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Article

A Preliminary Study on Changes in Macrobenthic Assemblages in the Fenced Experimental Plots for Restoring Tidal Marsh, Hogok-ri Tidal Flat, West Coast of Korea

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Abstract : This preliminary study on the changes of macrobenthic assemblages in experimental sediment fences was conducted as a part of tidal marsh restoration project. Intertidal sediment fences were designed to increase the efficiency of trapping sediments on unvegetated tidal flats in order to raise sediment elevation and to allow colonization of intertidal vegetation. Although increment of soil surface level was not observed over the first three months of the study, it was possible to obtain some effects of the sediment fence. Three months later, the particle sizes of the surface sediment at experimental plots became much finer compared to unfenced areas on the natural mudflats located in the same tide level as that of the plots. The difference was much greater on the plot with drainage canals than on the plot without ones. Species diversity of the experimental plots became much higher than that of natural sites. *Perinereis aibuhitensis* and *Glauconome chinensis* which were absent from initial community appeared with high density in the plot with drainage canals. Those species were significantly different in abundance between the experimental plot and the natural mudflat. Changes in species composition were not detected in another experimental plot without drainage canals.

Key words : restoration, tidal marsh, macrofauna, sediment fence, Hogok-ri tidal flat, west coast of Korea.

1. Introduction

Wetlands are one of the most important ecosystems on Earth, and valuable as sources, sinks, and transformers of a multitude of chemical, biological, and genetic materials (Mitsch and Gosselink 1993). Although the value of wetlands for fish and wildlife protection has been known for several decades, some of the other functions have been identified recently; e.g., function as "the kidneys" for wastes from natural and human sources, and "biological supermarkets" for the extensive food chain and rich biodiversity (Mitsch and Gosselink 1993). These values are now being recognized and translated into wetland protection laws, regulations, and management plans including wetland restoration. Restoration of tidal marshes may be conducted either to mitigate permitted impacts or to compensate and offset cumulative or historical impacts through proactive efforts (Cornelisen 1998). Recently, the need to restore damaged or degraded coastal ecosystems has received world-wide attention. It has forced the Korean government to pay more attention to restoration of wetlands and has caused to develop projects concerning coastal wetland restoration in Korea.

In the case of salt marshes, restoration efforts involving grading of upland soils or dredge spoils to tidal elevation and planting have been under way for decades (Race and Christie 1982; Broome *et al.* 1988; Zedler 1988; Levin *et al.* 1996). However, the information required to assess successful recovery of marsh function is collected only rarely until now. This is especially true for salt marsh infauna. Many of the faunal studies reported to date have been conducted, for example, *Spartina alterniflora* marshes of North Carolina, USA (Cammen 1976; Sacco *et al.* 1987, 1994; Moy and Levin 1991; Levin *et al.* 1996). However, while stands of vegetation have been successfully duplicated, less is understood about the establishment of faunal

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communities in created or restored tidal marshes (Posey *et al.* 1997). More than 13 to 16 years after creation, as Sacco *et al.* (1994) reported, several created marshes in North Carolina had not achieved macrofaunal densities, composition and feeding modes comparable to adjacent natural wetlands. Communities in the nature are composed of a large number of species adapted to unique environment around them. Changes in the community structure depend on a suite of different environmental variables to which each of the species in the community may respond differently (Warwick and Clarke 1991). When artificial marshes are established by grading of soil, sediments are necessarily disturbed and the organisms living there are subsequently affected. The grading might be a catastrophe for marsh organisms.

Soil surface elevation is one of the most important factors controlling the colonization, maintenance or deterioration of intertidal vegetation (Cahoon *et al.* 1995; Scarton *et al.* 2000), because the elevation affects salt concentration in sediments. Salt concentration determines the distribution of most plants in high tidal marsh, because the high concentration prohibits plants from absorbing water and inhibits their growth (Susan *et al.* 1982; Mc Nulty 1985). In case of high tidal marsh, the increment of elevation can diminish the concentration of sediment. If an artificial fence accumulates sediments on unvegetated tidal flats to the extent of which the elevation is equal to that of the vegetated habitats, the saline concentration can decrease due to reduced submergence frequency and the duration of submergence by tide. Therefore, it is possible for marsh plants to extend more seaward by this process, following reestablishment of marsh fauna. Actually, the sediment fences have been used for restoration of salt marsh in some countries such as the Netherlands, USA, Germany and Italy.

Intertidal sediment fences were designed to increase the efficiency of trapping sediments on unvegetated tidal flats, in order to raise elevation and to allow colonization of intertidal vegetation. The study investigates the variations of macrobenthic assemblages during two sampling periods (the interval of three months) as a preliminary study on a succession in fenced experimental plots accumulating the sediments for habitat extension or creation of salt marsh plants without grading soils, and is compared to the assemblages in an adjacent natural marsh.

