

**Stranded *Zostera marina* L.  
vs wrack fauna  
community interactions  
on a Baltic sandy  
beach (Hel, Poland):  
a short-term pilot study.  
Part I. Driftline effects of  
fragmented detritivory,  
leaching and decay rates\***

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**KEYWORDS**

Litterbags  
Decomposition  
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**Abstract**

The effects of the beach community structure of macro- and meiofauna on the process of beach wrack decay were investigated by means of a simple field colonisation experiment in a temperate, fine quartz sediment, sandy beach at the end of the Hel Peninsula in Poland. 1260 replicate litterbags of three mesh sizes (12 mm, 0.5 mm, 48  $\mu\text{m}$ ) containing fresh wrack were used to assess the role of faunal and non-faunal components in the breakdown of stranded *Zostera marina*. Wrack breakdown was determined during a three-year field study.

This paper presents the first part of the results of this field experiment, which refer to the effects of fragmentation detritivory, leaching and decay rates. Material was lost from the bags at a rapid rate, with only 22–32% of the original dry mass remaining after 27 days in the field. This degradation was not directly related to

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the faunal succession of the eelgrass tissue, which proceeded in two distinct phases throughout the study period. Exclusion of macrofauna from the wrack by the use of finer-mesh litterbags (< 1 mm) had no appreciable effect on the rate of dry matter loss. Microbial decay, and abiotic leaching and fragmentation are probably the major causes of seagrass weight loss from the litterbags.

## 1. Introduction

The intertidal zones of sandy beaches appear barren when compared with other intertidal habitats. Despite this superficial impression, many beaches support a diverse flora and fauna. A number of sandy littoral localities may be considered highly productive (Brown & McLachlan 1990). Beaches are important in the processing of organics, no matter whether these substances are derived from DOM, POM or larger animal and plant remains. Because of the lack of stable solid surfaces, and because of the abrasive action of moving particles, a sandy beach will not support either the abundant macrophytes found on rocky shores or the rooted vegetation that colonises more muddy areas (McLachlan 1983, Brown & McLachlan 1990). Exposed beaches are also characteristically lacking in resident primary production and attached macroalgae below the driftline. Nourishment is therefore generally supplied from offshore and constitutes the primary food supply for the beach's supralittoral fauna (Griffiths et al. 1983, Inglis 1989, Brown & McLachlan 1990, Colombini et al. 2000). Sandy beaches are examples of simple ecosystems, driven principally by the physical forces of waves, tides and sediment movement. Marine beaches receive a variety of organic materials from the sea: macrophyte wracks, dead animals, and dissolved and particulate organics flushed into the sand by waves. Biota associated with the interstitial system (bacteria, protozoa, meiofauna) and macrofauna process these organic materials and return nutrients to the sea. On most beaches, therefore, the interstitial system functions as a biological filter that mineralises organic material and thus cleanses the surf waters (Lenanton et al. 1982, McLachlan 1983, Berzins 1985, Brown & McLachlan 1990). In this process, decaying algae are reduced to CO<sub>2</sub> and nutrients. Each square metre of beach consumes some 30 kg of organic wastes per year (Węsławski et al. 2000).

On sandy coasts adjacent to rocky shores or beaches fringing seagrass meadows, the macrophyte input may be very high. In such cases, values of between 20 000 and 300 000 g C m<sup>-1</sup> y<sup>-1</sup> have been recorded (Griffiths et al. 1983, McLachlan & McGwynne 1986, Brown & McLachlan 1990), this material totally dominating the sandy-beach food chains. The macrophytes are fed upon directly by scavengers associated with the driftline, such as talitrid amphipods, isopods and insects. Much decomposition is also

accomplished by bacteria, but a diverse assemblage of beach fauna greatly influences the decomposer flora as a result of their feeding activities (Griffiths et al. 1983, Koop & Lucas 1983, Wachendorf et al. 1997).

