae, Meliaceae, Solanaceae, Polypodiaceae, Plantagnaceae, and Brassicaceae (Fig. 3-5). A high concentration and low percentage (20 % of the total pollen) of *Avicennia marina* was encountered in most of the swamp sediments, the first samples of the intertidal transect and the lower depth of studied core. Based on the field observations and palynological analysis, the concentration of *Avicennia* in the mangrove site suggested that it is less representative in relation to the *Avicennia* trees. Some causes can be explained: 1) the low pollen production of the mangrove tree, this can be confirmed by the low concentration of mangrove pollen depiction in cores 2 (Fig. 4) taken from swamp area. 2) In general, the distribution pattern of mangrove pollen, and other pollen families can reflect the role of transport processes by marine current at the studied site, which is ineffectual to spread the seed far from the zone of production. This may be due to the unique features of Abu Ghoson intertidal zone, which is nearly separated from off-shore by the conglomeratic barrier. The effective role of the marine current for pollen dispersal is discussed through the study of the other sediments from Wouri estuary (Boyé et al. 1975), sediments from Ivory coast (Fredoux 1980), and continental shelf marine sediments (VAn Campo & Bengo 2004). 3) Absence of mother trees in some areas may result in low or no propagules. The role of monsoon at the studied locality could be responsible for seed dispersal.

In common, the maximum concentration of pollen was recognised during the dry season consistently with the flowering period from December to March (Hutchinson & Dalziel 1972) and in January (Callèja et al. 1993). Hence, we expected the high concentration and absolute abundance of pollen grains in the studied sediments at least near the zone of plant production since the studied samples were collected on the 5th February. This also supported the poor pollen production of the studied mangrove site.

Table 2: Pollen family concentration per cm² in the core 2.

Pollen Family	Sample no.						
	1	2	3	4	5	6	7
Amaranthaceae/ Chenopodiaceae	32	311	511	2425	859	1200	769
Asteraceae	120	331	4310	274	815	524	630
Avicenniaceae	9	225	1210	742	2330	4520	3700
Brassicaceae	0	0	256	375	982	1211	318
Cypraceae	6	0	189	396	247	671	341
Ephedraceae	0	0	98	178	0	0	63
Euphorbiaceae	98	113	0	355	59	200	196
Fabaceae	231	0	567	389	712	1159	2508
Geraniaceae	0	0	149	185	359	216	265
Polypodiaceae	241	162	400	637	305	1310	530
Liliaceae	0	0	132	158	93	271	180
Oxalidaceae	8	89	129	165	210	143	223
Palmae	0	336	510	1298	559	1614	860
Plantagnaceae	0	158	299	611	743	520	780
Poaceae	1152	2195	942	5524	1259	631	7850
Polygonaceae	50	112	230	142	96	176	96
Porate	165	0	210	322	134	543	331
Solanaceae	0	0	133	345	252	163	388
Tiliaceae	0	92	118	353	150	269	188
Meliaceae	270	198	448	367	794	1520	643
Zygophyllaceae	0	0	99	183	395	543	260
Unid pollen	0	0	1210	2159	1199	859	624

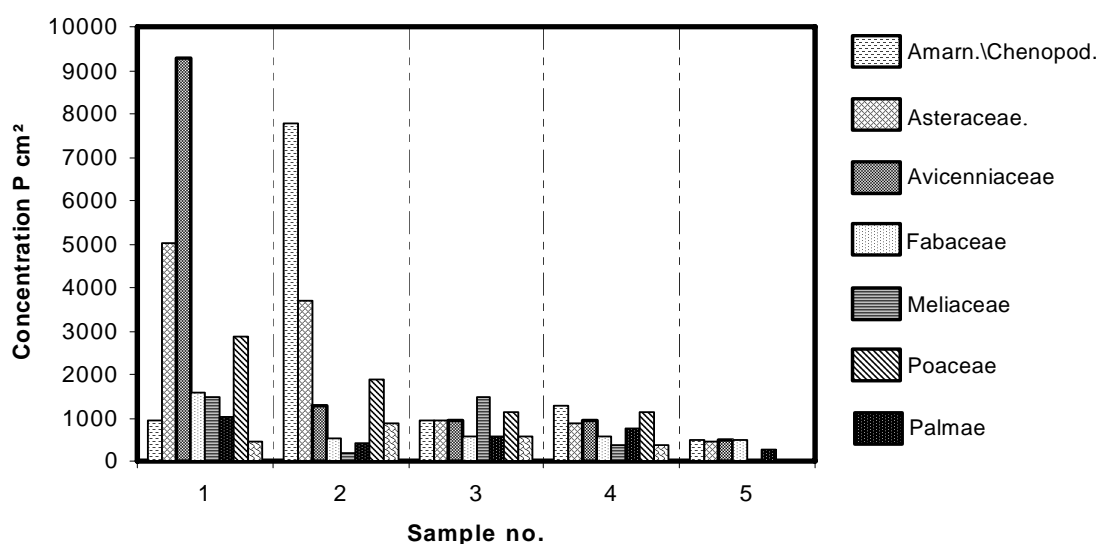


Figure 3: Productive samples pollen diagram (intertidal transect), showing the most abundant pollen family concentration in the Abu Ghoson area, Red Sea Coast.

Table 3: Pollen family concentration (per cm²) in the transect and swamp sediments; the concentration is calculated by Lycopodium marker spores.

Pollen family	Transect sample no.							Swamp sample no.												
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Amaranthaceae	929	7779	929	0	0	0	1301	496	0	0	5110	4248	0	0	588	2001	0	0	0	7690
Chenopodiaceae	5013	3717	929	0	0	0	867	465	0	0	4248	4469	0	0	2144	1282	0	0	0	1666
Asteraceae	9292	1296	927	0	0	0	929	496	0	0	6194	4778	0	0	1411	4574	0	0	0	1282
Avicenniaceae	557	0	248	0	0	0	310	0	0	0	235	429	0	0	0	0	0	0	0	0
Bisaccate	1301	432	0	0	0	0	0	0	0	0	2168	1062	0	0	823	715	0	0	0	641
Brassicaceae	465	346	0	0	0	0	0	0	0	0	774	743	0	0	353	429	0	0	0	256
Casuarina	465	259	0	0	0	0	0	0	0	0	2323	956	0	0	470	715	0	0	0	1410
Cypraceae	372	173	0	0	0	0	0	0	0	0	774	319	0	0	0	0	0	0	0	0
Ephedraceae	929	432	0	0	0	0	0	0	0	0	1858	1062	0	0	588	572	0	0	0	641
Euphorbiaceae	1580	519	557	0	0	0	557	496	0	0	1084	1487	0	0	823	715	0	0	0	769
Fabaceae	279	432	929	0	0	0	372	0	0	0	1394	850	0	0	588	2573	0	0	0	1025
Geraniaceae	372	259	557	0	0	0	0	0	0	0	1549	1274	0	0	588	715	0	0	0	256
Liliaceae	1487	173	1487	0	0	0	372	0	0	0	774	743	0	0	470	715	0	0	0	256
Meliaceae	650	432	372	0	0	0	0	0	0	0	774	850	0	0	235	429	0	0	0	256
Oxalidaceae	1022	432	557	0	0	0	743	372	0	0	3562	2124	0	0	588	2144	0	0	0	1410
Palmae	465	519	743	0	0	0	929	248	0	0	6969	5097	0	0	588	3574	0	0	0	897
Plantagnaceae	2880	1902	1115	0	0	0	1115	867	0	0	5420	3398	0	0	588	2144	0	0	0	3460
Poaceae	372	259	0	0	0	0	0	0	0	0	774	425	0	0	353	572	0	0	0	384
Polygonaceae	929	432	557	0	0	0	372	496	0	0	2633	1062	0	0	235	1429	0	0	0	0
Polypodiaceae	372	605	557	0	0	0	557	0	0	0	774	850	0	0	235	429	0	0	0	513
Porate	465	864	557	0	0	0	372	0	0	0	1084	531	0	0	235	429	0	0	0	256
Solanaceae	465	0	0	0	0	0	557	0	0	0	2942	2018	0	0	1529	1715	0	0	0	641
Tiliaceae	2230	691	372	0	0	0	372	0	0	0	2478	1911	0	0	823	715	0	0	0	513
Unidentified 1	1951	432	0	0	0	0	0	0	0	0	3097	1593	0	0	588	429	0	0	0	641
Unidentified 2	557	432	0	0	0	0	0	0	0	0	1549	850	0	0	353	1429	0	0	0	256
Zygophyllaceae																				

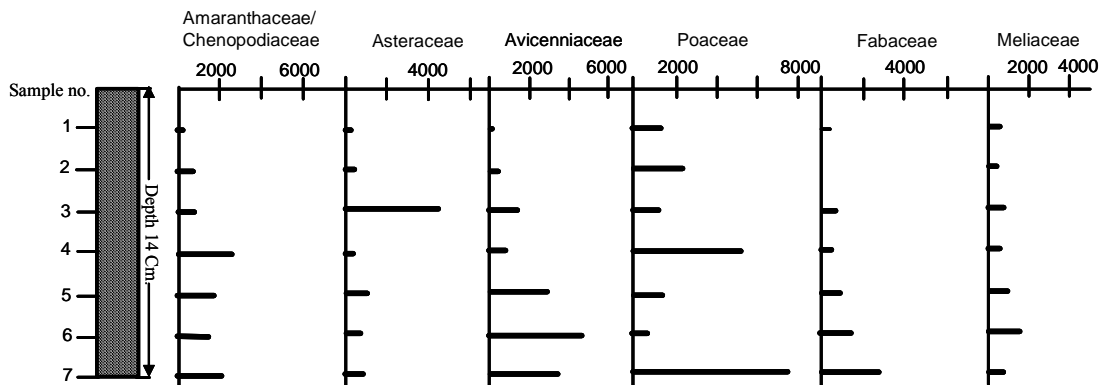


Figure 4: Pollen diagram showing the most abundant pollen families of core 2 sediments of mangrove site, in the Abu Ghoson area, Red Sea Coast: note that the concentration of pollen is defined as pollen per cubic cm using the Lycopodium marker spores.

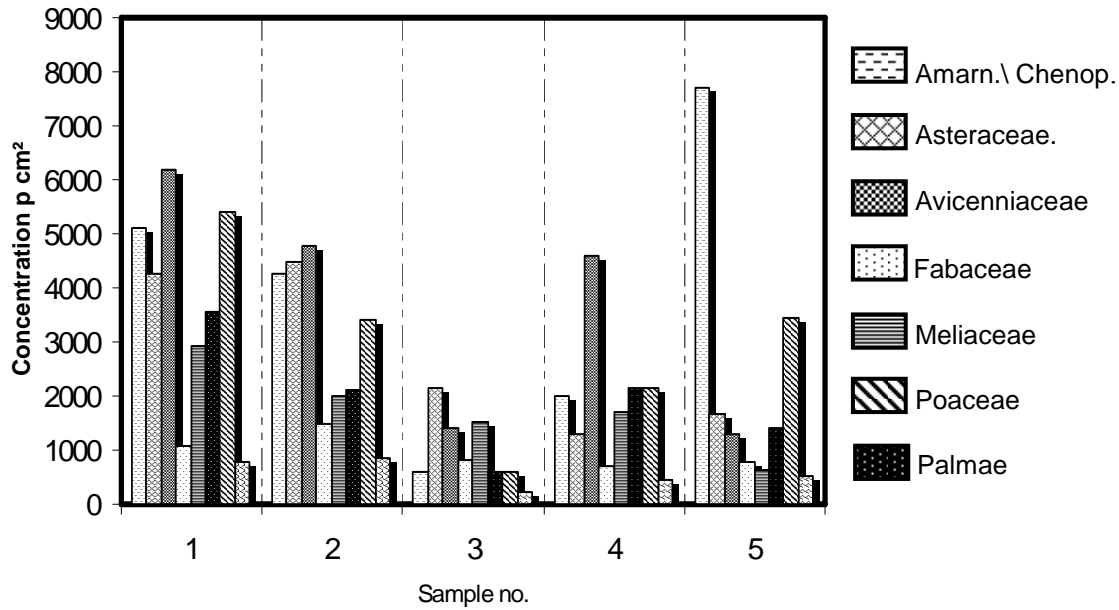


Figure 5: Productive samples pollen diagram (swamp transect), showing the most abundant pollen family concentration in the Abu Ghoson area, Red Sea Coast.

4 Mangroves restoration overview

4.1 Mangrove restoration

A realistic definition of restoration is given by Morrison (1990); “Restoration is the re-introduction and reestablishment of community-like groupings of native species to sites which can reasonably be expected to sustain them, with the resultant vegetation demonstrating aesthetic and dynamic characteristics of the natural communities on which they are based”. The restoration of mangroves has received a lot of attention world wide since the long neglect of ecological and environmental values of mangrove forests was based mainly on the planting of mangroves as the primary tool in restoration. However, a better approach to mangrove restoration would determine the causes for plant loss, remove these causes, and work with natural recovery processes to re-establish mangrove habitat (Lewis & Streever 2000). In addition, deterioration of mangroves has occurred throughout the world leading to coastal erosion, decline of fishery resources and other environmental consequences.

In Egypt, mangrove is an important scarce resource. The tourist industry has been developed in mangroves throughout the Red Sea Coast. Mangroves form a part of the attraction for tourism. The government of Egypt has at last recognised the close link between coral reefs, mangroves and tourism development, which helps contribute to the local and national economy. Accordingly, the Egyptian government started initiating conservation and protection measures for mangroves.

Mangroves in the study area have been affected by some hydrological changes due to as the impact of a new asphalt coastal road. The road interrupted the rare fresh water supply, a decrease in nutrients, and an increase in salinity (~ 45 ppt). In addition, the mangrove density and quality have been effected by man-induced change, for instance; resort building, cutting of trees, animal grazing and fishing. Many mangrove bushes of *Avicennia marina* along the Red Sea coast are threatened by extinction because of the physical stress from resorts, cut-off for playground buildings or for the supply of freshwater (Saenger 2002). Mangroves have many very valuable ecological benefits. Nevertheless, these benefits are hard to define and do not offer income for citizens or for the managers of the mangroves who are living in and around mangrove sites. For that reason, mangroves are often considered

to be “wasteland” rather than mangrove “wetland”. In order to correct this concept and capture the interest and understanding of all people in and around mangroves, it is important to try to convert some of these indefinable ecological values into a financial value. One way in which this can be done is during the development of the tourism industry in the mangroves, through careful planning and consideration of the mangroves unique environment. The tourism industry can provide considerable profits to the environment, local citizens, visitors to the mangroves and the agencies that are responsible for the management of the mangrove areas (FAO 2002). Nevertheless, first we should determine the causes of mangrove loss, remove these causes, and work first with natural recovery processes to re-establish the mangroves as realised by Lewis & Streever (2000).

Finally, it can be recommending that:

- The mangroves inimitability is a link between terrestrial and marine habitats and their ecological significance should be fully understood to awaken the interest and appreciation of mangrove residents, managers, and tourists.
- At the studied site, the threat to mangroves by extinction, hydrological changes, physical stress of resorts around the mangroves would require assessment. However, the damage is caused by browsing that can be recovered once the browsing pressure is lifted.
- Socio-economic study of mangrove values could enhance the re-establishment of mangroves.
- Raising awareness and changing the behaviour of tourists and people - especially those who have direct contact with mangroves.
- As it has often been said, “to care we must understand”, so special attention and support should be paid to the marine ecosystem, scientific research and conservation activities.

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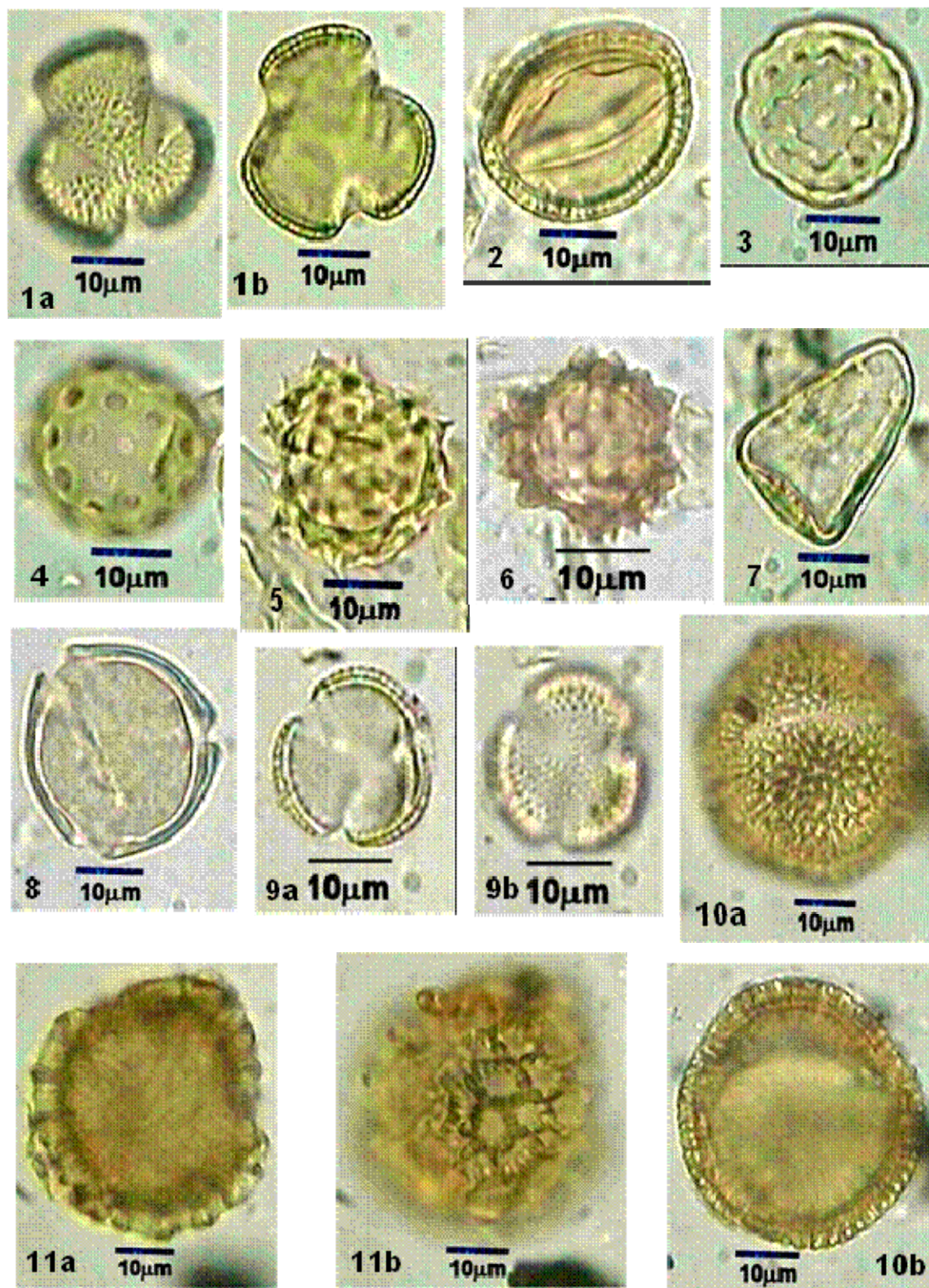


Figure 6: 1-2 *Avicennia marina* (Avicenniaceae); 3-4 Amaranthaceae./ Chenopodiaceae; 5-6 *Aaronsohnia factorovskyi* (Asteraceae); 7 *Cyprus conglomeratus* (Cypraceae); 8 Casuarina; 9a,b *Anastatica hierochuntica* (Brassicaceae) ; 10-11 *Erodium ciconium* (Geraniaceae).

