

# Comparison of Restored and Natural Seagrass Beds Near Corpus Christi, Texas

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**ABSTRACT:** Structural equivalence between seagrass restoration sites and adjacent natural seagrass beds on the mid Texas coast was assessed six times between April 1995 and May 1997. Throw traps and corers were used for quantitative sampling. Restoration sites were 2.7 to 6.6 yr old when first sampled and 3.7 to 8.2 yr old when last sampled. There were few significant differences in water column, seagrass, or sediment characteristics, in fish and decapod (nekton) densities, or in nekton and benthos community compositions between restored and natural seagrass habitats at any time during the study period. Differences in densities of dominant benthic invertebrates were regularly observed, with greater densities of more taxa observed in natural seagrasses than in restored beds. Densities of Class Oligochaeta and the polychaete *Prionospio heterobranchiata* are proposed as potential indicators of structural equivalence in restored seagrasses. This study indicates that seagrass restorations in the vicinity of Corpus Christi, Texas, exhibit minimal quantitative differences in community structure (except for benthos) relative to adjacent natural seagrass beds after 3 to 5 yr.

## Introduction

Seagrasses respond to natural stresses such as disease, herbivory, and storms by producing shifting mosaics of species, by altering density, and by expanding or contracting in spatial extent. Habitat loss occurs when human impacts such as propeller scarring, dredging, filling, nutrient enrichment, or reduction in water transparency are added to natural stresses (Short and Wyllie-Echeverria 1996). These losses may impact fishery resources that depend on seagrasses for physical refuge, foraging areas, and nursery habitat (Heck et al. 2003). Seagrass restoration by transplanting is employed to offset human-induced losses. The record of success in reestablishing seagrass habitat has been patchy, and few reported seagrass restoration projects have created enough acreage to achieve the usual goal of a 1:1 habitat replacement (Fonseca et al. 1998). Until recently, few assessments of success went beyond achieving a required vegetation coverage (Fonseca et al. 1996a,b, 2000; Sheridan et al. 2003).

Most seagrass in the southeastern United States grows in the Gulf of Mexico, with Florida and Texas supporting the greatest acreage (986,900 and 68,500 ha, respectively; Orth and van Montfrans 1990; Duke and Kruczynski 1992). Much of the coastline of these two states has been affected by human development, and seagrass restoration has been attempted at numerous locations (Altsman et al. 2004).

Newly restored seagrass beds are often isolated from naturally occurring beds and form patches of structured habitat in sands or muds with little or no inherent structure beyond sand waves and biogenic mounds or tubes. Structured habitats such as seagrass beds quickly attract mobile macrofauna; a naturally reseeded eelgrass (*Zostera marina*) bed on a dredged material placement site in Core Sound, North Carolina, supported densities of fishes and shrimps similar to densities in the edge of nearby natural beds after only 6 mo (Fonseca et al. 1990). This is the same response observed for structures put in otherwise featureless habitats, such as artificial reefs placed on bare sand (Hair et al. 1994; Carr and Hixon 1997) or fish aggregating devices deployed in the water column (Rountree 1989, 1990). The new habitat quickly provides refuge and may provide food. Research in Tampa Bay, Florida (Fonseca et al. 1996a,b), indicated that if restored seagrass beds persisted, then fish and decapod densities and diversities became indistinguishable from those in adjacent natural beds within 3 yr of planting. Three years was not enough to assure faunal parity between restored and natural seagrasses in Galveston Bay, Texas (Sheridan et al. 2003). As time passes, persistent restored habitats are expected to provide the full range of services typical of seagrass beds that were lost (Fonseca et al. 2000).

The objectives of this study were to compare structural attributes of natural and restored seagrass beds of various ages near Corpus Christi, Texas, and to estimate time required for structural equivalence as persistent beds age. Structural and

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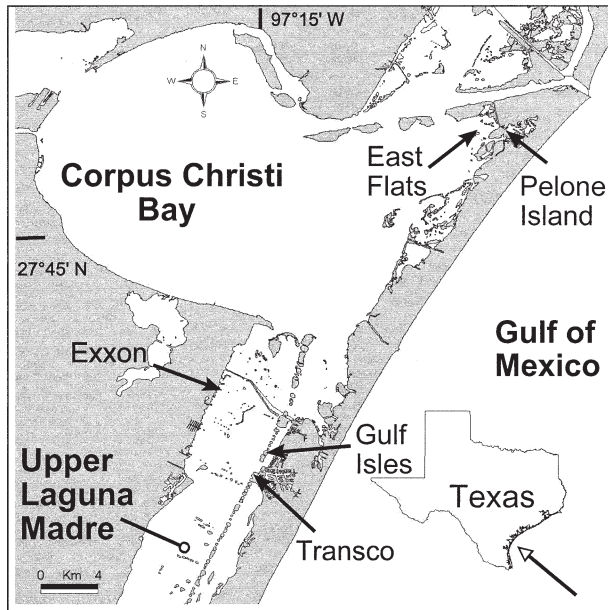


Fig. 1. Location of sampling sites in Upper Laguna Madre and Corpus Christi Bay, Texas.

functional attributes are defined as follows: structural attributes of seagrass habitats describe the distribution and abundance of biotic and abiotic components, and functional attributes (which were beyond the scope of this study) concern system processes such as energy flow, biogeochemical cycling, trophic relationships, growth rates, and materials exchange (Odum 1972). The general null hypothesis was that there was no significant difference in water, sediment, plant, or animal components between these two types of seagrass habitat. Many structural attributes have been suggested as indicator criteria for judging restoration success and habitat equivalence (Short et al. 2000).

## Materials and Methods

### STUDY AREAS

Permit application files maintained by the Texas General Land Office were examined to locate successful restoration projects near Corpus Christi that were required by the U.S. Army Corps of Engineers under Section 404 of the Clean Water Act. Success was defined by the permitting agencies as achievement of at least 70% seagrass coverage within a specified time period, usually 1 to 3 yr. Seagrass coverage was estimated periodically by project contractors using transects, quadrats, or cores until the minimum coverage was met. Four projects were found that were termed successful (or nearly so) and were relatively close to each other (within 30 km; Fig. 1). Each restoration site was paired with the closest adjacent natural seagrass

bed. Site names and natural seagrass locations were Gulf Isles and Upper Laguna Madre (27°36.5'N, 97°15'W), Exxon and Upper Laguna Madre (27°35.8'N, 97°15.4'W), Transco and Upper Laguna Madre (27°35.8'N, 97°15.4'W), and Pelone Island and East Flats of Corpus Christi Bay (27°49'N, 97°06'W). During the course of this study, all sampling areas in restored and natural habitats were vegetated by shoalgrass *Halodule wrightii*, and some sites supported scattered stargrass *Halophila engelmanni* or wigeongrass *Ruppia maritima*.

