

The Use of Ecological Data in the Implementation and Management of Seagrass Restorations

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ABSTRACT

Effective restoration of seagrass systems can be implemented best through incorporation of basic ecological data into decision-making processes. The data are comprised mainly of seagrass population growth and coverage models. Careful selection of a site to be transplanted and monitored under strict performance standards offers functional restoration of a system. Environmental factors such as light, temperature, salinity, tidal range, and sediment stability may be used to determine a priori whether seagrass growth could be supported at a given site. On-site restoration directly addresses a return of ecosystem productivity. From an ecological perspective, off-impact-site restoration often results in the selection of sites unacceptable as a mitigative tradeoff. Our data suggest that unvegetated patches among natural meadows should not be transplanted as a method to compensate for destruction of existing meadows. Most important for effective management is the utilization of seagrass-population-growth models in the planning process. These data allow 1) selection of the appropriate species for the site in question, 2) setting of precise and testable performance standards, 3) accurate planning of the transplanting operation, including spacing between plantings, 4) development of a monitoring program, and 5) an objective determination by regulatory agencies as to when compliance has been achieved.

INTRODUCTION

For the most part, seagrass transplanting as a management tool is not working. Isolated cases of success or partial success can be found, but these are overshadowed by many costly failures. This lack of success is due largely to general disregard for and lack of scientific information on environmental requirements of transplant species.

What we propose is not new, insofar as approaches to a problem go, but it is relatively new to seagrass management. We must be able to predict the growth responses of seagrasses under transplanting conditions. This predictive capability is requisite for managers to develop reliable expectations for the technology of seagrass transplantation. Predictable responses can allow us to set performance standards, plan transplanting operations, minimize costs (by avoiding unsuitable planting

areas), and quantitatively assess compliance with those standards.

The pressure to make management decisions has grown disproportionately to the increase in information regarding seagrass habitat management. With much emphasis placed on methodology of transplanting and much less on collection of environmental data, we have been at a loss to explain successes and failures in a quantitative fashion. Such a lack of information was lamented by Lindall et al. (1979) and was identified as an important factor preventing our assessment of environmental impacts to estuarine systems in the southeast.

The most serious problem facing management of seagrass restorations is locating appropriate planting sites. Selection of a seagrass transplant site must be made to achieve "zero habitat loss" (Lindall et al., 1979); but the same "incomplete understanding" problem cited by Lindall et al.

(1979) is present even now. Areas particularly appealing for seagrass transplanting have been sparsely covered (and, therefore, planted as an attempt at enhancement) or subtidal and unvegetated. Experience and observation have shown that both types of areas have shortcomings as mitigation alternatives. Selection of either of these areas does not recognize that seagrass meadows form a pattern of coverage in relation to local environmental conditions (den Hartog, 1971; Fonseca et al., 1983; Kenworthy et al., 1982; Moliner and Picard, 1952; Thayer et al., 1984).

Sparse, patchy patterns of seagrass distribution quite commonly emerge from the heterogeneity of environmental conditions. Some of these stochastic events may be short-lived (e.g., dredging, storm erosion); others may be of greater duration (e.g., decreases in water quality due to shore-based development); still others may also be short-term, but more a matter of degree (e.g., storm erosion from a squall versus that from a hurricane). Other patchy seagrass distributions may not result from short-term events, but from less dynamic influences, such as sediment depth. In the case of *Thalassia testudinum* Banks ex König, this factor partially controls meadow development (Zieman, 1972). The type of environmental condition contributing to an observed seagrass distribution also may determine whether a successful mitigation project can be implemented in that area. For example, a *Halodule wrightii* Aschers. or *Syringodium filiforme* Kützinger meadow may be quickly established on a disturbed site, but insufficient sediment depth could prevent the eventual development of *Thalassia*.

