Early vegetation succession and management options on a brackish sediment dike

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Abstract

Port authorities worldwide are confronted with a continuous stream of dredged sediment that needs to be disposed. One of the solutions being considered for the Port of Antwerp (Belgium) is the beneficial use of constructing landscape dikes with dredged material. This paper examines whether spontaneous development, whether or not combined with mowing or sowing, is a valuable alternative to afforestation for revalorizing brackish sediment dikes. Early vegetation succession was followed on an experimental dike along the river Scheldt. The pioneer vegetation was closely related to flood-mark communities of the *Atriplicetum littoralis*. The most important abiotic variables for determining consecutive development were the salt gradient originating along the topographic gradient, and the mowing management. When halophytic pioneer species have disappeared, the successional pathways on dredged-sediment dikes are very similar to those described for other hyper-eutrophicated soils, such as abandoned arable fields. Zero-management results in species-poor *Urtica–Elymus* stands. Mowing and cut removal leads to ruderal grassland related to the *Artemisietea*. Grass species need to be sown to obtain target communities of the *Arrhenatherion*. The consequences of these findings for the construction, design and management of future landscape dikes are discussed.

Keywords: DCA; *Atriplicetum littoralis*; Dredged material; Beneficial use; Upland disposal; Landscape dike; Spontaneous development

1. Introduction

Sedimentation of polluted material in harbor docks, and in access channels to the docks, is a considerable and ongoing problem faced by port authorities worldwide. This is a particular problem in the Port of Antwerp (Belgium) due to its location in the zone of the river Scheldt with maximum sediment load (Baeyens et al., 1998). Approximately 400,000 t of dry matter have to be dredged and stored each year (Anon., 1995; Wartel et al., 2002).

Traditionally, the sediment is stored in confined upland disposal sites or underwater pits (Anon., 1995). This policy is untenable for the Port of Antwerp, due to the limited amount of space available in this densely

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populated part of Belgium. Considerable efforts are being made to find new solutions to cope with the continuous stream of dredged material. Csiti and Burt (1999) provide an extensive overview of possible solutions and applications for dredged material. One of the options being considered for the Port of Antwerp is the beneficial use of landscape restoration or landscape development (Luyssaert et al., 2001). In theory, landscape dikes have several advantages over traditional disposal sites: (1) use of space is optimized by the vertical storage of dredged material; (2) dikes can be used as a buffer between incompatible adjacent land uses (e.g. industrial compared to dwelling sites); (3) dikes can be permanently integrated in the ecological infrastructure of the port area, unlike traditional disposal sites that often have a high but temporary ecological value due to constant exploitation; (4) passive recreation or development into amenity areas is possible (e.g. Luyssaert et al., 2001).

Essentially, landscape dikes are disposal sites for dredged material that may or may not be confined by retention dikes. Therefore, many of the reasons for 'aesthetic dissatisfaction' identified by Mann et al. (1975) for classical confined disposal sites also apply to landscape dikes: the unnatural appearance of the disposed material and its retention structures (i.e. sandy dikes), size of the area covered (albeit much less than classical sites), visual incompatibilities with adjacent natural or man-made environments and interference with existing land-use patterns. Consequently, public acceptance is often low. Revalorization of such sites by afforestation is a generally applied measure and is considered to be a useful tool for landscape development (e.g. De Vos, 1994; Vandecasteele and De Vos, 2002; Luyssaert et al., 2001).

Contrary to afforestation, the intentional spontaneous development of disposal sites is rarely considered. For freshwater disposal sites, this is partly because new sites are rapidly colonized by willows. These are well known for accumulating heavy metals in their leaves, hence increasing the risk of bioaccumulation and the spreading of pollutants in the ecosystem (e.g. Vandecasteele et al., 2002, 2004). An additional concern is that spontaneously developing sites are likely to give the impression of being neglected and so are 'likely to be an assault on the gardenesque aesthetic that prevails' (Harrison and Davies, 2002). Verlinden (1980) noted that the spontaneous vegetation of raised sites is mostly ignored because it is considered to be trivial and ephemeral anyway. Nevertheless, he found valuable vegetation types at several hydraulically raised sites in the Port of Antwerp area with distinct moisture, salt and texture gradients. The importance of spontaneous vegetation succession is also increasingly recognized in ecosystem restoration projects of man-made or disturbed habitats (e.g. Rebele, 1992; Box, 1996; Jochimsen, 1996; Prach et al., 2001; Prach and Pyšek, 2001).

In this paper, we examine the potential of spontaneous development, whether or not combined with a cut and removal management, on brackish sediment dikes as an alternative or complementary revalorization measure to afforestation. The possibilities for using sowing mixtures are also considered. Two major questions are addressed: (1) Which plant species and plant communities colonize the new substrate and how do they develop during the first years? (2) What are the driving forces behind this succession? The results are translated into measures to be taken during the design, construction and management phase of a landscape dike in order to enhance its diversity and ecological value.

In addition to ecological and aesthetical values, ecotoxicological risk assessment is prerequisite when using contaminated sediments in landscape development. Although a thorough discussion of this issue is beyond the scope of the present paper, it has important implications for the management of dredged-sediment dikes.

2. Material and methods

2.1. Site description

An experimental dredged-sediment dike of 300 m × 100 m was constructed in the Port of Antwerp on the right bank of the Scheldt. The dike is situated north of the locks of Berendrecht and Zandvliet at a sand raised site known as Magershoek, close to the Belgian–Dutch border. Construction started in 2000 and was finished in the spring of 2001. A sandy retention dike was mechanically filled with brackish dredged material from an adjacent hydraulically filled disposal site that had ripened and settled for about 5–6 years. The dredged sediments originate from the harbor docks at the right bank of the river. The fill