Gulf Isles (also known as Padre Isles) is a former 7.7 ha dredged material island originally deposited over seagrasses that was scraped down in May 1991 but allowed to vegetate naturally. A low-elevation ring of dredged material was left around the site, and the ring was broken through in six places to permit circulation. Gulf Isles achieved 70% seagrass coverage in June 1992 (after 1.1 yr). Exxon is a former 0.3 ha oilfield well pad originally built over seagrasses that was scraped down, planted with shoalgrass in September 1993, and partially replanted in March 1994. Exxon reached 70% seagrass coverage by July 1994 (0.8 yr after initial planting). Transco is a former 4.2 ha dredged material island placed over seagrasses that was scraped down and planted with shoalgrass in April 1990. As with Gulf Isles, some of the original dredged material at Transco was left in place as an outer protective ring. The ring was breached by three circulation exits that were connected by channels 0.5 m deeper than the rest of the site. These channels tended to trap debris and prevent seagrass growth, as did a large poorly-flushed area landward of the channels. Transco had only reached 55% seagrass coverage as of August 1993 (after 3.3 yr) but was not expected to improve much more due to debris accumulation. Pelone Island is a former 0.6 ha dredged material island originally deposited on nonvegetated tidal flats that was scraped down and planted with shoalgrass in June and September 1988. It is not a true restoration site but a creation site (Lewis 1989), where subtidal habitat was created from what was originally intertidal habitat prior to placement of dredged material. Pelone Island achieved 70% seagrass coverage in October 1990 (after 2.1 yr). Gulf Isles, Transco, and Exxon all border directly on natural seagrass beds in Upper Laguna Madre, where comparative samples were collected. Pelone Island is northeast of natural seagrasses in the East Flats of Corpus Christi Bay and is separated from its comparative sampling area by 2 km of navigation channels, intertidal sand flats, and islands.

### SAMPLING METHODS

Aerial photographs were used to divide each restored or natural site into a series of  $10 \times 10$  m grids that were then selected randomly prior to each sampling period. Ten replicate grids were selected at each site per visit, and a sample set (see below) was collected within vegetated sections of a grid. If a grid contained no seagrass, an alternate grid with seagrass coverage was randomly selected.

Densities of fishes and decapods (also termed nekton) were compared by deploying and clearing a quantitative  $1\text{-m}^2 \times 0.8\text{-m}$  deep throw trap constructed of solid aluminum sheets and bars (modified from Kushlan 1981). The throw trap was swept repeatedly with a rigid, form-fitting, 3-mm mesh seine until no organisms were recovered in three consecutive sweeps (six sweeps minimum). Fishes and decapods were frozen or preserved in 10% formalin-seawater, then identified and counted in the laboratory. Some specimens were identified only to family due to damage or larval life stage. Brown shrimp *Farfantepenaeus aztecus* and pink shrimp *F. duorarum* were combined as *Farfantepenaeus* spp. due to difficulties in separation at small sizes.

After deploying the throw trap but prior to clearing nekton, a  $1\text{ m}^2$  quadrat divided into 16 sections was placed adjacent to the throw trap on a randomly selected side. Densities of benthic organisms and seagrasses were estimated using a  $182.5\text{ cm}^2$  ( $15.2\text{ cm}$  diameter) corer taken to a depth of 10 cm in a randomly chosen section of the quadrat. The core contents were sieved through a 0.5-mm mesh screen, and retained materials were preserved in 10% formalin-seawater with rose bengal added to assist sorting. A second randomly placed core was taken to a depth of 5 cm for sediment analysis and refrigerated. Sediment samples were only collected during April or May each year. Seagrass coverage was then estimated by presence or absence in each of the 16 sections of the quadrat. Water temperature, salinity, and depth were also measured within the throw trap prior to clearing it, using a stick thermometer, refractometer, and meter stick, respectively. Temperature and salinity were measured only in the first trap per site during April 1995 and in all traps in May 1997; otherwise, these measurements were made only in the first and last traps. Depth was measured near each corner and in the center of each trap and was recorded as an average. In the laboratory, benthic animals were removed from the preserved seagrasses, identified, and counted. Some specimens were identified only to family or genus due to damage. Oligochaetes, actiniarians, hemichordates, insects, nemerteans, ophiuroids, sipunculids, and turbel-

larians were only identified to phylum, class, or order and may have consisted of more than one species. Remaining seagrasses were sorted into shoots or roots (including rhizomes), rinsed in fresh water, dried at  $60^\circ\text{C}$  for 48 h, then weighed. Each sediment core was analyzed for organic content and proportions of rubble, sand, silt, and clay following methods of Dean (1974) and Folk (1980). Since organic materials were not removed prior to grain size analyses, the terms rubble, sand, silt, and clay actually refer to rubble-, sand-, silt-, and clay-sized particles. Exclusive grain size limits were: rubble  $>2.0\text{ mm}$ , sand  $0.0625\text{--}2.0\text{ mm}$ , silt  $0.0040\text{--}0.0624\text{ mm}$ , and clay  $<0.0040\text{ mm}$  (Folk 1980). Rubble, consisting only of shell and seagrass fragments, was usually  $<2\%$  of any sample, and is not discussed further.

Potential seasonal and annual differences in community characteristics were addressed by collecting sample sets six times: April, July, and October 1995; May and October 1996; and May 1997. Shoalgrass, the primary seagrass found at both restored and natural sites, exhibits a seasonal growth pattern in Texas of increasing shoot biomass in the spring and decreasing shoot biomass in the fall, while belowground biomass of roots and rhizomes is low in the spring and high in the fall (Dunton 1990, 1994). The root:shoot biomass ratio (RSR), a potential indicator of seagrass health, was expected to change seasonally as leaves lengthen and multiply and as rhizome carbohydrate reserves build up during spring and summer then decline during fall and winter (Pulich 1982; Dunton and Tomasko 1994). Fish and decapod densities were expected to be relatively high in spring and fall and lower in summer, and species compositions were expected to change seasonally (Simmons 1957; Hellier 1962; Hoese and Jones 1963; Stokes 1974). Benthic communities were expected to exhibit a unimodal abundance pattern with a peak in late winter through spring and a nadir in fall, although some studies indicate a secondary fall peak (Harper 1992; Montagna and Kalke 1992).

Pelone Island, Gulf Isles, and Transco were visited during the first five sampling periods. Exxon (the youngest site) was only available in May 1996 and May 1997, and it was the only site visited in May 1997. All samples from a given site pair were collected on the same day, and all site pairs were completed in two consecutive days.

