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Shifting baselines in the Ems Dollard estuary: A comparison across three decades reveals changing benthic communities

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ABSTRACT

At a time when there is a growing discussion about the natural state of estuaries, a comparison of macro-zoobenthos communities from two surveys conducted 30 years apart in the Ems Dollard estuary, in the eastern Wadden Sea, The Netherlands, provides a unique opportunity to compare changes over time. As expected, our comparison revealed a gradient in species composition from land (the Dollard) to sea (the Outer Ems) at both points in time, with brackish species in the Dollard and more marine species in the Outer Ems (Wadden Sea). Total richness increased over time; however, this mainly reflected the immigration of new species and sampling differences. In the Dollard, total biomass declined over time, most likely reflecting de-eutrophication in this area. Strikingly, at the meeting point between the sea and the brackish Dollard, i.e. the Inner Ems, the community composition changed from one dominated by bivalves (1970s) to one dominated by worms (since 2009). This change involved a reduction in total biomass, mainly of *Mya arenaria*, and immigration of polychaete worms (*Marenzelleria viridis* and *Alitta succinea*). In the Outer Ems, an increase in total biomass was observed, associated with the recent successful recruitment of *Cerastoderma edule*. This comparison highlights that historical data provides useful insights at large spatial scales. However, a full understanding of the complex dynamics of estuaries requires an analysis of continuous long-term monitoring series.

1. Introduction

Estuaries are generally defined as transitional coastal ecosystems that are found between the land and sea (Levin et al., 2001), which are characterized by varying and often unpredictable hydrological, morphological and chemical conditions (McLusky, 1993). Even though estuaries perform numerous and important ecosystem functions (Levin et al., 2001), historically (and even now) they are undervalued for the ecosystem services they provide. Thus they have been subject to numerous human exploitations including land reclamation, waste water drainage (e.g. from agriculture or industry), and dredging (de Jonge et al., 2014; Essink, 2003; McLusky and McCrory, 1989; McLusky, 1999). Indeed, European coastal estuaries are some of the most severely degraded coastal systems worldwide (Lotze et al., 2006).

Understanding baseline conditions in estuaries is often complicated by a lack of historical data (Lotze et al., 2006). Without a view to what happened in the past, we ignore the possibility that today's ecosystems

look very different to what they were, i.e. a shifting baseline (Pauly, 1995). Thus, a first step in understanding the true dynamic and/or natural state of estuaries, is to compile available historical data to describe the baselines of these systems (Andersen et al., 2017; Kolbe and Michaelis, 2001; Villnäs and Norkko, 2011). Ecological baselines are by definition the point at which data describing a system is available, because even though changes might have occurred prior to the first point of measurement, without earlier information we cannot infer what changes occurred.

In the Wadden Sea, an intertidal system that extends from The Netherlands to Denmark, there are four major areas with a significant freshwater input: the remaining IJsselmeer (the former Zuiderzee), the Ems Dollard, Weser and Elbe estuaries. These estuaries have been subjected to human pressures over a long period of time. Of the three, the Ems Dollard estuary is well known for changes associated with land reclamation, eutrophication and sediment dredging. From the 16th to 19th centuries, the Dollard basin was largely reclaimed for agricultural

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Fig. 1. Map with important physical locations and the three defined compartments of the Ems Dollard estuary.

land. Starting in the mid-19th century, large potato starch and cardboard factories were built in the surroundings of the Dollard. The organic and nutrient rich wastewater from these factories was dumped into the estuary until the 1990s mainly via the Westwoldse Aa (Arkel, 1977; Eggink, 1965; Essink, 2003). As this problem was so severe, it was then called the “biggest waste problem in the world” (Ribbius, 1961).

Nowadays, the waste water entering the Dollard is cleaner due to measures taken by the government to clean-up the waterways from the 1980s to the 1990s (Esselink et al., 1989; Essink, 2003). More specifically, the biochemical oxygen demand (BOD) declined from 25×10^6 kg BOD₅ in the 1970s to 10×10^6 kg BOD₅ in the 1980s to 0.5×10^6 kg BOD₅ in the 1990s (Essink, 2003). In the 1980s, as the era of industrial shipping developed, and ships got bigger, the channels in the estuary were, and are still, dredged for new harbours and for mega-ships to sail through. The deepening of the channels has caused significant changes to the hydrography and sediment dynamics (de Jonge et al., 2014). Possibly, the tidal asymmetry has changed, with increased landward transport of the fine sediments. The Inner Ems has also become an efficient sediment trap where high concentrations of fine suspended sediment are known to accumulate (Talke and De Swart, 2006). Furthermore, the estuarine turbidity zone in the Ems Dollard has become hyperturbid since the 1990s (van Maren et al., 2015).

The Ems Dollard is the only Wadden Sea estuary where quantitative ecological monitoring occurred on an estuary-wide scale in the 1970s (the first baseline study for this system). The study of multiple biological, hydrodynamic and chemical aspects of the estuary was completed during the Biological Research of the Ems Dollard Estuary (BOEDE; Anonymous, 1985; Baretta and Ruardij, 1988). This program was initiated to study the effects of waste discharge into the estuary from the potato and cardboard industries (de Wolf, 1977, 1978). During the BOEDE program, macrozoobenthos were one of the indicators used to study ecological change in the Ems Dollard. Macrozoobenthos are considered useful for monitoring the ecological health of estuaries because they cannot escape adverse conditions, are often short-lived (< 5 years in most cases, Beukema et al., 1999; Hewitt et al., 2005), have clear environmental associations (e.g. Compton et al., 2009; Kraan et al., 2010; Ysebaert and Herman, 2002), and are known to respond to stressors in the environment (Pearson and Rosenberg, 1978).

Furthermore, they provide important ecosystem services like nutrient cycling, sediment stabilization and facilitation of other species (de Goeij et al., 2001; Donadi et al., 2013; Donadi, 2014; Lohrer et al., 2004; Thrush et al., 2006). Macrozoobenthos are also an important food source for a variety of fish and shorebird species. The Ems Dollard is known to be an important nursery and feeding area for juvenile fishes (Jager et al., 1993; Jager, 1999, 2001), and also a feeding area for migratory shorebirds (Prop et al., 2012). In addition, a recent study has shown that the macrozoobenthic communities in the Ems Dollard are unique in the context of the Dutch Wadden Sea (Compton et al., 2013).

The BOEDE program lasted from 1973 to 1982 (Anonymous, 1985; Baretta and Ruardij, 1988). Two macrozoobenthic surveys across the entire estuary were conducted over this period (Arkel, 1977; van Arkel and Mulder, 1979, 1982). After the BOEDE project, some additional sampling occurred in 1988 in the Hond and Paap area by the Rijkswaterstaat (Kleef, 1988). Since 2009, a Wadden Sea wide monitoring study began sampling in the Ems Dollard estuary: the SIBES program (Synoptic Intertidal Benthic Surveys, Bijleveld et al., 2012; Compton et al., 2013). A comparison of these programs, which are 30 years apart, provides a unique opportunity to explore the ecological changes across the Ems Dollard estuary. Thus in this study, we compare the macrozoobenthos communities from the BOEDE and SIBES sampling programs. Our aim is to describe any changes in these communities, while taking into account any methodological differences that have occurred between these two programs.