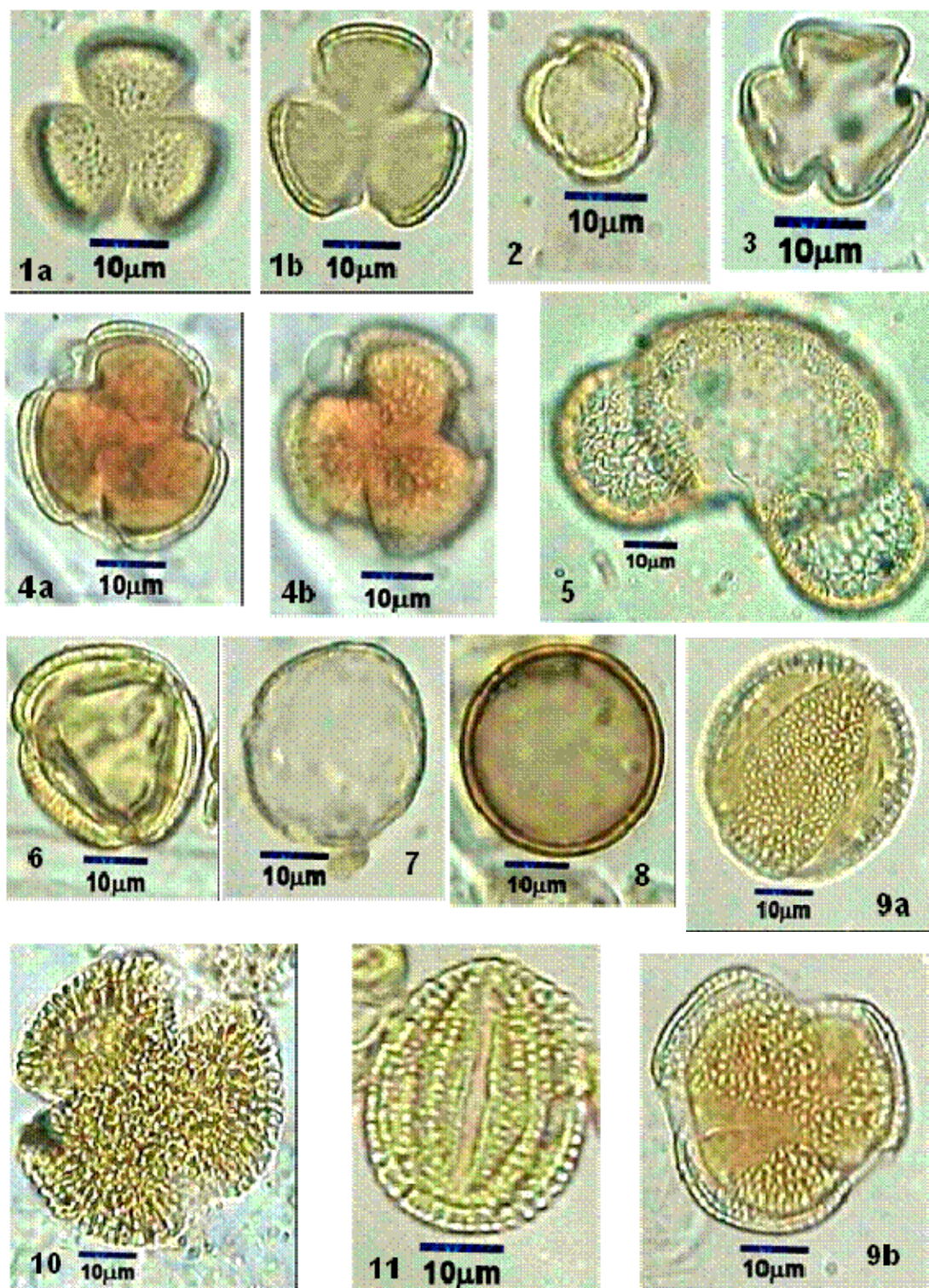


Figure 7: 1a,b *Oxalis* cf. *pes-caprae* (Oxalidaceae); 2 *Sparganium* cf. *diandra* (Solanaceae); 3 Polygonaceae; 4a,b *Fagonia* spec. (Zygophyllaceae); 5 *Podocarpus* spec.; 6 Rhamnaceae; 7 *Plantago amplexicaulis* (Plantaginaceae); 8 Plantaginaceae; 9a,b *Convolvulus* spec. (Convolvulaceae); 10 *Limonium* cf. *axillare* (Liliaceae); 11 *Euphorbia* spec. (Euphorbiaceae).



Long-term vegetation dynamics after land-use change in Wadden Sea salt marshes

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Abstract

After the establishment of the National Park "Schleswig-Holstein Wadden Sea" in Northern Germany, sheep grazing was strongly reduced, or abandoned and the maintenance of the artificial drainage system in the abandoned areas neglected. We studied the long-term development of plant species density and evenness in salt marshes of the "Hamburger Hallig". In this area, a grazing experiment including intensively grazed, moderately grazed and abandoned plots, was established in 1991 and modified in 1995 and 1999. Species density increased on all permanent plots except for the high salt-marsh plots ungrazed since 1992, which had the highest species density of all plots in 1992 and showed a slight but non-significant decrease in species density between 1995 and 2004. In the intensively grazed salt marsh, the increase in species density was delayed until 1998. Vegetation evenness E1/D showed high inter-annual fluctuations with a strong decrease during the early years of the experiment but hardly any significant trend from 1995 to 2004. The decreasing species density between 1999 and 2004 on plots dominated by *Elymus athericus* indicated that this species was able to outcompete other species at high salt-marsh sites with well-aerated soils. In contrast, no decrease in species density could be observed on sites dominated by *Atriplex portulacoides*. In the low salt marsh and in less drained parts of the high salt marsh, soil water-logging and low sedimentation rates may impede large-scale species dominance and consequentially prevent a decrease in species density.

1 Introduction

Salt marshes are habitats with extreme environmental conditions due to regular flooding by seawater. Gradients of waterlogging and salinity from the pioneer zone over the low salt marsh to the high marsh induce a distinct vegetation zonation depending on the tolerance ranges of different plant species (Adam 1990). Intensive grazing of salt marshes by sheep or cattle can lead to a downward-shift of vegetation zones (Ranwell 1968, Bakker 1989) and to a loss of grazing-sensitive species because only a few plant species (e.g. *Salicornia* spp., *Puccinellia maritima*) tolerate frequent biomass loss and trampling (Kiehl et al. 1996). In the beginning of the 1990ies the vegetation of mainland salt marshes along the North Sea coast of Schleswig-Holstein (Northern Germany) was uniform with low structural diversity (Fig. 3) and characteristic grazing-sensitive species (e.g. *Atriplex portulacoides*) became endangered after decades of intensive sheep grazing (Kiehl et al. 1996, Stock et al. 1997).

After the establishment of the National Park "Schleswig-Holstein Wadden Sea", the formerly intensive sheep grazing in mainland salt marshes was strongly reduced or abandoned and the maintenance of the artificial drainage system was neglected in the abandoned areas (Stock 1997, Stock & Kiehl 2000, Bakker et al. 2005). To study the effects of different grazing intensities, large-scale experiments were established in different areas (Dierssen et al. 1997, Stock & Kiehl 2000). Short-term investigations showed that grazing-sensitive species were able to spread quickly within four or five years after grazing abandonment or reduction (Kiehl et al. 1996). During this period, plant species density was lowest in intensively grazed low salt marshes but did not differ between moderately grazed and ungrazed plots (Kiehl 1997). A meta-analysis of long-term vegetation changes in Wadden-Sea salt

marshes showed, however, that grazing abandonment can have a negative effect on species density due to the dominance of competitive plant species such as *Elymus* spp. in the high marsh or *Atriplex portulacoides* in the low marsh (Bos et al. 2002). In contrast, some examples from long-term ungrazed salt marshes indicate that species-rich vegetation mosaics can also persist over long periods (Schwabe & Kratochwil 1984, Kiehl et al. 2000). The speed of vegetation succession in ungrazed Wadden-Sea salt marshes depends on drainage conditions and on sedimentation rates in relation to sea level rise (Leendertse et al. 1997, Olf et al. 1997, Schröder et al. 2002, Kiehl et al. 2003). In mainland salt marshes on clayey soils, species-poor stands of highly competitive *Elymus* spp. develop mainly at well-drained sites with high sedimentation rates and probably high nitrogen input (e.g. Andresen et al. 1991, Heinze et al. 1999), whereas soil water logging and high salinities can prevent dominance of single species on less drained sites with low sedimentation rates (Kiehl et al. 2003).

The aim of our study was to analyse long-term vegetation succession from 1992 to 2004 after the establishment of a large-scale grazing experiment in the salt marshes of the Hamburger Hallig. We addressed the following questions: i) What is the long-term effect of grazing reduction/cessation and of neglecting the drainage system on species density and evenness of salt marsh vegetation? ii) Does species density decrease after grazing abandonment due to the spreading of potentially dominant species (e.g. *Elymus* spp., *Atriplex portulacoides*)? iii) What is the effect of large-scale abandonment on the vegetation of adjacent salt marshes, which are still intensively grazed?

2 Study area and methods

The salt marshes of the "Hamburger Hallig" (coordinates: 8° 51' E, 54° 37' N) are located in the National Park "Schleswig-Holstein Wadden Sea" and cover an area of 1047 ha (Stock et al. 2005). They have been grazed intensively by sheep (> 10 sheep ha⁻¹) until 1991. A grazing experiment including intensively grazed, moderately grazed and abandoned plots was established in 1991 and modified in 1995 and 1999 (Stock & Kiehl 2000, Schröder et al. 2002). In 1992, 38 permanent plots of 4 m² were installed in differently grazed areas of the salt marsh (Table 1). In 1995, seven additional plots were installed in the intensively grazed salt marsh because grazing had to be stopped for logistical reasons on those permanent plots, which had been grazed intensively from 1991 to 1994.

Table 1: Design of the grazing experiment: number of permanent plots in differently grazed parts of the salt marsh. All plots had been grazed intensively by sheep (> 10 sheep ha⁻¹) until 1991. Values in parentheses represent the number of plots from 1999 onwards, after two additional permanent plots had been installed near each of the "old" plots.

	Low salt marsh	High salt marsh	Total no. of plots
ungrazed since 1992	11 (33)	8 (24)	19 (57)
ungrazed since 1995 (grazed intensively by > 10 sheep ha ⁻¹ until 1994)	4 (12)	3 (9)	7 (21)
moderately grazed (1991-1994: 1.5 sheep ha ⁻¹ , 1995-2004: 0.75 sheep ha ⁻¹)	6 (18)	6 (18)	12 (36)
intensively grazed (> 10 sheep ha ⁻¹), permanent plots installed in 1995	-	7 (21)	7 (21)
			45 (135)

After several years of succession, the vegetation became more heterogeneous than expected at the beginning of the experiment. To account for this spatial heterogeneity, two additional permanent plots were established in the direct vicinity of each of the already existing plots in 1999. So, in total 135 permanent plots were sampled at 45 locations from 1999 onwards.

The percentage cover of all plant species was estimated yearly on all permanent plots from 1992 to 2004 according to Londo (1976). Species density and the evenness index $E_{1/D}$ (Smith & Wilson 1996) with values between 0 (lowest evenness) and 1 (maximal evenness) were calculated for each year. Changes in species density and evenness were analysed for the 45 "old" permanent plots by a linear mixed effect model with plot as random effect and time and group as fixed effects. Analyses were computed by the lme function in the R package lme4 (Bates & Sarkar, version 0.9975-10). For these analyses, data from 1995 to 2004 were used, because the still intensively grazed plots were sampled first in 1995 (see above). Additionally, differences in species density between 1999 and 2004 were calculated for all 135 plots. To test for significant differences in changes of species density between 1999 and 2004 in relation to the species, which was dominant in 2004 (*Elymus athericus*, *Atriplex portulacoides*, *Festuca rubra*, *Puccinellia maritima* or *Spartina anglica*), the non-parametric Kruskal-Wallis H-test was used.

3 Results

3.1 Dynamics of species density and evenness under different grazing regimes

Species density increased on all permanent plots except for the high salt-marsh plots ungrazed since 1992, which had the highest species density of all plots at the beginning of the study (1992) and showed a slight but non-significant decrease in species density between 1995 and 2004 (Fig. 1, Tab. 2). On plots with similar grazing regime, the increase in species density was generally higher in the low salt marsh than in the high salt marsh. Within one salt marsh zone, the increase was more pronounced on continuously grazed plots and lowest on sites that became abandoned already in 1992 (Tab. 2). In the intensively grazed salt marsh, the increase in species density was delayed and only started in 1998 (Fig. 1).

Table 2: Rates of annual change in species density and evenness value between 1995 and 2004, estimated by a linear mixed model. Data represent means \pm 1 standard error. Significant changes ($p < 0.05$) are marked by an asterix.

	Species density, mean change [no. of species year ⁻¹]	Evenness, mean change [Evenness year ⁻¹]
Low salt marsh ungrazed since 1992	0.19 \pm 0.06 *	0.0013 \pm 0.0038
Low salt marsh ungrazed since 1995	0.31 \pm 0.07 *	-0.0071 \pm 0.0050
Low salt marsh moderately grazed	0.43 \pm 0.07 *	-0.0078 \pm 0.0044
High salt marsh ungrazed since 1992	-0.05 \pm 0.06	-0.0148 \pm 0.0041 *
High salt marsh ungrazed since 1995	0.11 \pm 0.08	-0.0205 \pm 0.0055
High salt marsh moderately grazed	0.18 \pm 0.07 *	-0.0076 \pm 0.0044
High salt marsh intensively grazed	0.36 \pm 0.04 *	-0.0044 \pm 0.0030

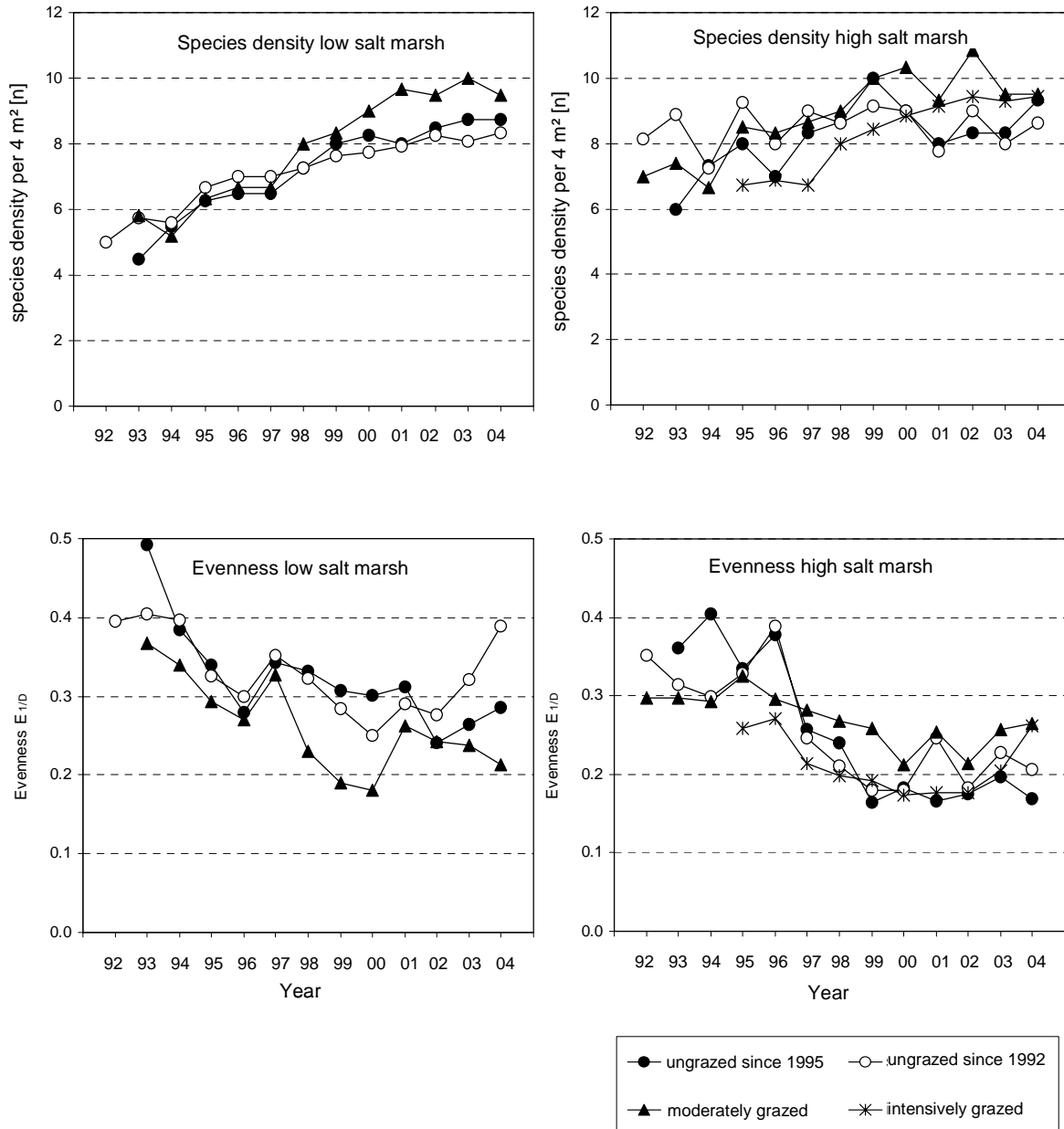


Figure 1: Changes in species density (4 m²) and evenness $E_{1/D}$ between 1992 and 2004 on differently grazed plots of the low salt marsh (left) and the high salt marsh (right).

Evenness values showed high inter-annual fluctuations with a strong decrease during the early years of the experiment but hardly any significant trend from 1995 to 2004 (Fig. 1, Tab. 2). Only in the ungrazed high salt marsh, a further decrease in evenness could be observed for this period.

3.2 What is the effect of dominant species on species density?

Species density decreased between 1999 and 2004 on most of the plots, which were dominated by *Elymus athericus* in 2004, but remained similar or showed even a slight increase on plots dominated by other species (Fig. 2). The decrease on plots dominated by *Elymus* was significantly different in comparison to the other plots ($p < 0.01$, $5 \leq n \leq 65$, $df = 4$, H-test). Only one plot dominated by *Elymus* showed an increase in species density between 1999 and 2004. No significant differences between median changes of species density were found between plots with dominance of *Atriplex portulacoides*, *Festuca rubra*, *Puccinellia maritima* or *Spartina anglica*.

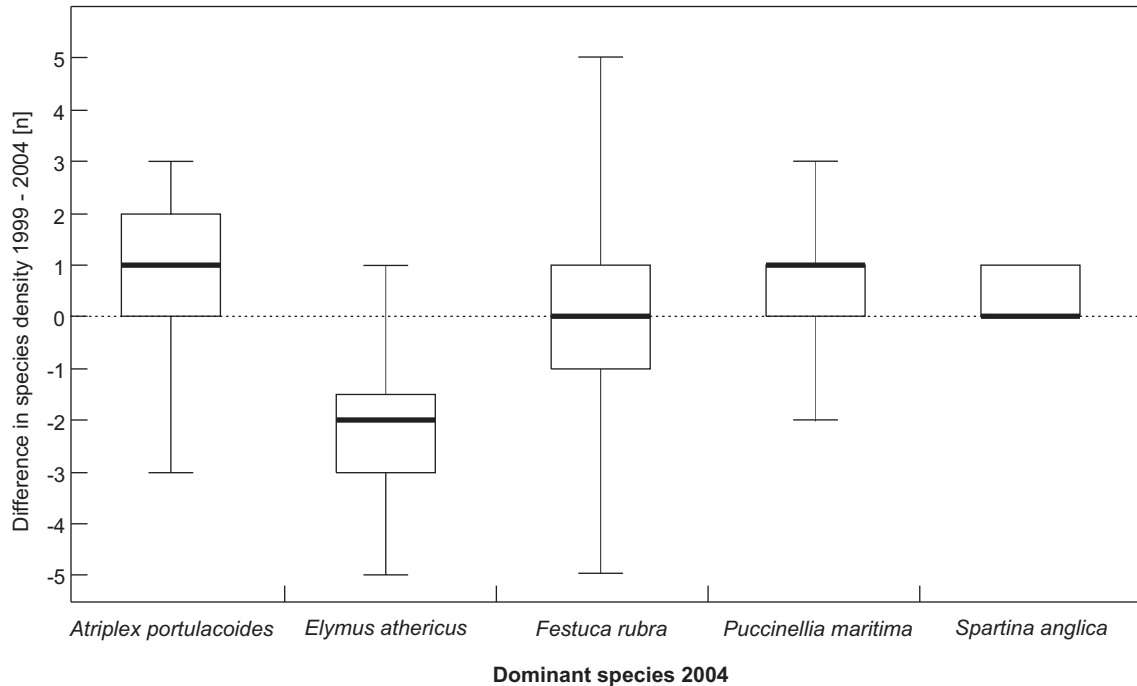


Figure 2: Difference in species density between 1999 and 2004 on permanent plots dominated by different plant species in 2004. Data represent medians (black line), upper and lower quartiles (box), minima and maxima (whiskers).

4 Discussion

Thirteen years after land-use change, species density in the salt marshes of the Hamburger Hallig increased considerably on most of the permanent plots due to the spreading of grazing-sensitive species (e.g. *Atriplex portulacoides*, *Artemisia maritima*). Up to now, a quick increase of these species after large-scale management change has been shown only for moderately grazed and ungrazed plots (Kiehl et al. 1996). During the early years after land-use change, species density remained low on intensively grazed plots because low seed availability and the selective grazing of the sheep hampered the colonisation of grazing-sensitive species (Kiehl 1997). In the present study, however, we found a delayed increase in species density even on formerly species-poor intensively grazed plots. This increase, which started seven years after the large scale land-use change in the study area (1998) and continued until 2004, was only possible because of the successful reproduction of grazing-sensitive species on adjacent moderately grazed and ungrazed plots (Fig. 4), which led to a high seed availability. As halophyte seeds are easily dispersed by flooding water (Koutstaal et al. 1987, Wolters et al. 2005) and no dispersal barriers exist in our study area, seed availability obviously became so high on the intensively grazed plots that the establishment of grazing-sensitive species was possible even under high grazing pressure.