### DATA ANALYSIS

Differences between restored and natural seagrass habitats within each sampling period were assessed with one-way analysis of variance (ANOVA). Balanced cell sizes ( $n = 10$  samples per habitat type per visit) were maintained except for one lost

nekton sample (Gulf Isles Restored, April 1995) and five lost benthos samples (Gulf Isles Natural [2] and Transco Natural, April 1995; Gulf Isles Restored and Pelone Island Natural, May 1996). The null hypothesis was that there would be no significant differences between restored and natural seagrass beds for the following characteristics: water temperature, salinity, and depth; seagrass cover, root and shoot biomass, and RSR; organic, sand, silt, and clay contents of sediments; densities of total fishes, total decapods, and dominant nekton species (those with  $\geq 1$  individual  $m^{-2}$  in either habitat type); and densities of total annelids, total nondecapod crustaceans, total molluscs, and dominant benthic taxa (those with  $\geq 10$  individuals  $core^{-1}$  in either habitat type). Positive relationships between means and standard deviations were typical for seagrass, nekton, and benthos densities, indicating heterogeneity of variances that would violate ANOVA assumptions.  $\log_{10}(x + 1)$  transformation was used to obtain homogeneity of variances. Proportional data including seagrass coverage and sediment characteristics were arcsine transformed prior to ANOVA. Tables of ANOVA results present transformed data, whereas other tabular information and figures present actual data. Similarity of nekton or benthos community compositions during each sampling period was compared using pooled samples, unweighted pair-group average cluster analysis of Euclidean distances, and nonmetric multidimensional scaling (MDS) of the resultant distance matrices. Data were fourth root transformed prior to such analysis to reduce emphasis on numerical dominants and to differentiate between habitat types and time periods having few or many rare taxa (Clarke and Green 1988). Stress values  $< 0.2$  were maintained in MDS to ensure valid representation of sample relationships (Clarke 1993). Analyses were conducted using STATISTICA personal computer software (StatSoft, Inc., Tulsa, Oklahoma).

## Results

### WATER COLUMN, SEAGRASSES, AND SEDIMENTS

There were few significant differences in mean water column characteristics between natural and restored seagrass habitats at any time during the study period (Fig. 2). None of these differences were consistently attributable to any specific site. The July 1995 differences in temperature were recorded across all three site pairs, whereas the October 1996 difference in depths was due primarily to shallow water at Pelone Island relative to natural seagrasses at East Flats. Only one site pair (Exxon) was sampled in May 1997 and, at that time, near-shore mats of seagrass debris in the natural sea-

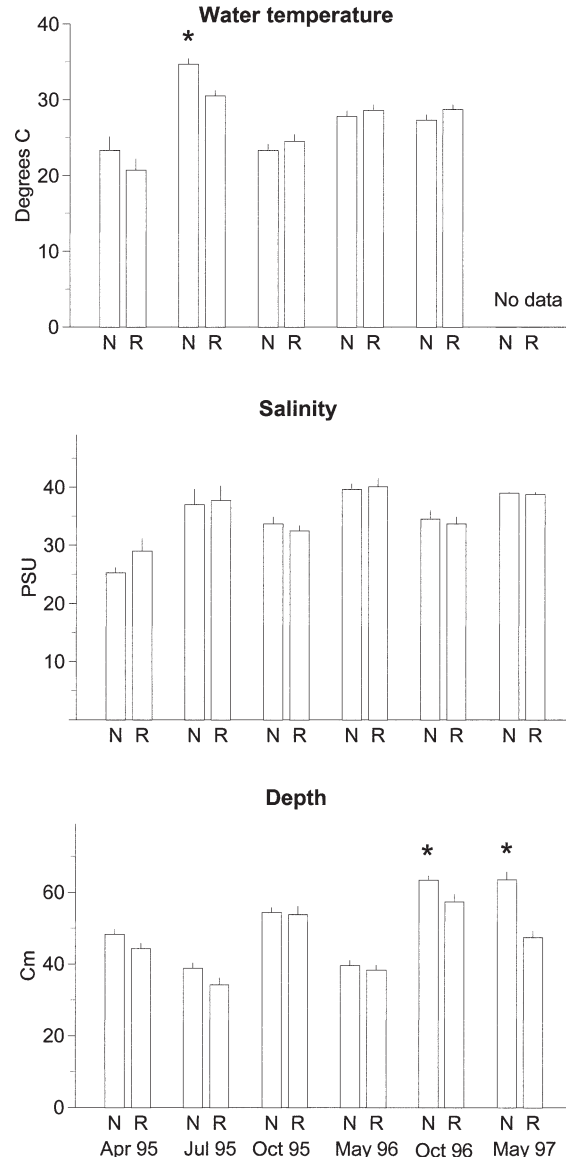
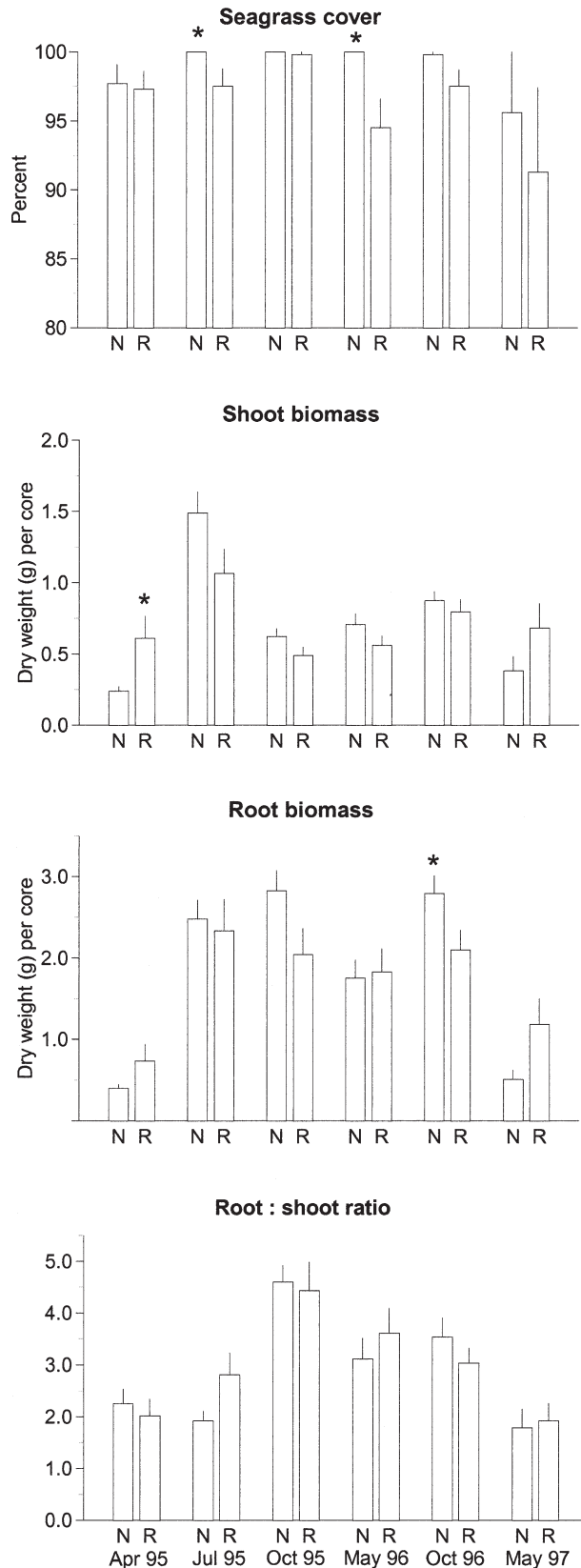


Fig. 2. Water column characteristics (mean  $\pm$  SE) of natural (N) and restored (R) seagrass habitat types. Temperature and salinity ANOVA df: April 1995 = 1, 4; July 1995, October 1995, October 1996 = 1, 10; May 1996 = 1, 14; May 1997 = 1, 18. Depth ANOVA df: April, July, and October 1995, October 1996 = 1, 58; May 1996 = 1, 78; May 1997 = 1, 18. \* = significant difference (ANOVA,  $p < 0.05$ ) between N and R on a given date.

grass bed forced sampling in significantly deeper waters.