2. Material and methods

2.1. Study area

The Ems Dollard estuary is located within the Wadden Sea ($53^\circ 20' 25''$ N and $6^\circ 58' 24''$ E, Fig. 1) and covers an area of 1071 km², of which 53% consists of intertidal flats. Most flats lie along the shore, but a large tidal flat (the Hond and Paap) divides the estuary into two parts creating two channels: Oost Friesche Gaatje and the Bocht van Watum. This system has a shared, and contested, border between The Netherlands and Germany (Fig. 1) and is co-managed by the two countries. The German part lies in the Nationalpark Niedersächsisches Wattenmeer.

Table 1

Total area of each compartment (km²), and the area of the intertidal flats (%). The average exposure time (Exposure; –), average salinity at the bottom of the estuary (Salinity; PSU), median grain size (MGS; μ m) and silt (%; < 63 μ m), and the associated confidence intervals ($\pm$ CI), are given.

Compartment	Total area	% flats	Exposure	Salinity (bottom)	MGS	Silt
Outer Ems	817	53	0.4 $\pm$ 0.02	30 $\pm$ 0.08	135 $\pm$ 2.4	9.8 $\pm$ 1.0
Inner Ems	151	38	0.3 $\pm$ 0.01	23 $\pm$ 0.12	102 $\pm$ 3.8	28.0 $\pm$ 2.2
Dollard	103	81	0.5 $\pm$ 0.01	15 $\pm$ 0.04	83 $\pm$ 2.8	40.5 $\pm$ 1.5

The Ems Dollard estuary is characterized by a wide-ranging salinity gradient (ranging from 6 Practical salinity units (PSU) starting in the Dollard to 35 PSU at sea, Table 1) and a large tidal gradient from land to sea. The tidal range is 2.2 m near Delfzijl (Inner Ems) and 3.2 m near Emden (Ysebaert et al., 1998). The major source of freshwater inflow is the river Ems, with a mean discharge of $100 \text{ m}^3 \text{ s}^{-1}$, ranging from 25 to $390 \text{ m}^3 \text{ s}^{-1}$ (Ysebaert et al., 1998). The Westerwoldsche Aa, a small canalized river discharging into the Dollard, has a mean river runoff of $12.5 \text{ m}^3 \text{ s}^{-1}$ (Ysebaert et al., 1998). The chemical, physical and biological properties of the Ems estuary have been described by Baretta and Ruurdij (1988) and de Jonge (1995).

Based on a review of previous definitions and based on the known salinity gradient and ecological differences (Baptist and Philippart, 2015), three different compartments of the Ems Dollard have recently been redefined (Fig. 1). The compartments include the marine part of the estuary (Outer Ems, mean salinity 30 PSU), the middle part of the estuary (Inner Ems, mean 23 PSU) and the brackish part of the estuary (the Dollard, mean 15 PSU). As the character of the Dollard north of the Geise-leitdamm dam consists mainly of intertidal areas, it was considered part of the Inner Ems and not the Dollard (Baptist and Philippart, 2015). We use the definition of these compartments in this paper.

2.2. Historical sampling program - BOEDE

The first BOEDE survey of macrozoobenthos was conducted from 1974 to 1975 across the Ems Dollard estuary (survey 1, Godthelp and van Arkel, 1974; van Arkel and Mulder, 1979, Table 2). The second survey was conducted from 1979 to 1981 (van Arkel and Mulder, 1982; van Gerven and van Arkel, 1979). In 1973, sampling was explorative and completed by foot with a standard benthic corer (1-mm sieve, core surface 0.06 m^2). However, upon realization that sampling by foot was so arduous in this muddy area, the team developed their own sampling device that enabled them to sample at greater speed and across vast distances (van Arkel and Mulder, 1975): the flushing sampler (1-mm sieve, core surface 0.06 m^2). The flushing sampler was used to sample the benthos by boat at high tide. The team plunged the flushing sampler into the muddy sediments, and then turned a pump on. This pump flushed the sediments through a 1-mm sieve until the core reached a depth of 30 cm. Upon reaching this depth, all organisms collected were retrieved from the sampling device. This method was compared with standard cores, and was shown to be just as effective in sampling as a standard core (van Arkel and Mulder, 1975). This sampling method was used from 1974 onwards, with further modifications in 1979 to enable sampling through shell layers (Mulder and van Arkel, 1980). All benthic samples were frozen and taken back to the research institute for identification and analysis.

The BOEDE was a first of its kind for monitoring macrozoobenthos

Table 2

Total number of sample points taken in each sampling program (total), the matched number of sample points between the two programs at a 500 m distance radius (matched), and the months of sampling.

Year	Location	Matched	Total	Sampling months
1974	Entire estuary	127	163	March, April, May, August, September, October, November
1975	Dollard	27	30	April, June
1979	Inner Ems and Dollard	80	94	Summer
1981	Outer Ems	36	36	May, June
2009	Entire estuary	216	415	October
2010	Entire estuary	212	406	August, September
2011	Entire estuary	176	374	July, August, September
2012	Entire estuary	203	398	August, September
2013	Entire estuary	193	386	August
2014	Entire estuary	243	654	July, August, September

across a broad geographic spatial scale. The flushing sampler made this possible, as it was deployed by boat. However, before the availability of GPS, identifying sampling locations was an arduous undertaking. First aerial photos were examined to identify the sampling locations and draw planned transects. Transects differed in their total length and were spaced $> 1.5 \text{ km}$ apart from each other. In the field, each transect line was identified from local geographic features and triangulation. Along a transect, sample points were positioned $\sim 200 \text{ m}$ apart, where the distance between them was measured using a meter wheel that was dragged behind a boat. As finding one's sampling location was costly in time, sampling of the entire Ems Dollard estuary took a period of a year.

Analysis of defrosted macrozoobenthic samples involved identifying the species to the lowest taxonomic level possible, and determining the ash free dry mass (AFDM_{flesh}) of individuals. The AFDM_{flesh} of an individual was determined by first drying samples at 60°C for three days, and then weighing them. Following weighing, the samples were then ashed at 560°C and then after cooling in a desiccator, weighed again. The difference between the dried and ashed mass gives the AFDM_{flesh}. In the case of bivalves, the shells and the flesh were separated to determine the AFDM of the flesh (individuals larger than $\sim 6 \text{ mm}$). Although the length of individuals was also measured this information has been lost. Worms were only counted if there was a head in the sample, but all fragments of the same worm were put together to determine the AFDM_{flesh}. Identifications were quality checked by a well-known Dutch marine ecologist (W.J. Wolff) in the field of macrozoobenthic taxonomy (Wolff, 1974). Raw data was filled in on punch cards and sent to an external location for digitization. Unfortunately, over the last decades the digitized data from the 1979 survey has been lost. As a consequence the data from this survey had to be extracted from previous reports (van Arkel and Mulder, 1982; van Gerven and van Arkel, 1979).

In addition to the BOEDE sampling, the Hond and Paap area of the Ems Dollard was sampled in 1988 by the Rijkswaterstaat RIKZ (Kleef, 1988). Notably, the macrozoobenthic specimens in this study were stored in formalin and then analysed to the lowest taxonomic level, as well as determining their AFDM_{flesh}. Unfortunately, the data for this study was not digitally available. Thus we extracted the results from this study out of the available report (Kleef, 1988).