Planting into sparse, existing meadows is an inappropriate management choice. A meadow may be patchy and may remain patchy for many reasons. If heterogeneity of environmental conditions is a major factor controlling the process of seagrass meadow development (at least within the time scale of seagrass transplant establishment; ca. 150-250 days), then managers will have no basis for predicting success of transplanting into unvegetated spaces within existing meadows. Consequently, these transplants would not provide a persistent, increased cover; the meadow would eventually revert to ambient patchiness with no net gain of habitat. Although we have planted only on disturbance sites in Florida, our data on temperate meadow plantings bear out the unpredictability of success in sparsely or patchily vegetated areas (Fonseca et al., in press).

Seagrass populations respond quickly to sto-

chastic or predictable local changes in environmental conditions. Temperate seagrass populations are often best described as fluctuating around an equilibrium level. A substantial portion of this fluctuation is seasonal and, thus, predictable. Tropical and subtropical meadows, especially those out of the influence of continental air masses in south Florida, tend not to exhibit a population fluctuation as extreme as that of temperate meadows. This is especially true for *T. testudinum* meadows. In contrast to the general 2-year life history of *Zostera marina* L. (Thayer et al., 1984), individual *T. testudinum* shoots may live for > 10 years (Patriquin, 1973). Thus, any fluctuations in population levels tend to be more gradual, unless interrupted by acute, catastrophic events (e.g., storms). Such stochastic, episodic events are usually easily identified. Disturbances attributable to these types of events provide excellent opportunities for mitigation actions that can potentially speed up recovery rates by years (Fonseca et al., 1985).

Generally, transplanting into presently unvegetated areas also is a poor management practice. An area is unvegetated for a reason. If the lack of vegetation at a given site can be ascribed to an acute event, then the site may well be considered a candidate for mitigation action in the future (e.g., transplanting). If the reason cannot be documented, then the manager is again faced with an unacceptable level of uncertainty as to the possible success of the transplant. The risk becomes even less acceptable if natural, persistent seagrass meadows are being traded off for transplanted meadows with uncertain probabilities of establishment. This can be prevented if up-front mitigation is required. Up-front mitigation requires that a replacement habitat is generated before destruction of an existing natural habitat is permitted.

Obtaining data to determine the environmental factors responsible for seagrass fluctuations and distributions may be expensive. Site specific, historical coverage data are needed to plan an effective mitigation project. Unfortunately, these data are frequently lacking. Documentation of meadow extent and cover over time is the best means of determining whether planting into unvegetated areas of an existing meadow is likely to provide true mitigation in the context of "zero net habitat loss," or if the planting will provide only a temporary pulse of cover to the system.

Because obtaining the essential data may not be economically or logistically feasible, we have concluded that transplanting of seagrasses should only

be performed into areas where a definable perturbation has occurred (Fonseca et al., 1985). We place emphasis on the use of the past tense in reference to the perturbation. Mechanical disturbance such as dredging must have ceased and physical stability returned to the site before planting can begin. Loss of seagrasses due to diminished water quality (light penetration, temperature, salinity) caused by non-point sources would be more difficult to control (Orth and Moore, 1983). The problem is only accelerated by the loss of seagrass that acts as a filter of suspended material and adds stability to local sediments (Fonseca et al., 1982a, 1983, 1985; Ginsburg and Lowenstam, 1958; Schubel, 1973; Scoffin, 1970; Ward et al., 1984). However, those areas that have had an acute episode of perturbation are often available for effective mitigation through transplantation. A good example of such a situation is the mitigation of seagrass meadow disturbance associated with construction of the bridges through the Florida Keys (Mangrove Systems, Inc., 1985).

Our selection of acutely impacted areas as acceptable seagrass restoration sites generated a need for predictable growth responses of seagrasses under transplanting situations. A clear understanding of how fast an area can be restored with a given seagrass species, and set of environmental conditions, allows the development of a mitigation plan instead of a mitigation concept. Our procedure has been to develop population growth models for seagrasses in order to plan the logistics of planting. The question is not whether seagrasses can be transplanted, but whether restoration can be executed under standards that allow objective assessment of performance and compliance.