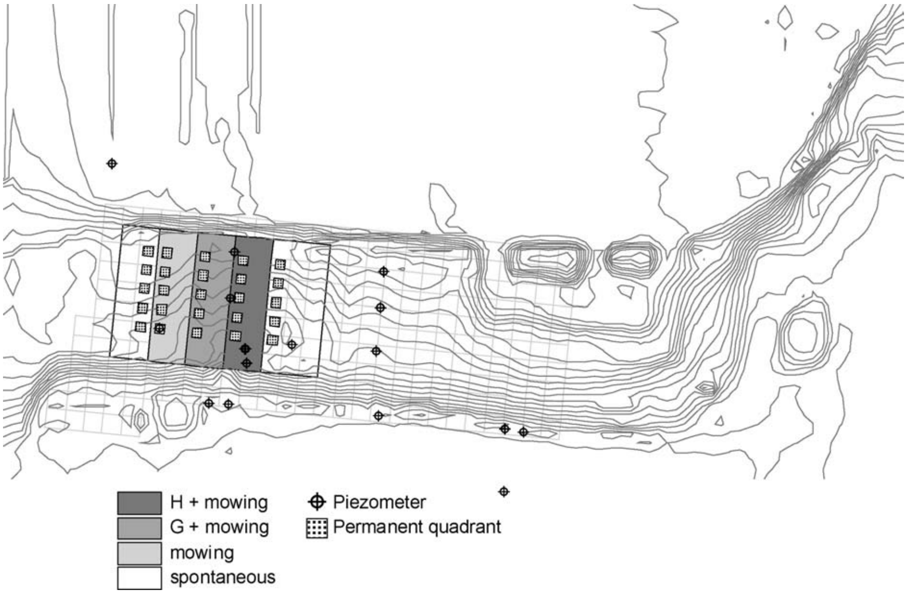

Fig. 1. Schematic overview of the experimental dike. Location of permanent plots is indicated by dotted squares. H: sown with mixture of grasses and herbs and G: sown with mixture of grasses. The eastern part of the dike is used for an afforestation experiment.

depth of the original hydraulic disposal site was about 9 m. No intermediate drainage layers were applied and therefore the dredged material was still poorly ripened when used for the dike. The material was an anoxic gray to black clay with high viscosity. The mean height of the dike is 8 m above ground level, but there is an altitudinal gradient ranging from 9.2 m in the north to 6.5 m in the south, corresponding with an average slope of 3% (Fig. 1). In the southern part there is a topographic depression that is flooded in winter and early spring. This had an important impact on the results and will be further referred to as the southern depression. The base of the dike is about 1 m beneath the average local groundwater table. No sand layers, liners or cover layers of any kind were used.

The general soil properties of the dike were sampled and measured in June 2001 at two depths (0–15 and 30–45 cm). Al, Fe, Mn, K, Mg, Ca, and Na were extracted with $BaCl_2$ (CEC) and subsequently determined using flame atomic absorption spectrometry. The total concentrations of Al, Fe, Mn, Cd, Cu, Pb, and Zn were digested in aqua regia and analyzed with an ICP-AES. Electrical conductivity (EC) was measured potentiometrically in a 1/5 sediment/water solution. Cl was determined in the same solution with an ion-specific electrode. Loss on ignition (LOI) was determined after 4 h calcination at 450 °C. Total nitrogen was determined using the modified method of Kjeldahl (Bremner, 1996). The potential pH (pH-KCl) was determined in a 1/5 sediment/KCl solution with an ion-specific electrode. Soil texture was determined using the pipette method (Labex 8903-11-2-1). The aggregates were destroyed by adding H_2O_2 and HNO_3 and dispersed by adding polyphosphate sodium to separate the individual sediment particles. Available P was extracted with NH_4 acetate and measured colorimetrically.

Groundwater dynamics were monitored in a piezometer network consisting of 10 piezometers at an average total depth of 4 m and a filter length of 2 m. Water levels were measured every 2 weeks using a sounding device with an acoustic and light signal.

2.2. *Experimental design*

Five contiguous zones of 20 m × 50 m were established parallel to the altitudinal gradient and subjected to four treatments (Fig. 1): (1) spontaneous development (two zones east and west of the experimental area), (2) mowing, (3) sowing with grass mixture and mowing and (4) sowing with herb/grass mixture and mowing. Practical constraints obstructed the establishment of a randomized design: accessibility for mowing equipment, delay during the construction phase due to which spontaneous development did not start simultaneously on all locations, and a change in the building plan due to the poor building qualities of the material. Plots are therefore pseudo-replicates (Hurlbert, 1984), which means that the differences between management types are only indicative and cannot be formally inferred by statistical hypothesis testing. The fact that spontaneous development did not start at the same time in all plots eventually did not affect the results: all first year records clustered together in our analyses.

In each of the five contiguous zones, permanent plots of 5 m × 5 m were established (Fig. 1). The distance between the plots within each zone was 5 m. Records were made according to the Braun–Blanquet method using the cover scale of Barkman et al. (1964). Recording commenced at the start of August 2001 and was repeated in mid-June and mid-September 2002 and 2003, and in mid-June 2004. The permanent plots in the eastern spontaneous parts were not recorded in June 2002 and September 2003.

Both non-commercial seed mixtures consisted of species naturally occurring on clay-rich soils, salt-tolerant species and species with an attractive flowering aspect. The exact composition is given in Table 1. The mixture mostly contained late successional species of the *Arrhenatherion elatioris*, which we considered to

Table 1
Composition of seed mixtures (as weight percentage) and overview of presence and characteristic coverage of each species in the different recording campaigns

		A2001	S2002	A2002	S2003	A2003	S2004
Grasses (100%)							
Agrostis stolonifera	20%	II	V^{II}	V^{II}	V^{II}	IV^{II}	III^{I}
Agrostis capillaris	10%						
Festuca rubra	20%	III	III	V	V^{I}	V^{+}	IV
Festuca pratensis	10%		IV^{I}	II	V^{I}	IV	V
Poa pratensis	10%						
Cynosurus cristatus	10%		IV	I	IV		IV
Arrhenatherum elatius	10%		IV	IV^{I}	IV^{II}	IV^{+}	IV^{III}
Puccinellia distans	10%	IV	V^{+}	V^{+}	III	I	I
Grasses (80%)–herbs (20%)							
Agrostis stolonifera *Agrostis capillaris*	16.25% 11.25%		V	V^{I}	V^{+}	V^{+}	I^{III}
Festuca rubra	11.25%	II		II	IV	III	III
Festuca pratensis	11.25%		IV	I	IV	I	V
Cynosurus cristatus	11.25%		IV		III		II
Poa pratensis	6.25%						
Trisetum flavescens	6.25%						II
Arrhenatherum elatius	6.25%		IV	IV^{+}	V	V^{+}	IV^{III}
Leucanthemum vulgare	3%				I		I
Anthriscus sylvestris	1%						I
Daucus carota	1%	IV		IV	II	III	II
Heracleum sphondylium	1%					I	II
Pastinaca sativa	2%						
Centaurea jacea	3%		II	I	IV	IV	V
Tragopogon pratensis	1%						
Hypochaeris radicata	1%		I			I	
Cichorium intybus	2%	IV	III	II	IV	II	I
Leontodon autumnalis	1%						
Achillea millefolium	1%						
Tanacetum vulgare	1%						
Potentilla anserine	2%						

Presence classes correspond with number of sown plots in which the species was found (maximum 5); characteristic coverage classes are given in superscript (no symbol: <5; +:≤10; I: ≤20; II: ≤40; III: ≤60); A, autumn; S, spring.

be a target community for a nutrient-rich clay dike. The mixtures were sown by hand after mechanical plowing and harrowing of the soil at the start of May 2001. A long drought period after the sowing may have negatively influenced the establishment of certain species. Plots sown with just grasses will be referred to as 'G plots' in the text, and plots sown with herbs and grasses as 'H plots'.