2.3. Contemporary sampling program - SIBES

During the Synoptic Intertidal Benthic Surveys (SIBES) from 2009 to 2014 (Bijleveld et al., 2012; Compton et al., 2013), sampling locations were accessed mainly by boat (core surface 0.0173 m^2) but sometimes by foot (core surface 0.0178 m^2) over the course of summer (Table 2). Similar to the BOEDE samples, the SIBES sediment cores ($\sim 25 \text{ cm}$ depth) were also sieved on a 1-mm round mesh in the field, after which organisms were stored either frozen, in the case of larger shellfish, or in formalin for later analysis, in the case of the remaining organisms on the sieve. Samples were taken at 500 m intervals across the entire intertidal part of the estuary with an additional 15% of samples taken randomly at interspersed distances between grid points. The German part of the Outer Ems, at the entrance of the Dollard estuary, was sampled over a period of ~ 9 days in September 2014 using the same approach.

In the laboratory, samples stored in formalin were stained using rose Bengal dye (C.A.S. no. 632-68-8) for 24 h, then flushed, and identified under a stereo-microscope before their AFDM was determined (for methods see Compton et al., 2013). All molluscs were identified to species level. All other smaller organisms (mainly polychaetes and crustaceans) were identified to the lowest taxonomic level possible, called the operational taxonomic unit (OTU). In the case of worms, the fragments of the worms were noted and summed together (in the case of multiple fragments from a single worm) and put together to determine the AFDM_{flesh}. The AFDM_{flesh} of the organisms was determined in the same way as during the BOEDE program. Oligochaetes and very

small polychaetes were excluded from this study, as these small organisms were not measured in the BOEDE and RIKZ sampling programs. Small gastropods and bivalve recruits (< 6 or 8 mm, depending on the species) were weighed with multiple individuals in a single crucible and with the shell and flesh together. The shell was separated from the flesh in big bivalves (> 6 or 8 mm, depending on the species) so that the $\text{AFDM}_{\text{flesh}}$ could be measured.

Prior to analysis, outliers in the $\text{AFDM}_{\text{flesh}}$ values of the bivalves were identified with a linear regression of $\log_{10}$ of $\text{AFDM}_{\text{flesh}}$ on $\log_{10}$ of shell length. If residuals exceeded the prediction interval they were defined as outliers and an $\text{AFDM}_{\text{flesh}}$ was estimated from their length based on the regression equations determined previously. Because length estimates of polychaetes were often unavailable, we inspected outliers using a dotplot. Values deemed impossible relative to the remaining data were removed. Missing values of $\text{AFDM}_{\text{flesh}}$ for the worms were estimated by calculating the median biomass of an OTU based on the data available. The median was calculated for each species based on either (1) each year and each compartment of the Ems Dollard when available; alternatively, (2) for each year if compartment data were unavailable; or, as a last option, (3) based on all data available if neither all compartments nor all years had data available.

2.4. Environmental data

Median grain size measurements of sediment median grain size were available from the SIBES sampling (for methods, see Compton et al., 2013). Sediments were measured at every sample point in all years. Here, we calculated the average sediment grain size from 2009 to 2014 and displayed this graphically based on the Wentworth scale (Wentworth, 1922).

Salinity and exposure time were obtained by means of simulation with the General Estuarine Transport Model (GETM, Burchard and Bolding, 2002). GETM was used to simulate the hydrodynamics, temperature and salinity for the entire Wadden Sea on a resolution of 200×200 m (Grawe et al., 2015). The model output was used to obtain estimates of mean exposure time (i.e. the mean fraction of time that the seabed is exposed to the air) and salinity (practical salinity units, PSU) over the period 2009–2011 (Folmer et al., 2016a). For further details regarding the simulations we refer to Grawe et al. (2015) and to (Folmer et al., 2016b) for postprocessing of simulation data.

2.5. Analysis

Species that could not be identified to species level, or species with difficult identification keys, were considered as a single genus or family: *Corophium* spec., Bathyporeiidae, *Harmothoe* spec., *Littorina* spec., Hydrobiidae, *Scoloplos* (*Scoloplos*) *armiger*. The *Corophium* spec. are a mix of *C. volutator* and *C. arenarium*. The Hydrobiidae are a mix of *Peringia ulvae* and *Ecrobia ventrosa*. *Scoloplos* (*Scoloplos*) *armiger* is a species complex of which several cryptic species occur in the Wadden Sea (Luttikhuis et al., 2011). Species only found once in the SIBES sampling over the six-year period were excluded. This rule did not count for BOEDE, as we were not sure if their laboratory methods led to an underestimation of species observed. To determine the biomass of each species at each sample point (AFDM g m^{-2}), the $\text{AFDM}_{\text{flesh}}$ (g) of all individuals were summed together at a sample point and then divided by the total surface area sampled.

Due to the different sampling extents between the two programs, i.e. SIBES sampled much more extensively, we corrected for differences in sampling extent by selecting those sampling points in the SIBES program that lay within 500 m of the BOEDE sampling points. In a couple of cases, the BOEDE points also fell in areas not sampled by SIBES. These points were also excluded. To complete this exercise, we used the functions available within the rGeos package in R (Bivand et al., 2016). After matching the sample points between the two programs, 154 matched sample points in the first BOEDE survey and 116 matched

sample points in the second BOEDE survey remained. Over the six years of SIBES sampling, 207 sample points per year matched the samples taken during the BOEDE program.

To describe general trends in the Ems Dollard estuary over time, the environmental gradients found in this area were broadly described. Unfortunately, we did not have access to historical data describing sediment grain size, salinity or exposure times at the time of the BOEDE sampling to make a comparison of changes in the environment over time. To explore how species were distributed along the most dominant estuarine gradient of salinity (contemporary information) ranked boxplots were used.

To compare how species richness varied between the three compartments of the Ems Dollard estuary, given the same amount of sampling, the vegan package in R (Oksanen et al., 2007) was applied. Species accumulation curves (SAC) were used to compare diversity properties of community data sets using an accumulator function. The classic method finds the mean SAC and its standard deviation from random permutations of the data, or subsampling without replacement (Gotelli and Colwell, 2001). To compare the species richness (γ richness) between the compartments in the two programs, we compared the observed richness and we also extrapolated species richness, i.e. the number of unobserved species, using the function specpool. Functions used to extrapolate richness include the Chao estimator, the jackknife and the bootstrap. These are popular and commonly used methods used to extrapolate species richness (Colwell and Coddington, 1994; Palmer, 1990). These functions use the incidences in sample sites, and give a single estimate for a collection of sample sites. Functions estimate the standard errors of the estimates, and assume no variance in the observed richness.

To examine how species composition has changed in each compartment over time, the contribution of each species to the total biomass (%) was estimated. Based on the percentage of occurrence in the data, we identified the common OTUs (Table 3) along the estuarine gradient and plotted these in stacked barplots.