2. Methods

Experimental design

This study was carried out in two sediment fences

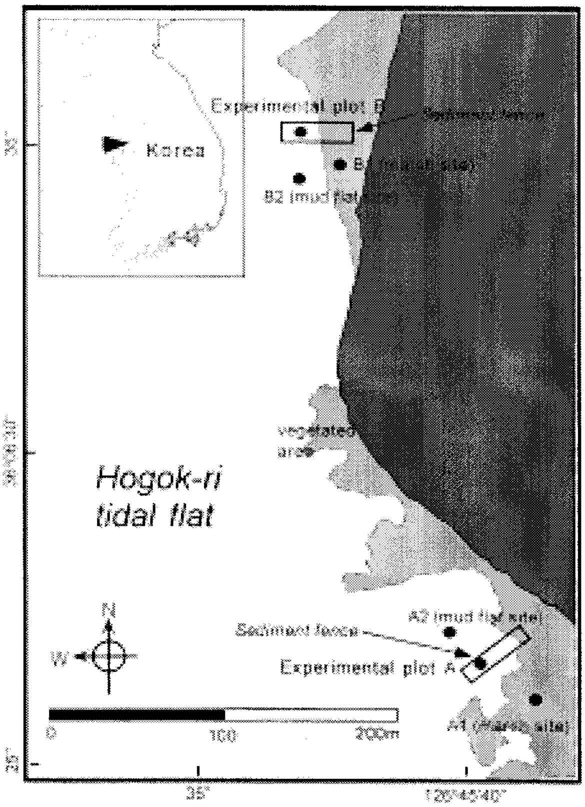

Fig. 1. Location and layout of the natural marshes and experimental study sites in Hogok-ri tidal flat in Namyang Bay, west coast of Korea.

(experimental plots A and B) to extend habitats for salt marsh plants by increasing elevation of substratum and in an adjacent natural marsh and mudflat of high tidal marsh located at Hogok-ri tidal flat in Namyang Bay, west coast of Korea (Fig. 1). Vegetation at the marsh was mostly composed of *Phragmites communis* and *Suaeda japonica*. Two experimental plots were established between 3.17 and 3.42 m msl tidal elevations on the tidal flat in May 2000 to raise substratum elevation by the sediment fence surrounding 400 m^2 area with twigs supported by metal poles of 4 cm diameter (Fig. 2). The elevation of surface sediment of two plot sites (A, B) was the same as that of two mud flat sites (A2, B2), where the elevation was 3.25 m msl. By comparison, the elevation of the marsh sites (A1, B1) was 3.35 m msl. In order to compare deposition rates of sediment between the experimental plot A and B, small drainage canals were made in the plot A, but not in the B.

Faunal sampling and data analyses

Macrofauna were collected by 5 replicate can cores ($0.026\ m^2 \times 5$) at each site. Before establishment of