Mowing took place once in 2001 (late summer), once in 2002 (late summer) and twice in 2003 (early and late summer) using a tractor with a mowing beam. Cuttings were removed immediately after mowing.

2.3. Data analyses

Unconstrained ordinations were carried out with Canoco 4.5 on species-cover percentage data. The vegetation matrix consisted of 135 records and 126 species. Logarithmic transformation was applied to cover percentage data to avoid extreme influence of dominant species. This is comparable to transforming cover scales into an ordinal scale (e.g. Lepš and Šmilauer, 2003). Detrended correspondence analysis (DCA) was performed with default options (detrending by 26 segments, non-linear rescaling). DCA is an unconstrained ordination, which extracts axes of maximum variation present in the species data only (Lepš and Šmilauer, 2003). The length of the first gradient was 3.45, which means that we are in the twilight zone where both unimodal and linear response models are usually adequate (Lepš and Šmilauer, 2003). Ordination axes were ecologically interpreted by calculating the Spearman rank order correlation between DCA scores of samples and weighted Ellenberg indicator values (EIV) for salinity (S), soil reaction (R), nitrogen (N) and moisture (M). Cover percentage was used as a weighting factor (weighted EIV = ($\sum$EIV × cover percentage)/$\sum$cover percentage). Note that EIV express ecological preferences of plant species on an ordinal scale (1–9 for S, R, N and 1–12 for F; Ellenberg et al., 1992) and should therefore not be interpreted in terms of a specific measurement unit. Many studies have demonstrated the usefulness of indicator values for the interpretation of ordination axes (e.g. Ter Braak and Wiertz, 1994; Lameire et al., 2000; Ejrnaes et al., 2003). Missing values and indifferent species, marked as X by Ellenberg et al. (1992), were ignored.

The following categorical data were used as supplementary variables to further facilitate the ecological interpretation: management (mowing, sowing and spontaneous), year of vegetation record (2001–2004), topography (levels 1–5) and season (spring or autumn). These were all coded as dummy variables. Supplementary variables are passively plotted in ordination space and are not used to calculate the ordination axes. Detailed topographic measurements were made shortly after the construction of the dike. Nevertheless, we chose not to code topography as a continuous variable because the ground level of poorly ripened dredged material can drop by up to several decimeters due to settling phenomena. This settling did not usually occur uniformly across the site and was not continuously monitored. Therefore, coding it as a categorical variable seemed to be appropriate.

In order to partial out the effect of seasonality, a partial DCA was performed with 'season' as a covariable. This had little influence on the ordination and is therefore not discussed further.

A Two Way Indicator Species Analysis (TWINSPAN) was carried out in PC-ORD 4.0 to define the ordination clusters. TWINSPAN is a divisive classification method that first classifies samples and then species based on the sample classification (Hill, 1979). TWINSPAN uses the concept of a pseudo-species to maintain the quantitative information present in the data. A species is subdivided into several pseudo-species if its cover percentage exceeds the so-called cut levels for that pseudo-species. Default cut levels were used (0, 2, 5, 10 and 20%). The method is explained in greater detail by Hill (1979) or Lepš and Šmilauer (2003).

Due to identification problems, *Epilobium tetragonum*, *Epilobium ciliatum* and *Epilobium spec.* were treated as one species (*E. spec.*); *Juncus bufonius* and *Juncus ambiguus* were merged as *J. bufonius*. Taxonomic nomenclature follows Lambinon et al. (1998).

The program ASSOCIA was used for the syntaxonomical identification of the vegetation records (Van Tongeren, 1998). Based on the weirdness and incompleteness indices, the most suitable syntaxonomical unit was withheld from the first three ASSOCIA results. A high weirdness index means that many species atypical for the syntaxonomical unit occur in the record. A high incompleteness index means that many species usually occurring in the syntaxonomical unit are missing. The identification procedure is further explained in Wamelink et al. (2002). ASSOCIA was developed for syntaxonomical identification based on vegetation records in the Netherlands. However, as our study site is very close to the Dutch border there would appear to be no objection to using it here. Syntaxonomical nomenclature follows Schaminée et al. (1996, 1998).

3. Results

3.1. General soil properties

The general soil properties are summarized in Table 2. The dredged sediment is characterized by high pH and very high nutrient levels comparable to those in highly productive sludge-amended or otherwise fertilized agricultural soils (Gough and Marrs, 1990; Rebele, 2001). The available P concentration, in particular, is very high. As the sediments originate from the brackish part of the estuary, Cl and conductivity are also high. Values for Cl and conductivity are significantly lower in the 0–15 cm layer than in the 30–45 cm layer. The soil texture was largely homogeneous, although local mixing with sand from the retention dike did occur.

Compared to natural alluvial soils (e.g. Vandecasteele et al., 2003) there is a distinct enrichment of the soil with heavy metals. Cadmium levels in particular are critical and restrict the possible use of the dredged sediments, due to the Flemish decree on soil sanitation (Vlarebo, 1996).

In summary, the substrate can be characterized as a brackish, neutral to slightly alkaline, hyper-eutrophicated, and moderately polluted clay soil.