The yearly mean total biomass was estimated for each compartment. Due to the data often being skewed, we calculated bootstrapped means and used these to estimate 95% confidence intervals around the yearly mean estimate. The 95% confidence intervals were estimated by taking a random subsample of the data (100% of sample points, with replacement), estimating the mean as above, doing this 1000 times and using these values to estimate the 95% confidence intervals (based on the 2.5 and 97.5 percentiles). To observe whether trends in mean total biomass were the same if the taxonomic resolution was the same between both programs, the same procedure above was followed for those OTUs only shared between BOEDE and SIBES (see Appendix S1). This is called the “matched” mean biomass estimate.

To examine changes in biomass observed in each OTU over time, we first estimated the mean biomass of each OTU at a sample point given the sampling in the BOEDE and SIBES programs (including the zeros). The mean was estimated by summing the total biomass of each OTU and then dividing this by the total number of sample points for all years combined; after matching the sample points (BOEDE $n = 270$, SIBES $n = 1243$). Confidence intervals were estimated from bootstraps of the mean value, as described above. This information was tabulated.

To estimate if there has been a statistical change in the biomass of OTUs found in both programs, we used a delta-gamma generalized linear modelling approach (Lecomte et al., 2013; Stefánsson, 1996), i.e. we first modelled the presences and absences of each OTU as a function of sampling program, and then we modelled the biomass of each OTU, excluding the absences in the data, using a generalised linear model with gamma distributed errors. Thus the dependent variable was biomass (AFDM g m^{-2}) and the independent variable was the fixed factor: sampling program. The gamma distribution was used to account for the skewed data and non-integer data associated with biomass.

All analyses were completed in R (R version 3.2.4, 2016-03-10). Graphics were predominantly made using the ggplot package

Table 3

List of the operational taxonomic units (OTU) identified in the BOEDE and SIBES programs. The frequency of occurrence in all samples is shown for both programs (occur, %), the total number of individuals found in both programs over all surveys (n), the mean biomass across sample (mean, AFDM g·m⁻²) and bootstrapped confidence intervals ($\pm$ CI). The OTUs with an asterisk indicate alien species found after the BOEDE surveys. *A. succinea* is the only OTU not found in BOEDE that is now commonly found in SIBES samples (**).

Class	OTU	BOEDE				SIBES			
		n	Occur	Mean	CI	n	Occur	Mean	CI
Gastropoda	<i>Crepidula fornicata</i> *					2	0.2	0.0009	0.0023
	<i>Hydrobiidae</i>	72	26.7	0.0801	0.0385	401	32.3	0.5517	0.2980
	<i>Littorina spec.</i>	1	0.4	0.0015		2	0.2	0.0029	0.0117
	<i>Retusa obtusa</i>	4	1.5	0.0008	0.0012	5	0.4	0.0002	0.0004
Bivalvia	<i>Cerastoderma edule</i>	30	11.1	0.2246	0.1259	125	10.1	1.5563	0.8714
	<i>Ensis leei</i> *					22	1.8	0.0338	0.0380
	<i>Kurtiella bidentata</i>	1	0.4	0.0002		1	0.1	0.0000	
	<i>Limecola balthica</i>	158	58.5	0.6182	0.1188	652	52.5	0.9965	0.2179
	<i>Magallana gigas</i> *					4	0.3	0.1289	0.3717
	<i>Mya arenaria</i>	84	31.1	2.6704	1.2354	168	13.5	0.2685	0.2017
	<i>Mytilus edulis</i>	1	0.4	0.0189		10	0.8	0.0217	0.0467
	<i>Petricolaria pholadiformis</i>					1	0.1	0.0001	
	<i>Scrobicularia plana</i>	1	0.4	0.0130	0.0226	5	0.4	0.0301	0.0730
	<i>Bathyporeiidae</i>	20	7.4	0.0007	0.0004	53	4.3	0.0031	0.0030
Crustacea	<i>Urothoe spec.</i>	37	13.7	0.0037	0.0021	102	8.2	0.0116	0.0097
	<i>Carcinus maenas</i>	3	1.1	0.0006	0.0008	27	2.2	0.0099	0.0128
Malacostraca	<i>Corophium spec.</i>	97	35.9	0.2256	0.0694	753	60.6	0.2529	0.0479
	<i>Crangon crangon</i>					27	2.2	0.0165	0.0235
	<i>Gammaridae</i>	1	0.4	0.0000		4	0.3	0.0003	0.0007
Polychaeta	<i>Hemigrapsus takanoi</i> *					3	0.2	0.0100	0.0269
	<i>Alitta succinea</i> **					400	32.2	0.4255	0.1691
	<i>Arenicola marina</i>	52	19.3	0.4262	0.1431	107	8.6	0.5682	0.3984
	<i>Capitella capitata</i>	27	10.0	0.0003	0.0001	118	9.5	0.0068	0.0041
	<i>Eteone longa</i>	57	21.1	0.0037	0.0017	208	16.7	0.0094	0.0033
	<i>Harmothoe spec.</i>	9	3.3	0.0064	0.0045	32	2.6	0.0095	0.0090
	<i>Hediste diversicolor</i>	129	47.8	0.4717	0.1126	450	36.2	0.4576	0.1553
	<i>Heteromastus filiformis</i>	130	48.1	0.1951	0.0463	523	42.1	0.1370	0.0471
	<i>Lanice conchilega</i>	6	2.2	0.0160	0.0268	47	3.8	0.0882	0.0692
	<i>Marenzelleria viridis</i> *					250	20.1	0.2252	0.1020
	<i>Nephtys hombergii</i>	52	19.3	0.0715	0.0244	131	10.5	0.1172	0.0616
	<i>Nephtys spec.</i>					15	1.2	0.0033	0.0050
	<i>Phyllodoce spec.</i>	22	8.1	0.0026	0.0021	71	5.7	0.0092	0.0072
	<i>Polydora cornuta</i>					68	5.5	0.0330	0.0572
	<i>Scolecopsis (Scolecopsis) squamata</i>	1	0.4	0.0000		1	0.1	0.0044	
	<i>Scoloplos (Scoloplos) armiger</i>	71	26.3	0.0292	0.0168	322	25.9	0.2587	0.1207
	<i>Spiophanes bombyx</i>					5	0.4	0.0002	0.0004
	<i>Streblospio shrubsolii</i>					39	3.1	0.0005	0.0003

(Wickham, 2009).

3. Results

The Ems estuary was characterized by fine sand in the Outer Ems (125–250 μ m) and very fine sand (63–125 μ m) in the Inner Ems and Dollard (Fig. 2). Salinities were lowest in the center of the Dollard (6–15 PSU), with brackish/marine salinities characterizing the Inner Ems and parts of the Dollard (15–25 PSU). Marine salinities characterized the northern part of the Hond en Paap and the Outer Ems (> 25 PSU). The greatest range in exposure times were found in the Dollard (0.3–1, Fig. 2). By contrast, exposure times were shorter at the Hond en Paap and the Inner Ems (0.3–0.5).