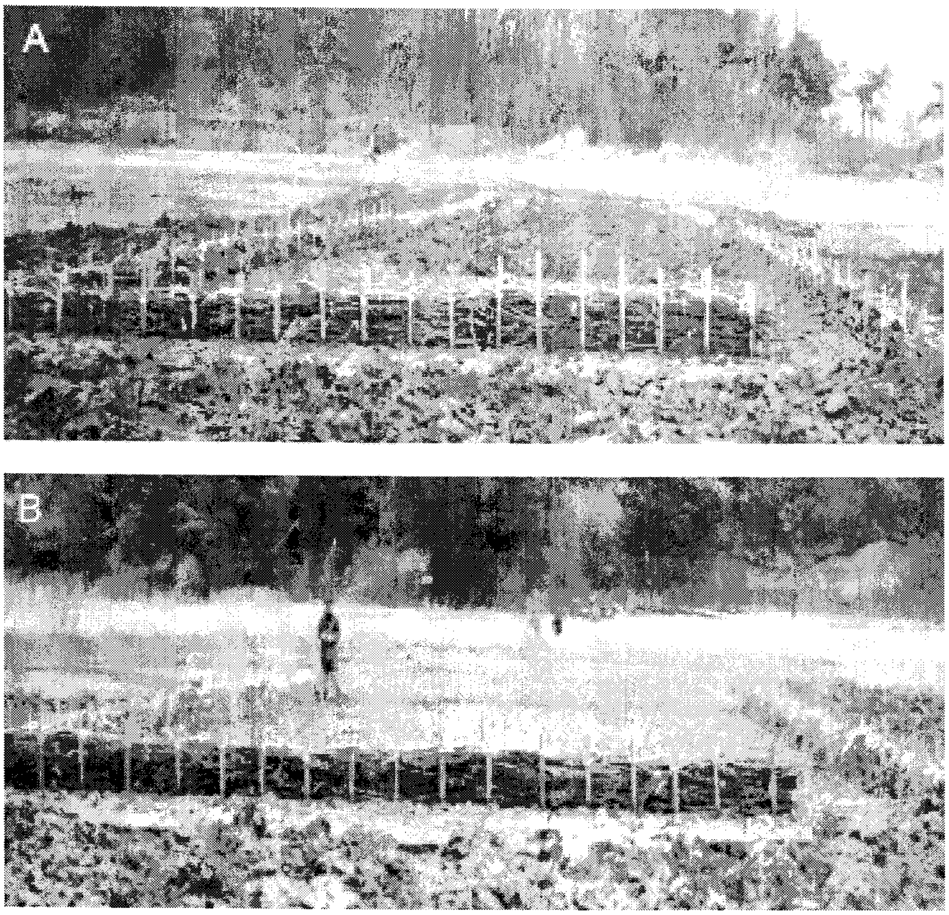

Fig. 2. Photographs of two experimental plots A and B. Photos were taken just after establishing the plots.

experimental plots (April 2000), samples were taken from natural area of two mudflat-sites (A2 and B2) and two marsh-sites (A1 and B1) as reference sites (Fig. 1). Mudflat-sites A2 and B2 were selected under the assumption that both sites A and B had equal conditions to those of A2 and B2, respectively, before establishing experimental plots. Natural marsh-sites, A1 and B1, were selected as reference sites in order to compare macrofauna assemblages with those of the experimental plots (A and B) after the plots were set up. Three months later (July 2000), samples were taken from middle elevations within two experimental plots (A and B) and also from the reference sites (A1, B1, and A2) (Fig. 1). Sampling at B2 was not carried out in July 2000.

Sediment samples were sieved through 0.5 mm mesh screen by washing with seawater, and residuals on the screen were preserved in 10 % neutralized formalin solution. In the laboratory, the samples were sorted into major faunal groups and the animals were identified to the species level if possible and counted.

The significance of differences among sites or seasons for total abundance, species number, species diversity (H') and mean density of dominant species was tested using one-way ANOVA.

3. Results

Sediment characteristics

Three months later, any change in soil surface elevation was not detected.

Before the sediment fence was established (April 2000), the surface sediment of Hogok-ri salt marshes and mudflats consisted mainly of silt and clay. Particle sizes were about the same at both natural sites, where the mean size ranged from 6.77 ϕ to 7.17 ϕ (Fig. 3). The sizes at both experimental plots A and B were also similar to those at natural sites just after establishing the fences (April 2000). Three months later (July 2000), particle

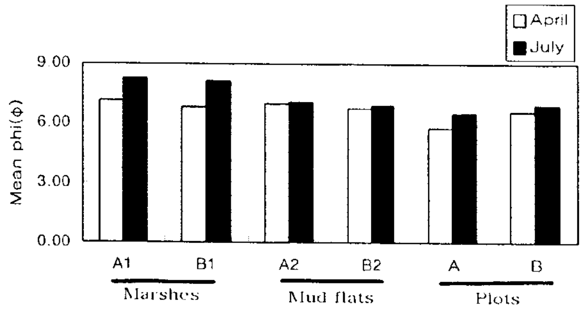

Fig. 3. Variation of particle sizes in the surface sediment at two different seasons from the Hokog-ri study sites.

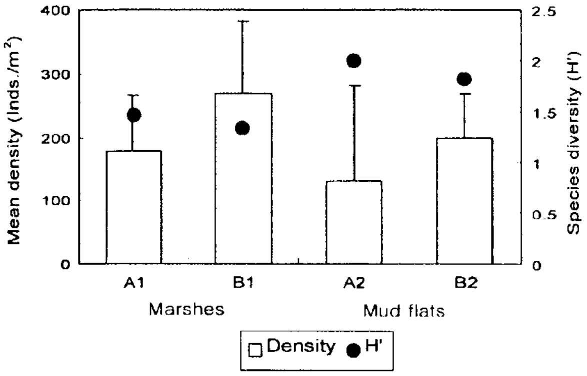

Fig. 4. Mean density and species diversity of macrobenthos occurred on Hogok-ri natural marshes and upper mud flats in April 2000 before setting up experimental plots.

sizes in the surface sediments of natural marshes (A1 and B1) and two experimental plots had become much finer. Sediment fence had effect on particle sizes of surface sediments, especially in the plot A in which small drainage canals were made. The sediments of the natural mudflats (A2 and B2), however, were almost unchanged for those in April 2000 (Fig. 3).