The present study examines the degradation rates of eelgrass *Zostera marina* L. from litterbags in the supralittoral zone of Hel Beach, Poland, during a short-term study of the driftline effects of beach community structure on the process of wrack decay. A brief account of the stranded wrack degradation and oxygen consumption of this site was given in Jędrzejczak (1999), Urban-Malinga & Opaliński (1999), and Urban-Malinga & Opaliński (2001). The hypothesis is that the rate of seaweed breakdown on the shore is directly related to the successional influence of different major faunal and non-faunal beach parameters. This paper addresses the question of how the rates of ecosystem processes (mainly the decomposition of organic matter and associated processes) change along the strandline.

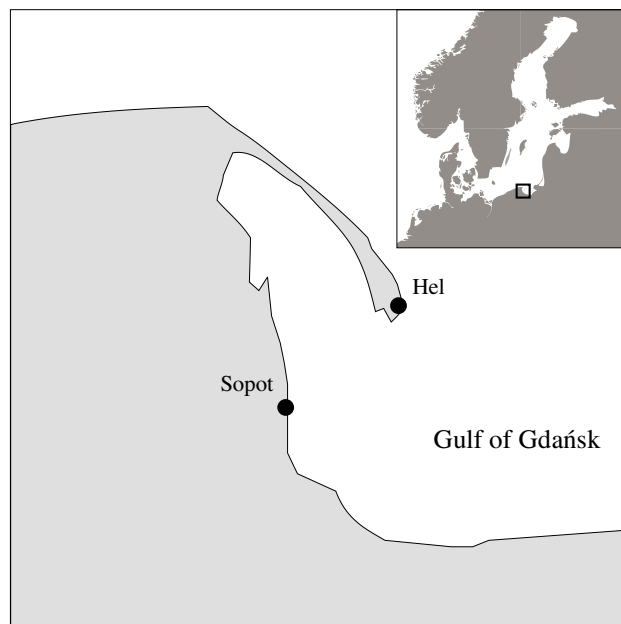
This work was motivated by research within the LITUS Project (Interaction of Biodiversity, Productivity and Tourism in European Sandy Beaches). The basic objectives of the Project are to assess the vulnerability of a sandy beach's biodiversity and the functioning of a beach ecosystem, as well as a better understanding of the interaction between tourism, natural changes, and physical marine factors of such an ecosystem. The LITUS Project provides a scientific, socio-economic and technological basis for understanding changes in European sandy beaches (Węśławski et al. 2000).

## 2. Materials and methods

### Study area

The study site of Hel Beach (54°36'N, 18°49'E) is located on the northern coast of Poland (Gulf of Gdańsk, Baltic Sea), at the end of the Hel Peninsula, about 3.5 km north of Hel (Fig. 1), in a former military area, now part of the Marine Landscape Park, where human impact is relatively low. It was selected as an example of a relatively undisturbed Polish beach. Hel is a dissipative non-tidal beach, characterised by a low beach gradient, fine to very fine sediments, and a surf zone with numerous longshore bars and troughs, from which a broad sandy beach (80 to 120 m in width) emerges. Backed for most of its length by a well-developed, stable dune system (30 m wide and 4 m in height) forming the upper limit of storm-accumulation, the Hel Beach is exposed to moderate to heavy wave action from the open Baltic Sea. The sandy shore, completely bare of vegetation, is separated from the dune by a scarp corresponding to the storm driftline.

Wrack deposits on the beach are composed almost exclusively of the seagrass *Z. marina* L. and the algae *Pilayella littoralis* (Lyngbye) Kjelm.,