3.2. Groundwater dynamics

Fig. 2 shows the time series of the groundwater levels in some representative piezometers. Two different patterns are apparent. The first occurred in piezometers located at the higher topographic levels of the dike and

Table 2
Overview of soil properties (samples taken in June 2001)

Parameter	0–15 cm			30–45 cm		
	n	Mean	S.D.	*n*	Mean	S.D.
Kjeldahl N ($mg\,kg^{-1}$)	25	2106	740	4	2132	1451
P available ($mg\,kg^{-1}$)	29	138	52	5	106	71
pH-KCl	29	7.6	0.1	28	7.7	0.1
Conductivity* ($mS\,m^{-1}$)	29	147	78	28	180	85
Cl* ($mg\,kg^{-1}$)	29	808	675	28	1439	962
LOI (%)	29	8.5	2.5	4	8.3	5.2
Total Al ($mg\,kg^{-1}$)	29	24185	8732	14	24434	13287
Total Fe ($mg\,kg^{-1}$)	29	27522	4741	14	25718	8042
Total Mn ($mg\,kg^{-1}$)	29	585	198	14	483	279
Total Cd ($mg\,kg^{-1}$)	29	8.3	2.8	14	6.4	3.8
Total Cu ($mg\,kg^{-1}$)	29	73	31	14	65	41
Total Pb ($mg\,kg^{-1}$)	29	102	31	14	83	46
Total Zn ($mg\,kg^{-1}$)	29	448	168	14	365	214
Al CEC ($mg\,kg^{-1}$)	29	(<160)	–	28	(<160)	–
Fe CEC ($mg\,kg^{-1}$)	29	(<110)	–	28	(<110)	–
Mn CEC ($mg\,kg^{-1}$)	29	(<90)	–	28	(<90)	–
K CEC ($mg\,kg^{-1}$)	24	463	138	4	536	314
Mg CEC ($mg\,kg^{-1}$)	24	992	376	4	1108	698
Ca CEC ($mg\,kg^{-1}$)	24	4114	1196	4	3437	1889
Na* CEC ($mg\,kg^{-1}$)	24	1022	745	4	1961	1364
<2 μm (%)	10	32.6	5.15	–	–	–
50 μm< x <2 μm (%)	10	35.7	8.4	–	–	–
>50 μm (%)	10	31.7	12.5	–	–	–

Measurements of P available did not meet laboratory standards but are indicative. *n*, number of samples; S.D., standard deviation; (<detection limit). Parameters marked with an asterisk are significantly different between the two sampled layers (at $p = 0.05$).

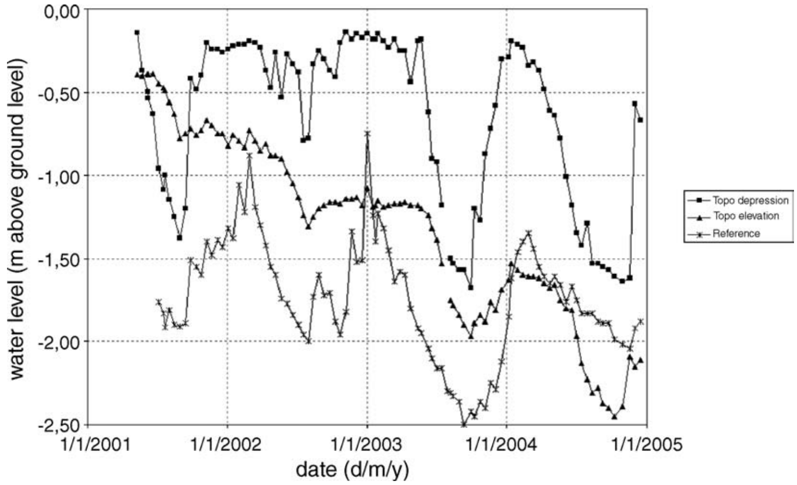

Fig. 2. Groundwater dynamics of piezometers on the experimental dike: type 1 piezometer from topographic elevation, type 2 piezometer from topographic depression and reference piezometer from raised surroundings of the dike.

is characterized by a stepwise decrease of the water table. In late spring and summer when evapotranspiration by the lush vegetation is high, the water level starts dropping. In autumn and winter, loss by evapotranspiration is low and the groundwater table stabilizes until the next growing season.

The second pattern was seen in the piezometers located in the southern depression. The water level drops during the same periods as in the type 1 piezometers. After the growing season, however, there is a rapid rise back to the original level just below the surface, where it stabilizes during winter and early spring. This results in flattened peaks in the time series. For comparison the figure also shows a seasonal pattern of a piezometer in the sandy raised area around the test dike: the water rises and falls more gently, resulting in sharper peaks in the time series.

3.3. Ordination of vegetation records

Fig. 3 shows a sample plot of the DCA with the first two ordination axes (DCA1 and DCA2). DCA1

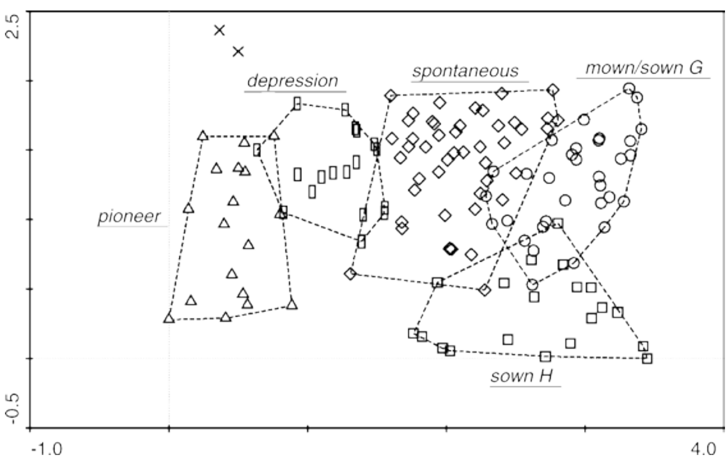

Fig. 3. DCA sample plot of the first two ordination axes. Meaningful TWINSPAN groups are superimposed on the diagram. Eight of 126 plots were removed from the diagram for clarity. See text for further discussion.

explains 11% of the species variability and DCA2 explains an additional 5.7%.

Five more or less distinct clusters can be distinguished. The cluster ‘pioneer’ contains all but two first year records, i.e. the pioneer vegetation in a strict sense. The sown plots all lie in the lower half of this cluster. Cluster ‘depression’ mainly contains the records from the southern depression in 2002, regardless of the management zone (spontaneous, mown or sown) in which they were located. According to the TWINSPAN analysis, the two outliers at the top left of the cluster also belong to this group. They are treated separately for floristic reasons (see further). ‘Spontaneous’ contains the records from plots with zero management, as well as the southern depression plots from 2003 onwards. ‘Mown/sown G’ contains the records from mown but unsown plots and most G plots. The fifth cluster ‘sown H’ mainly contains the records from the H plots.