Macrozoobenthic sampling was conducted across the entire estuary in the BOEDE and SIBES sampling programs (Fig. 3). The taxonomic resolution of the BOEDE program was lower than that of SIBES both when comparing the observed species richness (γ -diversity_{BOEDE} = 25, γ -diversity_{SIBES} = 37) and the expected species richness (Table 4, γ -diversity_{BOEDE} = 33, γ -diversity_{SIBES} = 38). The species accumulation curves also showed that the BOEDE sampling did not reach its asymptote, whereas the SIBES sampling has reached an asymptote for the entire Ems Dollard estuary, indicating all possible species have been sampled (Table 4, Fig. 4). A comparison of expected diversity in each of the compartments of the Ems Dollard showed that species richness was high in the Outer Ems (γ -diversity_{BOEDE} = 33, γ -diversity_{SIBES} = 36) and relatively similar to that of the Inner Ems in the case of SIBES (γ -

diversity_{BOEDE} = 17, γ -diversity_{SIBES} = 37), but lowest in the Dollard (γ -diversity_{BOEDE} = 14, γ -diversity_{SIBES} = 30) (Table 4). In the Inner Ems and Dollard, γ -diversity differed between the two programs and was now higher than previously observed.

A comparison of the ranked boxplots with respect to salinity showed that the occurrences of the OTUs (Fig. 5), with respect to salinity, did not change greatly between the two programs. OTUs found to be euryhaline during BOEDE remained so in the SIBES program, e.g. *Hediste diversicolor*, *Heteromastus filiformis*, *Eteone longa*, *Limecola balthica*. OTUs more commonly associated with lower salinities in BOEDE remained so during SIBES, i.e. *Mya arenaria* and *Corophium spec.*

The community composition of the Ems Dollard showed a gradient from land to sea (Fig. 6). In the Outer Ems, the community was dominated by the bivalve *Cerastoderma edule* and the worms *Hediste diversicolor* and *Arenicola marina*. In the Inner Ems, the community was dominated by the bivalves *Mya arenaria*, *Limecola balthica*, gastropods *Hydrobiidae*, and the worms *A. marina*, and *Alitta succinea*. In the Dollard, the most common species were: the bivalves *M. arenaria*, *L. balthica*, the worms *H. diversicolor*, *A. succinea*, *Heteromastus filiformis*, and the crustacean *Corophium spec.* The members of each community, in each compartment of the Ems Dollard, remained relatively consistent over time, with the exception of the Inner Ems. In the Inner Ems, the community was dominated by the bivalve *M. arenaria* in the 1970s (> 60% of the biomass), but was recently dominated by a polychaete worm previously not recorded in this area *A. succinea* (> 10% of the biomass), whereas *Mya arenaria* only contributed 2–11%. Notably, an

Fig. 2. Sediment characteristics, salinity (PSU) and exposure times (fraction of time exposed) derived from either contemporary data or hydrodynamic models. The bathymetry available from the Rijkswaterstaat (2011–2014) was used as a background layer for each map. Data used to create these figures is described in the [Material and methods](#).

independent survey of the Inner Ems in 1988 (Kleef, 1988), also observed that *M. arenaria* dominated the community (> 10% of the biomass). In the Outer Ems and the Dollard, the community members did not change, but instead the relative contribution of the members to the total biomass changed. An exception was observed in the Outer Ems in 1974, when deeper living species like *A. marina* and *M. arenaria* were

not observed.

The mean total biomass for each sample year, in each of the compartments, showed interesting trends (Fig. 7, all), even when accounting for differences in the taxonomic resolution between the two programs (Fig. 7, matched). In the Outer Ems, mean total biomass was higher in contemporary times relative to the 1970s (mean and confidence intervals: average_{BOEDE} 4 ± 2 AFDM g·m⁻² vs. average_{SIBES} 14 ± 4 AFDM g·m⁻²). In the Inner Ems, mean total biomass was higher in the 1970s relative to now (average_{BOEDE} 17 ± 1 AFDM g·m⁻², average_{RIKZ} 108 AFDM g·m⁻² and average_{SIBES} 6 ± 1 AFDM g·m⁻²). In the Dollard, total biomass was also higher in the 1970s relative to now (average_{BOEDE} 6 ± 2 AFDM g·m⁻² and average_{SIBES} 3 ± 0 AFDM g·m⁻²). The differences in total biomass between all OTUs versus the matched OTUs, when accounting for taxonomic resolution, were always smaller than the differences observed between the programs (Fig. 7).

A comparison of the changes in the total biomass of common OTUs separately, showed that the probability of occurrence and average biomass of OTUs in SIBES differed relative to the BOEDE program (Fig. 8, Table 5). In the Outer Ems, 9 of the 11 OTUs showed a significantly lower probability of occurrence during SIBES. In the Inner Ems, 3 of the 11 OTUs showed a significantly higher positive probability of occurrence. In the Dollard, one of 8 OTUs had a higher probability of occurrence (Table 5). In terms of changes in average biomass, 10 of 11 OTUs had a higher average biomass in the Outer Ems during SIBES. In the Inner Ems, 3 of 11 OTUs had a higher average biomass, and a single bivalve *M. arenaria* had a lower average biomass. In the Dollard, two OTUs had a significantly lower biomass (*M. arenaria* and *Corophium* spec.).

4. Discussion

A comparison of the macrozoobenthos communities across the entire Ems Dollard estuary from the 1970s with 2009–2014, revealed that the Ems Dollard could now be in a “new ecological state” relative to 30 years ago (Scheffer et al., 2001). Specifically, three main changes were observed by comparing these snapshots in time: (1) a change in species richness, due to new immigrants and sampling differences, (2) a decrease in total biomass in the Inner Ems and Dollard, and (3) a shift from a bivalve- to polychaete worm-dominated community in the Inner Ems, i.e. the Hond en Paap area. Below the differences between these two programs are discussed in more detail.

4.1. Species richness and environmental associations

The total number of species found in the BOEDE program was lower than in SIBES, even though the area of the core was larger than that used during the SIBES surveys (0.06 m² cf. ~0.0173 m²). Increased species richness in the SIBES program could be associated with the addition of alien species that have found their way to the Ems Dollard after the 1980s. These include the razor shell *Ensis leei*, the worm *Marenzelleria viridis*, the gastropod *Crepidula fornicata*, the crab *Hemigrapsus takanoi* and the bivalve *Magallana gigas* (Essink et al., 1998; Wolff, 2005, 1999). Another reason for the difference in the total number of species observed could be due to the sample handling used during BOEDE. Specifically, the samples from BOEDE were frozen, whereas the contemporary samples were either stored on formalin, in the case of small organisms, or frozen, in the cases of bigger organisms. As sieve mesh size was the same between the two programmes, we do not think that sieve mesh size contributed to these differences. Instead, freezing samples would have destroyed small organisms making them harder to identify. Indeed, the SIBES program measured many smaller organisms that were not identified or commonly measured during BOEDE, e.g. *Polydora cornuta*, Gammaridae. The actual observed number of species identified in SIBES is in fact higher than we report here but we grouped a number of taxonomic groups to enable

Fig. 3. Sampled points in the BOEDE and SIBES sampling programs. The blue points indicate the points used for this comparison, and the black points are those left out, as they did not match between the two surveys. The defined compartments of the estuary (Outer Ems, Inner Ems and Dollard) are highlighted in the background.

comparability between the two programs in this study (see [Material and methods](#) and Appendix S1).