Significant differences in seagrass characteristics between habitat types were also infrequent (Fig. 3). Differences in coverage during July 1995 and May 1996 were primarily due to 100% coverage for all natural habitat samples, whereas there was small-scale patchiness at restored sites. The lowest mean coverage for restored seagrass was 91.3%, and the





maximum mean difference between habitat types was only 5.5%. The coverage estimates for restored sites are biased (sampling was not conducted in nonvegetated areas) and should not be construed as representing coverage for entire sites. The only site with no known large bare areas was Exxon, which is a finger-like projection from the shoreline with no obstructions or depressions that could capture debris or drift algae preventing seagrass growth. Pelone Island had several deep areas that filled with debris and fine particulate material and remained nonvegetated. Both Gulf Isles and Transco were surrounded by rings of emergent scrubshrub habitat, and the inner margins of these rings tended to accumulate debris that prevented seagrass growth. As noted before, Transco also had channels and backwaters that remained nonvegetated due to accumulated debris.

Differences in sediment characteristics were noted only in April 1995 (Fig. 4). At that time, natural seagrasses had significantly greater organic, silt, and clay contents and significantly lower sand content than did restored seagrasses. The sediment data set is relatively sparse, and there may have been seasonal variations that were not documented due to spring-only sampling.

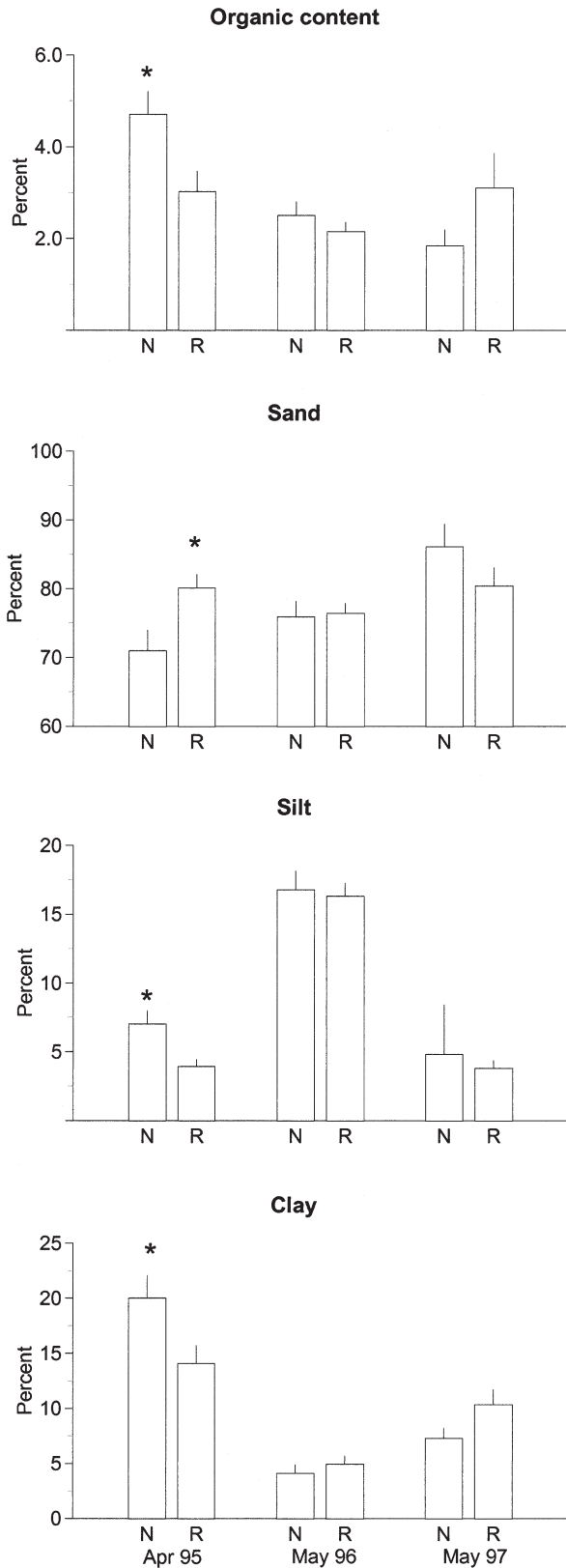
#### FISHES AND DECAPODS

Throw trap samples yielded 43 taxa of fishes and 34 taxa of decapods among 31,788 individuals (a complete list of organisms is available from the author). More taxa (62 versus 44) and more individuals (20,334 versus 11,454) were collected from restored seagrasses than from natural seagrasses, respectively. Differences in nekton taxonomic richness were due to rare organisms, as most of the taxa found in only one of the habitat types consisted of fewer than 10 individuals and many were of a single individual. The most abundant taxa were generally collected in both restored and natural seagrasses.

Numerically dominant fishes and decapods were defined as those species exhibiting mean densities of  $\geq 1 \text{ m}^{-2}$  in either natural or restored seagrass beds in a given sampling period. Differences in densities of dominant taxa, total fishes, and total decapods within each sampling period are presented in Table 1. Only eight taxa were considered dominant in four or more of the six sampling pe-

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Fig. 3. Seagrass characteristics (mean  $\pm$  SE) of natural (N) and restored (R) seagrass habitat types. Note truncated y-axis for seagrass cover. ANOVA df: April, July, and October 1995, October 1996 = 1, 58; May 1996 = 1, 78; May 1997 = 1, 18. \* = significant difference (ANOVA,  $p < 0.05$ ) between N and R on a given date.



riods (code goby *Gobiosoma robustum*, rainwater killifish *Lucania parva*, blue crab *Callinectes sapidus*, gulf grassflat crab *Dyspanopeus texana*, penaeid shrimps *Farfantepenaeus* spp., zostera shrimp *Hippolyte zostericola*, brackish grass shrimp *Palaemonetes intermedius*, and daggerblade grass shrimp *P. pugio*), and none of them exhibited consistent trends in density related to seagrass bed type. Another 12 species were irregularly abundant (gulf menhaden *Brevoortia patronus*, sheepshead minnow *Cyprinodon variegatus*, darter goby *Gobionellus boleosoma*, pinfish *Lagodon rhomboides*, clown goby *Microgobius gulosus*, gulf pipefish *Syngnathus scovelli*, bigclaw snapping shrimp *Alpheus heterochaelis*, lesser blue crab *Callinectes similis*, thinstripe hermit *Clibanarius vittatus*, white shrimp *Litopenaeus setiferus*, marsh grass shrimp *Palaemonetes vulgaris*, and arrow shrimp *Tozeuma carolinense*), and again none of them exhibited consistent trends in density related to seagrass bed type. The same was true for total decapod densities. The only potential trend detected was that total fishes were significantly more abundant in restored seagrasses in three of six sampling periods; otherwise, fish densities were similar.