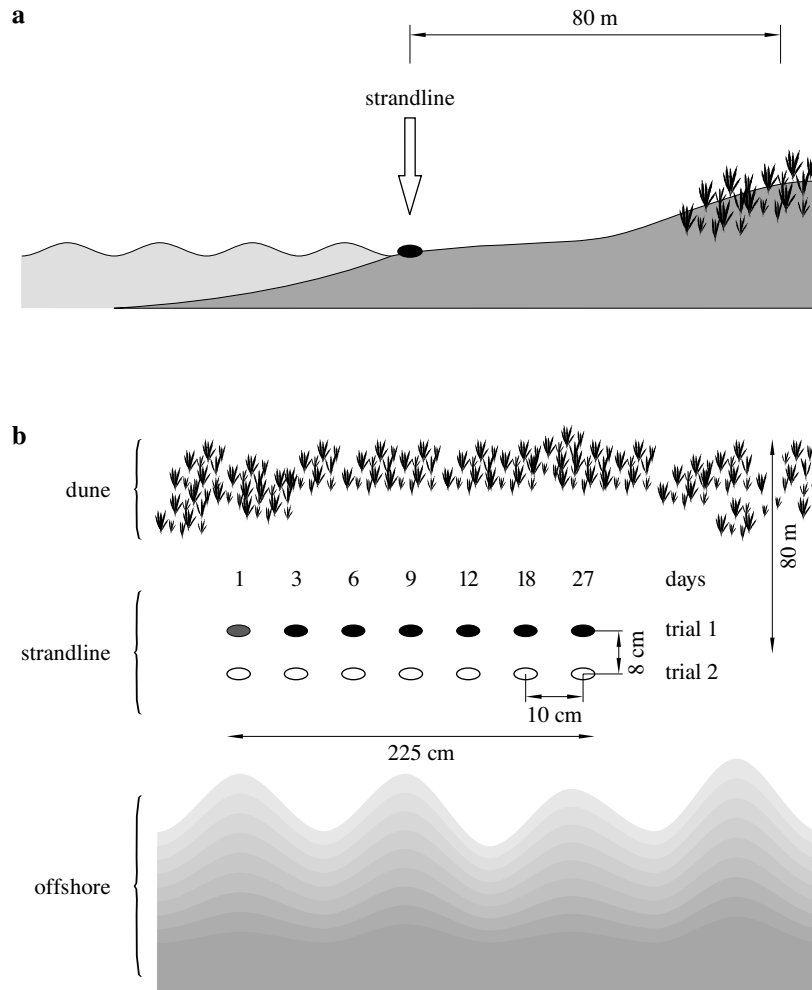
**Fig. 1.** Location of study beach site

*Enteromorpha* spp. and *Cladophora* spp. The input of these plants fluctuates widely but is generally associated with seasonal phases and vegetation capacity, e.g. seasonal amounts of the large brown algae *Fucus vesiculosus* are found on this beach. However, the shape, position and cover of the debris on this beach have not yet been reported from this site. The wrack deposits vary in size from small, isolated mounds to great heaps, lying in belts parallel to the shoreline. Because there are no tides, during the entire period of stranding, the wrack undergoes dehydration, ages and, in the case of small mounds, becomes covered by wind-blown sand. Only during a storm is the beach completely devoid of debris. After this, accumulation restarts and continues until the next storm blows up.

### **Field studies – litterbag design and placement**

The method used followed Inglis (1989), modified to local conditions.  $225 \times 105$  mm litterbags were made from fine nylon mesh with either 0.5-mm or  $48\text{-}\mu\text{m}$  apertures, designed to exclude the beach macro- and meiofauna respectively. Litter bags of a coarser mesh size were produced from larger plastic-coated steel mesh,  $235 \times 120$  mm with 12 mm wide openings, and filled with wrack. The latter bags were designed to allow all of the supralittoral zone fauna access to the enclosed wrack material. Each fine-mesh litterbag was separately enclosed inside a coarse-mesh litterbag.

Two parallel experimental trials were done in May, July and September during a three-year study (1999–2001). Each trial ran for 27 days and was set up on the strandline (Fig. 2). In the first trial, loss of wrack dry weight was compared between coarse-mesh and 0.5-mm mesh bags, while in the second trial, coarse-mesh and 48- $\mu\text{m}$  mesh bags were compared.



**Fig. 2.** Profile of the Hel Beach site, showing the supralittoral zone sampled and the exposition of litterbags on the sandy beach; beach-dune profile (a) and view from above (b)

Fresh *Zostera* was collected, blotted dry, and 60-g wet weight portions of leaves placed within the bags. Mesh sizes were assigned randomly, and the bags identified with plastic tags. Bags of each mesh size were randomly

positioned in areas around the driftline from which existing wrack had been removed. Each bag was then lightly covered with sand. There were five replicate samples in the field during each period. Litterbags were sampled at random 1, 3, 6, 9, 12, 18 and 27 days post-placement, not in a spatially consecutive way. 1260 litterbags were used in the all experimental studies (3 years  $\times$  3 seasons  $\times$  4 mesh fractions in 2 trials  $\times$  7 samples  $\times$  5 replicates).