Species composition and abundance of natural marshes and mudflats

In April 2000, the benthic fauna collected on Hogok-ri natural marshes and upper mudflats were composed of a total of 16 species, 9 species of which occurred on the marshes and 14 species on the mudflats. Polychaetes and crustaceans were major faunal groups occupying 75.2 % of the total abundance (Table 1). Crustaceans were the dominant groups at A1 and A2 sites, and polychaetes were at B1 and B2. The trend in species number was different according to habitat types; crustaceans were most abundant at A2 and B2 of mudflat-sites and polychaetes were at A1 and B1 of the marsh-sites (Table 1). The density of benthic invertebrates was higher at the marsh-sites (A1: 174 inds./m^2, B1: 269 inds./m^2) than at mudflat-sites (A2: 131 inds./m^2, B2: 201 inds./m^2), and was relatively higher at B1 and B2 than at A1 and A2 (Fig. 4). Species diversity, however, was higher at the mudflat-sites than at the marsh-sites.

Three months later (July 2000), macrofaunal species sampled at the natural sites was composed of 18 species, 14 species of which occurred on the marshes and 7 species on the mudflats. The most dominant group in terms of total abundance at the marsh and mudflat-sites was changed from polychaetes and crustaceans in April to

Table 1. The faunal group composition of macrobenthos occurred on Hogok-ri natural marshes and upper mud flats in April 2000 before setting up experimental plots. Numbers in parentheses are percentages.

1) Abundance (Inds./m^2)

Sites		Mollusca	Polychaeta	Crustacea	Others	SUM
A1	Marshes	38(21.7)	46(26.1)	92(52.2)	0(0.0)	174
B1		46(17.1)	177(65.7)	23(8.6)	23(8.6)	269
A2	Mud flats	0(0.0)	15(11.8)	108(82.4)	8(5.9)	131
B2		23(11.5)	62(30.8)	62(30.8)	54(30.0)	201
Total		105(13.9)	300(38.6)	285(36.6)	85(10.9)	775

2) No. of species (per 5 replicate cores)

Sites		Mollusca	Polychaeta	Crustacea	Others	SUM
A1	Marshes	1(20.0)	2(40.0)	2(40.0)	0(0.0)	5
B1		1(12.5)	3(37.5)	2(25.0)	2(25.0)	8
A2	Mud flats	0(0.0)	2(22.2)	6(66.7)	1(11.1)	9
B2		1(12.5)	2(25.0)	4(50.0)	1(12.5)	8
Total		2(12.5)	4(25.0)	8(50.0)	2(12.5)	16

Table 2. The faunal group composition of macrobenthos occurred on natural sites and experimental plots in July 2000, three months after experimental plots were established. Numbers in parentheses are percentages.

1) Abundance (Inds./m^2)

Sites		Mollusca	Polychaeta	Crustacea	Others	SUM
A1	Marshes	654(75.2)	100(11.5)	92(10.6)	23(2.7)	869
B1		1,085(84.4)	77(6.0)	100(7.8)	23(1.8)	1,285
A2	Mud flats	69(34.6)	77(38.5)	46(23.1)	8(3.8)	200
B2		-	-	-	-	
A	Experimental plots	177(34.8)	231(45.5)	92(18.2)	8(1.6)	508
B		0(0.0)	92(38.7)	123(51.6)	23(9.7)	238
Total		1,985(64.0)	577(18.6)	453(14.6)	85(2.7)	3,100

2) No. of species (per 5 replicate cores)

Sites		Mollusca	Polychaeta	Crustacea	Others	SUM
A1	Marshes	2(22.0)	2(22.2)	4(44.4)	1(11.0)	9
B1		2(18.2)	3(27.3)	4(36.4)	2(18.2)	11
A2	Mud flats	1(14.3)	2(28.5)	3(42.9)	1(14.3)	7
B2		-	-	-	-	
A	Experimental plots	1(12.5)	2(25.0)	4(50.0)	1(12.5)	8
B		0(0.0)	3(27.3)	6(54.5)	2(18.2)	11
Total		3(14.3)	4(19.0)	11(52.4)	3(14.3)	21