Despite the differences in observed species richness between the BOEDE and SIBES programs, both programs showed a gradient in species richness from the sea to land. In both, diversity was highest in the Outer Ems and lowest in the Dollard. Interestingly, total diversity in the Inner Ems was similar to the Outer Ems in the SIBES surveys,

whereas a gradient of high to low richness from sea to land was observed during the BOEDE sampling (Anonymous, 1985; Ysebaert et al., 1998). Diversity is likely, in part, to be associated with salinity tolerance, as few species can tolerate the low salinities of the Dollard (Essink and Bos, 1985; Ysebaert et al., 1998). Interestingly, the ranked boxplots showed that the occurrence of a number of OTUs did not change, with respect to the estimated contemporary salinity gradient, between the

Table 4
Observed (Sp(Obs)) and estimated species richness from the BOEDE and SIBES sampling programs for the entire estuary and each compartment (γ -diversity). Species richness was estimated using the Chao equation (chao), a jackknife estimator (jack1) and a bootstrapped estimation (boot). Standard errors of these estimates are shown. The number of samples used for these calculations are shown (n).

Program	Compartment	Sp(Obs)	chao	jack1	boot	n
SIBES	Entire	37	37 $\pm$ 0.44	38 $\pm$ 1.00	38 $\pm$ 0.89	1243
	Outer Ems	34	35 $\pm$ 2.28	37 $\pm$ 1.73	36 $\pm$ 1.26	310
	Inner Ems	34	38 $\pm$ 5.27	38 $\pm$ 1.99	36 $\pm$ 1.17	259
	Dollard	28	29 $\pm$ 1.46	31 $\pm$ 1.73	30 $\pm$ 1.19	674
BOEDE	Entire	25	40 $\pm$ 13.68	31 $\pm$ 2.82	27 $\pm$ 1.38	270
	Outer Ems	24	42 $\pm$ 23.26	30 $\pm$ 2.79	26 $\pm$ 1.38	64
	Inner Ems	16	17 $\pm$ 1.29	18 $\pm$ 1.39	17 $\pm$ 1.13	70
	Dollard	12	14 $\pm$ 3.72	14 $\pm$ 1.4	13 $\pm$ 0.92	136

Fig. 4. Species accumulation curves based on all available data collected from the BOEDE and SIBES surveys in the Ems Dollard estuary (see Table 3 and Appendix S1). Accumulation curves were derived by examining random permutations of the data and estimating species richness given a number of sample sites.

two surveys. Specifically, a number of OTUs were observed across a wide-range of salinities, i.e. *E. longa*, *L. balthica*, *H. filiformis*, *H. diversicolor*, *Hydrobiidae*, whereas others were more likely found in either higher or lower salinities. Interestingly, the species/OTUs with the widest distributions across the salinity gradient are known to be species complexes (cryptic species) at other localities (*E. longa* (Carr et al.,

2011), *H. filiformis* (Glasby et al., 2009), *L. balthica* (Luttikhuisen et al., 2003), *H. diversicolor* (Audzijonyte et al., 2008)), i.e. morphologically similar species that have erroneously been classified as a single species (Bickford et al., 2007; Nygren, 2013). This suggests not only that species diversity in the Ems Dollard area could be higher than previously recognized, but also that it might be more tightly linked to local salinity regimes than possible to disentangle at the moment.

The boxplots of species occurrence with respect to salinity are a rough guide to the species distributions, as they assume salinity alone is important. A modelling framework examining the distributions of individual species with respect to environmental gradients would provide an idea of how salinity interacts with other variables to define the distributions of species in this area (e.g. van der Meer, 1999; Ysebaert and Herman, 2002).

4.2. Total biomass over time

Total biomass during the past surveys was higher in the Inner Ems and Dollard than now. In the Outer Ems, the reverse trend was observed. These differences form a striking contrast with species richness estimations, which were lower for the historical compared to the contemporary data. Despite methodological differences these patterns in total biomass appear to be robust, as the larger organisms, which mainly determine total biomass, were measured well in both programs. Furthermore, when taking differences of taxonomic resolution into account the trends remained robust (see Fig. 6).

One methodological issue that plagues the BOEDE estimate of total biomass in 1974, is that the sampling device used differed to that in 1979. In the Outer Ems in 1974, shell layers hindered sampling to a depth of > 20 cm and thus samples were not taken deep enough to obtain the deeper living organisms like the soft-shell clam *M. arenaria* and the lugworm *A. marina*. Consistent with this method change, total

Fig. 5. The occurrence of operational taxonomic units (OTUs) with respect to the contemporary estimate of salinity (PSU) at the bottom of the estuary in the BOEDE and SIBES sampling programs (see Material and methods). The boxplots are ordered based on the median value of occurrence with respect to salinity. The circles indicate the mean value for each OTU.

Fig. 6. Changes in community composition over time for common occurring OTUs (see Table 3) reflected by the relative biomass (%) of each OTU in each sample year.

biomass was lower in 1974 relative to 1979. But this difference in sampling did not exist in the Inner Ems and Dollard, as here there were no shell layers. Note that the sampling device was adapted, by modifying it into a suction sampler, between 1974 and 1979 to deal with this problem (Mulder and van Arkel, 1980).

In the Outer Ems, an increase in total biomass was observed in recent years. This observed increase in total biomass was in part due to differences in sampling techniques, but also due to an increase in the biomass of the majority of OTUs found in this area (see Fig. 8). For example, cockle biomass was now higher than before. As cockle recruitment can be highly site dependent and related to temperature (Magalhães et al., 2016), this might explain these differences in time. In 2011, there was also a large recruitment of cockles *C. edule* in the Dutch Wadden Sea (Beukema and Dekker, 2015; Bijleveld et al., 2015), which would have driven total biomass even higher.

In the Dollard, the lower total biomass observed in recent years, might be associated with further de-eutrophication of the Dollard. In the Dollard, it is known that at the time of the BOEDE surveys, the south eastern part of the Dollard, near the river Westwoldse Aa, suffered severe anoxia where only a limited number of species survived (Essink, 2003; van Es et al., 1980). Furthermore, a significant relationship between total macrobenthic biomass and organic waste discharge in the previous year (BOD_5 from the Westwoldse Aa) was observed along transects on the Heringsplaat in the Dollard (Essink, 2003; Prop, 1998). This ties in with the prediction of a higher total biomass in areas of moderate organic enrichment along a gradient of high to low enrichment (Pearson and Rosenberg, 1978). A decline in organic enrichment suggests that the Dollard has shown a response to de-eutrophication over time (Essink, 2003; Prop, 1998). Interestingly, a considerably higher total biomass was observed in the Inner Ems during the previous surveys (Kleef, 1988; van Arkel and Mulder, 1979, 1982) compared to now. Whether the effect of de-eutrophication extends to the Hond en Paap in the Inner Ems is currently unknown. In contrast to the Hond en Paap, an increase in total biomass was observed in the Outer Ems suggesting that de-eutrophication did not extend to the Wadden Sea.

An alternative explanation for the decline in total biomass in the Inner Ems could be the decline in the total biomass of *Mya arenaria*. Two long-term data series have shown that *M. arenaria* has not declined in biomass in the Balgzand area of the Wadden Sea, whereas it has declined in the Dollard (Dekker pers. comm.). A decline in this species could suggest that either recruitment failure has occurred or that survival chances for this species have declined in this area, e.g. due to sediment disturbance via dredging (Essink and Bos, 1985; Wolff, 1973). We will discuss this further below.