In the laboratory the content of each litterbag was gently rinsed onto a 0.25-mm sieve to remove the adhering sediment and surface fauna. The remaining vascular material was then dried for 3 days at 60°C and weighed. Final results were expressed as the percentage of initial wrack dry weight remaining on completion of the experiment and were analysed using ANOVA. Initial weight was obtained by drying and weighing five replicates of 60-g portions of fresh *Z. marina* tissue.

### Decomposition model and statistics

Decomposition data derived from litterbag studies (change in dry weight through time) were expressed as the percentage of initial tissue dry weight remaining and were analysed using ANOVA. The final results were evaluated with a single exponential decay model (Jenny et al. 1949, Swift et al. 1979) according to the formula:

$$W_t = W_0 e^{-kt},$$

where

$W_t$  – dry weight [g] of litter sample remaining after time  $t$ ,

$W_0$  – initial dry weight [g] of litter sample,

$e$  – base of natural logarithm,

$k$  – decay coefficient [ $\text{day}^{-1}$ ],

$t$  – time [days].

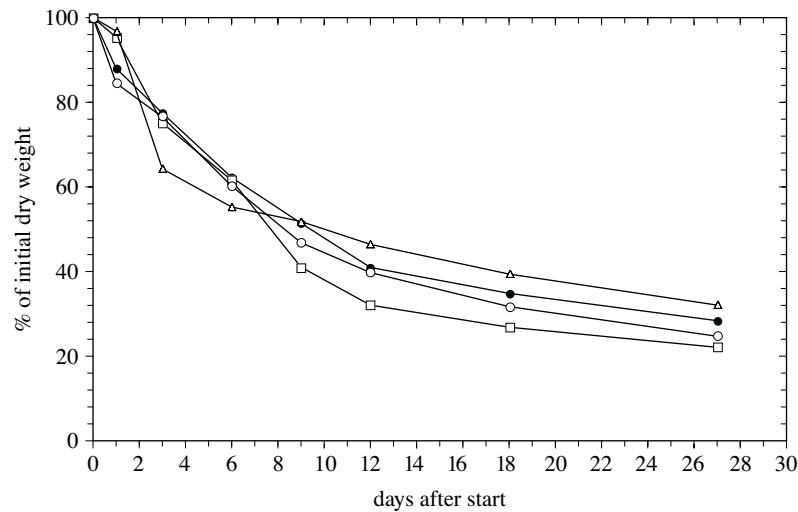
The decay coefficient  $k$  permits comparison of decomposition rates between organic material types and among studies. The single-exponential model does not discriminate between soluble versus refractory material, nor does it distinguish microbial contributions (Wieder & Lang 1982). F-tests were used to compare differences between individual site regressions for each mesh type (full model) and a reduced model, where data from all sites, for each mesh type were synthesised in one regression. F-tests were compared for each of three regressions: coarse-mesh bags, 0.5-mm mesh bags and 48- $\mu\text{m}$  mesh bags.

### 3. Results

Significant weight loss occurred in bags of all mesh sizes (Table 1). The pattern of weight loss was similar in both trials (Fig. 3) and the eelgrass disintegrated in an approximately linear fashion for much of the

**Table 1.** Analysis of the proportion (arcsine transformation) of seagrass dry weight remaining in the litterbags of different mesh size (mesh) on seven sampling occasions (days): data are for three sites (site) on Hel Beach

Source of variation	Trial 1				Trial 2		
	df	MS ( $\times 100$ )	F ratio	P	MS ( $\times 100$ )	F ratio	P
days	7	48.47	183.60	< 0.01	39.83	127.91	< 0.01
mesh size	1	1.60	9.69	< 0.01	0.90	5.95	< 0.05
site	2	1.40	0.18	> 0.05	0.78	0.12	> 0.05
days $\times$ mesh	14	0.52	3.16	< 0.05	0.96	6.38	< 0.01
days $\times$ site	7	1.35	0.55	> 0.05	0.08	0.21	> 0.05
mesh $\times$ site	2	1.54	0.02	> 0.05	0.43	0.06	> 0.05
days $\times$ mesh $\times$ site							
nonadditivity	1	3.12	0.71	> 0.05	8.17	2.74	> 0.05
balance	14	0.02			0.08		
Cochran's statistic		C = 0.22		> 0.05	C = 0.20		> 0.05