Nekton community assemblages were identified through classification and ordination of the pooled temporal data. Sampling period, not seagrass bed type, was the driving force behind community similarity (Fig. 5). With the exception of October 1996, habitat pairs within a time period more closely resembled each other than did habitat types across all time periods. Nekton community compositions were fairly similar during any sampling period, in spite of any previously described differences in faunal densities. Nekton densities during October 1996 exhibited a large number of significant habitat-related differences (Table 1), and pooled data indicated that restored shoalgrass held more than twice the number of individuals than natural seagrass (1,926 versus 949, respectively) and almost twice the number of species (31 versus 19).

#### BENTHOS

Benthic samples yielded 199 unique species or genera and another 79 composite taxa (phylum, class, order, family, or damaged) comprising 85,057 individuals (a complete list of organisms is available from the author). Annelids were the most

Fig. 4. Sediment characteristics (mean  $\pm$  SE) of natural (N) and restored (R) seagrass habitat types. Note truncated y-axis for sand. ANOVA df: April 1995 = 1, 58; May 1996 = 1, 78; May 1997 = 1, 18. \* = significant difference (ANOVA,  $p < 0.05$ ) between N and R on a given date.

TABLE 1. Results of one-way ANOVA comparisons of densities (mean number m<sup>-2</sup> and standard error [SE]) of dominant nekton in natural and restored seagrass habitats near Corpus Christi, Texas, by collection date. ANOVA df: April 1995 = 1, 57; July 1995, October 1995, and October 1996 = 1, 58; May 1996 = 1, 78; May 1997 = 1, 18. \* indicates significant difference (p < 0.05).

Date	Taxon	Natural			Restored			Natural			Restored		
		n	SE	n	SE	n	SE	n	SE	n	SE	n	SE
April 1995	Fishes	14.0	2.7	45.0*	6.9	Decapods	133.1	33.5	314.5*	79.7			
	<i>Brevoortia patronus</i>	0.4	0.3	11.1*	4.1	<i>Callinectes similis</i>	1.0	0.3	6.1	2.1			
	<i>Gobionellus boleosoma</i>	4.0	1.3	9.7	2.7	<i>Dyspanopeus texana</i>	1.9*	0.6	0.5	0.2			
	<i>Gobiosoma robustum</i>	0.9	0.2	1.0	0.3	<i>Farfantepenaeus</i> spp.	6.4	1.1	9.9	2.0			
	<i>Lagodon rhomboides</i>	2.5	0.7	4.8	1.1	<i>Hippolyte zostericola</i>	1.4	0.4	2.8	0.8			
	<i>Lucania parva</i>	2.7	0.7	15.3*	3.8	<i>Palaeomonetes intermedius</i>	56.8	15.6	143.5	41.6			
	<i>Microgobius gulosus</i>	1.5	1.5	0.1	0.1	<i>Palaeomonetes pugio</i>	64.3	18.5	144.2	40.8			
						<i>Tozeuma carolinense</i>	0.7	0.3	4.7*	1.6			
						Decapods	43.6	5.9	69.9	11.5			
						<i>Alpheus heterochaelis</i>	1.8	0.8	0.6	0.2			
July 1995	Fishes	10.1	1.5	24.4	5.5	<i>Callinectes sapidus</i>	0.0	0.0	1.8*	0.5			
	<i>Gobionellus boleosoma</i>	4.8	1.5	19.7	5.6	<i>Clibanarius vittatus</i>	1.8*	0.8	0.2	0.1			
	<i>Lagodon rhomboides</i>	1.0	0.4	0.5	0.2	<i>Dyspanopeus texana</i>	3.1	1.4	0.8	0.4			
	<i>Lucania parva</i>	1.3	0.3	2.3	0.6	<i>Farfantepenaeus</i> spp.	1.1	0.3	0.6	0.2			
	<i>Syngnathus scovelli</i>	1.4*	0.3	0.4	0.1	<i>Hippolyte zostericola</i>	1.2	0.4	3.1	1.7			
						<i>Litopenaeus setiferus</i>	0.0	0.0	1.5*	0.6			
						<i>Palaeomonetes intermedius</i>	8.5	2.7	18.7	5.7			
						<i>Palaeomonetes pugio</i>	23.5	4.2	31.3	5.1			
						<i>Palaeomonetes vulgaris</i>	1.6	0.9	10.6	4.7			
						Decapods	79.4	13.1	78.6	13.9			
October 1995	Fishes	10.1	1.1	19.6*	2.6	<i>Alpheus heterochaelis</i>	1.5	0.5	0.5	0.3			
	<i>Gobionellus boleosoma</i>	2.6	0.7	11.2	2.9	<i>Callinectes sapidus</i>	1.7	0.5	5.2	1.6			
	<i>Gobiosoma robustum</i>	3.4*	0.5	1.5	0.8	<i>Callinectes similis</i>	0.0	0.0	1.1*	0.6			
	<i>Lucania parva</i>	2.0	0.4	3.9	0.8	<i>Dyspanopeus texana</i>	11.1	3.1	1.8	0.6			
	<i>Syngnathus scovelli</i>	1.5	0.3	1.1	0.3	<i>Farfantepenaeus</i> spp.	19.2	3.0	12.1	1.6			
						<i>Hippolyte zostericola</i>	6.4	2.0	10.1	3.9			
						<i>Litopenaeus setiferus</i>	0.1	0.1	3.5*	1.2			
						<i>Palaeomonetes intermedius</i>	7.9	1.2	12.4	3.2			
						<i>Palaeomonetes pugio</i>	16.4	2.0	20.7	4.8			
						<i>Palaeomonetes vulgaris</i>	1.4	0.7	3.3	3.0			
May 1996	Fishes	10.2	1.8	11.3	1.7	<i>Tozeuma carolinense</i>	12.7	4.2	6.3*	3.1			
	<i>Brevoortia patronus</i>	0.0	0.0	1.6	1.2	Decapods	31.5	5.7	28.7	5.3			
	<i>Gobionellus boleosoma</i>	1.6	0.5	2.4	0.8	<i>Dyspanopeus texana</i>	1.4	0.3	0.6	0.1			
	<i>Gobiosoma robustum</i>	1.0	0.3	0.6	0.1	<i>Farfantepenaeus</i> spp.	5.6	0.9	6.1	1.3			
	<i>Lagodon rhomboides</i>	6.0	1.4	4.5	0.8	<i>Hippolyte zostericola</i>	5.4	1.6	2.6	1.2			
	<i>Lucania parva</i>	0.3	0.1	1.9*	0.6	<i>Palaeomonetes intermedius</i>	17.2	3.6	16.8	5.0			
						<i>Palaeomonetes pugio</i>	0.4	0.2	1.2	0.7			
						<i>Tozeuma carolinense</i>	1.4	0.4	0.8	0.4			
						Decapods	25.1	4.3	40.3	7.1			
						<i>Callinectes sapidus</i>	0.6	0.3	4.4*	1.3			
October 1996	Fishes	6.6	1.0	23.9*	3.8	<i>Callinectes similis</i>	2.7	0.9	8.0*	2.2			
	<i>Cyprinodon variegatus</i>	1.4	0.4	7.6*	1.6	<i>Farfantepenaeus</i> spp.	6.3	0.9	4.4	0.7			
	<i>Gobionellus boleosoma</i>	0.4	0.2	13.3*	4.1	<i>Hippolyte zostericola</i>	0.0	0.0	3.8*	2.0			
	<i>Gobiosoma robustum</i>	1.4*	0.3	0.6	0.2	<i>Palaeomonetes intermedius</i>	12.8	3.6	10.3	2.2			
	<i>Lucania parva</i>	2.6*	0.7	0.7	0.2	<i>Palaeomonetes pugio</i>	1.5	0.5	2.9	0.9			
						<i>Palaeomonetes vulgaris</i>	0.1	0.1	2.1	1.1			
						Decapods	6.8	2.2	21.5*	6.8			
						<i>Callinectes sapidus</i>	1.2	0.3	0.8	0.3			
						<i>Dyspanopeus texana</i>	0.0	0.0	5.1*	2.6			
						<i>Farfantepenaeus</i> spp.	1.1	0.2	0.9	0.3			
May 1997	Fishes	5.4	1.8	5.9	1.0	<i>Palaeomonetes intermedius</i>	4.1	1.9	13.8*	4.8			
	<i>Brevoortia patronus</i>	1.2	1.1	0.6	0.6	<i>Palaeomonetes vulgaris</i>	0.1	0.1	2.1	1.1			
	<i>Gobionellus boleosoma</i>	0.3	0.2	1.4	0.5	Decapods	6.8	2.2	21.5*	6.8			
	<i>Gobiosoma robustum</i>	1.3	0.6	1.7	0.5	<i>Callinectes sapidus</i>	0.0	0.0	0.8	0.3			
	<i>Syngnathus scovelli</i>	1.3	0.6	1.7	0.5	<i>Dyspanopeus texana</i>	0.0	0.0	5.1*	2.6			