A similar delayed increase in species density on intensively grazed plots adjacent to ungrazed plots could be observed in the Sönke-Nissen-Koog salt marsh, north of the Hamburger-Hallig area (Sieger 2004, Kiehl 2005). Our results indicate that large-scale land-use change does not only affect the vegetation of the respective areas directly but can also have strong effects on neighbouring areas, in which the management did not change. Without any land-use change, species density would have remained low under continuously high grazing pressure.

In the low salt marsh, species density increased both on grazed and ungrazed plots. Our results show that the dominance of the low-marsh species *Atriplex portulacoides* did not have any negative effect on species density in our study area up to now. This is in contrast to the findings of other authors who

stated that *Atriplex portulacoides* is able to outcompete other species in ungrazed salt marshes (Bakker 1989, Bos et al. 2002). In our study area, the neglectance of the drainage system has probably led to reduced aeration of the clayey soils, which prevented the dominance of *Atriplex* because this species is negatively affected by soil water logging (Beeftink 1977).

In the high salt marsh, species density also increased on most plots but showed a slight and not significant decrease on plots ungrazed since 1992. On the latter plots, species density had been highest (eight species per 4 m²) of all plots in the beginning of the study because grazing-sensitive species were already present with few individuals then. As the species pool in salt marshes is generally small due to the extreme environmental conditions (Adam 1990), a further strong increase of species density could not be expected. Nevertheless, first tendencies of a slight decrease in species density could be observed between 1995 and 2004. For these plots, the decreasing evenness values indicate increasing dominance of single species. For all other plots, evenness values decrease mainly due to the increasing species density because the dominance index 1/D of Williams (1964) is divided by the number of species in order to calculate the evenness index $E_{1/D}$.

The analysis of the difference in species density between 1999 and 2004 showed that *Elymus athericus* was able to outcompete other species in our study area. This confirms the findings of Bos et al. (2002). *Festuca rubra*, in contrast, as another competitive high marsh species was less competitive in our study area. Large-scale dominance of this species was probably also hampered by soil water logging (c.f. Gray & Scott 1977).

In summary, our results indicate that species density might decrease in the future in ungrazed plots on well aerated soils of the high salt marsh dominated by *Elymus* spp. In the low salt marsh and in less drained parts of the high salt marsh, soil water-logging and low sedimentation rates (Schröder et al. 2002, Kiehl et al. 2003) may impede species dominance and consequentially prevent a decrease in species density.



Figure 3: Intensively grazed salt marsh in the beginning of the 1990ies (Photo: K. Kiehl).



Figure 4: Ungrazed salt marsh of the Hamburger Hallig in 2002 (Photo: M. Stock).

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Ground beetles as ‘early warning-indicators’ in restored salt marshes and dune slacks

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Abstract

Populations of ground beetles and spiders are continuously monitored since 1990 in the dunes and salt marshes of the river Ijzer estuary (Belgium), where a recent nature restoration project took place within the framework of LIFE. Immediately after restoration measures, continuous (year cycle) pitfall and window trapping was performed during several years in restored or newly developed salt marsh and dune slack habitats and compared to target ('old' salt marsh) habitats. In this paper, we focus on ground beetle assemblages and species quality from these samplings, based on some 40,000 beetles identified to 96 species. Results show several beetles new to the study area as well as a marked increase of several target species with high conservation interest (Red-list species). However, many of these species could be rapidly lost again unless natural dynamic processes are kept ongoing. Historical beetle data show that many species that disappeared from the area during the past century have not yet been able to recolonise. This is especially true for salt marsh species and possibly due to dispersal limitation. Many dune slack species re-appeared but did not establish viable populations. Moreover, several ground beetle species indicate increased sand instead of silt deposits in new and old salt marshes. Further invertebrate monitoring therefore is a prerequisite for a well-founded long-term evaluation of the executed nature development measures. Such monitoring will be of much interest, both for an evidence-based nature conservation management, for fundamental ecological research, but also as a possible early warning system for the need of additional management measures in the future.

1 Introduction

1.1 History of the river Ijzer estuary: a story of continued habitat destruction

The relatively small river Ijzer is located in the western part of Belgium and has a short intertidal zone at the North Sea. A short history of this area is summarised on Fig. 1, comparing the most important phases of habitat destruction and degradation that took place from early medieval time until about 1986. About 1200, the estuary still consisted of many creeks, vast areas of salt marshes, extensive coastal dunes and, probably, many biologically very interesting dune-salt marsh transitions. By the end of the 18th century (as derived from the maps of de Ferraris, 1775), large natural areas had been lost, especially of salt marshes, mainly due to the limitation of sea tidal influences up to the sluices that had been constructed by that time besides the port of Nieuwpoort (situated at the lower right). Recent historical maps show an ongoing reduction of salt marsh area about 1842. At the beginning of the 20th century, when tourism along the Belgian coast started to expand, the first large areas of dunes were urbanised, along with the complete loss of the salt marsh along the left bank of the river. By 1955, habitat loss reached dramatic proportions, then also along the right bank of the Ijzer and prominently in the associated coastal dune area. A naval port was constructed and dunes were further reduced mainly due to building activities and camping sites. Moreover, a large part of the already highly reduced salt marsh was covered with dredged materials from maintenance channel deepening projects of the river. Shortly after that period, a new marina was constructed to the south of the persisting salt marsh relic, which was then even further reduced due to renewed deposition of dredging

material on its northern remnants. Nevertheless, the highly reduced natural remains of the estuary still contained many organisms of high conservation interest, including many invertebrate species of the Red Data Book. We therefore started a long-term ecological study of spiders and ground beetles in the area, on one of the few remaining sites left along our coast with at least some continuous gradients between seafront and inland dunes as well as a salt marsh relic.

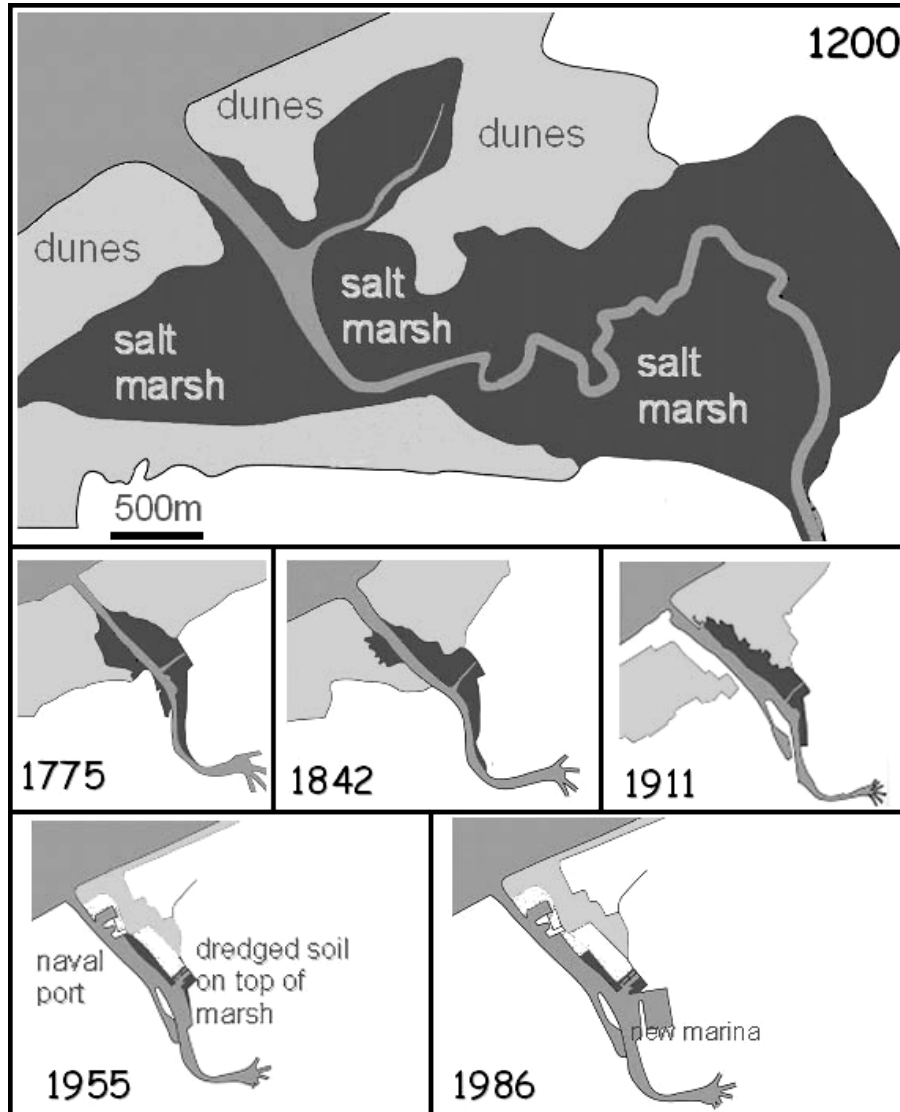


Figure 1: A short history of the river Ijzer estuary, illustrating the main episodes of destruction and degradation of salt marsh and dune habitats until the situation before the recent nature restoration project (all maps are drawn at approximately the same scale) (modified, after: Decler et al. 1995, Verhulst 1995).

1.2 Long-term invertebrate studies in the river Ijzer estuary

Since 1989, we have been continuously monitoring ground beetle and spider populations in the river Ijzer estuary coastal dunes and salt marsh (Belgium), mainly by means of uninterrupted long-term pitfall trapping on several sites. From this unique study area, along the right bank of the Ijzer estuary and the seaside beach front, no less than 140 ground beetle species are hitherto known based on these continuously performed samplings (Desender 1996, 2005b, Desender & Baert 1995). Diversity and assemblages have been studied on different scales by means of different sampling techniques, including pitfall traps, window traps, and air bell traps. The area moreover has been a favourite location for

many entomologists from about 1850 onwards, especially during the past century. Therefore, many historical data also exist on the former beetle fauna of this area. Most of this historical material is housed in the RBINSc (Brussels) and we recently completely checked and summarised these data in a distributional database.

Our long-term ecological study, as well as the presence of even older historical beetle data, offer an ideal framework to monitor and evaluate the effects of a recent nature restoration project that was started by the Flemish Government in 1998-2001 in the same area, within the framework of LIFE (Hoffmann 2004, Hoffmann et al. 2005, Maelfait et al. this volume). The general aim was to restore or create the natural ecological gradients typical for a coastal estuarine ecosystem. Most energy and restoration measures have therefore been directed towards an important increase of the pre-existing limited surface of old salt marsh, as well as towards increasing contact zones between mud flats, salt marsh, and coastal dune habitats. The project aims at restoring the situation of the Ijzer estuary around the beginning of the previous century, i.e. a period that is documented at least to a certain degree concerning the nature values (including ground beetles) of the area.



Figure 2: Recent aerial view of the river Ijzer estuary study area, including a new dune slack (site J) at the margins of a dune pond, and the old salt marsh relic (lower left with OS sites), separated by a low dike from new salt marsh (sites, H, K, S, NS, ZE); scale bar ~50 m.

In order to evaluate the effects of this recent restoration project, we sampled newly created or modified sites (dunes, dikes, banks of a dune pond and new salt marsh), while continuing our long-term sampling on a number of reference sites from the main target habitats in the area. Spider and beetle results of additional area-covering (short-term) sampling campaigns are reported elsewhere (Desender et al. 2006).

In this contribution, we focus on ground beetle assemblages and species quality from restored or newly developed salt marsh and dune slack habitats, based on 26 continuous (complete year cycle) trapping series collected on 12 sampling sites. Sampling sites are illustrated on a recent aerial view of the area in Fig. 2 and include old salt marsh (OS sites), new salt marsh (sites H, K, S, NS, ZE) and a new dune slack on the margins of a dune pond (J). Colonisation and turnover of assemblages are continuously monitored in these newly created sites and in the corresponding target habitats in order to evaluate the results of the restoration measures.

2 Results

2.1 Ground beetle assemblage and indicator species analyses in old and more recent salt marshes and dune slack during different sampling years

About 40,000 ground beetles, belonging to 96 species, were identified from the 26 year-cycles used in this paper. Based on the quantitative data for the most numerous species, a Detrended Correspondence Analysis (DCA) was performed with PCORD (McCune & Mefford 1999), after transforming data to relative densities within each species over the different sampling series (i.e. equal weighting each species). To prevent possible overruling noise from accidental species in this analysis, only carabid species with 26 or more individuals (equalling at least the number of separate sampling series used) were retained. A total of 42 ground beetle species fulfilled this criterion. Test runs with a more or less strict criterion nevertheless yielded a similar ordination as compared to the result for the 42 species, representing more than 95 % of the total pitfall and window trap catches. We refer to Maelfait et al. (this volume) for more information on this type of analysis and the rationale behind the use of pitfall data. Four of the analysed year-cycles are based on window trap catches and were entered as separate series in the analysis.

Figure 3 shows the ordination plot of year-cycle samples scores (upper graph, with overlay of main habitat type) and added species scores (lower graph) from this analysis. Based on the major groups of sampling series observed in this ordination (cf. added ellipses and overlay), the ground beetle species and their abundance are re-ordered in two-way Table 1 (number of individuals per series based on three sampling units or traps). Within each of the groups, sample series are ordered according to sampling year. An Indicator Value (IndVal) analysis (Duf rene & Legendre 1997) was performed based on this data and testing the hypothesis of indicator species for the four groups of samples (i.e. dune slack, window trap series, new salt marsh and old salt marsh). These results, along with the observed significance for each species indicator value, are presented in Table 2. Species with a statistically significant IndVal for one of the tested groups are also indicated in Table 1.

Along the first DCA ordination axis, we can easily discriminate three different groups of sites/series based on their ground beetle assemblages (Fig. 3): four consecutive year cycles in the developing dune slack vegetation (J01-J04) are regrouped at the left, whereas all 'old salt marsh' series cluster to the right. Early stages of developing new salt marsh as well as all window traps year cycles (two from the dune slack, two from new salt marsh) are found in more or less central position. Along the second axis, data on flying ground beetles (window traps) are discriminated from all pitfall trapping series, whereas at the same time dune slack samples are clearly positioned according to the year of sampling (2001 up to 2004, i.e. from the very first stages of development of the newly created site up to a more developed dune slack vegetation).

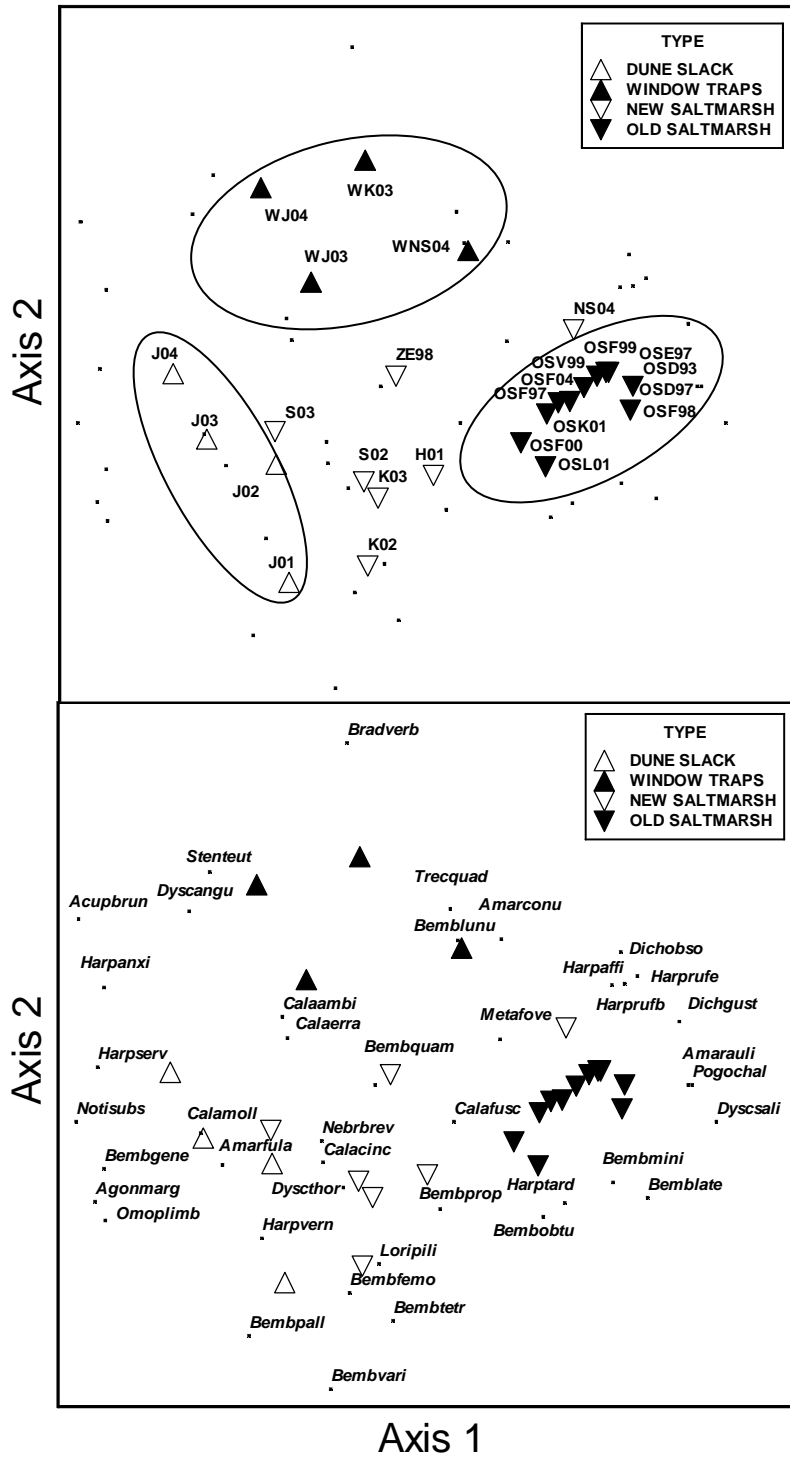


Figure 3: Plot of DCA ordination (detrended by 26 segments) sample and species scores, based on 26 site-year samples for the 42 most abundant ground beetle species: upper graph with added site labels and ellipses regrouping most important habitats/samples, lower graph with abbreviated species labels (see Table 2 for explanations).

Table 1: Re-ordered two-way table showing the number of ground beetles per species (corrected to three sampling units per series) in the 26 sampling series (site code followed by year of sampling between (19)93 and (20)04; preceded by 'W' for window trap catches) used for DCA (species names given in full in the same order in table 2); species in grey box have a significant Indicator Value for their respective group of sampling series, cf. table 2).

species	J01	J02	J03	J04	WJ03	WJ04	WKS04	ZE98	H01	S02	K02	S03	NS04	OSD93	OSE97	OSF97	OSF98	OSF99	OSV99	OSF00	OSK01	OSL01	OSF04				
Agonmarg	5	14	83	29				20	21	2	2	5	7	27	5	2		4	1				1	131			
Calanoil	6	117	159	227			3								1									607			
Harpvnr	2	8	3	8				3				1	1		1									29			
Notisubs	5	8	26						2			2												41			
Omopimb	98	113	146	276			3	18	4	29	9	3	1		2							4		632			
Bembpal	504	95	56	8				9	1						2									728			
Nebtrbrv	1	9	11	21											2									77			
Bembgene	3	3	17																					26			
Dyscsthr	302	927	1496	803				484	19	99	150	377	110	1106	2	34	62	18	79	18	57	46	600	130	7103		
Amarifula	12	26	15					6	2			8	8	2										78			
Harpnxi	11	15	62				3	1							1									98			
Acupbrun	1	20	54																					681			
Harpserv	2	42	455				42	141							1									83			
Dyscangu	3	3	21	51			3	12	18					1	1	1								130			
Calafusc	2	23	14	18				1	6	1	9	2		4	40	15	25	3						190			
Calambi	2	17	26	63				29	5	2	6	5	1	4	1									199			
Calacinc	5	20	14	27				74	1			15		2	1									172			
Calaeira	18	54	101					45	47	3		3		2	1	2	1	1						361			
Bembvari	15	2					6	145			41	2	15		3	4	4	1	2	1				68			
Bradverb															1									61			
Trequad	1	1	2				9	23	10	5	5	2	1	12	2	6	5	4						305			
Stenteut		5	53				165								4	2								111			
Bemblunu	5	8	5				6		13	2	18	6	6		4	2	2	6	8	27	10	16		190			
Amarconu							9	42			6	2	33	2	2									155			
Bembfemo	36	90	63	2			3	430	146	117	284	320	44	21	1	2	1	4						1584			
Bembquam	2	2	2				9	4	27	2	14	21	2	2	1									107			
Loripili		2	2				6		6		11	2	1											27			
Bembtetr	2	15	24	3				1	31	9	77	35	2		1		6	7		45	1	8		266			
Harpaffi		3	5					39	6		3	2	227	9	17	4	6	15	1	2				363			
Bembobtu	1	2							36	14	2	1	3	1	2	4	11	9	13	1	18	3	2	122			
Bembprop	1	3	5	2			3	30			12	2	1	3	1	1	3	1	1					79			
Pogochal									3	5	35	8	2	2096	1391	826	1659	560	2396	1050	270	142	347	504	1656	12948	
Dyscsali															7	4	66	2	72	3	2	3	15	4	178		
Amarauli	1							1							3	3	2	9	6	3	1			29			
Bemblate															6	10	39	1	1	1	29	24	4	131			
Harpard	3	3							2		5	6	3	3	1	4	4	5						30			
Dicligust								2						154	218	1097	239	418	645	129	189	176	250	97	5601		
Bembmini	3	2	3	2			6	232	3	185	230	646		146	32	105	144	206	341	7	89	33	70	829	3326		
Harpufib	2	2	1					4						1	1	23	2	12						53			
Harpufe								16			3		54	10	4	5	9	12	20	3	5	2		168			
Dichobso	1	1							4	2	2	1	741	67	91	324	92	180	267	74	34	56	30	29	2606		
Metarove								11	2					1	4	1	1	1	1	1	2			26			
Total	993	1522	2301	2327	105	228	190	930	1400	656	300	836	1078	212	6928	1810	1231	3454	1119	3494	2414	554	592	1293	1057	2868	39890

Table 2: IndVal (IV) analysis based on four groups corresponding to DCA results (compare with table 1); Maxgrp= group identifier for group with maximum observed IV: 1= dune slack, 2= window trap catches, 3= new salt marsh, 4= old salt marsh; (*)= proportion of randomised trials with indicator value equal to or exceeding the observed indicator value; $p = (1 + \text{number of runs} \geq \text{observed}) / (1 + \text{number of randomised runs})$; significant values in bold.