-: no data

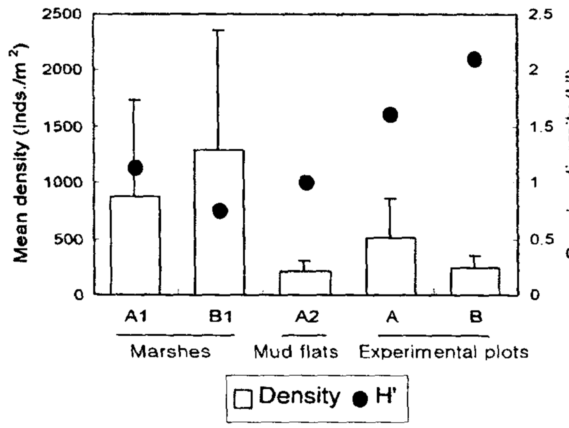

Fig. 5. Mean density and species diversity of macrobenthos occurred on natural marsh sites (A1, B1 and A2) and experimental plots (A, B, C) in July 2000, three months after experimental plots were established.

mollusks in July (Table 2). The crustaceans were major group in terms of the number of species at both marsh and mudflat-sites (Table 2). The mean density of the natural sites in summer which ranged from 869 to 1,285 inds./m^2 on the marshes and 200 inds./m^2 on the mudflat, was much higher than that in spring (Fig. 5). Especially, the density of a bivalve, *G. chinensis*, ranged from 850 inds./m^2 at the marshes to 70 inds./m^2 at the mudflats. The

Table 3. One-way ANOVA (F ratio values) of values obtained for inter-season (April and July) univariate analysis from natural sites (1 and 8 degree of freedom).

Sites		H'	Total ind.	Total spp.
A1	Marshes	ns	ns	46.29**
B1		ns	ns	6.45*
A2	Mud flats	ns	ns	ns

*P<0.5; **P<0.01; ns, not significant.

results of a one-way test of variances obtained for inter-season univariate analysis are shown in Table 3. There was no distinct difference between spring and summer in macrofaunal species diversity and total abundance ($p > 0.05$). However, the number of species of both marsh-sites in summer was significantly more than that of spring (A1-$p<0.01$; B1-$p<0.05$), but not significant for the mud-flat sites ($p>0.05$).

Before the plots were established, there was a distinct distribution pattern of dominant species according to habitat types at Hogok-ri coastal area. The most dominant species was *Perinereis aibuhitensis*, a polychaete, with the mean density of 52 inds./m^2. This species occurred only in the marshes, and its extremely high population density was found at station B1 (162 inds./m^2) of marsh areas. The next abundant species was a decapod species, *Cleistostoma dilatatum*, which inhibited in the habitats of

Table 4. Comparison of mean density of dominant species in natural sites and experimental plots before (April) and after (July) establishment of the plots.

	Dominant species	Mean density(Inds./0.13 m^2)				
Plot A	*Glauconome chinensis* (M)	0	3±2.7	23±10.4	5±7.1	81±72.3
	Perinereis aibuhitensis (P)	0	2±2.7	10±3.5	5±5.0	8±2.7
	Heteromastus filiformis (P)	6±2.2	5±3.5	20±15.8	0	10±7.1
	Macrophthalmus japonicus (C)	4±4.2	4±4.2	4±5.5	0	0
	Cleistostoma dilatatum (C)	6±4.2	0	5±5.0	8±5.7	2±2.7
	Sites ↑	A2		A	A1	
		April	*July*	*July*	*April*	*July*
		Mud flat		*Plot*	*Marsh*	
	Sites ↓	B2		B	B1	
Plot B	*Glauconome chinensis* (M)	0	No data	0	6±5.5	140±68.6
	Perinereis aibuhitensis (P)	0		4±4.2	21±18.8	8±8.4
	Heteromastus filiformis (P)	7±6.7		7±5.7	1±2.2	1±2.2
	Macrophthalmus japonicus (C)	4±2.2		0	1±2.2	0
	Cleistostoma dilatatum (C)	1±2.2		7±2.7	2±2.7	7±4.5

Shading legend: 0; 1 to 4; 5 to 9; 10 to 19; 20 to 49; above 50

(M: Mollusca P: Polychaeta, C: Crustacea)

Table 5. Results of one-way ANOVA (F ratio values) test for differences among the mean density of dominant species for experimental plots (A and B), natural marshes (A1 and B1) and mud flat (A2) in July 2000. 1 and 8 degree of freedom.

Dominant species	A-A2	A-A1	B-B1
Glauconome chinensis (M)	**17.35***	3.15 ns	**20.86****
Perinereis aibuhitensis (P)	**44.08****	2.67 ns	0.93 ns
Heteromastus filiformis (P)	4.28 ns	1.21 ns	4.79 ns
Macrophthalmus japonicus (C)	0.00 ns	2.67 ns	1.00 ns
Cleistostoma dilatatum (C)	4.97 ns	1.40 ns	0.00 ns

*P<0.005; **P<0.001; ns, not significant.

both marsh and mudflat with the mean density of 30.8 inds./m^2. The third abundance species was *Glauconome chinensis*, a bivalve, also occurred only in the marshes with the highest density of 46.2 inds./m^2 in B1 site. In addition, *Heteromastus filiformis*, a polychaete, and *Macrophthalmus japonicus*, a decapod, were also dominant species and occurred only in the mudflat-sites.