4.3. Gradient in community composition over time

Changing communities from marine to brackish zones are a

Fig. 7. Mean total biomass of all OTUs (and associated 95% confidence intervals) across the different surveys. Matched data points (grey circles) indicate those OTUs common to both surveys. Note that in 1988 the mean total biomass in the Inner Ems was reported at 108 AFDM $g.m^{-2}$ (Kleef, 1988), i.e. substantially higher than in any of the data depicted here.

Fig. 8. A comparison of the biomass (AFDM g.m⁻² log scale) of individual OTUs, across sampling points within a compartment, when comparing the BOEDE and SIBES surveys (all years together) shown by boxplots, with raw data within each box. The open circle represents the mean value.

Table 5

Changes in the occurrence and the biomass (AFDM g m^{-2}) of each OTU, found in both programs, as tested using a binomial model and a gamma model (a delta-gamma glm). The residual degrees of freedom (resid.df), the null deviance (nulldev), the residual deviance (resdev) and the p-value (pval) are shown for both models. The intercepts (bdB0, sbB1) are shown for each model. The relative positive (+) or negative (–) change in each species is indicated (Δ).

Comp	CLASS	OTU	RESULTS PA							RESULTS GLM with gamma errors						
			resid.df	nulldev	resdev	pval	bd_B0	sb_B1	Δ	resid.df	nulldev	resdev	pval	bd_B0	sb_B1	Δ
Dollard	Bivalve	<i>Limecola balthica</i>	808	1109.0	1108.4	0.46	0.54	0.53		456	721.7	721.5	0.7	1.35	0.94	
	Bivalve	<i>Mya arenaria</i>	808	818.9	811.2	0.01	0.29	0.35		163	452.4	439.8	0.1	2.95	0.54	–
	Crustacean	<i>Corophium spec.</i>	808	693.4	569.5	0	0.49	0.92	+	684	920.6	890.5	0.0	0.80	0.52	–
	Gastropod	<i>Hydrobiidae</i>	808	1039.3	1038.7	0.47	0.37	0.46		274	586.6	566.3	0.4	0.18	2.15	
	Polychaete	<i>Arenicola marina</i>	808	202.1	202.0	0.86	0.03	0.48		20	48.9	46.4	0.4	1.22	2.67	
	Polychaete	<i>Eteone longa</i>	808	613.3	611.5	0.18	0.16	0.41		100	68.8	60.5	0.0	0.03	2.11	+
	Polychaete	<i>Hediste diversicolor</i>	808	1122.8	1119.7	0.08	0.57	0.42		407	491.6	474.4	0.0	1.29	0.61	–
	Polychaete	<i>Heteromastus filiformis</i>	808	1118.7	1118.4	0.59	0.49	0.47		374	374.3	314.1	0.0	0.59	0.38	–
	Bivalve	<i>Cerastoderma edule</i>	327	258.9	258.5	0.52	0.16	0.44		42	119.5	115.3	0.3	2.36	0.50	
	Bivalve	<i>Limecola balthica</i>	327	438.8	435.2	0.06	0.49	0.37		125	189.7	189.0	0.4	1.43	1.18	
Inner Ems	Bivalve	<i>Mya arenaria</i>	327	303.3	287.6	0	0.34	0.22	–	55	280.0	226.9	0.0	22.49	0.13	–
	Crustacean	<i>Corophium spec.</i>	327	446.9	433.2	0	0.23	0.75	+	135	219.9	219.9	1.0	0.45	1.00	
	Gastropod	<i>Hydrobiidae</i>	327	405.8	362.7	0	0.03	0.95	+	99	300.3	299.6	0.6	1.97	1.94	
	Polychaete	<i>Arenicola marina</i>	327	223.0	208.3	0	0.24	0.19	–	33	53.8	51.8	0.3	3.14	0.62	
	Polychaete	<i>Eteone longa</i>	327	315.5	308.8	0.01	0.09	0.74		59	65.8	45.4	0.0	0.00	14.50	+
	Polychaete	<i>Hediste diversicolor</i>	327	273.4	272.3	0.3	0.19	0.41		46	105.5	96.5	0.0	0.36	2.91	
	Polychaete	<i>Heteromastus filiformis</i>	327	451.9	443.2	0	0.40	0.69	+	181	223.0	223.0	0.9	0.48	1.04	
	Polychaete	<i>Nephtys hombergii</i>	327	324.2	314.9	0	0.33	0.28	–	62	63.0	53.2	0.0	0.30	2.36	+
	Polychaete	<i>Scoloplos armiger</i>	327	345.7	344.5	0.27	0.17	0.59		70	130.3	109.2	0.0	0.15	5.87	+
	Bivalve	<i>Cerastoderma edule</i>	372	444.1	443.7	0.54	0.25	0.55		103	151.1	108.9	0.0	2.16	9.83	+
Outer Ems	Bivalve	<i>Limecola balthica</i>	372	502.9	489.7	0	0.80	0.25	–	223	449.8	329.8	0.0	0.39	8.70	+
	Bivalve	<i>Mya arenaria</i>	372	208.9	167.9	0	0.31	0.07	–	28	42.9	42.9	0.8	3.17	1.10	
	Crustacean	<i>Corophium spec.</i>	372	193.9	175.2	0	0.22	0.14	–	25	46.7	19.7	0.0	0.01	8.55	+
	Gastropod	<i>Hydrobiidae</i>	372	426.0	424.8	0.27	0.31	0.42		94	374.5	336.3	0.0	0.44	6.61	+
	Polychaete	<i>Arenicola marina</i>	372	438.3	422.3	0	0.48	0.24	–	100	180.9	140.6	0.0	1.83	4.71	+
	Polychaete	<i>Eteone longa</i>	372	438.3	426.6	0	0.45	0.27	–	100	126.0	84.9	0.0	0.01	5.04	+
	Polychaete	<i>Hediste diversicolor</i>	372	472.3	448.7	0	0.59	0.20	–	120	267.7	208.5	0.0	0.57	5.64	+
	Polychaete	<i>Heteromastus filiformis</i>	372	421.7	386.6	0	0.56	0.15	–	92	281.8	90.0	0.0	0.01	49.03	+
	Polychaete	<i>Nephtys hombergii</i>	372	466.3	459.9	0.01	0.45	0.33	–	116	143.7	121.7	0.0	0.43	3.04	+
	Polychaete	<i>Scoloplos armiger</i>	372	339.2	333.9	0.02	0.92	0.27	–	309	689.1	531.5	0.0	0.10	10.23	+

common feature of estuaries (McLusky et al., 1993; Remane, 1934; Wolff, 1973; Ysebaert et al., 1998). Previously, it has been shown that the Ems Dollard had a similar species composition to another estuary in The Netherlands: the Scheldt (Ysebaert et al., 1998) but a unique community composition within the Dutch Wadden Sea (Compton et al., 2013). Here, the gradient in community composition from land to sea was observed in both the historical and the contemporary data. However, we also showed that there have been changes in the community composition of the Inner Ems over time.