Species	Maxgrp	Observed Ind Val	IV from randomised groups		
			Mean	S.Dev	p (*)
<i>Agonum marginatum</i>	1	99.3	23.4	11.98	0.0010
<i>Calathus mollis</i>	1	90.3	41.7	13.46	0.0010
<i>Harpalus vernalis</i>	1	81.8	27.5	11.18	0.0010
<i>Notiophilus substriatus</i>	1	71.2	22.5	11.41	0.0010
<i>Omophron limbatum</i>	1	100.0	19.1	10.33	0.0010
<i>Bembidion pallidipenne</i>	1	94.5	34.9	14.17	0.0020
<i>Nebria brevicollis</i>	1	72.6	31.8	10.30	0.0030
<i>Bembidion genei</i>	1	69.7	22.6	11.57	0.0030
<i>Dyschirius thoracicus</i>	1	66.4	38.8	9.26	0.0080
<i>Amara fulva</i>	1	59.0	24.8	11.06	0.0110
<i>Harpalus anxius</i>	1	69.1	33.7	14.39	0.0120
<i>Acupalpus brunnipes</i>	1	54.8	22.9	11.61	0.0150
<i>Harpalus servus</i>	1	71.2	35.3	14.85	0.0200
<i>Dyschirius angustatus</i>	1	64.0	31.6	11.88	0.0200
<i>Calathus fuscipes</i>	1	48.6	31.1	9.81	0.0580
<i>Calathus ambiguus</i>	1	49.8	28.7	12.47	0.0720
<i>Calathus cinctus</i>	1	52.1	35.1	11.93	0.0960
<i>Calathus erratus</i>	1	46.6	36.2	13.30	0.1970
<i>Bembidion varium</i>	1	16.7	25.4	12.72	0.7740
<i>Bradycellus verbasci</i>	2	97.8	26.6	13.56	0.0010
<i>Trechus quadristriatus</i>	2	84.1	46.7	14.35	0.0050
<i>Stenolophus teutonus</i>	2	46.6	26.2	11.81	0.0630
<i>Bembidion lunulatum</i>	2	43.8	32.6	7.05	0.0780
<i>Amara convexiuscula</i>	2	27.8	29.9	12.78	0.4710
<i>Bembidion femoratum</i>	3	78.5	44.4	10.81	0.0030
<i>Bembidion quadrimaculatum</i>	3	54.3	34.0	11.30	0.0560
<i>Loricera pilicornis</i>	3	39.6	24.0	11.81	0.0770
<i>Bembidion tetracolum</i>	3	48.3	34.6	11.25	0.1150
<i>Harpalus affinis</i>	3	58.2	47.1	15.61	0.2480
<i>Bembidion obtusum</i>	3	39.2	35.9	11.18	0.3370
<i>Bembidion properans</i>	3	23.4	38.0	12.24	0.9480
<i>Pogonus chalceus</i>	4	76.2	33.4	9.97	0.0020
<i>Dyschirius salinus</i>	4	90.9	31.5	13.41	0.0050
<i>Amara aulica</i>	4	54.9	26.0	10.57	0.0190
<i>Bembidion laterale</i>	4	59.3	30.9	11.34	0.0220
<i>Harpalus tardus</i>	4	38.4	26.2	9.88	0.1030
<i>Dicheirotichus gustavii</i>	4	52.9	37.4	11.74	0.1150
<i>Bembidion minimum</i>	4	48.6	43.0	9.08	0.2410
<i>Harpalus rufibarbis</i>	4	33.9	30.9	12.44	0.3240
<i>Harpalus rufipes</i>	4	34.6	32.0	11.49	0.3360
<i>Dicheirotichus obsoletus</i>	4	30.3	39.7	11.48	0.7890
<i>Metabletus foveatus</i>	4	19.0	30.3	12.38	0.8330

An array of typical ground beetle species for each of the observed clusters of sampling series can be derived from the lower DCA graph as well as from the IndVal analysis and the ordered table with raw data, taking knowledge on the ecology of each species and its possible preference for other dune habitats into account. The last-mentioned aspect is especially important for the small-sized dune slack site,

where several ground beetles from other (dry dune) habitat types were observed in relatively large numbers (e.g. the typical marram dune species *Calathus mollis*, the dune grassland species *Calathus ambiguus*, *Calathus cinctus*, *Harpalus vernalis* and *H. anxius*). This suggests the occurrence of more or less strong edge effects and/or source-sink effects, an aspect dealt with in more details by Maelfait et al. (this volume).

The most typical carabid species for the dune slack (margins of new dune pond) were, among others *Agonum marginatum*, *Omophron limbatum*, *Bembidion pallidipenne*, *B. genei*, *Dyschirius thoracicus* and *Acupalpus brunnipes*. The last-mentioned species was also caught in large numbers during flight activity. Some of these species gradually build up a large population, whereas others have already nearly completely disappeared again from the area (e.g. *Bembidion pallidipenne*). *Bradycellus verbasci* and *Trechus quadristriatus* appeared most numerous in the window trap catches. Indicator species for early stages of new salt marsh development were *Bembidion femoratum* and *B. quadrimaculatum*, whereas *Pogonus chalceus*, *Dyschirius salinus*, and *Bembidion laterale* were most numerous in old salt marsh samples. A number of other salt marsh species occurred both in old salt marsh, some window trap series and most of the somewhat more developed 'new' salt marsh sites: *Bembidion minimum*, *Dicheirotichus gustavii*, and *D. obsoletus*. Assemblages, derived from the most recent new salt marsh series (NS04), therefore were already much more similar to 'old salt marsh' series as compared to earlier stages of developing new salt marsh. Nevertheless, the typical ground beetle salt marsh species *Dyschirius salinus* was still absent from restored or new sites.

2.2 Evaluating nature restoration based on historical and recent occurrence of dune slack and salt marsh ground beetles in the river Ijzer estuary

The river Ijzer estuary received already much attention in the past and repeatedly was visited by entomologists from about 1850 onwards. This old data, residing in the RBINSc collections (Brussels, Belgium), enables us to compare, for this area, the historical and recent occurrence of ground beetle species that are known as typical dune slack or salt marsh species.

Table 3 summarises this data and regroups, for each of the main target habitats, dune slack or salt marsh, the ground beetle species that are documented to have disappeared in historic times from the area (most recent year of observation mentioned), most probably as a consequence of the habitat destruction and degradation that took place mainly during the past century (1A and 2A species). A comparison with species that continuously remained present in the area or even appeared as 'new' during or since the recent nature restoration project (1B and 2B species) shows a strongly differing pattern depending on whether species are typical for dune slack or for salt marsh. All mentioned dune slack species (14) without exception have been observed in recently restored sites, whereas less than half of the salt marsh species (9 out of 20) managed to colonise or recolonise the area.

Most, if not all, of the very special carabid species that were lost in historical times from the area (some 20 species, a large majority of these seriously threatened in our region), were not or not yet able to re-establish viable populations in the Ijzer estuary restored habitats (cf. table 3, persistence in study area of 1A and 2A species). Salt marsh species with a preference for more coarse-grained sediments or with a wider range of preferred soil types (sand/silt), appear to show less problems in maintaining viable populations (several 2B species), also in restored sites.

Table 3: Typical dune slack and salt marsh ground beetles, with their historical and/or recent occurrence in the river Ijzer estuary, regrouped into (A) species that disappeared from the area before the onset of recent nature restoration and (B) those that have been continuously present or recently appeared as new to the area, along with quantitative data on their recent occurrence in the estuary, Red Data Book category for the region of Flanders (Desender et al. 1995), persistence in the area, and major soil type preference.

group	species	preferred habitat	year with most recent observation before restoration project	observed on recently restored sites	red data book category	persisting in study area	preferred soil type
1A	<i>Bembidion argenteolum</i>	dune slack	1883	2 ind.	rare	NO	sand
	<i>Bembidion doris</i>	dune slack	1850	1 ind.	not threatened	NO	sand
	<i>Bembidion pallidipenne</i>	dune slack	1938	abundant	near-extinct	??	sand
	<i>Broscus cephalotes</i>	dune slack	1926	2 ind.	not threatened	NO	sand
	<i>Cicindela maritima</i>	dune slack	1945	1 ind.	endangered	NO	sand
	<i>Dyschirius obscurus</i>	dune slack	1937	1 ind.	rare	NO	sand
	<i>Omophron limbatum</i>	dune slack	1909	abundant	not threatened	YES	sand
	<i>Acupalpus brunripes</i>	dune slack	new to area	abundant	vulnerable	YES	sand
	<i>Agonum marginatum</i>	dune slack	continuously present	abundant	not threatened	YES	sand
	<i>Bembidion velox</i>	dune slack	new to area	3 ind.	rare	NO	sand
1B	<i>Bradycellus distinctus</i>	dune slack	continuously present	abundant	rare	??	sand
	<i>Dyschirius thoracicus</i>	dune slack	continuously present	abundant	not threatened	YES	sand
	<i>Elaphrus riparius</i>	dune slack	continuously present	abundant	not threatened	YES	sand
	<i>Stenolophus teutonius</i>	dune slack	continuously present	abundant	not threatened	YES	sand
	<i>Anisodactylus poeciloides</i>	salt marsh	1880	1 ind.	extinct	NO	silt
	<i>Bembidion ephippium</i>	salt marsh	1934	NO	vulnerable	NO	silt?
	<i>Bembidion fumigatum</i>	salt marsh	1937	NO	rare	NO	silt
2A	<i>Bembidion tricolor</i>	salt marsh	1997	NO	rare	NO	silt
	<i>Bembidion maritimum</i>	salt marsh	1979	NO	endangered	NO	silt
	<i>Bembidion normannum</i>	salt marsh	1947	NO	vulnerable	NO	silt
	<i>Dyschirius chalcone</i>	salt marsh	1883	NO	rare	NO	silt
	<i>Dyschirius impunctipennis</i>	salt marsh	1949	NO	extinct	NO	silt?
	<i>Pogonus littoralis</i>	salt marsh	1875	NO	near-extinct	NO	silt
	<i>Pogonus luridipennis</i>	salt marsh	1865	NO	probably threatened	NO	silt
	<i>Pterostichus macer</i>	salt marsh	1982	NO	rare	NO	silt
	<i>Amara convexiuscula</i>	salt marsh	continuously present	abundant	rare	YES	silt/sand
	<i>Bembidion laterale</i>	salt marsh	continuously present	abundant	probably threatened	YES	sand
2B	<i>Bembidion minimum</i>	salt marsh	continuously present	abundant	not threatened	YES	silt
	<i>Bembidion nigropiceum</i>	salt marsh	new to area	2 ind.	new to fauna	NO	sand
	<i>Bembidion varium</i>	salt marsh	continuously present	abundant	not threatened	YES	silt
	<i>Dicheirotichus gustavii</i>	salt marsh	continuously present	abundant	rare	YES	silt
	<i>Dicheirotichus obsoletus</i>	salt marsh	continuously present	abundant	rare	YES	sand
	<i>Dyschirius salinus</i>	salt marsh	continuously present	NO	rare	??	silt
	<i>Pogonus chalcone</i>	salt marsh	continuously present	abundant	rare	YES	silt/sand

3 Discussion

Habitats of newly created, restored or developed sites in the Ijzer estuary (dikes, dunes, salt marsh as well as banks of freshwater habitats), can be characterised, during their first years of existence, by a large number of ground beetle species, typical for strongly disturbed habitats, more specifically for cultivated fields and ruderal sites on light soil. *Trechus quadristriatus* and *Bembidion femoratum* indeed prefer poorly vegetated disturbed sand to sandy loam soils, such as on cultivated fields, but also occur in rather wet situations such as highly dynamic riverbanks (Turin 2000). These ground beetles are excellent flyers and therefore were also abundant in our window trap catches in differing habitats (cf. Desender 2000).

In addition, an important number of Red Data Book species have been observed in these new or restored habitats. These include many typical dune and salt marsh species, occurring in the immediate surroundings, but also several dune slack species, recolonising the study area, such as *Bembidion pallidipenne* (near-extinct in Flanders) and *Bembidion argenteolum* (threatened). Another typical species for first stages of sandy riparian or dune slack habitats is *Omophron limbatum*, preferring margins of fresh or brackish water bodies of high quality, especially devoid of vegetation, including early stages of succession of coastal dune slacks. The first-mentioned beetle is extremely rare in our country and until now, it was only known from very few coastal locations. *Bembidion pallidipenne* appears to colonise nearly exclusively the first stages of brackish-fresh water riparian sites, particularly without vegetation, such as dynamic early stages of dune slacks. It needs constantly present dynamics and turnover in order to maintain a long-term surviving metapopulation structure. The species occurred in massive numbers in the Ijzer estuary during the first years after restoration measures. At present, this ground beetle has already disappeared nearly completely from the study area again, which could have important implications for options concerning future nature conservation management. Future monitoring will enable to conclude whether or not the individuals of such species are making up permanent (well-established) populations or, alternatively, only a temporary population, as a result of quickly changing habitat ecological characteristics or as a result of source-sink effects from adjacent high-quality habitats (cf. Maelfait et al., this volume). Such effects could especially manifest themselves in years with high abundance of particular species.

These beetles nowadays only persist in very few nature reserves of our region, but are typical pioneer species from coastal freshwater-brackish waterside habitats on sandy soil, such as dune slacks and dune-salt marsh transitions. Such species in general possess an excellent dispersal power (cf. Desender 1989a) and apparently rapidly colonised the new habitats that became available in the Ijzer estuary. To survive in the long run, they will need relatively continuously present dynamics at ecosystem level as well as sufficient populations functioning in a larger metapopulation. Many of these ground beetles however only occurred or were abundant during or immediately after the restoration measures took place, quickly disappearing after that initial stage. The rapidly decreasing numbers of *Bembidion pallidipenne*, after being initially very abundant on the new dune slack, are highly illustrative in this context. To conclude, many of these highly specialised dune slack beetles did not manage to establish more continuous and viable populations in the river Ijzer estuary.

In North-Western Europe, ecological restoration of salt marshes, assisting the recovery of degraded, damaged or destroyed salt marsh (SER 2004), has received increasing attention during recent years, mainly within the context of de-embankments and realignment of coastal defences (Garbutt et al. 2006, Wolters et al. 2005b). The recent nature restoration project in the river Ijzer estuary involved, related to wet target habitats, the removal of a large amount of top soil in order to recreate potential sites for salt marsh development and dune-salt marsh transitions, adjacent to an old salt marsh relic. As part of this project, a dune pond was created, giving rise to accompanying dune slack vegetation along its margins. At the onset of this recent nature restoration project, less than 4 ha of salt marsh remained in the area from about 60 ha that were present about a century ago (Goetghebeur 1976), while only one dune slack, moreover in late successional stage, occurred in the neighbouring dune area.

Restoration success in other salt marsh or dune slack studies has mostly been evaluated by plants or birds and there are only very few investigations that also used information on terrestrial invertebrates in this context. Yet invertebrates offer unmatched possibilities for such studies, being extremely diverse, with many highly specialised organisms and useful at small as well as larger scale. Native arthropod assemblages are abundant and considered functionally important in many ecosystems, and certainly in wet dune slacks and salt marshes (Gratton & Denno 2005). Successful restoration of (benthic) invertebrate communities of salt marshes has been reported to require consideration of both habitat (ecological) characteristics and dispersal ability of target species, even in created sites in close proximity to natural source areas (Armitage & Fong 2004, Warren et al. 2002).

Ground beetles are documented from about 1850 in the river Ijzer estuary and have been studied in detail and monitored continuously, along with spiders, for about the past 20 years (Baert & Maelfait 1999, Desender 1996, Desender et al. 1992, 2006). The study area has been recognised as a biodiversity hotspot, at least to recent Belgian standards, but is hoped to become even more interesting in the future. The large array of typical ground beetle species that disappeared from the area in historical times and have not yet been able to recolonise, points to a major problem of dispersal limitation. This is certainly true for salt marsh species and yet, somewhat paradoxically, nearly all of these species are known to possess a high dispersal power (Desender 1989a). In this process of recolonisation, availability and short distance of potential source areas are obviously of major importance, combined with the role of coincidence. Many of the most typical dune slack and salt marsh beetles from our region indeed are highly mobile as an adaptation to unstable conditions and inundation risks in the highly dynamic habitats they prefer (Desender 1989a, for interesting exceptions to this pattern, see also Desender 2005a, Desender et al. 1998, Dhuyvetter et al. in press). Typical dune slack invertebrates, as well as plants, therefore rely for their successful establishment on regular dispersal from other natural areas with persisting young dune slack successional stages (Bossuyt et al. 2003, Bossuyt & Hermy 2004, Desender et al. 1998, Esselink et al. 2000, Wolters et al. 2005a). To conclude, regional factors are crucial for a successful colonisation and establishment in nature restoration sites.

Our results in the nature restoration sites of the Ijzer estuary show the importance of such regional processes for ground beetles from dune slacks and salt marshes, be it at somewhat differing scales. Several rare or threatened ground beetles from mud flats and salt marshes, surviving in the salt marsh relic of our study site, *Pogonus chalceus*, *Dicheirotrichus obsoletus*, *D. gustavii* and *Bembidion laterale*, rather quickly expanded their populations in the adjacent, newly created, salt marsh areas, especially during the most recent year included in the above-mentioned analyses (2004). This is not or not yet the case for the species *Dyschirius salinus* and could be due to a lack of sufficient clay in the substrate of the new marshes. This beetle indeed is adapted to a very fine-structured salty substrate and avoids sandy sediments (Desender 1989a, Turin 2000). On the other hand, the relatively high abundances on newly created sites of species preferring more coarse-grained sediments, such as *Bembidion laterale* and *Dicheirotrichus obsoletus* (in contrast to the congeneric species *D. gustavii*, more typical for heavier soils), also point into the same direction. These beetles clearly prefer a more sandy substrate as compared to many other salt marsh or halophytic carabids (Desender 1989a). However, when comparing, at a larger scale, the actual salt marsh communities observed today to the extended list of carabid species that have been lost during the past century from the Ijzer estuary, the result is not yet very positive. Not a single of these salt marsh ground beetle species managed to (re)colonise the area yet. This strongly suggests a negative influence of dispersal limitation, in other words the lack of high quality salt marsh areas at sufficient proximity that could act as source areas for the re-establishment of species that disappeared during the past century from our study area.

Dune slack species in our region seem not (yet) to suffer much from such dispersal limitation, as most of the species recorded historically have already attempted to recolonise the restored sites of the river Ijzer estuary. There are indeed several dune slack areas of high interest at relatively shorter distances along the Belgian West coast, as compared to a complete lack of potential salt marsh source areas in this region. Unfortunately, only few of these colonisation attempts by dune slack carabid species re-

sulted in successful establishment of populations, which might be linked to the fast succession taking place in restored habitats, and/or the lack of sufficient dynamics, possibly also related to relatively small restoration scales, certainly applying to the restored dune slack.