The first ordination axis is strongly correlated with EIV for nitrogen (−0.80), EIV for salt (−0.67) and to a lesser extent with mowing (0.65) (Fig. 4; Table 3). This axis is also well correlated with the year of the records

Table 3
Inter-set correlation of supplementary variables with DCA1 and DCA2

	DCA1	DCA2
N	−0.80	0.01
M	−0.38	−0.38
S	−0.67	−0.02
R	−0.60	−0.33
Year	0.69	0.14
Topography	−0.26	0.36
Season	0.46	0.20
Spontaneous	−0.41	0.39
Mowing	0.65	−0.23
Sowing G	0.13	−0.09
Sowing HG	0.01	−0.70

Correlations for EIV, year and topography are Spearman rank order correlations. N, M, S and R: Ellenberg indicator values for nitrogen, moisture, salt and acidity.

(0.69) and therefore represents a distinct successional temporal gradient.

Table 4 gives the average weighted EIV per ordination cluster. Nitrogen indicator values were highest during the first year. In the following years, distinct differences were noted between the groups. Relatively

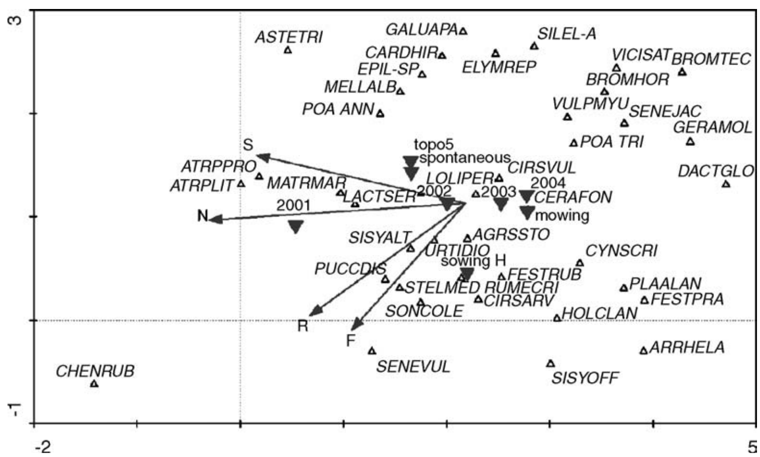

Fig. 4. DCA joint plot of species and supplementary variables (first two ordination axes). Only a selection of species and variables is shown. See text for further discussion (ASTETRI: *Aster tripolium*; AGRSSTO: *Agrostis stolonifera*; ARRHELA: *Arrhenatherum elatius*; ATRPLIT: *Atriplex littoralis*; ATRPPRO: *Atriplex prostrata*; BROMHOR: *Bromus hordeaceus*; BROMTEC: *Bromus tectorum*; CARDHIR: *Cardamine hirsuta*; CERAFON: *Cerastium fontanum*; CHENRUB: *Chenopodium rubrum*; CIRSARV: *Cirsium arvense*; CIRSVUL: *Cirsium vulgare*; CYNSCRI: *Cynosurus cristatus*; DACTGLO: *Dactylis glomerata*; ELYMREP: *Elymus repens*; EPIL-SP: *Epilobium spec.*; FESTPRA: *Festuca pratensis*; FESTRUB: *Festuca rubra*; GALUAPA: *Galium aparine*; GERAMOL: *Geranium molle*; HOLCLAN: *Holcus lanatus*; LACTSER: *Lactuca serriola*; LOLIPER: *Lolium perenne*; MATRMAR: *Matricaria maritima*; MELLALB: *Melilotus alba*; PLAALAN: *Plantago lanceolata*; POAANN: *Poa annua*; POATRI: *Poa trivialis*; PUCCDIS: *Puccinellia distans*; RUMECRI: *Rumex crispus*; SENEJAC: *Senecio jacobaea*; SENEVUL: *Senecio vulgaris*; SILEL-A: *Silene latifolia*; SISYALT: *Sisymbrium altissimum*; SISYOFF: *Sisymbrium officinale*; SONCOLE: *Sonchus oleraceus*; STELMED: *Stellaria media*; URTIDIO: *Urtica dioica*; VICISAT: *Vicia sativa*; VULPMYU: *Vulpia myuros*). Nominal variables are indicated by a grey triangle.

Table 4
Synoptic table of species composition and EIV of the ordination clusters

Cluster	2001	Depression	Spontaneous	Sown H	Mown/sown G	Outliers
Number of records	18	17	46	20	32	2
Average number of species	14.2	16	22.4	23.8	24	6
Ellenberg indicator values						
Moisture (M)	5.9	5.8	5.4	5.9	5.2	6.3
Acidity (R)	7.0	7.0	6.7	7.0	6.5	7.0
Nitrogen (N)	8.3	7.5	6.7	6.1	5.9	7.1
Salt (S)	2.5	3.9	1.9	0.8	0.3	7.4
Agrostis stolonifera	I	III	V^{+}	IV^{I}	IV^{II}	III
Anthriscus sylvestris				I		
Arrhenatherum elatius		I	II	V^{I}	IV^{I}	
Artemisia vulgaris	I		II		I	
Aster tripolium	V^{+}	V^{II}	IV^{+}	I	I	V^{V}
Atriplex littoralis	V	V	III	I	I	
Atriplex prostrata	V^{III}	V^{II}	V	III	I	V
Berteroa incana			I	I	III	
Bromus hordeaceus	I	I	II^{+}	III	III	
Bromus sterilis		I				
Bromus tectorum			II	II	III	
Calamagrostis epigejos		I				
Capsella bursa-pastoris		I	II	I	I	
Cardamine hirsuta		II	II^{+}	I	II	
Centaurea jacea			I	IV	I	
Cerastium fontanum s.l.		II	I	II	III	
Chenopodium album	IV			III		
Chenopodium ficifolium	III		I	II	I	
Chenopodium glaucum	III			I		
Chenopodium rubrum	V^{+}	I		I		
Cichorium intybus	II		I	III		
Cirsium arvense	III	III^{+}	V^{+}	V^{II}	V^{I}	
Cirsium vulgare		II	V	V	V	
Crepis biennis			I	II		
Crepis capillaris	I		I	I	II	
Cynosurus cristatus			II	I	II	
Dactylis glomerata			I	III	IV	
Daucus carota	II	I	I	III		
Elymus repens	III	III	V^{II}	II^{I}	V^{II}	
Epilobium hirsutum		II	I			
Erigeron canadensis		I	II		I	
Festuca pratensis			I	IV	III^{+}	
Festuca rubra	II	I	I	III	III^{+}	
Galium aparine		I	III^{+}		I	
Geranium molle			I	I	IV	
Heracleum sphondylium			I	I	I	
Holcus lanatus	I	II	IV	V^{III}	V^{+}	
Hypochaeris radicata			I	I	I	
Juncus bufonius	I	I	I			V
Lactuca serriola	II	IV	IV	I	I	
Leucanthemum vulgare				I		
Lolium multiflorum		I	I	I		
Lolium perenne		II	II	III	III^{+}	
Matricaria maritima	V^{II}	V^{II}	IV^{II}	V^{+}	IV	V
Medicago lupulina	I	I	I	I	II	