In the Outer Ems, a marine community, dominated by the common cockle *C. edule* and the lugworm *A. marina* was observed. Notably, the community of the Outer Ems is largely similar to the other tidal basins of the eastern Wadden Sea (from Pinkegat to the Outer Ems, Compton et al., 2013). These communities are all dominated by *C. edule*, followed by *A. marina* and the Baltic tellin *M. balthica*. By contrast to the Outer Ems, *C. edule* was generally absent from the inner part of the estuary. Experiments along the estuarine gradient of the Ems Dollard estuary showed that, when *C. edule* was transplanted into the inner estuary, it did not grow to the same extent as in the Outer Ems, and even died after an event with very low salinities (Essink and Bos, 1985). In the Dollard, a brackish community dominated by *H. diversicolor*, *L. balthica* and *M. arenaria* was observed. These species are well known for their ability to withstand low salinities (Wolff, 1973).

Changes over time in the Inner Ems have been dramatic. The community has shifted from one dominated by bivalves to one now dominated by polychaete worms, and more specifically two polychaete worms not observed in the 1970s: *Alitta succinea* and *Marenzelleria viridis*. This shift is corroborated by an independent survey conducted in 1988, that also found a community dominated by *M. arenaria*, but only a small proportion of polychaete worms (Kleef, 1988). The population succession of *M. viridis* has been followed intensively in the Dollard via a long-term monitoring programme (Dekker, 1992; Essink et al., 1998).

This invasive species arrived in the Dollard in 1985, exploded in numbers soon after (1990s), but then went into a dramatic population decline stabilising in numbers from 1996 onward (Essink and Dekker, 2002). By contrast, an explanation for the apparent boom in *A. succinea* is less clear. This species is cryptogenic and possibly also a cryptic species (Abbiati and Maltagliati, 1992). For example, this species first invaded the waters of Scandinavia in 1953 (Rasmussen, 1973). Records from The Netherlands show that it has been here since the early 19th century (de Beaufort, 1954; Wolff, 1973), but its historical distribution pattern within the Wadden Sea is undocumented. In the Heringsplaat, in the Dollard, *A. succinea* has recently shown an increase in total biomass (Dekker pers. comm.). The recent dominance of this species in the Hond en Paap (and Dollard) appears to raise more questions than answers.

An inspection of the individual OTUs from the Inner Ems showed that *M. arenaria* has declined significantly in total biomass from 1974 to now. Other species have not shown such a significant change in their average biomass. A potential methodological difference that could contribute to this difference in *M. arenaria* biomass is the sampling depth of the sediment corer. In the case of BOEDE, the sediment corer sampled till 30 cm, whereas this was only 25 cm in SIBES. A deeper sampling depth might have increased the chance of sampling deeper living *M. arenaria*. However, not many *M. arenaria* siphons were found in the SIBES samples, which would have been an indication of deep-living individuals. In support of our observation that the total biomass of *M. arenaria* has declined in the Inner Ems, another long-term dataset from the Heringsplaat in the Dollard has also shown a decline in *M. arenaria* over time (Dekker pers. comm.).

The relatively low biomass observed for *M. arenaria* in the Inner Ems and Dollard in recent times could suggest either recruitment failure in *M. arenaria* or human-mediated changes. Recruitment failure could be driven by milder winters, associated with climate change, as these

Fig. 9. Change in bathymetry in the Ems Dollard over the last thirty years. Data was sourced from the bathymetry profiles available from the Rijkswaterstaat.

have a negative impact on bivalve recruitment (Beukema, 1992). However, *M. arenaria* is only declining at a couple of sites (Dollard, Groninger Wad and near Norderney), and not Wadden Sea wide (Drent et al., 2017, Dekker pers. comm.). This suggests *M. arenaria*'s decline is quite specific to this area. Other local human-mediated changes that could be associated with *M. arenaria*'s decline could be de-eutrophication (as discussed above) or changes associated with dredging. Dredging has dramatically changed the morphology of the Ems Dollard estuary (Fig. 9), and has led to large increases in suspended particulate matter levels in the water column (de Jonge et al., 2014).

Suspension feeders, like *M. arenaria*, can be negatively affected by mild (3 cm) to catastrophic (up to 10 cm) increases in sediment loads (Norkko et al., 2002; Peterson, 1985), as suspension feeders have

troubles clearing excess sediment loads (Ellis et al., 2002). This species is also likely to be very vulnerable to sediment erosion, as it cannot move within the sediments as a larger adult (Wolff, 1973). Other environmental changes, associated with dredging, could include a change in local salinity, sediment grain size and/or inundation times. In the Jade tidal channel in Germany, macrozoobenthic communities in dredged areas differed to areas where no dredging had occurred (Gutperlet et al., 2015, 2017). Opportunistic and highly mobile species colonised the disturbed sediments after the dredging construction phase (Gutperlet et al., 2015). Historical environmental information should be collated and compared with current data to describe such potential changes.

The compilation of historical data is an important exercise, because

despite the contingencies between areas, consistencies might point to similar drivers of change at large spatial scales. Furthermore, by definition these are the first baselines describing such ecosystems, thus they can provide useful clues to significant change, whereas without such data a significant shift would go unnoticed and would be considered the “normal” ecological state (Pauly, 1995). Indeed, similar historical studies that compared macrozoobenthos data from ~1930–1950 with more contemporary data available in the Wadden Sea have shown similar trends to here, even though the Wadden Sea is not an estuary like the Ems Dollard (Christiane et al., 2017). Specifically, in the western Wadden Sea and Jade Bay a decline in molluscs was observed (Kraan et al., 2011; Schückel and Kröncke, 2013). At the Weser estuary, Jade bay and Königshafen, an increased number of aliens (Kolbe and Michaelis, 2001; Reise et al., 2008; Schückel and Kröncke, 2013; Schumacher et al., 2014) and cryptogenic species (Schückel and Kröncke, 2013) was observed. While these studies provide valuable insights, these insights are limited due to a lack of continuous long-term monitoring. Continuous long-term monitoring is important in understanding ecosystem change because estuarine systems are inherently variable in time (e.g. Essink et al., 1998; Essink, 2003) and space (e.g. Ysebaert et al., 1998). In the case of Königshafen, an increase in biomass was observed in 1980 to 1992 due to *M. viridis* (Kolbe and Michaelis, 2001), but from sites with continuous long-term monitoring it was observed that this species declined in biomass over time (Essink and Dekker, 2002). Thus while historical data provides immensely valuable insights into patterns at large spatial scales, they should supplement continuous long-term monitoring and not be considered as useful on their own.

5. Conclusions

In the Ems Dollard estuary, an ecological state change appears to have occurred, when comparing the system now relative to the first ever baseline study in this area. Mean total biomass in the Inner Ems and Dollard has decreased in recent times, and the community composition of the Inner Ems has changed from one dominated by bivalves to one dominated by polychaete worms. An increase in species richness was also observed in part due to the influx of alien and cryptogenic immigrants but mainly as a result of sampling differences. We conclude that while the comparison of sporadic historical data provides insights at large spatial scales, it should be supplemented by continuous long-term monitoring where possible to provide insights into the complex dynamics of estuarine ecosystems.

Supplementary data to this article can be found online at <http://dx.doi.org/10.1016/j.seares.2017.06.014>.

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