Sedimentation characteristics on restored sites impose further constraints on colonisation and successful establishment of ground beetles, because most of these beetles prefer specific soil conditions (Turin 2000). Many authors have acknowledged the importance of sand/silt balances in sedimentation for the outcome, and long term effects of recovery of restored salt marshes or dune slacks, thereby also stressing the need for long term strategic monitoring of indicator organisms (Crooks et al. 2002, Garbutt et al. 2006, Hughes & Paramor 2004, Mendelssohn & Kuhn 2003, Lammerts et al. 2001, Olf et al. 1997, Pethick 2002). Related to this, dredged material has been proposed for soil subsidy in order to combine dredging needs with coastal marsh rehabilitation and restoration (Weinstein & Weishar 2002). Long term monitoring of salt marsh or dune slack organisms also was mentioned as crucial in view of actual and future climate change and sea level rise (Lammerts et al. 2001, Olf et al. 1997, Weis & Weis 2003). Our results show that ground beetles appear to be excellent candidates as early warning indicators or monitoring tools in all of these contexts.

We obviously have documented the very first stages only of the development of restored and new sites in the Ijzer estuary. Some earlier studies have referred to the time and conditions needed for successful restoration of salt marsh communities: rapid recolonisation is expected for pioneer and low-marsh species; provided they are occurring in nearby source areas and restored sites are at appropriate altitude (Bernhardt & Koch 2003, Wolters et al. 2005B). Gratton & Denno (2005) reported a rapid recovery of native arthropod assemblages associated with restored brackish marshes after the removal of invasive *Phragmites australis*, but it is unclear how far such results can be generalised without reference to a regional-specific situation (see also Eertman et al. 2002, Weis & Weis 2003). Vegetation of restored wet dune slacks did not yet reach a stable state after 5 years (Grootjans et al. 2001). Brackish wetland vegetation and soil characteristics have been reported to take up to 90 years or even more than 200 years to develop and become equivalent to natural marsh (Craft et al. 2002). It is therefore necessary to continue our monitoring of old and new habitats of the Ijzer estuary nature reserve as a prerequisite for a well-founded long-term evaluation of the performed nature development measures. Moreover, further monitoring is of high fundamental scientific interest against the background of our long-term sampling data on beetles and spiders, sampling which is continued without interruption on several sites in this estuary. This should enable us in the future to make a clear distinction between directed changes in the area (mainly as a consequence of ecological processes, accompanying measures for conservation management) and changes due to year-to-year population dynamic fluctuations (whether stochastic or not) in carabid beetle or spider populations.

Another important aspect that was not yet taken into account in the current monitoring concerns population genetics of a number of target invertebrate species. To this end, we have formerly studied, on a regional scale, a number of ground beetles and other terrestrial arthropods for their genetic diversity and differentiation in salt marshes, including the Ijzer estuary marshes (cf. Desender 1985, 1989A, B, Desender et al. 1998, Desender & Verdyck 2001). It would be of much interest to repeat such studies in the future for species that occurred earlier in the area or not, in order to compare population genetics from newly created or restored populations to those from the old salt marsh relic or to those from other (potential source) areas. Such population genetic information not only might enable to the tracing of the origin of colonising species, but also could be of major importance to judge population viability in the long run.

Our results, less than five years since nature restoration measures were taken, show a number of new ground beetle species and assemblages, but warrant that many of these could be rapidly lost again unless natural dynamic processes are kept ongoing. It is, moreover, still unclear whether the newly created salt marsh habitats will evolve as hoped for, because of the observation that newly deposited sediments in this salt marsh are relatively coarse-grained (sand instead of silt), whereas, at the same time, the old salt marsh remnant seems to be more and more under the influence of sand deposits.

Moreover, many salt marsh species, present in the area about a century ago, did not re-appear yet, strongly suggesting dispersal limitation. To conclude, ground beetles appear once again as powerful ecological indicators. Further invertebrate monitoring is therefore imperative not only for a better understanding of the patterns and processes generated by the nature restoration measures, but also as a possible early warning system for the need of additional management measures in the future.

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Colonisation and source-sink dynamics in spiders and ground beetles after dry dune habitat restoration along the Belgian coast

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Abstract

We monitored the spider and ground beetle assemblages of old dune and newly created dune-like habitats in the Ijzer estuary by means of four years of continuous pitfall sampling (2001-2004). The new sites built with dune sand were rapidly colonised by good dispersing species. These populations thrived so well during the first years after colonisation that they acted as sources, for which the old dune habitats were the sinks. That temporal collateral effect of nature restoration did not seem to cause persisting damage in the old dune habitats once the source populations had disappeared.

Because general stochastic environmental fluctuations, like cold winters, seem to cause important year-to-year variation in population size of a number of species, it is advisable to sample developing and restoring habitats at the same time as their targets.

The newly created habitats appeared to offer opportunities to enlarge the population size of several species of dune living ground beetles and, to a lesser degree, spiders. A multitude of more specialised dune species could not (as yet?) install viable new populations. A continuing sampling effort will be required to monitor the development, so that additional nature restoration or management measures can be taken when bio-indicated to be needed.

1 Introduction

1.1 Recent nature restoration project in the river Ijzer estuary

At the turn of the century, a major restoration project was realised on the right bank of the estuarine part of the river Ijzer, Flanders, Belgium (Deboeuf & Herrier 2002, Hoffmann 2004, Herrier et al. 2005). The first two phases consisted of the demolition of the buildings and roads of the former naval basis and the removal of the jetties and quays of the former military harbour and slipway. The excavated sandy soil from the quays was used to build dune-like hills above the pits left by the removal of the buildings as well as a dune-like dike along the tidal mud flat created after the removal of the harbour and the slipway. These works ended mid-way though March 2001 and immediately afterwards a multidisciplinary monitoring scheme was started (Hoffmann et al. 2005).

1.2 Studies on dry dune spiders and ground beetles in the Ijzer estuary

In this paper we report the results of the first four years of monitoring of two newly created dune-like habitats. We assess the spider and carabid fauna assemblages having colonised these newly created habitats in comparison with the assemblages occurring in the adjoining old dune habitats, which were already sampled from 1990 (Desender 1996, 2005). Desender et al. (this volume) give details on the former history of the IJzer estuary. We compare the spider and carabid assemblages of five sampling sites (Fig. 1). Two are situated on the fore-dune, one on the seaward side (site A), and one on the landward side (site C). These sites belong to the association *Ammophiletum arenarii* with a ground cover of marram grass tussocks of about 50 %. At the more protected site C, some 5 to 10 % more is covered with low growing grasses, herbs, and *Euphorbia paralias*. The third site (site E) is a grey

dune belonging to the phytosociological order *Cladonio-Koelerietalia* (Provoost et al. 2004). During the sampling period 2001-2004 it had the characteristics of the alliance *Polygalo-Koelerion*, closed grassland with a well-developed sod layer. It was short-grazed by sheep. During the 1990's the site had much more open grey dune vegetation dominated by lichens and mosses and with a poorly developed organic soil layer, in other words it was rather a *Tortulo-Koelerion*. The change in carabid and spider assemblages due to this vegetation change will be treated elsewhere (Baert et al., in prep). The three before mentioned old dune sites are designated as priority habitats in the EU Habitats Directive. Therefore, we use them here as targets to assess the quality of the newly created sites with dune sands: sites F and G. These new sites are situated 50 m apart on a newly built dike about 500 m inland from the fore-dunes (Fig. 1). The dike was built with excavated sand with a content of finer soil particles (clay, organic matter) of about 10 %. At site G, on top of that, about 0.5 m of mineral sand with a 5 to 10 % content of shell fragments was added and planted with marram grass tussocks. By 2004, G showed a ground cover of about 50 % of marram tussocks with in between some 5 % grasses and herbs. From only about 10 % ground cover during the growing season in 2001, site F evolved in 2004 to a cover of about 95 % of sod forming grasses, mosses and herbs, kept short by sheep grazing.



Figure 1: Localisation of old (A, C, E) and newly created (F, G) dune sand sites along the river IJzer.

1.3 Year-cycle pitfall trapping as a sampling method for studies on spiders and carabids

All sampling was done with pitfall traps, i.e. glass jars with a content of half a litre, a depth of 10 centimetres and a diameter of 9.5 centimetres. These traps are dug in the soil with their upper rim just beneath the mineral soil surface and half-filled with 4% formaline solution as a fixative and a few

added drops of detergent to lower surface tension. At each sampling site, three pitfall traps are installed about five metres apart. They are emptied and refilled at fortnightly intervals. For each trap, once per fortnight, all adult male and female spiders and ground beetles were identified and counted. Each site was sampled in this manner for at least a complete year-cycle, starting from the beginning of April and ending at the end of March the next year. The total number of males and females caught during such a complete year-cycle is the figure used to assess the relative abundance of a species in a series of sites to be compared. In other words, these year-cycle numbers are used to determine the habitat preference of a species and to ascertain the year-to-year changes in abundance of a species at a particular site, sampled for several year-cycles.

Capture rates of pitfall traps not only depend on population densities (abundance) of the species caught, but also on intra- and interspecific differences in soil surface activity levels and in trappability as influenced by habitat structure (Greenslade 1964, Maelfait & Baert 1975, Baars 1979, Halsall & Wratten 1988, Topping & Sunderland 1992, Sunderland et al. 1995, Maelfait 1996, Antvogel & Bonn 2001). Therefore, they are not suited for comparisons between species in terms of their abundance. However, resulting from a long enough sampling period (a year-cycle) in not too structurally different sampling sites, pitfall capture rates give reliable estimates of the relative abundance of each particular species over the sampling sites (Maelfait & Baert 1975, Baars 1979, Desender & Maelfait 1986, Maelfait 1996, Retana & Cerdá 2000). When used for ordinations or classifications, capture rates have therefore to be relativised per species, as we did hereafter in the program PC-ORD used for Detrended Correspondence Analysis, DCA (Jongman et al. 1995, McCune & Mefford 1999).

1.4 Climate

The Belgian coast has a mild Atlantic sea climate. The mean winter and summer temperatures measured in nearby climatologic stations during the year of sampling were 4.9 and 17.3 °C. The number of frost days per month between July 2000 and July 2004 are shown in figure 2.

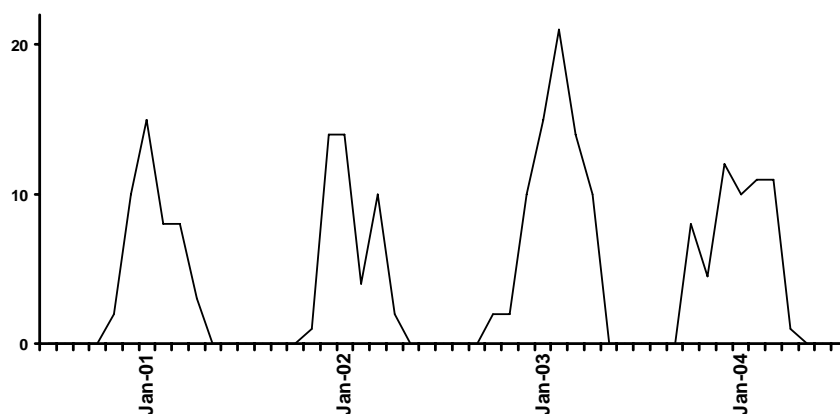


Figure 2: Number of frost days per month from July 2000 to July 2004.

2 Results

The scores of sites and species obtained after a DCA of the spatio-temporal distribution of the most abundantly caught species are plotted along the first and second axis in Fig. 3 for spiders and Fig. 4 for carabid beetles. The eigenvalues of the first, second and third axes are for spiders: 0.55, 0.26, and 0.07; for carabids: 0.53, 0.22, and 0.08.

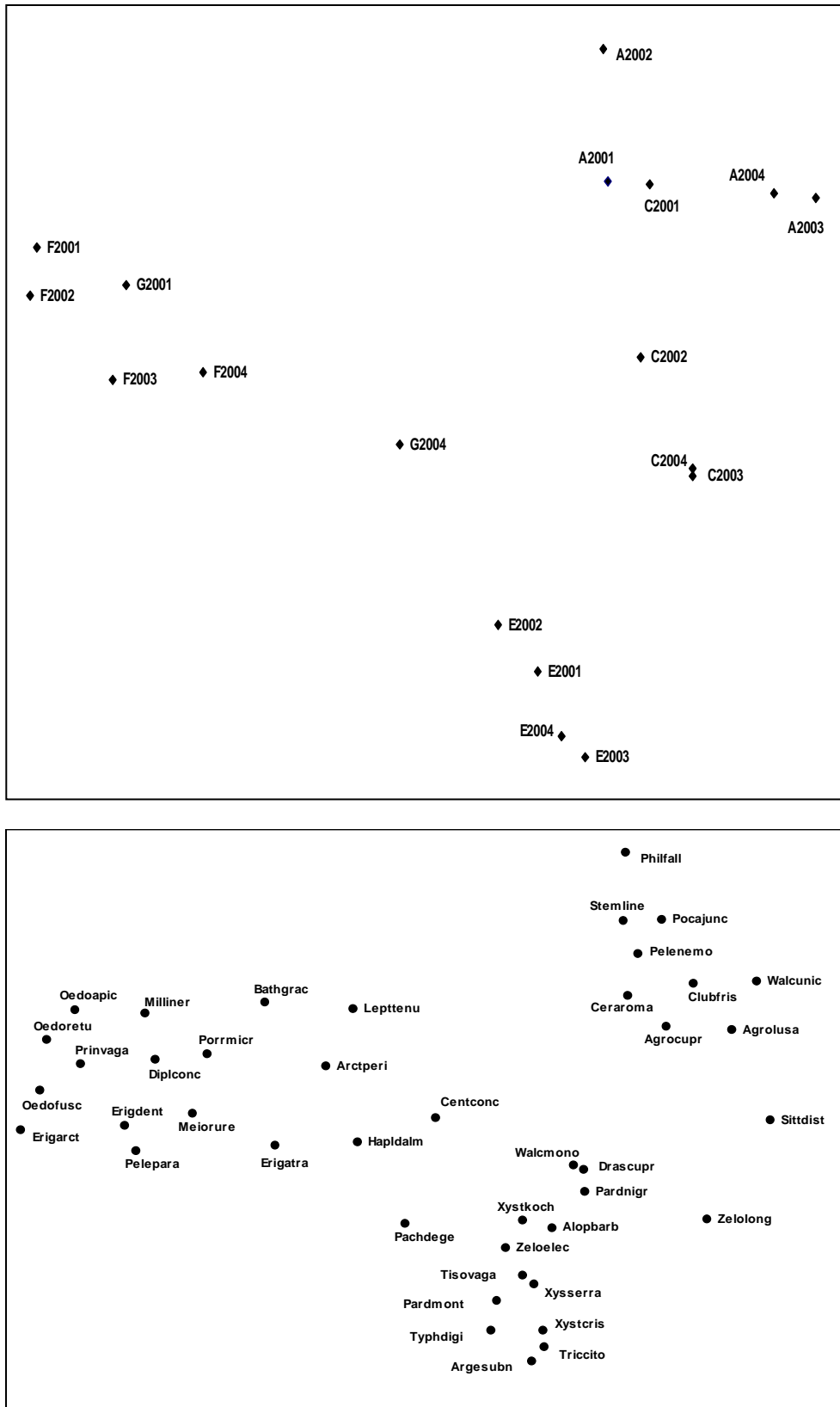


Figure 3: Scores of the year-sites (above) and the species (below) along first and second axis after DCA ordination of the spatio-temporal distribution over old and new dune sand sites of the 42 most abundant spiders. For full species names of abbreviations used here: see Table 1.

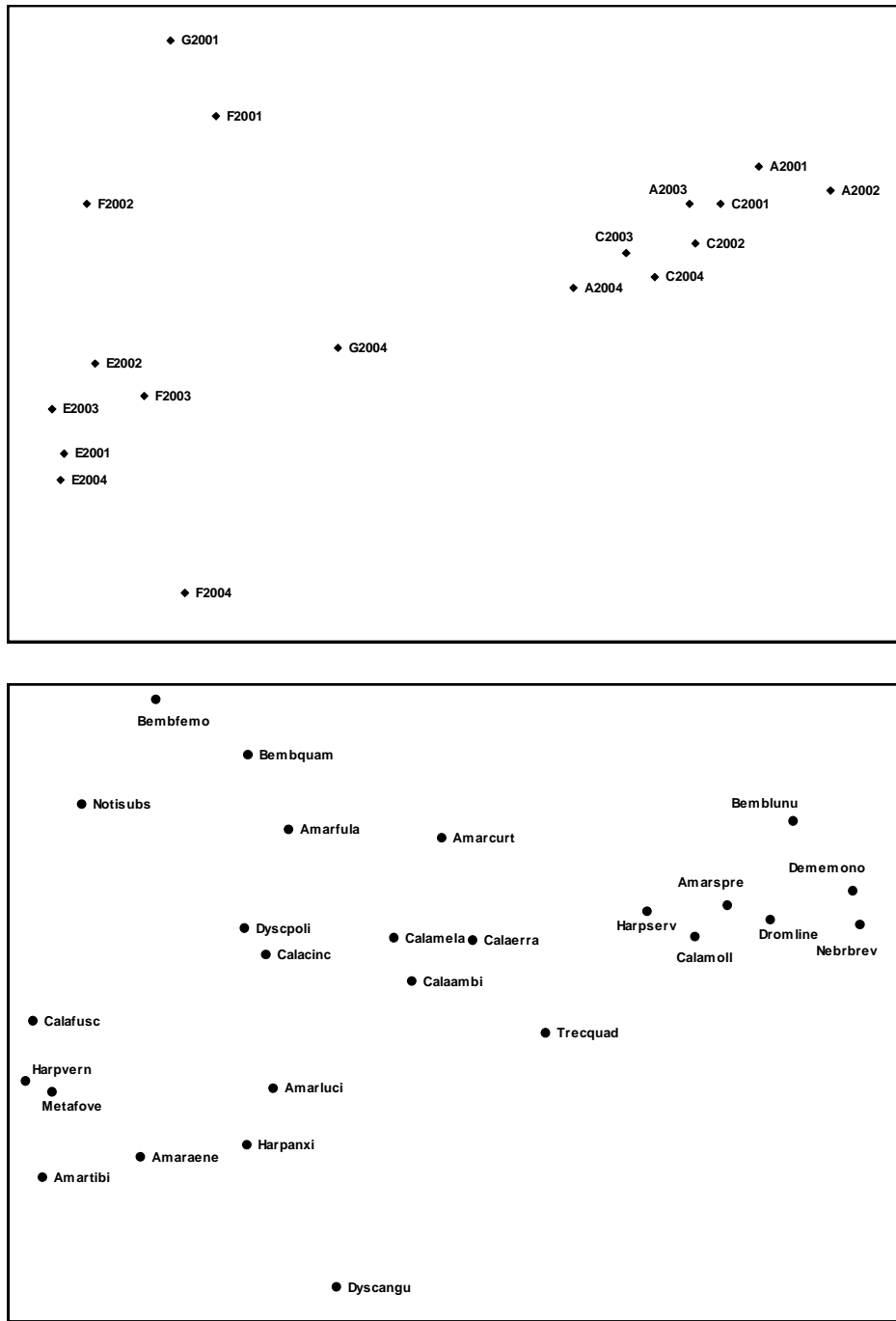


Figure 4: Scores of the year-sites (above) and the species (below) along first and second axis after DCA ordination of the spatio-temporal distribution over old and new dune sand sites of the 26 most abundant carabid beetles. For full species names of abbreviations used here: see Table 2.

On the basis of these ordinations, but also derived from the captures made in the old dune sites during the five years before the construction of new dune sand habitats, the year-sites and species are ordered in Table 1 (spiders) and Table 2 (ground beetles). The median values of the five yearly captures before 2001, i.e. from 1996-2000, in dune sampling sites A, C and E are mentioned in Tables 1 and 2. To obtain a more convenient arrangement of the results (and the discussion), each species was attributed to one of the Classes 1a to 3c (Column “Class” in Tables 1 and 2) according to its spatio-temporal distribution over the year-sites. However, when considered needed, idiosyncrasies of particular species are also taken into account.

Table 1: Yearly capture rates of the most abundant spider species per sampling year-site, ordered and grouped according to the DCA ordination of Fig. 2 and the capture rates made in the old dune habitats in the five years before 2001 (median value of 5 yearly capture rates in columns E, C and A). Red-listed species based on Maelfait et al. (1998). Code: abbreviation for species name as used in figure 3. Class: explained in text.