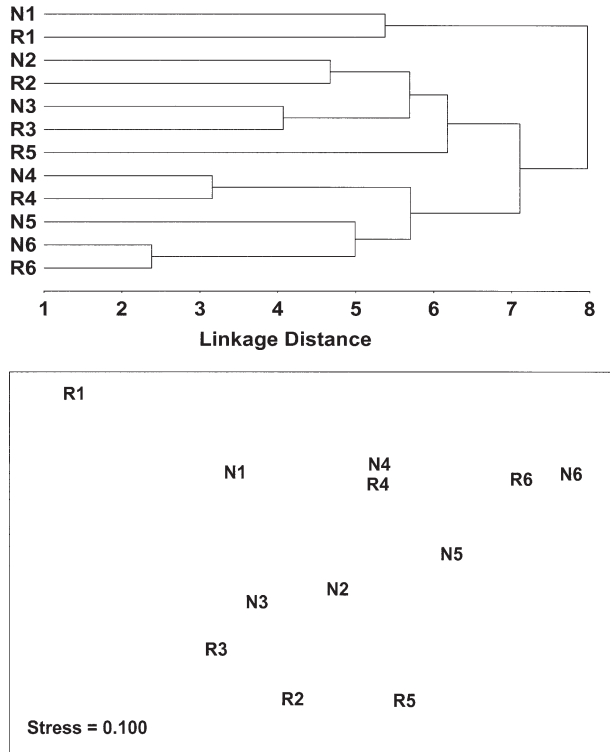


Fig. 5. Cluster analysis (upper) and MDS ordination (lower) of nekton communities by habitat type and sampling date using unweighted pair-group average linkage of a Euclidean distance matrix. N = natural seagrass, R = restored shoalgrass. Sampling dates are sequentially numbered (1 = April 1995, 2 = July 1995, 3 = October 1995, 4 = May 1996, 5 = October 1996, and 6 = May 1997).

abundant group (76.0% of all organisms), followed by nondecapod crustaceans (14.2%), molluscs (8.4%), and other taxa (1.4%). Similar numbers of unique species or genera were recorded from natural and restored seagrasses (134 versus 122, respectively). Fifty-seven percent of all individuals were collected from natural seagrasses. All taxa found in only one seagrass bed type averaged <10 individuals core<sup>-1</sup>, and many averaged <1 individual core<sup>-1</sup>.

Numerically dominant benthic organisms were defined as those taxa exhibiting mean densities of  $\geq 10$  core<sup>-1</sup> in either natural or restored seagrass beds in a given sampling period. Differences in densities of dominant taxa, total annelids, total crustaceans, and total molluscs within each sampling period are presented in Table 2. Only three taxa were considered dominant in four or more of the six sampling periods (Class Oligochaeta and the polychaetes *Streblospio benedicti* and *Prionospio heterobranchiata*). When significant differences between habitat types were observed, oligochaetes and *P. heterobranchiata* were always more abundant

in natural seagrasses whereas *S. benedicti* was always more abundant in restored seagrasses. Another 10 species were abundant on one or two occasions. When significant differences between habitat types were detected, the polychaete *Capitella capitata* was more abundant in restored seagrasses whereas *Leitoscoloplos foliosus*, *Exogone dispar*, *Syllis alosae*, *Polydora ligni*, *Axiiothella mucosa*, *Chone* cf. *americana* (polychaetes), *Grandidierella bonnieroides*, *Harrietta faxoni* (crustaceans), and *Anomalocardia auberiana* (bivalve) were more numerous in natural seagrasses. Other dominant organisms exhibited no significant differences between seagrass bed types, including *Syllis cornutus*, *Janua* sp., *Heteromastus filiformis*, *Grubeosyllis clavata*, *Fabricinuda trilobata*, *Naineris laevigata* (polychaetes), *Erichsonella attenuata*, *Deutella incerta* (crustaceans), *Cerithium lutosum* (gastropod), and Nemertinea (Table 2). Total annelid densities were rarely significantly different between habitat types. Total crustacean densities were significantly greater in natural seagrasses during three collecting periods, as were total mollusc densities during four collecting periods.