**Fig. 3.** Changes in the mean dry weight of *Zostera marina* leaves in litterbags placed in the supralittoral zone of Hel Beach: coarse mesh bags in the first trial (●-●); coarse mesh bags in the second trial (○-○); 0.5-mm mesh bags in the first trial (□-□); 48- $\mu$ m mesh bags in the second trial ( $\Delta$ - $\Delta$ )

experimental period. Within 27 days, the material within the litterbags had lost between 68 and 78% of its initial dry weight. The material in the coarse-mesh and 0.5-mm mesh bags in the first trial fragmented at about

the same rate, as did the tissue in the coarse- and 48- $\mu\text{m}$  mesh bags in the second trial (Table 1). Exclusion of fauna, therefore, appeared to have had little effect on the rate of seagrass breakdown.

#### 4. Discussion

The breakdown of plant material is heavily influenced by both site- and time-specific environmental conditions (Tenore et al. 1984, Harrison 1989). Differences between the decay of the same litter under identical climatic conditions, but at different sites, are normally minor (Table 2). These may manifest themselves not only in abiotic fragmentation of the plant material, but also in the structure and function of the local consumer assemblage and in the rate at which the plant material is decomposed by microbial components at specific sites. Aside from consumption, the feeding activities of detrital consumers can accelerate the decomposition of plant material through the spread of microorganisms. In addition, sediment-living organisms can make the material more available for decomposition through their burrowing activities (Harrison & Mann 1975a, b, Harrison 1977, Robertson & Mann 1980, Bedford & Moore 1984, Inglis 1989).

With the use of litterbag techniques it may be argued that exclusion of one element of the biota could result in a change in the functional role of a second, nonexcluded assemblage (Swift et al. 1979). This is unlikely to have affected the conclusions of this study because the scavenging macro- and meiofauna inhabited the wrack at different instances, and therefore will probably not have significantly affected each other (Brown & McLachlan 1990). However, Inglis (1989) reported that even if the observed increase in meiofaunal numbers in the 85- $\mu\text{m}$  mesh bags resulted from a release in predation pressure by macrofaunal species, this does not appear to have had any effect on the rate of kelp degradation. The same is likely to have occurred in the present study.

In describing the decomposition of saltmarsh grasses, Valiela et al. (1985) identified three distinct phases. Initially, organic matter is lost at a rapid rate, which corresponds largely to the leaching of hydrolysed compounds from the material (up to 5–40% of initial mass loss). This is followed by two phases where weight loss becomes successively slower. Decomposer activity is said to be the major source of weight loss in the second stage (to 40–70% of initial mass loss), whilst the slower decay of refractory materials characterises the third. In the present study loss of vascular dry weight from the litterbags was essentially linear, despite distinct temporal changes in the wrack fauna. However, Tenore et al. (1984) have suggested that algal detritus does have a decay resistant fraction. Because the present study was only a short-term one, it seems probable that only the first two stages were



**Table 2.** Details of published field study of seagrass decay. Rates are expressed as loss of organic material or (†) total dry weight

Species	Plant part	Age (state)	Treatment	Decay conditions				Decay rate [% × day <sup>-1</sup> ]	Reference
				Time [days]	Litter bag mesh [mm]	Location	Buried		
<i>Zostera marina</i>	leaf	old	fresh	90	1	subtidal		0.4	Robertson & Mann (1980)
						9	subtidal	0.6	
		mixed (living)	fresh	150	2.5	subtidal		1.8	Josselyn & Mathieson (1980)
							intertidal	0.7	
		green	fresh	210	1	subtidal		0.4 <sup>†</sup>	Pelikaan (1984)
		dead	fresh	365	1	subtidal		0.3	Gallagher et al. (1984)
	green	fresh	27	0.048	intertidal		0.2	present study	
					0.5	intertidal	3.5		
	rhizome and root	living	fresh	180	2	subtidal	+	0.2–0.3	Kenworthy & Thayer (1984)
<i>Thalassia testudinum</i>	leaf	yellow- green	fresh	105	4	subtidal		0.8–1.7	Newell et al. (1984)
	rhizome	living	fresh	100	2	subtidal	+	0.04	Kenworthy & Thayer (1984)
	root	living	fresh	100	2	subtidal	+	0.8	