Code	Spider species	G2001	G2004	F2001	F2002	F2003	F2004	E2001	E2002	E2003	E2004	C2001	C2002	C2003	C2004	A2001	A2002	A2003	A2004	E	C	A	Class	Red list	
eriarct	<i>Erigone arctica</i>	96	5	182	3548	1800		4	14	7	1	8	1	1	39	51	1		2			1a	-		
oedrusc	<i>Oedothorax fuscus</i>	103	86	352	23			1				2	5		2					2			1a	-	
oedretu	<i>Oedothorax retusus</i>	8	24	75	6			1				1	6		2				1				1a	-	
oedapic	<i>Oedothorax apicatus</i>	253	283	626	8	1		1	2	1	1	2	2		23	119			7	1		1	1a	-	
privaga	<i>Pririgone vagans</i>	4	1	10	25	5		2				3			4	2							1a	-	
milliner	<i>Collinsia inerrans</i>	6	2	8	21	4	2	1				3			4	3							1a	-	
dipconc	<i>Diplostyla concolor</i>	6	3	11	18	29	10	2		1	1	1			3	6			1			1	1b	-	
meiure	<i>Microneta rurestris</i>	6	43	6	44	7	26	1				5			1					1	1		1b	-	
pormirr	<i>Porhonna microphthalmum</i>	6	4	6	10	7	13	3				3			1	4			1	1			1b	-	
pelpara	<i>Paleopsopsis parallela</i>	2	57		85	518	889																1b	-	
erident	<i>Erigone dentipalpis</i>	56	10	118	711	188	101	96	177	16	23	6	1	1	3	7				83	1	1	1c	-	
batgrac	<i>Bathyphanes gracilis</i>	45	7	64	112	17	35	14	43	3	12	2	9	3	44	54			9	13	6	11	21	1c	-
eriatra	<i>Erigone atra</i>	56	18	172	414	95	49	260	448	58	113	2	3	1	2	15			3	321	8	7	1c	-	
leplenu	<i>Tenuiphantes tenuis</i>	52	93	63	168	23	120	22	57	8	51	15	10	1	20	89	120	27	99	19	49	97	2a	-	
hapdalm	<i>Haplodrassus dalmatensis</i>	7	72	4	24	48	66	15	36	12	17	15	25	22	9	18	2	1	8	67	21	4	2a	yes	
cenconc	<i>Centromerita concinna</i>	2	98	2	6	12	26	33	63	11	33	26	17		3	23	2	1	1	47	9	4	2a	-	
xyskoch	<i>Xysticus kochi</i>	27	11	6	18	54	51	57	69	12	57	12	25	15	12	18	7	5	9	14	10	3	2a	-	
dracupr	<i>Drassodes cupreus</i>	16	2		2	4		1	3	2	4	3	10	11	13	1	2	4	3	1	25	2	2a	-	
alobarb	<i>Alopecosa barbipes</i>	2	11	2	2	3	15	13	20	23	15	4	7	10	15	10	2	2	2	6	32	2	2a	yes	
steline	<i>Stenonyphantes lineatus</i>	47	1	3	1	5	10	6	3	5	5	32	83	31	22	141	267	64	67	5	7	27	2a	-	
pelnemo	<i>Parapaleopsis nemoralis</i>	5	1	1	1	6	3	3	3	2	7	23	19	3	6	14	56	13	46	15	158	73	2a	yes	
arceper	<i>Arctosa petita</i>	20	7	3	10	14	8	1				8	15	7	2	4	3	2	3	2	3	6	2b	yes	
cerroma	<i>Styloctetor romanus</i>	3						2		1	1	98	35	18	13	10	3	2	3	2	1		2b	yes	
pacdege	<i>Pachygnatha degeeri</i>	4	2	2	3	3	2	23	5	11	2	1			2					2			2c	-	
zelelec	<i>Zelotes electus</i>	1	10	3	2			16	16	12	10	1	2		4				15	6			2c	yes	
parmont	<i>Pardosa monticola</i>	9	31	3	3	12	20	96	87	90	193	7		2	4	1			1	17			2c	yes	
typdigi	<i>Typhochrestus digitatus</i>	1	3	1	2	2	14	7	14	10	108	1								14			2c	-	
walmono	<i>Walckenaeria monoceros</i>					1		22	34	20	31	9	14	6	1	9	11	6	3	39	21	18	3a	-	
agrucupr	<i>Agroeca cuprea</i>			1				14	3	3	7	7	10	7	3	20	10	15	35	5	11	20	3a	yes	
parnigr	<i>Pardosa nigricaps</i>	1	1	3				12	3	6	6	1	6	11	11	5	1		4	3	1		3a	-	
zeilong	<i>Zelotes longipes</i>									1	2	2	3	13	9			3	6	1	1		3b	yes	
sitdist	<i>Sitticus distinguendus</i>			1				1	3	3	2	1	3	3	2	1		24	10	2	2		3b	yes	
agriusa	<i>Agroeca usatica</i>	1						2	5	1	6	2	5	1	6		8	4	38	1	6	3	3b	yes	
clufiris	<i>Clubiona frisla</i>							1	13	14	4	13	14	4	11	25	13	26	12	5	13	18	3b	yes	
waluni	<i>Walckenaeria unicornis</i>							3	2	7	14	2	7	14	16				16	1	2	2	3b	-	
pocjuunc	<i>Pocadicnemis juincea</i>							2	2	2	1	2	2	7	7	27	41	14	18	1	2	26	3b	-	
phifall	<i>Philodromus fallax</i>							19	1	1	1	19	1	1	1	14	27	3				1	3b	yes	
tisvaga	<i>Tiso vagans</i>	1		1				29	11	9	19												3c	-	
xyserra	<i>Xysticus erraticus</i>							17	17	9	21									5			3c	yes	
xysoris	<i>Xysticus cristatus</i>			2		2		17	15	27	23			2	2	1	1	1	1	11			3c	-	
trictro	<i>Trichopterna cito</i>			1				39	20	36	61			1	1	1				22			3c	yes	
argsubn	<i>Argenna subnigra</i>			5				20	19	37	41			1	1	1				24			3c	yes	

Class 1a species appear in site G and F in 2001, occur in high numbers in F in 2002 and 2003 (G not sampled in these two years) and disappear (or almost so) in 2004. All spider species of this group show the pattern as exemplified for *Erigone arctica* (Fig. 5). During the years of high abundance in G and F, these species also appear in considerable abundance in A, but also in E and C, where these species were virtually absent during the five years before the creation of the new dune sand habitats F and G.

Table 2: Yearly capture rates of most abundant carabid beetle species per sampling year-site, ordered and grouped according to the DCA ordination of Fig. 2 and also on the basis of the capture rates in the old dune habitats during five years before 2001 (median value of 5 yearly capture rates in columns E, C and A). Red-listed species based on Desender et al. (1995). Code: abbreviation for species name as used in figure 4. Class: explained in text.

Code	Carabid species	G2001	G2004	F2001	F2002	F2003	F2004	E2001	E2002	E2003	E2004	C2001	C2002	C2003	C2004	A2001	A2002	A2003	A2004	E	C	A	Class	Red Data Book
Bembfemo	<i>Bembidion femoratum</i>	170	1	80	73			1								3	1					1a	-	
Bembquam	<i>Bembidion quadrimaculatum</i>	8		15	10	3		5								3	1					1a	-	
Amarfulva	<i>Amarula fulva</i>	6	15		3	13	3	1				1				1						1b	-	
Dyscpoli	<i>Dyschirius politus</i>	2	20		12	33	1									2	2					1b	yes	
Dyscangu	<i>Dyschirius angustatus</i>	5	25	5	25	63	74					3	3	3	5	3	8	15				1b	yes	
Nebrev	<i>Nebria brevicollis</i>	3	3	2				2	1			1	4	1	1	49	3	1				1	-	
Notisubs	<i>Notophilus substriatus</i>	1	16	23	113	67	3	25	29	13	23	1				1	1			5		1c	-	
Treacquad	<i>Trechus quadristriatus</i>	4	11	7		4	15	1				1				8	1	6	21		6	16	-	
Calacinc	<i>Calathus cinctus</i>	52	112	56	106	75	133	241	215	26	54	30	37	17	1	13	30	28	3	221	13	1	yes	
Calaaambi	<i>Calathus ambiguus</i>	9	339	20	14	43	36	1	4	7	13	15	14	9	1	2	2			19	2		yes	
Calamela	<i>Calathus melanocephalus</i>	1	3	4	2	3	1	8	7	6	9	3	3	1	1	3	5	1		3	1	1	-	
Calaaerra	<i>Calathus erratus</i>	2	410	3	6	23	3	9	20	21	29	4	11	26	7	2	6	46	3	27	71	10	-	
Calamoll	<i>Calathus mollis</i>	37	1243	22	28	30	79	280	154	38	53	772	641	729	329	434	944	330		89	200	75	2a	yes
Harpsev	<i>Harpalus servus</i>	8	65	5	4	14	5	2	4	1	6	21	20	52	51	2	17	18	27	7	43	9	yes	
Amarluci	<i>Amarula lucida</i>	4	34	2	1	7	14	33	30	9	47	1	4	2	1	4	5	18		13	1	2	yes	
Harpaxxi	<i>Harpalus anxius</i>	5	89	6	25	63	59	45	36	23	33	4	2	3	9	1	3	4	13	17	25	15	-	
Amarcurt	<i>Amarula curta</i>	2	5	2	3	1		5	9	4	5	1				1	1	6	10	5	1		-	
Dromline	<i>Dromius linearis</i>	4		1		2		1				5	10	3	3	9	12	13	7	3	1		-	
Dememono	<i>Demetrius monostigma</i>	2										9	3	2	1	17	37	18	9	3	1		yes	
Amarraene	<i>Amarula aenea</i>	2	35	5	19	28	36	23	21	8	45					1	1	3		11	1	1	-	
Calafusc	<i>Calathus fuscipes</i>	2	32	7	61	45	21	96	189	224	131					1				32	1		-	
Harpvern	<i>Harpalus vernalis</i>	7	7	6	7	24	9	133	64	64	93					1				45	4	1	2c	yes
Metafove	<i>Metabletus foveatus</i>	2	14	13	12	11	20	151	45	19	33					1				77	10	1	-	
Amararti	<i>Amarula tibialis</i>	1			8	4	34	13	10	29		2				2	1			3			yes	
Amarspre	<i>Amarula spreta</i>							4				12	5	3		3	21	10					-	
Bembturnu	<i>Bembidion lunulatum</i>	9		12	17	5	1	5	3			12	5	3		126	84	105	18		2	15	-	

As the 1a species, the 1b species did not have populations in the old dune habitats before 2001. In contrast to the former class, the species of 1b built up viable populations by 2004 in the new sites. This is especially the case for the spider *Pelecopsis parallella* and the ground beetle *Dyschirius angustatus* in site F.

Class 1c species occurred before 2001 in the old dune habitats, especially in the dune grassland E. In F they display the same year-to-year variation as the 1a species, except that the class 1c species have established populations in 2004 in G and/or F. Class 2a contains species that occurred before 2001 and during the period 2001-2004 in A, C and E. These species were able to colonise G and/or F during that 4-year period. The species making up the class 2b are spiders and carabids of the fore-dune ridge that succeeded in the colonisation of one or both of the new sites.

Class 2c species are inhabitants of habitats like site E (stabilised grey dunes) that succeeded to colonise G and/or F.

Classes 3a, 3b, and 3c are species of the fore-dunes and/or dune grassland (A and C and/or E) that were not able to colonise the newly created habitats.

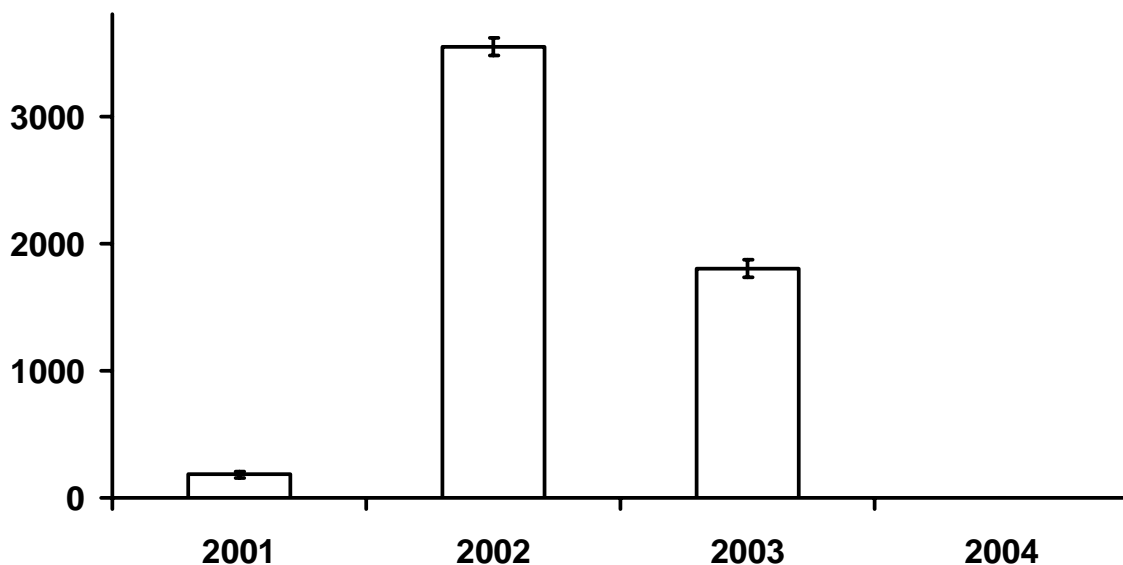


Figure 5: Yearly capture rates (\pm 95 % confidence intervals) of the spider *Erigone arctica* in the newly created dune sand site F.

3 Discussion

All spider species of classes 1a and 1b, with the exception of *Pelecopsis parallella*, are small linyphiid spider species known to be good aeronautic dispersers. These species occur in high densities in unstable, poorly vegetated, temporal habitats such as regularly inundated riverbanks, arable land, intensively exploited hayfields, and pastures (Bell et al. 2005, Bonte et al. 1998, 2002, 2004, De Keer & Maelfait 1987A, B, 1988, 1989, Maelfait & De Keer 1990, Maelfait et al. 2004). Their colonisation capacities are well illustrated by their rapid massive appearance in G and F. While the species of class 1b, which appear a bit later than 1a species, have populations in 2004 in G and F, the 1a species already disappeared again. The vegetation apparently attained a too high ground cover by that year for such species. *Pelecopsis parallella*, a species of which there are no observations of aerial dispersal, found especially in F a very suitable habitat. It is this far more widely distributed species and not its rare sister species of dry dune habitats *Parapelecopsis nemoralis* that colonised the new sandy habitats G and F.

The spider species of class 1c are excellent aerial dispersers with a wide distribution, including man-made habitats. They differ from the class 1a species because they need low vegetation (grasses, low herbs) to attach their webs used to catch prey. In addition, the carabid *Notiophilus substriatus* is a species bound to short grassy vegetations (Desender et al. 1995, Turin 2000). These species are also quite abundant in E. This suggests that colonising individuals in G and F could as well have come from these old dune habitats as from nearby agricultural land.

All the carabids of 1a, 1b, and 1c are also good aerial dispersers, i.e. full-winged (macropterous) species with well-developed flight muscles (Desender 1989, 2000). As for the spiders, all these species, with the exception of two *Dyschirius* species, are widely distributed species in disturbed habitats. The two *Dyschirius* species are red-listed and are bound to more or less humid, patches of sand devoid of vegetation (Desender et al. 1995), as occurring in young dune slacks, a rare habitat type along our coast. As can be seen in Table 2, they were doing very well in at least of one of the two new habitats in the period 2001-2004. However, as observed in the captures of 2005 and 2006 (Desender, pers. comm.) this was only a temporary situation. It is expected that the species will again locally become extinct when highly dynamic, open sand situations cannot be kept in the area due to expected vegetation succession (cf. Desender ET AL., this volume).

The relatively high numbers caught in C, E and especially in A of several species of the classes 1a and 1b in 2001 and 2002 are most probably the result of an overflow of the rapidly growing populations in F (and presumably in G, not sampled in 2002). Mass effects (Leibold et al. 2004, Leibold & Miller 2004) by species with a high aerial dispersal propensity, i.e. by ballooning spiders and flight by carabids with fully developed hind wings (and functional flight muscles) appear to have been important in the first year of the restoration process. During the first year, newly created habitats acted as a sink for dispersers coming from highly productive source populations in their vicinity, such as agricultural habitats, a dredging sludge dump area of several hectares excavated during the sampling campaign and possibly also from the dune grasslands. Later on, the newly created habitats became a source of individuals of good dispersing species immigrating in the long existing sites. The presence of these species does not seem to have caused damage or lasting changes in the assemblage composition of old dune sites as these fugitive species disappeared again by 2004 and as no other former occurring species were lost (Figs. 3, 4, and columns A, A2001 to A2004 of Tables 1 and 2). These results confirm the strong effect that dispersal ability may have on spider (meta-) community composition of grey dunes (Bonte et al. 2004, Bonte et al. 2006). Here we observe that this effect rapidly fades away once nearby local disturbances come to an end.

As the species of class 1a and *Meioneta rurestris* (1b), the 1c species showed a sharp decline in numbers between 2002 and 2003. This decimation has probably been caused by the relatively harsh conditions of the winter 2002-2003 (Fig. 2) to which these species appear to be susceptible. Such high winter mortality can also be observed for several other species and sites, e.g. for the spider *Tenuiphantes tenuis* at all sites. Why particular species appear to be more vulnerable to harsh winter conditions than others would require investigations that are more detailed. This striking influence of climate and possibly other general stochastic environmental factors implies that the estimation of the distance-to-target of developing and restoring assemblages can best be done by sampling them simultaneously with the assemblages of the target habitats.

Of the remaining 29 spider and 18 carabid species, respectively 14 and 16 were able to colonise the new habitats after 4 years (classes 2a, 2b and 2c of tables 1 and 2), while 15 spider and 2 carabid species failed to do so (classes 3a, 3b and 3c of tables 1 and 2).

Colonisation was most successful for species with a wider distribution in dune habitats, i.e. species occurring in E, C, and A (classes 2a and 3a); for spiders: 8 species out of a total of 11 could colonise, for carabids: 9 out of 9. Colonisation was less successful for spider species of the dune grasslands like E (classes 2c and 3c: spiders: 4 out of 9, carabids: 5 out of 5). The new habitats did not yet seem suitable enough for or could not be reached yet by several species of the fore-dunes (2b and 3b): only 2 out of 9 spider species and 2 out of 4 carabids could invade the new sites.

Overall, by 2004, 16 out of the 18 typical dry dune carabid species of the area could establish populations in the new dune-like sites created by the nature restoration project, while this was only the case for 12 out of 29 typical dry dune spiders. For seven endangered spider species (column Red Data Book in table 2) one or both new habitats appeared suitable enough to be permanently colonised. However, for 9 other spider species of the Red Data Book this was not or not yet the case. For carabids, all 8 Red Data Book species were typical dry dune carabids of the area established populations and two new Red Data Book species came in, at least temporally (see above).

This leads to the conclusion that the newly created dune-like habitat can be considered a valuable enlargement of the natural habitats for several typical dune living species, mostly carabids, to lesser degree spiders. Of especially this last taxonomic group, the more specialised dry dune species could not install populations in the new habitats. Further monitoring will be required to evaluate if this situation gradually improves or if additional nature restoration and/or management measures will have to be considered.

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Rapid improvement of grey dunes after shallow sod cutting

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Abstract

Grey dunes are an important habitat in Dutch coastal sand dunes, harbouring various Red Data Book flora and fauna species. However, during the last decades, dune grasslands belonging to this habitat have suffered from severe grass encroachment due to prolonged stabilisation, atmospheric pollution, and rabbit decline. In the Amsterdam Water supply Dunes shallow sod cutting was applied in 2002 on a small scale in dune grasslands of the syntaxa *Taraxaco-Galietum veri* and *Phleo-Tortuletum ruraliformis*, with high cover of the tall grass species *Calamagrostis epigejos* and *Elytrichia maritima*. In this article, we describe the first results of sod cutting, which was executed in order to recover dune grassland vegetation and the accompanying characteristic fauna species.

Soil and biomass were studied and permanent plots were used for vegetation survey and pellet counting of rabbits. In addition, several insect groups were monitored in pitfalls and plots, and along transects. The dune grasslands of the *Taraxaco-Galietum veri* show a strong increase of characteristic plant species and a recovery of rabbits, butterflies and grasshoppers. In the dune grasslands of the *Phleo-Tortuletum ruraliformis* vegetation responds more slowly to sod cutting, probably due to lower nutrient availability. Number of rabbits and the grasshopper *Oedipoda caerulescens* increased as well. The longer-term effects remain to be seen, but the results indicate that a rapid improvement of vegetation and fauna can be reached after shallow sod cutting of grass-encroached grey dunes.