Benthic community assemblages were identified through classification and ordination of the pooled temporal data. Sampling period was the driving force behind community similarity, not whether the seagrass was restored or natural (Fig. 6). There were no exceptions: habitat pairs within a time period more closely resembled each other than did habitat types across all time periods. Benthic community compositions of restored and natural seagrasses were similar during any sampling period, in spite of any previously described differences in densities of benthic taxa.

### Discussion

Seagrass loss remains a continuing problem worldwide because much of the loss is directly or indirectly attributable to human activities on or near the coast (Short and Wyllie-Echeverria 1996). The increasing development and use of coastal lands and waters will bring more stresses to seagrass beds. Conservation and protection are currently recommended over restoration because seagrass beds are fragile ecosystems that are hard to duplicate (Fonseca 1992; Kirkman 1992). Increasing coastal development is likely to cause an increasing demand to restore or mitigate for current and future seagrass destruction. Restoration (as opposed to no action) may lead to improved ecosystem productivity, since abundance, growth, and survival of macrofauna are enhanced in seagrasses over those observed in unstructured sand or mud habitats (Heck et al. 2003).

Dredging for navigation purposes, and associated in-bay placement of dredged material, are



TABLE 2. Results of one-way ANOVA comparisons of densities (mean number core<sup>-1</sup> and standard error [SE]) of dominant benthos in natural and restored seagrass habitats near Corpus Christi, Texas, by collection date. ANOVA df: April 1995 = 1, 55; July 1995, October 1995, and October 1996 = 1, 58; May 1996 = 1, 76; May 1997 = 1, 18. \* indicates significant difference ( $p < 0.05$ ).

Date	Taxon	Natural		Restored		Taxon	Natural		Restored	
		n	SE	n	SE		n	SE	n	SE
April 1995	Annelida	205.6	17.5	319.4*	38.1	<i>Sireblosipio benedicti</i>	5.4	1.7	125.9*	33.8
	<i>Capitella capitata</i>	37.2	8.7	109.9*	23.3	<i>Syllis cornutus</i>	25.5	3.8	27.7	7.1
	<i>Leitoscoloplos foliosus</i>	10.3*	3.8	0.1	0.1	Crustacea	32.6	5.9	45.4	8.5
	Oligochaeta	30.0	5.5	20.8	4.3	Mollusca	15.9*	3.3	4.5	1.3
	<i>Prionospio heterobranchiata</i>	72.0	10.7	5.2	2.0					
July 1995	Annelida	62.8	5.9	87.0	13.4	<i>Sireblosipio benedicti</i>	1.9	0.8	13.3*	8.4
	<i>Exogone dispar</i>	12.7*	1.8	7.9	1.5	Crustacea	10.1*	1.3	6.1	0.8
	Oligochaeta	30.2*	7.7	9.1	2.6	Mollusca	6.2	0.9	14.6	4.1
	<i>Prionospio heterobranchiata</i>	23.8	3.6	23.9	4.5					
	Annelida	213.3	26.4	185.5	27.7	<i>Sireblosipio benedicti</i>	3.6	1.9	70.8*	20.2
October 1995	<i>Grubeosyllis clavata</i>	15.3	2.2	26.7	5.7	<i>Syllis allosae</i>	10.0*	2.9	3.1	0.9
	<i>Janua</i> sp.	63.8	21.3	20.8	16.9	Crustacea	30.8	3.1	30.4	4.4
	Oligochaeta	23.8*	2.8	15.4	4.6	Mollusca	41.5	9.1	27.8	5.0
	<i>Prionospio heterobranchiata</i>	72.5*	9.2	11.1	4.1	<i>Cerithium lutosum</i>	32.3	8.6	21.2	5.2
	Annelida	144.7*	17.6	96.9	15.1	<i>Prionospio heterobranchiata</i>	43.8*	5.8	14.2	3.0
May 1996	<i>Capitella capitata</i>	17.9	2.9	21.8	3.6	<i>Sireblosipio benedicti</i>	3.3	1.3	22.0*	6.5
	<i>Heteromastus filiformis</i>	14.7	5.7	11.1	4.2	Crustacea	24.4*	3.9	13.9	2.9
	Oligochaeta	29.2*	4.0	9.2	2.4	Mollusca	10.5*	1.7	5.3	1.0
	Annelida	109.4	13.8	141.7	29.1	<i>Sireblosipio benedicti</i>	0.4	0.3	65.7*	25.4
	<i>Capitella capitata</i>	2.1	0.8	11.1*	3.0	Crustacea	30.4	3.3	40.6	15.9
October 1996	Oligochaeta	17.6	4.1	9.6	2.9	<i>Granditrella bonnieroides</i>	20.5*	2.9	10.8	2.1
	<i>Polydora ligni</i>	10.9	2.0	9.8	1.8	<i>Hargeria rapax</i>	2.8	1.1	12.4	8.5
	<i>Prionospio heterobranchiata</i>	25.7*	4.1	5.9	1.2	Mollusca	22.5*	6.5	7.9	1.3
	Annelida	333.0	70.4	227.7	53.5	<i>Prionospio heterobranchiata</i>	52.8	11.6	58.7	16.0
	<i>Axiolthella mucosa</i>	23.8*	6.6	7.4	5.5	<i>Syllis allosae</i>	9.3	2.5	11.5	2.3
May 1997	<i>Capitella capitata</i>	34.0	5.7	24.8	4.2	Crustacea	100.1*	22.5	39.8	8.0
	<i>Chone</i> cf. <i>americana</i>	20.8*	5.8	12.8	4.7	<i>Deutella incerta</i>	15.1	4.0	6.5	2.0
	<i>Fabricinuda trilobata</i>	13.7	12.9	0.0	0.0	<i>Erichsonella attenuata</i>	19.6	4.9	13.8	4.5
	<i>Grubeosyllis clavata</i>	99.2	57.3	66.1	29.8	<i>Harrieta faxoni</i>	29.9*	9.7	6.9	1.5
	<i>Naineris laevigata</i>	9.5	4.6	10.4	3.2	Mollusca	149.8*	30.9	66.8	19.4
May 1997	Oligochaeta	25.1*	5.6	8.2	2.4	<i>Anomalocardia auberiana</i>	124.6*	32.6	33.1	13.0
	<i>Polydora ligni</i>	12.7*	4.0	2.3	0.6	Nemertinea	12.9	3.1	7.3	3.3