discernible here. This corresponds well with the findings of Inglis (1989), who analysed the faunal succession of the kelp on a New Zealand sandy beach. It is also in agreement with my own study (Jędrzejczak 1999) done on carrion breakdown at the same Hel Beach site. However, Inglis (1989) and Colombini et al. (2000) reported that any such refractory materials may be washed back into the surf zone by spring tidal flushes, making them no longer available to the supralittoral community.

Short-term studies, both in the laboratory and on the beach itself, have led to suggestions that faunal succession does not relate directly to the degradation of the eelgrass tissue, which proceeded linearly for the entire study period (Koop & Lucas 1983, Inglis 1989). In the present study material was lost from the bags at a rapid rate, with only 22–32% of the original dry mass remaining after 27 days in the field. Exclusion of the macrofauna from the wrack, using litterbags of finer mesh bags (< 1 mm), had no appreciable effect on the rate of dry matter loss. These findings are in accordance with Inglis (1989).

However, Hackney (1987) demonstrated that the degree of breakdown is greater on the surface than beneath the soil with results ranging from 80% of the remaining dry mass after one year to 45% after 1.5 years. Furthermore, Hackney & de la Cruz (1980) and Blum (1993) indicated on salt marshes that decomposition is not affected by redox conditions or conditions associated with a low redox potential. However, saturation and aeration play a major role within the sandy sediment on the beach (Brown & McLachlan 1990). Large volumes of seawater are filtered by intertidal sand, which occur by swash flushing of unsaturated sand. The water table may rise after rain and may even show weak tidal fluctuations near the sea. The water flow through the body, driven by waves, is able to transport oxygen, and hence help to maintain biological activity in the porous media (Massel 2001). Urban-Malinga & Opaliński (1999) further noted that the biotic oxidation is the dominant component of the total oxidation (up to 57%) at the waterline, whereas abiotic oxygen consumption is very important under the water and in the splash zone (70 and 77% of the total oxidation, respectively). Oxygen consumption decreases gradually from under the water to the backshore and seems to be a function of distance from the waterline. Thus Wachendorf et al. (1997) attribute the decay differences between sites to differences in the moisture regime. After one year of a study of plant decomposition on a lake shore bordered by deciduous woodland, they noted that the percentage of total mass loss in 20- $\mu\text{m}$  litterbags, i.e. losses due to microbial and microfaunal respiration, as well as leaching, were 88% at the dry station, but only 48% at the wet station. These authors claimed that leaching and microbiological activity exert separate

influences on decay, the latter value being estimated from measurements of microbial biomass. The calculated respiration of microorganisms in the field was estimated to be 70% and 18% of the total loss at the dry station and wet station respectively. They concluded that leaching contributed an 18% loss at the dry station and a 30% loss at the wet station. The mass loss at the wet stations was mainly due to the soil fauna, followed by loss due to leaching. However, the breakdown at the dry station was predominantly due to microbial respiration. Microbial utilisation of carbon and nitrogen from wrack deposits has been estimated by Koop & Lucas (1983) at 28 and 94% of the original values, respectively. The findings of Jędrzejczak (1999) seem to confirm that microorganisms and leaching play a more significant role than the meio- and macrofauna community in the overall degradation process, although the significance of meiofauna rises as the exposure time elapses.

The present study shows that numerous factors affect beach wrack degradation due to the fragmentation detritivory, leaching and decay rates. Without doubt, the type of material, temperature, internal nitrogen content and decomposer activity, in that order, significantly alter the rate of weight loss. Microbial decay, abiotic leaching and fragmentation are, however, thought to be of primary importance in the breakdown of seaweeds stranded in the supralittoral zone of sandy beach. The differences in processing of organics may be explained by variations in moisture regime and nutrient status, and not by variations in the decay processes themselves. The effects of beach community structure of macro- and meiofauna on the process of beach wrack decay are described and discussed in Part II.

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