1 Introduction

1.1 Grass encroachment

Until the 1980's large parts of the Dutch coastal sand dunes were covered by dune grasslands, belonging to the *Koelerio-Corynephoretea* (cf. Weeda et al. 1996). They are part of the grey dunes, which are considered a priority habitat according to the European Habitats Directive (Council of the European Commission 1992). In the last decades, grass encroachment caused fragmentation and a severe decline of the area of grey dunes in most Dutch coastal dune areas. This led to loss of biodiversity (Veer & Kooijman 1997) and a decline of vulnerable and endangered Red Data Book flora and fauna species, such as the plant species *Polygala vulgaris* and *Thymus pulegioides* and the butterfly species *Aricia agestis* and *Issoria lathonia*. They depend on a varied vegetation structure with moss patches and bare sand (Brouwer et al. 2005), and finally disappear due to the expansion of tall grasses.

Grass encroachment in coastal sand dunes in the Netherlands is enhanced by several reasons. Over many decades, site managers used to counteract wind dynamics and stabilised drifting sand and blow-outs by planting marram grass (*Ammophila arenaria*), shrubs, and coniferous forest (Van Til et al. 1999). The outbreak of myxomatosis in the 1950's and 1960's followed by rabbit haemorrhagic disease in the 1990's led to a collapse of the rabbit population. They usually play an important role in the maintenance of a short and open turf in many grasslands of the *Koelerio-Corynephoretea* (Weeda et al. 1996). Rabbit decline caused a strong decrease of small-scale dynamics in the grey dunes. Furthermore, prolonged high loads of nitrogen in the second half of the 20th Century contributed to growing nutrient availability in the generally poor grassland systems (Bobbink et al. 1998), and led to im-

proved conditions for competitive grass species such as *Calamagrostis epigejos*, *Elytrichia maritima* and *Ammophila arenaria* (Kooijman & Van der Meulen 1996, Kooijman et al. 1998).

In the Amsterdam Water supply Dunes (AWD) along the Dutch mainland coast the area of dune grasslands has decreased substantially since 1980, especially in the middle dunes, which are most sensitive for grass encroachment (Van Breukelen & Van Til 2005, Kooijman & Besse 2002). The effects of several nature conservation measures in dune grasslands were studied, such as mowing and haymaking, top soil removal and extensive cattle and sheep grazing (Veer & Kooijman 1997). However, these management tools do not always seem to contribute to a decline of grass encroachment and an increase in area of dune grassland, or are rather expensive. Mowing and haymaking has to be applied frequently and is a difficult measure in the undulating dune landscape. Top soil removal is an expensive measure, which sometimes promotes scrub expansion, especially of *Hippophaë rhamnoides*. Moreover, a transition in the vegetation structure from open to closed canopy has been determined within the dune grasslands under grazing management in the AWD (Van Breukelen & Van Til 2005). This closed vegetation is probably unfavourable for above-mentioned Red Data Book species.

1.2 Shallow sod cutting

In 2002, we started a research project as part of a nature conservation experiment, which was executed within the framework of the Dutch survival plan for forest and nature and subsidised by the Dutch government. Shallow sod cutting (5 cm top soil), a new measure in coastal sand dunes, was applied on a small scale in about 4 hectares of grass-encroached grey dunes at the locations Rozenwaterveld and 'infiltration area' in the AWD in October 2002. This measure is intermediate between mowing and top soil removal: it might be more sustainable than mowing and is less expensive than top soil removal.

Today, the Rozenwaterveld is, in many places, dominated by the grass *Calamagrostis epigejos* and the dwarf shrub *Rosa pimpinellifolia*. The dune grasslands belong to the Taraxaco-Galietum veri. In the infiltration area *Elytrichia maritima* has increased during the last decades, which caused grass encroachment in dune grasslands of the Phleo-Tortuletum ruraliformis. A monitoring plan was worked out, focussing on the recovery of the dune grassland vegetation and the return of characteristic fauna species.

1.3 Research methods

The research was executed in permanent plots of 6 m², in which the effects of sod cutting were studied on soil and biomass in the first year after the measure was applied. Vegetation relevés were made, using Londo's scale for permanent plots (Londo 1975). The development of the vegetation was analysed for structure and for ecological species groups (cf. Van der Meijden et al. 2000) of dune grasslands and ruderal vegetation, which is the result of grass encroachment. Mosses and lichens were added to this list of vascular plants, according to their distinctiveness in a local vegetation typology of the AWD (Van Til & Mourik 1999).

Rabbit (*Oryctolagus cuniculus*) activity was estimated by the amount of pellets produced in the permanent plots in four weeks, which can be used as a measure for the population density (Bankert et al. 2003). Butterflies were only studied at the Rozenwaterveld, as the flower-rich dune grasslands of the Taraxaco-Galietum veri harbour many (Red Data Book) species in comparison with the grasslands of the Phleo-Tortuletum ruraliformis of the infiltration area, which are poor in butterflies. Species were divided into an ecological group characteristic for open dune grasslands and another group with more preference for uniform vegetation structure (Wallis de Vries & Raemakers 2001). They were counted along transects of 50 metres nine times a year, in spring and summer. Numbers of individuals of different grasshopper species were counted in large plots three times in summer, during time intervals of ten minutes. Carabid beetles were caught once a week in pitfalls from March to October. Among the carabid beetles, various target species were distinguished for dune grasslands (cf. Turin 2000).

The monitoring was set up in a random block design (Lepš & Šmilauer 2003), with the treatments sod cutting, control (grass encroached; no management) and reference (remnants of dune grassland; target type). Vegetation, rabbits, and butterflies were also studied in the summer before the application of sod cutting. Data were statistically analysed after log transformation. In case of a normal distribution, ANOVA was applied (Anonymus 1999). Depending on whether there was (un)equality of error variances the additional tests Tamhene or Tukey HSD were used for multiple comparisons. If the data did not meet the conditions of a normal distribution, the non-parametric Kruskal-Wallis test was chosen.

2 Results

The soil survey of summer 2003 made clear that there was neither a significant rise in pH of the top soil shortly after sod cutting nor a significant decline in decalcification depth. The Rozenwaterveld has a lower pH (5.5) and a larger decalcification depth (6 – 15 cm) than the infiltration area (pH > 7; decalcification depth 0 – 2 cm). This is mainly caused by the difference in age of the dune sand: the Rozenwaterveld (RV) was formed in the 12th – 13th Century, whereas the infiltration area (IA) originates from the 18th – 19th Century (Jelgersma et al. 1970). Both living biomass and dead organic material show a strong, significant decline after sod cutting, resulting in a more favourable light climate for dune grassland species. Although the soil characteristics indicate that the sod cutting sites at the Rozenwaterveld are still sensitive for grass encroachment (Koojiman & Besse 2002), litter input and litter mineralisation have diminished, due to which nutrient availability has probably become low.

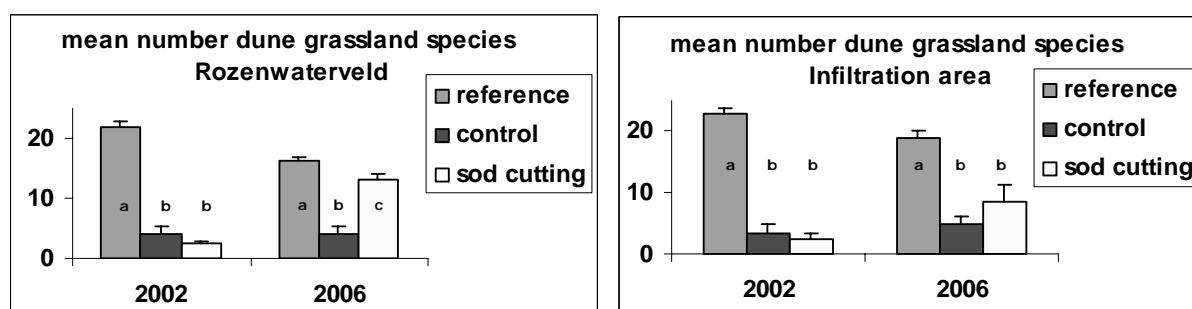


Figure 1: Mean number of dune grassland species per plot at the Rozenwaterveld and in the infiltration area shortly before (2002) and four years after shallow sod cutting (2006). Error bars represent standard error. Significant differences between treatments (sod cutting, control, and reference) are indicated with different letters.

Analysis of the species composition of the vegetation for the period 2002 – 2006 (Fig. 1) illustrates that there was no significant difference in number of dune grassland species between sod cutting and control plots shortly before treatment in 2002 in both areas. The number in the reference (dune grassland) plots was significantly higher. In 2006, the fourth year after sod cutting, dune grassland species such as *Viola curtisii*, *Erodium cicutarium* ssp. *dunense*, *Lotus corniculatus*, *Saxifraga tridactylites* and *Phleum arenarium* have returned. Their average number has increased significantly in both areas (RV: 13 species; IA: 9 species) in comparison with 2002 (RV: 3 species; IA: 2 species). Nevertheless, at the Rozenwaterveld and in the infiltration area the number was still significantly lower than in the reference situation (RV: 16 species; IA: 19 species). In the infiltration area there was in 2006 not yet a significant difference with the control plots (5 species), while at the Rozenwaterveld mean species number was significantly higher in comparison with the control plots (4 species). The coverage of dune grassland species in moss and herb layer after four years was still very low (RV: 17 %; IA: 2 %) compared with the reference plots (RV: 95 %; IA: 88 %).

The mean coverage of the tall grass species *Calamagrostis epigejos* decreased significantly at the Rozenwaterveld from 41 % in 2002 to 4 % in 2006. *Rosa pimpinellifolia*, which often shows a strong

increase in grass-encroached sites, diminished significantly in coverage (from 68 % to 28 %) and height (from 40 cm to 15 cm). The vegetation structure has become very diverse with a mosaic of bare sand (25 %), moss patches, low grass and herbs and dwarf shrubs. In the infiltration area, mean cover of *Elytrichia maritima* decreased significantly from 70 % to 5 %, although it reaches higher density at some steep slopes where the sod-cutting machine was not very effective. The vegetation development is generally slower, which has resulted in a still high cover of bare sand (50 %).

In 2002, mean number of rabbits at the Rozenwaterveld were significantly higher (25 per hectare) in the reference plots in comparison with the control sod cutting plots (≤ 1 per hectare). In the period 2002 – 2006 rabbits showed a significant increase and remarkable recovery after sod cutting (Fig. 2). Mean number increased from less than 1 rabbit per hectare to 61 per hectare. The plots in the infiltration area had a large variety (Fig. 2), which explains that there was no significant difference between all treatments. Nevertheless, the mean number of rabbits had become significantly higher four years after sod cutting (0 vs. 60 per hectare). Besides, in both areas there was also an increase in rabbit numbers in the control plots, which can be explained by the likewise growing numbers in the reference sites.

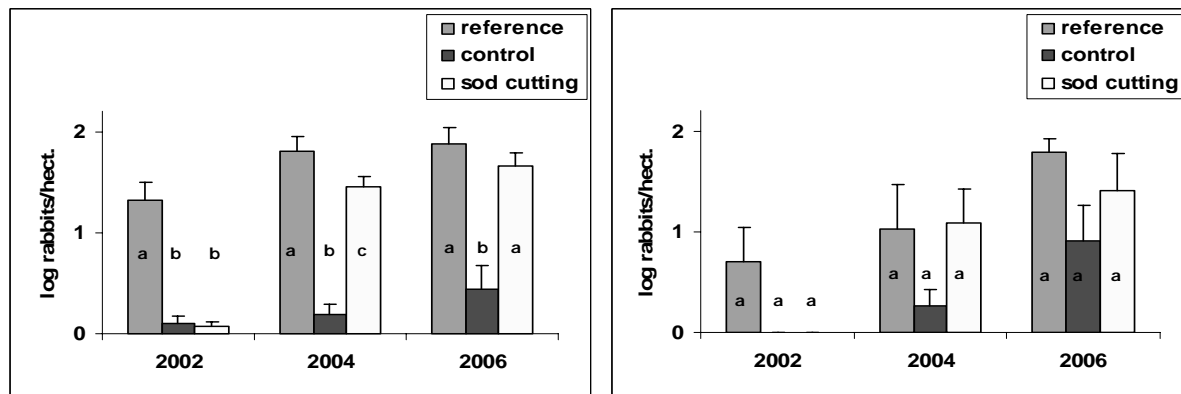


Figure 2: Mean number of rabbits (log rabbits per hectare) at the Rozenwaterveld (left) and in the infiltration area (right) shortly before (2002), two years (2004) and four years after shallow sod cutting (2006). Error bars represent standard error. Significant differences between treatments (sod cutting, control, and reference) are indicated with different letters.

Four grasshopper species were found at the Rozenwaterveld, which are characteristic for dune grassland (Kleukers et al. 1997; Table 1), among which the Red Data Book species *Oedipoda caerulescens*. At the Rozenwaterveld, there was a significant increase from 2003 to 2005 for all species, and in the third year after sod cutting (2005), they all had significantly higher numbers in comparison with the control sites. There was no significant difference with the reference situation. In the infiltration area, *Oedipoda caerulescens*, the only species which was investigated here, also showed a significant increase after sod cutting.

Table 1: Mean number of individuals of grasshopper species characteristic for dune grasslands at the Rozenwaterveld in the third year after sod cutting (2005). Statistical analysis was performed with Kruskal-Wallis test).

Species	Reference	Control	Sod cutting	Significance
<i>Chortippus mollis</i>	58	25	63	$p < 0.01$
<i>Myrmeleotettix maculatus</i>	77	1	113	$p < 0.01$
<i>Oedipoda caerulescens</i>	7	0	20	$p < 0.01$
<i>Chortippus brunneus</i>	101	43	122	$p < 0.05$

Carabid beetles were studied at the Rozenwaterveld in the first year after sod cutting (2003). The dune grasslands of the Taraxaco-Galietum veri seemed to be rather species-poor. Mean number of species and mean number of individuals were significantly higher in the reference plots (10 species, 65 indiv.) in comparison with sod cutting (5 species, 19 indiv.) and control plots (5 species, 24 indiv.). This difference was especially caused by larger numbers of common species such as *Calathus fuscipes*. There was no significant difference found between sod cutting and control plots. Shallow sod cutting had not yet contributed to the return of target species such as *Amara lucida*, *Harpalus neglectus*, and *Notiophilus germinyi*.

The infiltration area was investigated in the second year after sod cutting (2004). The open dune grasslands of the Phleo-Tortuletum ruraliformis were found to be more species-rich (reference: 13 species). They harboured more target species such as *Masoreus wetterhalli*, *Harpalus anxius*, and *Amara curta*. Nevertheless, the differences between sod cutting, control and reference plots were mostly not significant. Only the number of pioneer species (e.g. *Cicindela hybrida* ssp. *hybrida* and *Harpalus servus*) was significantly higher after sod cutting.

Several butterflies were found at the Rozenwaterveld that are characteristic for dune grasslands, among which Red Data Book species such as *Issoria lathonia*, *Fabriciana niobe*, and *Aricia agestis* occurred. In 2002, before sod cutting, mean number of dune grassland species and individuals per transect of 50 metres was significantly lower in the sod cutting (1 species, 2 indiv.) and control plots (1 species, 1 indiv.) in comparison to the reference plots (5 species, 18 indiv.). In the third year, 2005, both mean numbers of species and individuals were significantly higher in the sod cutting plots (4 species, 11 indiv.) in comparison with the control situation (1 species, 2 indiv.; see figure 3). The same goes for the Red Data Book species, which seem to profit from sod cutting. There was no significant difference between sod cutting and reference plots (5 species, 13 indiv.). Mean number of the butterfly species that prefer ruderal vegetation (mainly *Maniola jurtina* and *Thymelicus lineola*) was significantly lower after sod cutting in 2005 (2 indiv.), compared with the untreated control situation (10 indiv.).

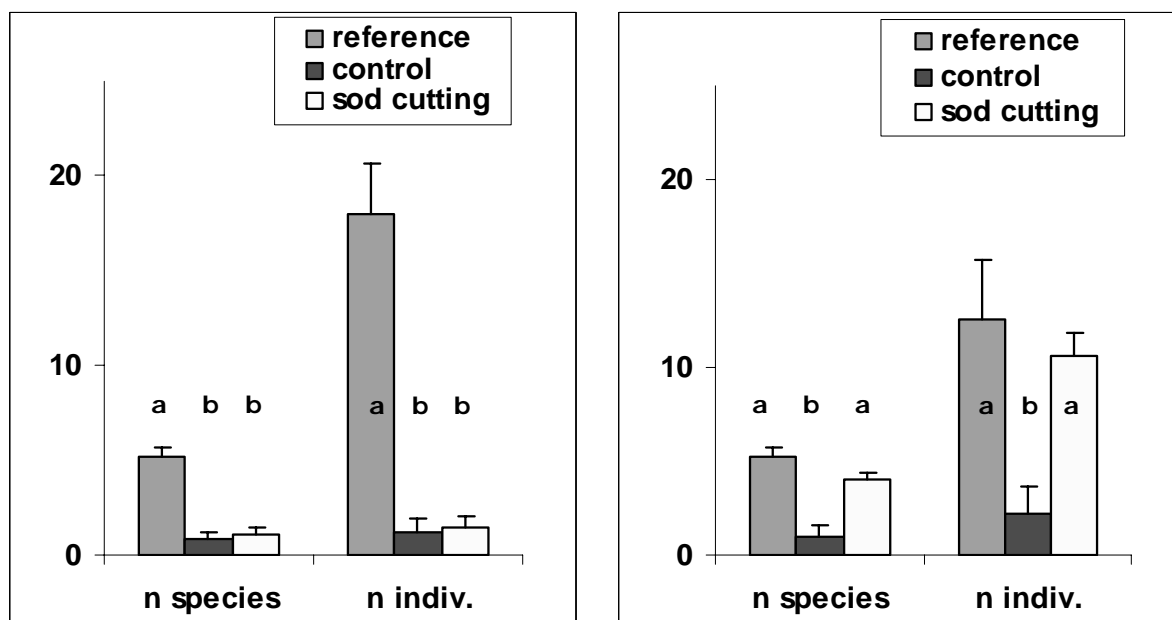


Figure 3: Mean number of species and individuals of dune grassland butterflies at the Rozenwaterveld shortly before (2002) and three years after shallow sod cutting (2005). Error bars represent standard error. Significant differences between treatments (sod cutting, control, and reference) are indicated with different letters.

3 Discussion and conclusion

The results of this project show clearly that both vegetation and fauna of grey dunes start to recover after shallow sod cutting of grass-encroached dune grasslands. Large-scale sod cutting is often considered unfavourable, especially for the fauna (Brouwer et al. 2005). The success of this project can be explained by the small-scale application and the shallowness of the measure, and the presence of still species-rich remnants of the original habitat in the vicinity. Besides, the development within a few years towards a diverse vegetation structure with bare sand and moss patches has probably contributed to the quick improvement of the insect fauna, especially at the Rozenwaterveld.

There is however a difference between the two areas. The more rapid response of the vegetation at the Rozenwaterveld in comparison with the infiltration area might be explained by several reasons. First of all the organic top soil is thicker and has not been removed completely by shallow sod cutting, as a result of which the deeper part of the soil seed bank (5 – 10 cm) is still present. Research in the AWD made clear that seeds from several dune grassland species might be present in this layer, such as *Veronica officinalis*, *Aira praecox* and *Thymus pulegioides* (Bekker & De Vries 2001). In the infiltration area the organic layer is generally less than 5 cm thick, so here the seed bank has been completely removed. Besides, at the Rozenwaterveld rabbits are present in much higher numbers and have expanded their activity more quickly into the sod cutting sites. They contribute to the dispersion of grassland species by pellets (Malo & Suarez 1996, Cosijns et al. 2005). Finally, sod cutting at the Rozenwaterveld did not lead to a change in pH of the topsoil. The intermediate pH of 5.5 (IA: pH > 7) together with the larger decalcification depth of approximately 9 cm (IA: 0 cm) and the higher organic matter content of the remaining top soil (RV: 2.6 %; IA: 0.9 %) probably provides much better conditions for germination, establishment and growth.

The question is whether the fauna species are able to settle down definitively, or whether we observed temporary visitors. Although oviposition was not studied, the establishment at the sod cutting sites of plant species on which caterpillars of several butterfly species of the Red Data Book depend, such as *Aricia agestis* (*Erodium cicutarium* ssp. *dunense*) and *Issoria lathonia* (*Viola curtisii*), is hopeful. Many grasshopper species characteristic of grey dunes forage on different (small) grass species, which also appear after sod cutting. However, carabid beetles do not respond quickly to the conservation measure applied, although they are considered as good colonisers (Turin 2000).

The results of vegetation and butterflies illustrate remarkable differences in number of dune grassland species and individuals throughout the years, considering the reference plots. These differences are probably caused by fluctuations in weather conditions. The year 2002 started wet, while spring 2006 was very dry, resulting in the absence of various annuals, such as *Cerastium semidecandrum* and *Erophila verna*. This explains the lower number of dune grassland species found in the summer of 2006 (Fig. 1). The butterflies show decreasing numbers over a much longer period. Even in years with good weather conditions (2005) the situation is getting worse in comparison with previous good years (e.g. 2002; Fig. 3). The underlying reasons are not yet clear, although the situation for the butterflies characteristic of dune grasslands is generally deteriorating on a regional scale (Wallis De Fries 2004).