Macrobenthic assemblages at the experimental plots

After three months, both experimental plots A and B, as transition areas between marshes and mudflats, exhibited medium values of macrofaunal mean density (Fig. 5). However, a total species was similar to that observed in the natural sites (Table 2). Species diversity was about 1.6 in plot A and 2.1 in plot B, which was much higher than that of natural sites (Fig. 5). The mean density of dominant species in the experimental plots was compared with that in marshes and mudflats (Table 4). Although the density of dominant species in marsh-sites and mudflat-sites under the natural condition was more or less fluctuating with seasons, the distribution pattern of these species differentiated according to habitat types was relatively constant, i.e., *P. aibuhitensis* and *G. chinensis* were dominant in the marsh-sites, *H. filiformis* and *M.*

japonicus in the mudflats and *C. dilatatum* in both sites.

Three months after the plots were established, there was temporal variability in the benthic community at the plot A. Five species being abundant in natural sites concurrently occupied on the plot A. *G. chinensis* and *P. aibuhitensis* were absent from the community at plot A in April (under the assumption that both sites A and B have equal conditions to those of sites A2 and B2, respectively, before establishing the experimental plots) but showed mean densities of 23 inds./0.13 m^2 and 10 inds./0.13 m^2 in July, respectively. *H. filiformis* had higher abundances at the plot A during summer compared to the preceding April. *M. japonicus* and *C. dilatatum* showed constant densities. However, the dominant species was not yet changed at the plot B (Table 4). Table 5 is the result of ANOVA test for differences among the mean density of dominant species for the experimental plots and natural sites in July. Two numerically dominant species sampled in July exhibited significant differences in abundance between experimental plots and natural mudflat-sites. Mean densities of *G. chinensis* and *P. aibuhitensis* were significantly different between the plot A and site A2 of mudflat but did not show significant differences between the plot A and marsh-site A1. *G. chinensis* was also significantly different in abundance between plot B and marsh-site B1. Neither *H. filiformis*, *C. dilatatum* nor *M. japonicus* exhibited significant differences in abundance between the sites in July 2000.

4. Discussion

Macrofauna at natural sites

The macrofaunal abundance in Hogok-ri tidal marsh was more or less different from those in other salt marshes of Korea. The benthic invertebrates sampled in natural marshes during two sampling periods included a total of 33 species. This value was lower than that of other tidal marshes. For example, the benthic community of Tonggom salt marsh in Ganghwa Island had 44 species (Lee 1999). The mean density of benthic fauna in natural sites, which was 447 inds./m^2, was also low compared with that of Tonggom marsh (671 inds./0.25 m^2). However, the composition of dominant species was in good agreement with that of Tonggom marsh. *P. aibuhitensis*, *G. chinensis*, *C. dilatatum* and *M. japonicus* were abundant in both areas. The difference in the composition between two areas was a small gastropod, *Assimenea lutea*, i.e., not abundant in Hogok-ri marsh but dominant in Tonggom marsh. There were only little changes in macrofaunal composition and abundance on mudflat-sites between spring and summer. Abundance of macrofauna, however, became five times higher mainly due to increment of the density of *G. chinensis* on natural marsh-sites in July. Other species did not show any clear seasonal variation in abundance but this species markedly exhibited increases in density. The abundance of *G. chinensis* mounted up as 81±72 inds./0.13 m^2 at A1 marsh-site and 140±69 inds./0.13 m^2 at B1 marsh-site in July. However, this species hardly occurred on the mudflat-sites in both spring and summer. This species was supposed to be recruited in late spring (Lee 1999).