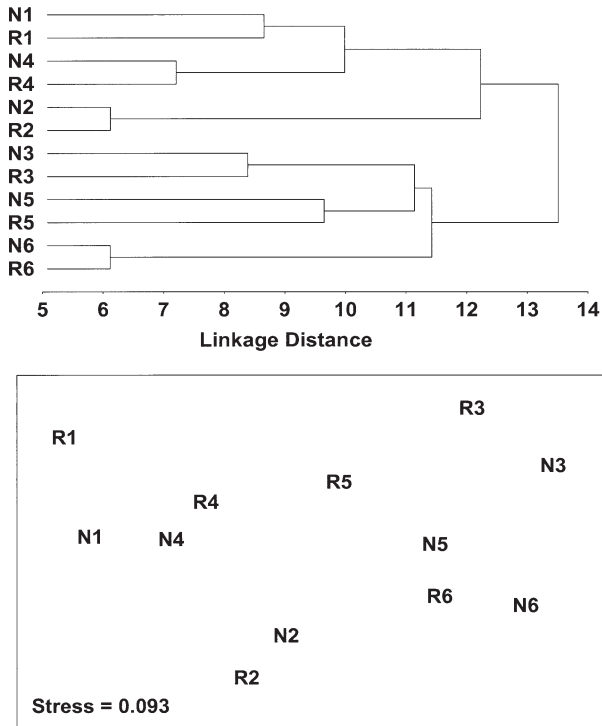


Fig. 6. Cluster analysis (upper) and MDS ordination (lower) of benthos communities by habitat type and sampling date using unweighted pair-group average linkage of a Euclidean distance matrix. N = natural seagrass, R = restored shoalgrass. Sampling dates are sequentially numbered (1 = April 1995, 2 = July 1995, 3 = October 1995, 4 = May 1996, 5 = October 1996, and 6 = May 1997).

among the most recognizable human activities that continue to affect the central Texas coast (Tunnell et al. 1996). Although not quantified, there are many dredged material placement sites alongside navigation channels in the Corpus Christi Bay system that were established on seagrass beds and that could be scraped down to return them to their original subtidal state. The present study indicated that if restorations in the area are conducted properly (meaning that topographic, seagrass, water column, and sediment characteristics approach those of natural beds and persist through time), then most quantitative differences in faunal community structure between restored and natural seagrass beds will be minimal after 3 to 5 yr. This observation is tempered by the fact that the restored sites did not always provide one for one replacement; seagrass coverage was usually less, sometimes much less, than the site footprint. Sampling was restricted to vegetated areas that usually harbor higher densities of organisms than do nonvegetated substrates (Heck et al. 2003; Sheridan et al. 2003; Sheridan 2004).

There are not enough data available on the early

stages of restorations in the Corpus Christi Bay area to judge whether structural equivalence occurs within 3 yr of site preparation. Studies in Florida demonstrated that fauna in restored beds reached densities and species compositions similar to those of reference natural seagrasses in 1 to 2.3 yr for nekton (Fonseca et al. 1996b) and in 2 to 4 yr for annelids (Bell et al. 1993). Recent research in northern Texas indicated that 3 yr was not enough time for restored and natural shoalgrass beds in Galveston Bay to reach floral or faunal parity (Sheridan et al. 2003). Differences may persist at longer time scales as well. The present study reports that numerically dominant benthic invertebrates had not reached equivalence in restored and natural seagrasses after 8 yr. Brown-Peterson et al. (1993) compared natural and restored seagrasses (in this case, seagrasses recolonizing dredged material) 31 yr after dredging and found differences in water temperature, salinity, depth, seagrass characteristics, and fish species compositions, although there were no consistent differences in total fish density or number of fish species between habitat types.

The persistence of seagrass coverage at the threshold shoot density that attracts macrofauna has been proposed as a crucial component of long-term restoration success (Fonseca et al. 1996b). Fonseca et al. suggest that the basic data requirements are seagrass persistence and shoot density; once those have stabilized, faunal components will follow. A longer monitoring list proposed by the Pacific Environmental Research Laboratory (1990) includes a minimum of site topography, sediment organic matter, inorganic nitrogen in sediments and pore water, seagrass coverage and shoot length, and fish and benthic invertebrate densities. The present study followed the latter course, but by the time comparisons were undertaken most indicators had reached parity. Results from the present study suggest that the benthic community was still in transition and that certain benthic organisms should be evaluated for their use as success indicators. Densities of oligochaetes and *P. heterobranchiata* were consistently greater in natural seagrass beds than in restored beds. Depressed oligochaete densities in restored habitats relative to natural habitats have been reported in restored seagrasses by Sheridan et al. (2003), in seagrasses that were recolonizing dredged material by Sheridan (2004), and in restored salt marshes by Levin et al. (1996). Unfortunately, none of these studies identified oligochaetes beyond Class level, so it remains uncertain whether more than one species was involved in each case. Densities of *P. heterobranchiata* were also reported to be consistently greater in natural seagrass beds than in seagrasses recolo-

nizing dredged material (Sheridan 2004). A potential indicator species for restored habitats that have not reached structural equivalence is the polychaete *S. benedicti*, which is often considered an opportunistic species that colonizes and dominates disturbed habitats (Levin et al. 1996). The present study indicated densities of *S. benedicti* were consistently greater in restored shoalgrass, and Levin et al. (1996) also noted its dominance in restored salt marshes. This was not the case for restored shoalgrass in Galveston Bay or for shoalgrass recolonizing dredged material in more southerly parts of Laguna Madre, where *S. benedicti* densities were usually lower (often significantly lower) than in natural beds (Sheridan et al. 2003; Sheridan 2004).

There are some actions needed for assessment of restoration sites and for improving the state of knowledge concerning the value of restored versus natural seagrasses in Texas and elsewhere. Resource conservation agencies must assure that seagrass restorations persist for periods longer than those typically needed to achieve target seagrass coverage; in part, this will come from proper site planning and requirements for replanting if early transplants fail. Site planning is critical; three of the four sites visited (the oldest being 8 yr old at the final visit) appeared healthy but will likely never completely revegetate due to construction anomalies, so full benefits will not be restored. The final appearance of restoration sites should closely resemble the topography of nearby reference seagrasses. The questions of what to monitor, and for how long, depend on what is required in mitigation permits or what is intended to be restored to the public. Pacific Environmental Research Laboratory (1990) suggested regular checks of selected criteria in restored marshes for up to 20 yr, while recognizing that most resource agencies would be constrained by funding. The present study indicated that the minimum list of variables to monitor would include seagrass shoot density or biomass, persistence of seagrass coverage, and biannual assessment of nekton and benthos in restored seagrasses for 5 yr, all in comparison to reference natural beds. Trends in these data would be indicative of a trend toward structural equivalence, and the 5-yr duration could be relaxed if equivalence is reached beforehand. Full structural restoration probably does not occur for up to 8 yr (Sheridan 2004), but monitoring the entire suite of community variables for such a period may be cost-prohibitive.

Structural equivalence of restored and natural seagrasses is not the only topic of concern. In the past, monitoring programs have not addressed functional equivalence, which would compare ecosystem processes such as energy flow, biogeochem-

ical cycling, trophic relationships, or growth rates. Bell et al. (1993) provide the only published comparison of secondary production rates between restored and natural seagrasses that were equivalent in seagrass biomass and sediment characteristics, and this was for a single species of polychaete. Given the wide variety of taxa in a given habitat, the scope of such studies will necessarily be reduced below that of structural surveys. Manipulative studies of key ecosystem processes using a small suite of characteristic or indicator species will provide much needed information concerning functioning of restored seagrass systems.

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