Whether the measure is sustainable at the Rozenwaterveld and whether the improvement will continue in the infiltration area remains to be seen. Gradually it becomes clear that rabbits play an indispensable role in the recovery of the grey dunes. Fortunately, the overall trend of rabbits in the AWD has changed from stable and low in the period 1995 – 2003 to a strong increase in recent years (CBS, 2006), especially in sites where nature conservation measures are applied. They keep the vegetation short by grazing and open by burrowing, so the light climate becomes and remains favourable for critical flora and fauna species and competitive grass species are suppressed. Further research on vegetation and fauna will be necessary to monitor future developments.

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Restoration of an ancient dune system enhancing landscape perception

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Abstract

The Doñana National and Natural Parks (SW Spain) present two distinct substrates: the marshes, a silted-up former estuary, and the sands, a Pleistocene detritic formation of gravels, which has been repeatedly covered by dune mantles. The last historical dune-building period occurred during the Little Ice Age (XVI to XIX Centuries) with repeated pulses of dune advance and stability reaching to the El Partido stream watershed to the N of the Parks. The successional response of vegetation developed a cork oak forest with scattered wild olives and elms and dense scrub vegetation. Ashes and willows grew along stream banks. Early in the XX Century scrub was cleared and umbrella pines were planted, although some of the area was reserved for crops. By 1970, to further expand agriculture, most trees in the area had been removed and the old dune morphology was partially erased by wind blow. In 2004, the Spanish Ministry of Environment bought some 4,000 ha of El Partido watershed for ecological restoration in the framework of the Doñana 2005 Project, a large hydrology and landscape plan for the restoration of the Doñana Parks. The landscape restoration of the watershed now under way includes forestation and plantation of native scrub to reconstruct plant communities in order to support threatened vertebrate populations. However, the old dunes are subject to a different treatment with the purpose of recovering their primeval vegetation and to make them more noticeable in the landscape. The paper briefly describes the vegetation, hydrology, and ecological history of the area focusing on the restoration of the ancient dunes and the enhancement of their visual perception.

1 Introduction

1.1 The stabilised dunes of Doñana

The Doñana National and Natural Parks are situated on the southwest coast of Spain to the west of the Guadalquivir River, which flows into the Gulf of Cadiz in the Atlantic Ocean. Littoral currents built up sand deposits in the littoral zone, building a wide spit at the mouth of the Guadalquivir River estuary where marshes evolved during the Holocene. Oncoming winds favoured the transport of coastal sand deposits inland forming sand sheets. The rolling morphology that evolved during alternating episodes of dune activity is recognisable 15-20 km inland (Lario et al. 2002). At present, active dune fields are limited to the coastal strip, and sand substrates with old dune morphology dominate the landscapes to the west and north (García Novo & Merino 1993). Four main land units have been traditionally distinguished in the Doñana National Park: inland marshes, mobile dunes, stabilised dunes, and the Vera, a transition band between the dunes and the marshes (García Novo 1997, García Novo & Merino 1997). At a small scale, the plant composition of the stabilised dunes of Doñana is arranged in parallel bands along dune slopes following local topography (González Bernáldez et al. 1975). Geomorphology controls the vegetation pattern at different scales, through water availability from dune ridges to slacks, exhibiting a sequence of xerophytic scrub, mixed scrub, and heath. Drier crests exhibit *Thymus mastichina*, *Lavandula stoechas*, *Scrophularia frutescens*, and *Halimium commutatum*. Mid slope shrublands are dominated by *Halimium halimifolium* and *Stauracanthus genistoides* and in floodable depressions *Calluna vulgaris*, *Erica scoparia*, *E. umbellata*, and *Myrtus communis* are often associated with *Pteridium aquilinum*. Superimposed, cultural interactions operate as model agents at a small scale and a short term (Granados Corona et al. 1988), interacting with succession and climatic shifts at larger scales of time and space (Muñoz Reinoso & García Novo 2005).

Prescribed fire regimes have promoted pyrophytic Mediterranean shrubs, which now dominate the scrub vegetation. Only a few thickets of mature scrub survive, being composed of fruit-bearing species such as *Quercus coccifera*, *Arbutus unedo*, *Phillyrea angustifolia*, *Pyrus bourgaeana*, *Pistacia lentiscus*, *Rhamnus oleoides* ssp. *lycioides*, *Crataegus monogyna* and *Chamaerops humilis*. Close to the coastal zone *Retama monosperma* forms dense stands often becoming the only woody species. A few remnants of the native juniper woodlands (*Juniperus oxycedrus* ssp. *macrocarpa* and *J. phoenicea* ssp. *turbinata*) survive in dunes close to the coastal fringe. *Pinus pinea* plantations started in the Doñana area in 1737, spreading from a single site to become the main woodland of the Parks at present. Retama, oaks, junipers, and pines are now dispersed by wild vertebrates.

1.2 Study area

The area for restoration covers about 2300 ha towards the north-east border of the Doñana National Park, in the floodplain of the El Partido stream, a tributary of the Doñana Marshes (see Fig. 1).

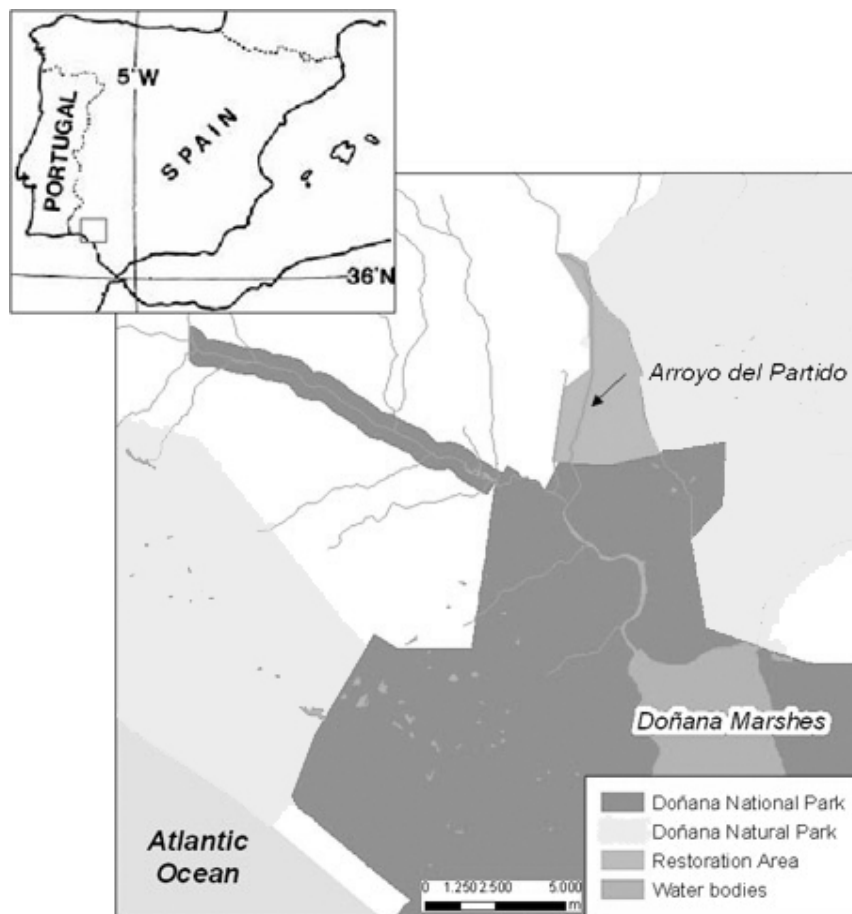


Figure 1: Location of the El Partido stream restoration area.

The El Partido floodplain is covered by sandy sediments deposited by floods. The morphology of transgressive dunes developed during the Little Ice Age (XVI-XIX Centuries) is noticeable (García Novo et al. 2006, Rodríguez Ramirez et al. 2002, Sousa & Sarcía Murillo 2003).

The Mediterranean-type climate receives some oceanic influence. Average annual rainfall is 600 mm, with two peaks: November-December and March with little rain from June to September and a rainless August with high summer temperatures (July and August average temperature: 23 °C). Winter temperatures are mild with a minimum in January (average temperature of 9 °C) and only a few frost days in winter months.

1.3 The El Partido Stream Restoration Project

The Spanish Ministry of the Environment launched in 1998 the Doñana 2005 Project with the goal to restore the hydrology of the Doñana National Park (Saura Martínez et al. 2001) as a basis for conservation. It comprises six key interventions, one of them being the restoration of the El Partido Stream watershed. The stream exhibits a marked torrential regime, due to a rainfall pattern with intense spells. Early in the 1980's the last 7 km of the stream were channelled, and the surrounding flood plain was levelled, tilled and converted into arable land or left as pastureland. Over the last 20 years, the hydraulic impact of collecting the dispersed braided channels of the area into a single straight course led to major erosion of the channel bed and the creation of a large depositional delta (now over 300 ha) on the Doñana National Park marshes (Mintegui Aguirre et al. 2003).

The restoration of the El Partido watershed is being carried out at different spatial levels using an interdisciplinary approach, with the following aims:

1) to prevent the further silting-up of the marsh by avoiding bed erosion and the transport of sand; 2) to recover the former aquatic habitats; 3) to create functional ecosystems for target species (Iberian lynx, imperial eagle and their prey, namely rabbits); 4) to restore some of the original ecosystems of the flooding area, applying the principles of population ecology, community ecology, and landscape ecology (Carotenuto et al. 2006, García Novo et al. 2006).

One of the restoration targets are the former transgressive dunes, which have undergone tree plantation and crop cultivation resulting in dune-field fragmentation and the blurring of dune morphology. This paper describes the restoration of the vegetation in the ancient dune field incorporating landscaping actions aiming at the perception of dunes from the public.

The floristic composition and structure of the vegetation in the stabilised dunes of the nearby National Park have long been studied by many scientists such as González Bernáldez et al. (1971, 1975), Allier et al. (1974), García Novo (1990), Granados Corona et al. (1988) and Muñoz Reinoso (2001), to name a few. The availability of aerial pictures from 1946 onwards, supported the interpretation of ecosystem and landscape changes and the mapping of previous vegetation types.

2 Restoration of an ancient dune system enhancing landscape perception

The first restoration step has been the survey of the vegetation and morphology of the El Partido stream floodplain and the identification of preserved reference ecosystems.

The present day landscapes in the El Partido floodplain are abandoned crop fields, cultivated until 2002-03. Post-cultural grasslands, with ruderal forbs and tall grasses cover the abandoned fields. Some relicts of the former vegetation such as isolated cork oaks, pines, kermes oaks, pear trees and scattered *Chamaerops humilis*, *Pistacia lentiscus* and *Crataegus monogyna*, survived in the fields. Succession to woodland is dominated by scrub, often arranged in dense isolated clusters. A few transgressive dunes retained their geomorphology but are of a low height (0.5-1 m above the surrounding topography). They preserve vegetation similar to the stabilised dunes of the National Park, mainly composed of *Halimium halimifolium*, *Lavandula stoechas*, and *Rosmarinus officinalis*. This vegetation is accompanied by scattered *Quercus coccifera* trees, planted orchards of *Pyrus bourgaeana*, and some *Chamaerops humilis* or *Rubus ulmifolius* stands in the lower part of the dune slopes.

The large distance (almost 10 km) to the sources of seeds of dune shrub species from the National Park, and the scarcity of remnants of mature vegetation, prevents an adequate seed supply to the area, making succession to more mature stages of dune vegetation virtually impossible. In addition, the long-standing human intervention in the area has obliterated the original morphology leaving only a few of the ancient dunes clearly visible.

This explains why it was decided to undertake the ecological restoration of dune vegetation incorporating elements to enhance the visual perception of dune morphology for future visitors of the area. Strictly speaking, the plantation of dune species not found in the area at present cannot be regarded as

ecological restoration. However, two arguments deserved our consideration: the plantation of shrubs meets the restoration objectives of the area in relation to endangered vertebrate populations since their main prey, the rabbit, will be strongly favoured. And secondly, plantation restores the natural dune landscape lost in the area due to past human intervention.

The addition of sand to return dunes to their former height has been dismissed. This artificial sand contribution could drive the system further away from its self-sustained goal and damage the populations of surviving dune species that belong to stabilised dune stages and hence are not adapted to burial. The opening of ditches around the dunes to lower the soil water table and to restrict the root-access to phreatic waters was also considered, but rejected, because it would create an artificially sustained system and affect the actual dune morphology.

2.1 Preservation of visually distinctive dunes

Only those dunes retaining a noticeable morphology and suitable dune vegetation (*Halimium halimifolium*, *Rosmarinus officinalis*, *Lavandula stoechas*, and a few others) have been considered for intervention.

Plantations:

Dune vegetation shows a well-defined pattern according to substrate stability. Where sand movement ceases, succession favours scrub development. In some species, such as *Retama monosperma*, it has been shown that rabbits have a role as active dispersers of the species, which in turn favours the development of dense burrows under *Retama* canopy (Dellafiore et al. 2006).

Table 1 includes the list of species used in plantations. Only a few trees were planted and at specific locations. Other species were planted in clumps with the exception of *Retama monosperma*, which was planted delineating the dune contours and favour its identification in the El Partido stream watershed. In addition, the low-height plantations at the base of dune slopes favours access of rabbits to its fruits and the eventual dispersion of the species in the area.

Table 1: List of dune species to be used in restoration.

Dune species	Distance to phreatic table
<i>Juniperus oxycedrus</i> ssp. <i>macrocarpa</i>	Low
<i>Juniperus phoenicea</i> ssp. <i>turbinata</i>	High
<i>Halimium halimifolium</i>	Intermediate
<i>Cistus libanotis</i>	High
<i>Lavandula stoechas</i>	Intermediate
<i>Rosmarinus officinalis</i>	Intermediate
<i>Thymus mastichina</i> ssp. <i>donyanae</i>	High-intermediate
<i>Retama monosperma</i>	Intermediate

From a landscape perspective, the plantation of these species will be accomplished in a way that enhances the visual perception of the dune and its constituent parts. To favour this perception, the plantation will highlight the different parts of the dune (dry, intermediate and humid) by locating the drier species in the upper parts of the dune (*Juniperus phoenicea* ssp. *turbinata*) and *Juniperus oxycedrus* ssp. *macrocarpa* and other species along the slopes of the dune and delimitating its perimeter. Figure 2 shows an example of the restoration of one of the remaining dunes.

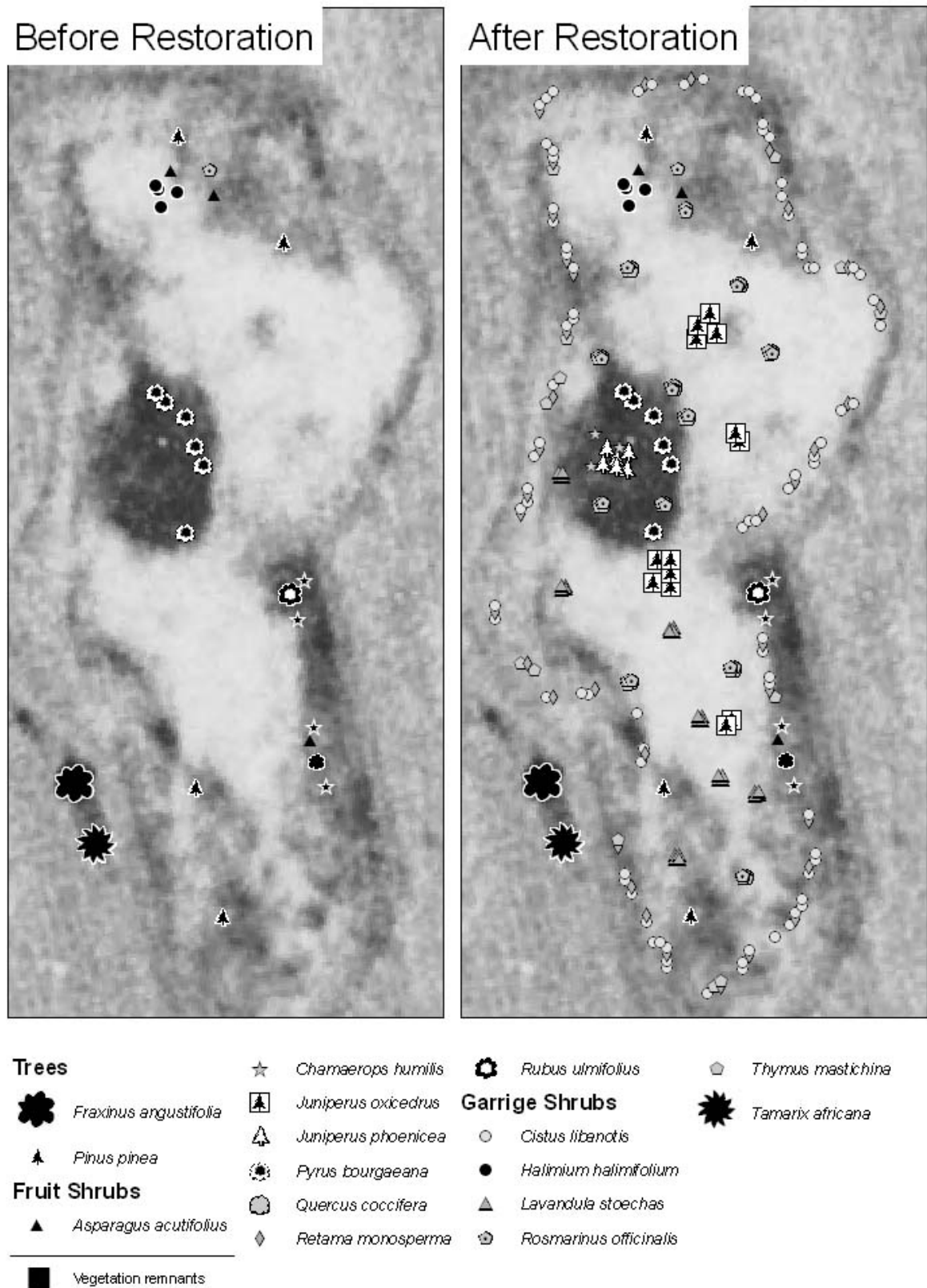


Figure 2: Vegetation present in the dune before and after the restoration.

The plantation of individuals of *Cistus libanotis*, *Thymus mastichina* and *Retama monosperma* along the base of the dune will visually delimit its perimeter. If *Retama monosperma* grows above its desired height, pruning or even elimination of some individuals could be considered once the other dune species have reached their desirable development. *Juniperus oxycedrus* ssp. *macrocarpa* will be

planted on the tallest parts of the dune to add height to dune vegetation and serve as focal points. This pointed-shaped species will be easily differentiated from the rounded shaped umbrella pines present in other parts of the restoration area. Individuals of *Juniperus phoenicia* ssp. *turbinata* and *Chamaerops humilis* will point out the location of the inner depression, and small patches of *Lavandula stoechas*, *Halimium halimifolium* and *Rosmarinus officinalis* will be planted along the intermediate part of the dune.

Besides the difference in the species shape, their flowering colour will also constitute a plant trait that will help in visitors' interpretation of the dune parts during the different flowering seasons.

2.2 Enhancement of perception

On sandy substrates with an almost flat morphology, the aim of restoration was to make the dune remnants more noticeable in the landscape. This will be accomplished by leaving dune borders as bare sand boundaries in sharp contrast amidst scrub; preserving the remaining vegetation (typical vegetation of stabilised dunes), and the plantation of the open vegetation type that prevails in the Doñana sand mantles (see Table 1).

Where the ancient dune landscape has been erased, that is, where there is no dune morphology left, the restoration includes the afforestation and plantation of native Mediterranean shrubland to reconstruct plant communities and to enhance vertebrate populations. Existing remnants of woody vegetation covering about 7.66 ha (2.2 %) will be restored, adding about a 3-fold surface (26.8 ha, 7.8 %) of new plantations. In addition, 545 new vegetation patches, initially covering 20.53 ha (about 6 % of the area) will be planted, following composition and structure of the natural scrub. Each patch combines 5-10 perennial species: a core of a few trees, an inner area with some fruit-bearing shrubs, and the outer fringe with flower-bearing shrubs. Riverbanks will also be vegetated (for further details, please refer to García Novo et al. 2006, García Novo & Marín Cabrera 2006).

Restoration of dune landscapes may be focused on sand transport, geomorphology, coastal vulnerability, biodiversity purposes, visitors use, and many other objectives. This paper suggests how the restoration of vegetation in an ancient dune field can incorporate the perception of the remaining ancient dune landscape, without impairing recovery of the natural ecosystem properties.

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