Table 4 (*Continued*)

Cluster	2001	Depression	Spontaneous	Sown H	Mown/sown G	Outliers
Melilotus alba	III	I	II^{I}	I	II	
Oenothera parviflora	I	I	I		I	
Plantago lanceolata	I	I	II	V	V	
Poa annua	I	II	II	I	I	III
Poa trivialis		II	IV^{+}	III^{+}	III^{I}	
Polygonum amphibium				I		
Polygonum aviculare	I	II	I		I	
Polygonum lapathifolium	IV			I		
Polygonum persicaria	II	I	I	I	I	
Puccinellia distans	II	II	II	I^{+}	I	
Rumex crispus	II	V	IV	V^{+}	V	
Rumex palustris	I	I	I			
Scirpus maritimus		I				III
Senecio inaequidens	I	II	I			
Senecio jacobaea		I	I	I	IV	
Senecio vulgaris	III	III^{+}	III^{+}	IV	II	
Silene latifolia (subsp. alba)	I		II		III	
Sisymbrium altissimum		I	III	I	I	
Sisymbrium officinale		I	II	III	I	
Sonchus asper	III	I	I			
Sonchus oleraceus	II	III	IV	IV^{+}	III	
Spergularia salina	I	I^{+}	I			
Stellaria media	I	II	II	II	II	
Taraxacum officinale s.s.			I	I	II	
Tragopogon pratensis			I		I	
Trifolium arvense		I	II	I	II	
Trifolium dubium			I	I	II	
Trifolium pratense				II	I	
Trifolium repens			I	I	I	
Triglochin maritima		I				III
Trisetum flavescens				I		
Tussilago farfara			I		I	
Urtica dioica		II^{I}	III^{I}	III	II	
Veronica arvensis			I		II	
Vicia hirsuta		I	I	I	III	
Vicia sativa	I	II	IV	I	V^{+}	
Vicia villosa ssp. villosa					I	
Vulpia myuros		II	III^{+}	III	III^{I}	

Presence classes—no symbol: absent; I: ≤20; II: ≤40; III: ≤60; IV: ≤80; V: ≤100. Characteristic coverage classes are given in superscript (no symbol: <5; +:≤10; I: ≤20; II: ≤40; III: ≤60; IV: ≤80; V: ≤100). Only the sown species and species occurring in more than one plot are shown.

high nitrogen indicator values were recorded in the depression plots and to a lesser extent plots with spontaneous vegetation development. In the mown parts, nitrogen indicator values were low, which was confirmed by the negative correlation between mowing and N (−0.73). Salt indicator values were also high in the first year. Then distinct differences developed, ranging from high salt indicator values in the depression, over intermediate values in the spontaneous parts, to low values in the mown parts.

The second axis is best correlated with the management variable ‘sowing H’ (−0.70) (Table 3) and mainly separates the H plots from the other mown parts. Correlations with other environmental variables are very low. The mown and G plots are not separated in the ordination plot.

Higher order axes were poorly correlated with the environmental variables and contribute little to our ecological understanding of the vegetation records.

3.4. Floristics and syntaxonomy

About 150 plant species were recorded on the dike, 126 of which occurred in the plots and 13 of which were sown (Table 4). Average species richness and floristic composition of the ordination clusters is summarized in Table 4 and visualized in the ordination plot (Fig. 4). The pioneer vegetation (cluster '2001') of the brackish dike was dominated by very tall, salt-tolerant, mostly annual nitrophilous species (*Matricaria maritima*, *Atriplex prostrata*, and to a lesser extent, *Aster tripolium*, *Chenopodium rubrum* and *Atriplex littoralis*). The subcluster consisting of the sown plots is distinguished by the occurrence of several *Chenopodium* species. Syntaxonomically, all pioneer plots had distinct affinities with the communities of the *Atriplicetum littoralis*.

The vegetation records in the cluster 'depression' had an even more halophytic nature: cover percentages of *A. tripolium* strongly increased, *Spergularia salina* and *Atriplex littoralis* locally reached relatively high cover percentages and the cover of *Atriplex prostrata* remained high. Compared to the pioneer vegetation, a shift occurred from the *Atriplicetum littoralis* towards the *Puccinellio-spergularia salinae* (*Asteretea tripolii*). The two outliers at the top left of the ordination plot have the most pronounced halophytic vegetation, which is illustrated by the presence of *Scirpus maritimus* and *Triglochin maritima*.

In the cluster 'spontaneous' *A. tripolium* and *Matricaria maritima* persisted and locally reached high cover percentages. *Atriplex littoralis* and *Atriplex prostrata* were better represented than in the mown parts but nevertheless decreased. *Elymus repens* was the dominant grass. A development towards ruderalized, tall herb vegetation is very distinct in this group. Syntaxonomically, the records of this cluster occupy an intermediate position between the *Atriplicetum littoralis* (mainly in the eastern spontaneous zone), and ruderal communities of the *Artemisietea vulgaris* (more in the western zone). Despite floristic differences, the two spontaneously developing zones cluster together in the TWINSPAN.

In general, mowing resulted in rapid increase of grass cover and rapid disappearance of halophytic pioneers. Unsown mown plots and G plots are not well separated in the ordination plot, yet there were differences in species composition. G plots were characterized by relatively high presence and cover of the sown grasses, whereas in mown plots spontaneously emerging grasses were more important.