Some effects of sediment fence in experimental plots

It takes much time to restore ecosystem of natural salt marsh by grading upland soils or dredging spoil and planting marsh plants, since the deposition necessarily brings about severe environmental disturbance. These systems typically resemble heavily disturbed habitats immediately after their creation because they contain few inhabitants and substrates may differ from natural condition (Levin *et al.* 1996). This study was a small-scale experiment to restore ecosystem of salt marshes using sediment fences to increase substratum level comparable to one of the marsh habitat without grading of soils. In the Netherlands, one of the first countries in which this technique was adopted, sediment fences were built in the last 30 years, and accretion rates of up to 3.5 cm/year have been measured following colonization by plants (Kamps 1962; Bouwersma *et al.* 1986). Fences have been used in Germany (Lieberman *et al.* 1997), in Louisiana (Boumans *et al.* 1997) and in Italy (Scarton *et al.* 2000) as well. Scarton *et al.* (2000) reported, 28 months after the fences were established, 5.7 cm of sediment was accumulated on the intertidal flat in the lagoon of Venice (2.5 cm/year). Although the increment of such level was not observed over the first three months of the study period, we could obtain some effects of the sediment fences. The particle size of surface sediments at the experimental plots became much finer when compared with unfenced area on natural mudflat located in the same tidal level as that of the experimental plots (Table 1). It was believed that the enhancement of fine sediment deposition was a result of decreased wave energy which increased sediment deposition, reduced resuspension, and enhanced consolidation of the surface sediments (Boumans *et al.* 1997). However, the result of the changes in grain size was not in agreement with that reported by Scarton *et al.* (2000) for Venice Lagoon in Italy. They informed that the grain size did not show any significant differences between protected areas by the fence and unprotected one after 28 months. Through our results that the degree was much larger on the plot A with the drainage canal than on the plot B without one, it was considered that physical energy of tidal wave was rather weakened by the drainage canals, but more data might be needed to demonstrate this conclusively.

There are some studies in which variations of sediment characteristics and composition of plant communities give rise to changes in macrobenthic community of salt marsh (Phleger and Bradshaw 1966; Vince *et al.* 1981; Levin *et al.* 1996). Although we analysed macrofaunal data during two seasons, there were some changes in macrobenthic assemblages in the experimental plot A. With the effects of sediment fence, the experimental plot became a mixed habitat with the characteristics of both marsh and mudflat after three months. The species diversity of the plot became much higher than that of natural sites because most of species inhabited in marsh and mudflat concurrently occurred in the plot. In particular, *P. aibuhitensis* and *G. chinensis* which were absent from the community of mudflat appeared with high density in the plot A in July (Table 4). The emergence was considered due to the changes of habitat environments such as particle sizes and vegetation. The nereid polychaete *P. aibuhitensis* occurs commonly in the Yellow Sea (Lee *et al.* 1992; Lee and Choi 1992). It has been found only in the intertidal zone, mostly from the upper intertidal zone. Lee and Choi (1992) reported that this species was recruited from July to September on the upper mudflat in the western coast of Korea. The species occupied in the experimental plot A in July, showing significantly different in abundance between the plot A and the mudflat-site A2 (Table 5). With the result that the species was not recruited on mudflat area in July, it was suggested that *P. aibuhitensis* selected the experimental plot A as a preference habitat. Although *G. chinensis* showed maximum density in natural marsh-site, the density was higher in the plot than the mudflat-site. There was significant difference between the plot A and the mudflat-site A2, as well. In the experimental plot B, however, the composition of macrobenthic fauna did not yet changed when compared with initial community (Table 4). Dominant species did not show changes in abundance. Unlike the plot A, *P. aibuhitensis* and *G. chinensis* also had similar density to that of initial community. Especially, the abundance of *G. chinensis* was significantly different between the plot B and the marsh-site B1. Although it is difficult to conclude what brings about the differences in colonization patterns between the plot A and the plot B, the drainage canal is thought to be an important factor.

This preliminary study was investigated for two sampling periods of first three months, so that the data may be insufficient for detecting the changes in macrofaunal assemblages in the experimental plots. Our study was also restricted to show vegetation characteristics that might affect infaunal colonization rates and colonist composition in artificial marsh sediments (Rader 1984; Lana and Guiss 1992), hydrology in sediment fence, and effects of drainage canals on sediment deposition. However, more data have been accumulated for these factors and we are going to do further studies.

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References

Boumans, R.M.J., J.W. Day, G.P. Kemp, and K. Kilgen. 1997. The effect of intertidal sediment fences on wetland surface elevation, wave energy and vegetation establishment in two Lousiana coastal marshes. *Ecol. Eng.*, 9, 37-50.

Bouwersma, P., J.H. Bossinade, K.S. Dijkema, J.W.T.M.V. Meegen, and R. Reenders. 1986. The progression in height and area of the saltwater marshes associated with the reclamation projects in Friesland en Groningen. R.I.N., rapport 86/3, The Netherlands.

Broome, S.W., E.D. Seneca, and W.W.Jr. Woodhouse. 1988. Tidal salt marsh restoration. *Aquat. Bot.*, 32, 1-22.