A clear floristic difference was noted between the H and G plots. Obviously, the sown herbs were mainly present in the H plots. *Cichorium intybus*, *Centaurea jacea* and *Daucus carota* were best represented and the latter two were clearly expanding. The other sown herbs occurred either accidentally or not at all. Although the composition of the grass mixture in H and G plots was very similar, a difference in the abundance of grasses was recorded. In the G plots *Agrostis stolonifera* and especially *E. repens* were the dominant species. In the H plots *Holcus lanatus* was by far the most dominant species. *Puccinellia distans* was well represented in 2002 but had largely disappeared in 2003. This was undoubtedly a consequence of desalination of the soil and closing of the vegetation coat. The syntaxonomical position of the sown records is ambiguous, which is reflected by high weirdness and incompleteness indices in the ASSOCIA analysis. Closest affinities were with the ruderal vegetation of the *Artemisietea* (*Dauco-Melilotion* and *Erigeronto-Lactucetum*), but the increase of typical grassland species indicated a development towards the *Arrhenatherion elatioris*.

Dredged disposal sites are notorious for the explosive growth of common nettle (*Urtica dioica*) and creeping thistle (*Cirsium arvense*) due to the hypereutrophic soil conditions. Highest cover percentages of thistle occurred in the sown parts, and lowest in the mown parts. In the wet and saline southern depression, cover of *Cirsium* remained low during the first years. Establishment of *U. dioica* was delayed until the second year. It remained largely restricted to the spontaneous parts, where it occurred in several dense growth centers with high cover by the start of 2004. Under a mowing regime, *Urtica* remained inconspicuous.

4. Discussion

4.1. Environmental gradients

The first ordination axis was negatively correlated with N indicator value and positively correlated with mowing regime. This should not be interpreted as a consequence of the removal of nutrients by mowing. Cutting management in productive grasslands will remove about 200 kg N ha^{-1} (e.g. Bakker, 1989; Oomes, 1990), which is less than 1% of the total amount of nitrogen in the top meter of the soil. Atmospheric deposition amounted to 44.2 kg ha^{-1} in Flanders in 2002 (Mira-T, 2003) and reduced the effect of N removal.

Rather than depleting nutrient amounts, cutting and removal influenced litter accumulation on the soil and the light climate for plants. Spontaneous development without mowing resulted in a dense suffocating mass of litter that created optimal conditions for nitrophilous pioneers. *E. repens*, the dominant grass in the spontaneous parts after a few years, also has high biomass and litter production and similarly reduces the light availability at ground level (Hansson and Fogelfors, 1998).

The salt gradient expressed by the first ordination axis was partly determined by topography and applied management. The most obvious difference was between the southern depression, where the high salt indicator values of the pioneer vegetation persisted and even increased, and the other parts of the dike. The lowest values were encountered in the mown and sown plots. Mechanical soil harrowing prior to sowing probably improved the upper soil structure and led to faster leaching of the salts.

The development of the salt gradient is explained by the groundwater dynamics of the dike. Hydraulic conductivity is generally low in clay soils. Replenishment of the groundwater table by precipitation was therefore small, which led to a general decreasing trend of the groundwater table in the dike (first type of piezometer time series). The precipitation surplus during low evapotranspiration periods was instead rapidly evacuated from the top of the dike to the topographic depressions via the thin permeable upper layer and many soil cracks, which are very typical for drying dredged sediments (USACE, 1987). Consequently, salts were leached out from the upper layers and washed to the depressions. The low permeability of the deeper soil layers is also demonstrated by the salt concentrations, which were significantly higher in the layer from 30 to 45 cm compared to the top layer (Table 2). Rapid evacuation of precipitation to the depressions leads to the second type of piezometer time series.

Although the southern depression is waterlogged during several months in winter and early spring, the correlation between the variables topography (i.e. topo5 for the depression) and moisture value is low (0.16). Ellenberg's moisture value is obviously not a good indicator of periodical flood. Schaffers and Sýkora (2000) found it best indicated the average lowest moisture level in summer. This is low in both depression (type 2 piezometers) and non-depression (type 1 piezometers) plots.

4.2. Floristics and succession

Total species richness on the experimental site was relatively high compared to sites with similar nutrient availability. Rebele (2001) found 78 species on hypereutrophicated abandoned sewage fields compared to 113 unsown species in our vegetation plots. This high diversity is due to the salt gradient, which allows the establishment of halophytic species that are obviously lacking in the sewage fields.

Fig. 5 summarizes early vegetation succession on brackish sediment dikes and the effects of several management interventions. The pioneer vegetation is closely related to the *Atriplicetum littoralis* but also shows affinities with associations of the *Bidention tripartitae*. The latter alliance is common on wet parts of freshwater disposal sites along the Scheldt.

Development of the spontaneous vegetation after the pioneer stage depends on the salt and water content of the substrate and on the management. Under seasonally wet and high salt conditions, halophytic communities related to the *Puccinellio-Spergularion salinae* develop. Under natural circumstances these are not very persistent due to the intrusion of stoloniferous grasses (Schaminée et al., 1998). How these zones will evolve in the long term is uncertain, but a plausible late successional stage is a vegetation type dominated by *Calamagrostis epigejos* in the dryer or *Phragmites australis* in the wettest parts of the dike. These are the vegetation types encountered on an old nearby disposal site of brackish infrastructural sediment (unpublished data).

From a floristic point of view, the halophytic community in the depressions is the most valuable vegetation type that can be created on a brackish sediment dike because it contains rare species. Landscape dikes should be designed to enhance the directing salinity gradient by intentionally creating topographic depressions. Traditional engineering practices, such as mixing the dredged soil with sand for rapid drainage and desalination (De Vos, 1994), are diametrically opposed to such ecological objectives. Due to its higher water and

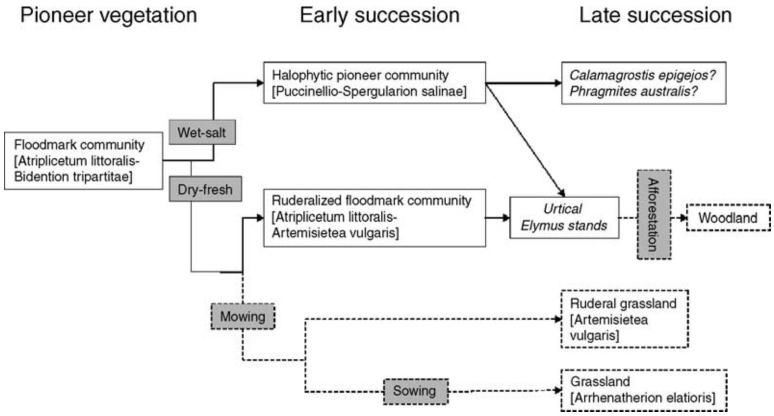

Fig. 5. Schematic overview of succession pathways on brackish dredged-sediment dikes. Late succession vegetation types are hypothetic. Pathways indicated by dotted lines involve an either or not recurrent management intervention.

salt content, poorly ripened dredged material is preferable to well-ripened material. However, it poses many technical problems for the construction of a dike, e.g. realizing the desired height. Technical feasibility and ecological targets should both be carefully assessed before a decision is taken.

Unless specific measurements, such as liners or the compaction of layers, are taken during the construction phase, permanent wet and salt conditions on a topographically raised site, such as a dike, are unlikely to persist in the long term. After a certain delay, desiccation and desalination will start and the depressions will probably follow the succession pathway described below for the dry parts of the dike.

In the high, dry, and faster desalinated parts of the dike the pioneer communities soon start to ruderalize. After an intermediate stage with vegetation related to the *Artemisietea vulgaris*, they will develop into monotonous *Urtica*/*Elymus* or *Cirsium*/*Elymus* stands. This evolution starts after about 3–4 years with zero management. It has been described by many authors on highly productive soils (Hansson and Fogelfors, 1998; Rebele, 2001; Kleijn, 2003). Hansson and Fogelfors (1998) described the intermediate *Artemisietea* related stage as 'neglected grassland with low floristic value'.

Several management choices can be made to avoid trivializing of the vegetation: cut and removal regime, whether or not preceded by sowing, or afforestation of the site.

Mowing and cut removal leads to a type of productive grassland with many ruderal elements of the Artemisietea. At best a sustained mowing/removal management will result in a form of *Arrhenatherion* grassland, which we consider to be a target community; at worst in *Elymus*-dominated grassland. The effects of mowing are very similar to those described for nutrient-rich, abandoned arable fields. There a mowing regime first leads to *Elymus* dominance and later to meadowland with other dominant grass species. On nutrient-rich soils, at least two cuts a year are necessary to suppress nitrophilous competitors, such as *E. repens* (Hansson and Fogelfors, 1998).

Many attempts to restore grasslands on ex-arable fields fail because early successional species establish vigorously and persistently whereas late successional species remain absent (Kleijn, 2003). This is mostly attributed to the high residual soil fertility and the absence of seed sources of the target species (Walker et al., 2004). Similar causes will slow down or prevent natural establishment of target communities of the *Arrhenatherion* on a dredged-sediment dike. Applying seed mixtures can promote or enable the establishment of late successional species. According to Kleijn (2003), a suitable cover may be achieved by sowing non-competitive grass species immediately after a field has been abandoned. These nurse or crop species mitigate the effect of persistent perennials and enable other species to establish (Hansson and Fogelfors, 1998). A suitable species resistant to the high initial salt concentrations might be *Puccinellia distans*. It was applied in a small percentage in our experiment (10% of the seed mixture) but established well in the first year. As it disappears when the soil desalinates, it might be a good crop species for brackish soils.

A small selection of herbs can be added to enhance the flowering aspect after the flowery pioneers have disappeared. However, sowing mixtures, especially those with herbs, are usually very expensive. Moreover, total species richness was not higher in the sown than in the mown parts after a few years. The use of herb mixtures should therefore be carefully considered and opportunities for establishment of the selected species should be optimal. In our experiment *C. jacea*, *C. intybus* and *D. carota* were the only species capable of competing with the spontaneous herbs.

4.3. Afforestation or not?

Smit and Olff (1998) did not recommend afforestation on hyper-eutrophicated substrates because the natural establishment of woody species on such soils is low. Decleer (1999) on the other hand stated that nutrient status of the top layer of dredged, sediment-derived soils should determine the choice between afforestation and spontaneous development, nutrient-rich soils being better suited for afforestation.

Our results show that the appropriate management choice at least partially depends on the salt and water content of the substrate. In freshwater conditions, waterlogged sediment soils are often rapidly colonized by willows, whereas dry parts develop into monotonous stands of nitrophilous herbs. Elder (*Sambucus nigra*) is probably the only woody species able to establish and thrive in these tall herb vegetations (Vandecasteele et al., 2002). On brackish sediments willows and other pioneer trees remain absent in the pioneer stage due to the high salt levels (Luyssaert et al., 2001). By the time the soil has become desalinated and the germination conditions would be favorable for willows, the unattractive tall herb vegetation has closed and tree establishment is no longer likely. Afforestation is then a good option, especially if the development of amenity areas is intended.

The inability of willows to colonize brackish sediments has a beneficial side effect. Willows are known to accumulate large concentrations of cadmium and zinc in their leaves and hence to increase the risk of bioaccumulation in the ecosystem through their leaves and litter (Vandecasteele et al., 2002, 2004). Several species, such as common ash (*Fraxinus excelsior*) or maple (*Acer pseudoplatanus*), do not accumulate heavy metals and are more appropriate for afforestation (Mertens et al., 2004).

4.4. Undesired species

Common nettle (*U. dioica*) and creeping thistle (*C. arvense*) are undesired species in view of the development of sludge dikes into amenity areas. Moreover, control of creeping thistle is statutory in Belgium (Cornelis and Hermy, 2003). Management of sludge dikes should therefore be aimed at reducing their impact. *Cirsium* obviously takes advantage of the soil disturbance caused by either sowing or mowing damage. Disturbance of the soil should clearly be avoided. Although *Cirsium* reached relatively high cover percentages in the sown parts, it did not really dominate the grassland. Under mowing management it has a low and relatively open growth form, allowing other species to develop. Bakker (1960) found *C. arvense* to be poorly adapted to wet reduced soils, which explains its absence in the wet depression. *Urtica* establishes later and is largely restricted to the spontaneous parts. Unlike to *Cirsium*, it leaves little or no room for other species and tends to become a much bigger problem. Mowing and cut removal proved to be an effective way of controlling this species.

5. Conclusion

Hyper-eutrophic soils derived from dredged sediment are generally considered to have banal and unpleasant spontaneous vegetation. Rapid afforestation is therefore a generally applied revalorization measure. Our results show that, depending on the abiotic conditions, other management options can also result in valuable vegetation types. Zero management is recommended where water and salt levels allow valuable halophytic vegetation. Except for the occurrence of halophytic species in the early stages, succession of brackish dredged-sediment derived soils on dryer parts is remarkably parallel to that of other hyper-eutrophicated sites, such as abandoned arable fields. Many of the management recommendations are similar. Cutting and removal result in ruderal grassland. Rapid sowing is necessary to suppress perennial weeds and allow the establishment of more species-rich target communities related to the *Arrhenatherion*. Introduction of late successional herbs was successful for only a minority of species.

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