Cahoon, D., D. Reed, and J. Day. 1995. Estimating shallow subsidence in microtidal salt marshes of the southeastern United State: Kaye and Barghoorn revisited. *Mar. Geol.*, 128, 1-9.

Cammen, L.M. 1976. Abundance and production of macroinvertebrates from natural and artificially established salt marshes in North Carolina. *Am. Midl. Nat.*, 96, 487-493.

Cornelisen, C.D. 1998. Restoration of coastal habitats and species in the Gulf of Maine. Gulf of Maine Council on the Environment.

Kamps, L.F. 1962. Mud distribution and land reclamation in the eastern Wadden shallows. Rijkswaterstaat, Communications Baflo, 4.

Lana, P. da Cunha and C. Guiss. 1992. Macrofauna-plant-biomass interactions in a euhaline salt marsh in Paranagua

Bay (SE Brazil). *Mar. Ecol. Pro. Ser.*, 80, 57-64.

Lee, H.G. 1999. An ecological study of the macrobenthos on salt marshes in Tonggom, Kanghwa-do, Korea. Master's thesis, Inha University, Korea. 106 p. (In Korean).

Lee, J.H. and J.W. Choi. 1992. Population dynamics of *Perinereis aibuhitensis* in the intertidal mudflat of the west coast of Korea. Report BSPE 00255-466-3. Korea Ocean and Research Development Institute. 78 p. (In Korean).

Lee, J.H., J.G. Je, and J.W. Choi. 1992. Taxonomical review of *Perinereis aibuhitensis* Grube, 1878 (Nereidae: Polychaeta) in Korea. *Korean J. System Zool.*, 8, 1-10.

Levin, L.A., D. Talley, and G. Thayer. 1996. Succession of macrobenthos in a created salt marsh. *Mar. Ecol. Pro. Ser.*, 141, 67-82.

Lieberman, N., A. Matheja, and C. Zimmermann. 1997. Foreland stabilisation under waves in shallow tidal waters. Thiruvananthapuram, December 7-10, 1997, 1236-1245.

McNulty, I.B. 1985. Rapid osmotic adjustment by a succulent halophyte to saline shock. *Plant physiol.*, 78, 100-103.

Mitsch, W.J. and J.G. Gosselink. 1993. Wetlands. Van Nostrand Reinhold, New York, 920 p.

Moy, L.D. and L.A. Levin. 1991. Are *Spartina* marshes a replaceable resource? A functional approach to evaluation of marsh creation efforts. *Estuaries*, 14, 1-16.

Phleger, F.B. and J.S. Bradshaw. 1966. Sedimentary environments in a marine marsh. *Science*, 154, 1551-1553.

Posey, M.H., T. Alphin, and C. Powell. 1997. Plant and infaunal communities associated with a created marsh. *Estuaries*, 20(1), 42-47.

Race, M.S. and D.R. Christie. 1982. Coastal zone development: mitigation, marsh creation and decision making. *Environ. Manage*, 6, 317-328.

Rader, D.N. 1984. Salt marsh benthic invertebrates: small-scale patterns of distribution and abundance. *Estuaries*, 7, 413-420.

Sacco, J., F. Booker, and E.D. Seneca. 1987. Comparison of the macrofaunal communities of a human-initiated salt marsh at two and fifteen years of age. p. 282-285. In: *Proc. International Wetlands Symposium (NWF) Oct. 5-10, 1987*, eds. by J. Zelazny and S. Feierabend. National Wildlife Federation, Washington DC.

Sacco, J., E.D. Seneca, and T. Wentworth. 1994. Infaunal community development of artificially established salt marshes in North Carolina. *Estuaries*, 17, 489-500.

Scarton, F., J.W.Jr. Day, A. Rismondo, G. Cecconi, and D. Are. 2000. Effects of an intertidal sediment fence on sediment elevation and vegetation distribution in a Venice (Italy) lagoon salt marsh. *Ecol. Eng.*, 16, 223-233.

Susan, L.U., R.W. Pearcy, and D.E. Bayer. 1982. Plant water relations in a San Francisco Bay salt marsh. *Bot. Gaz.*, 143, 368-374.

Vince, S.W., I. Valiela, and J.M. Teal. 1981. An experimental study of the structure of herbivorous insect communities in a salt marsh. *Ecology*, 62, 1662-1678.

Warwick, R.M. and K.R. Clarke. 1991. A comparison of methods for analysing changes in benthic community structure. *J. Mar. Biol. Ass. UK*, 71, 225-244.

Zedler, J.B. 1988. Salt marsh restoration: lessons from California. p. 123-138. In: *Rehabilitating Damaged Ecosystems*, ed. by J. Cairns. CRC Press, Boca Raton, FL.

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