METHODS

Study sites were located on perturbed areas in north and south Florida. Two study sites were located in St. Andrews Bay, near Panama City, and Panama City Beach (Figure 1). One site, near Redfish Point (RF) on the west side of Smack Bayou (Figure 1A), was situated in a quiescent backwater fringed by marsh. Large semiconical depressions, down to 4 m deep, were found interspersed with large, unvegetated, subtidal sand areas. The presence of large fragments of metal surrounded by seagrass cover, and the location of the site immediately adjacent to a military installation lead us to speculate that the disturbance had been caused by detonations of high explosives. The

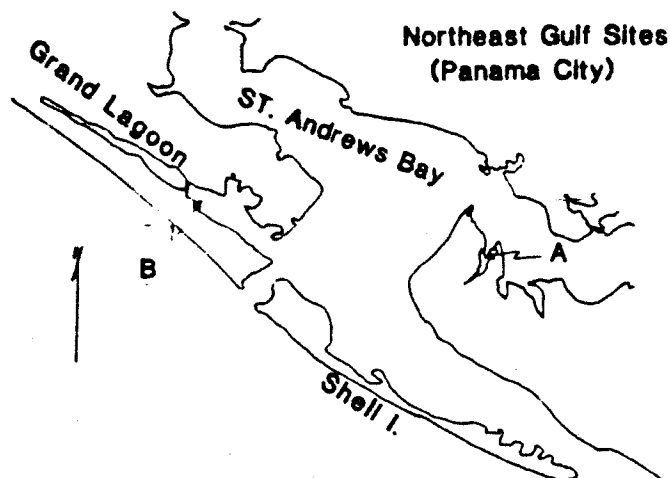


Figure 1. Locations of study sites in the northern Gulf of Mexico. A. Redfish Point site; B. Treasure Island site.

configuration was strikingly similar to other similarly impacted sites (Fonseca and Kenworthy, personal observation).

The other north Florida site (Treasure Island, TI) was located on an old (> 2 yr) dredge material island 300 m east of the bridge that bisects Grand Lagoon (Figure 1B). Approximate longitude and latitude for these sites are 85°40'W and 30°10'N. Dimensions of study plots are given in Table 1.

The remaining study sites were located in Monroe County (Florida Keys), Florida. The four sites, from north to south, were Rock Harbor, Channel Five, Boog Powell Marina spit removal site, and Stock Island power plant (Figure 2A, B, C, D,

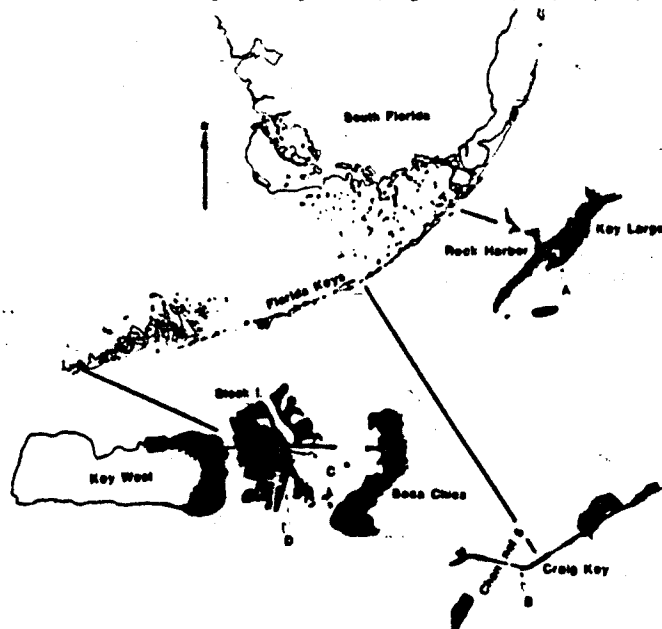


Figure 2. Locations of study sites in the Florida Keys. A. Rock Harbor; B. Channel Five; C. Boog Powell spit removal (unshaded); D. Stock Island power plant.

TABLE 1. PLANTING SITES, NORTHEAST GULF OF MEXICO.

Site Name	Figure 1 Referenced Location	Date planted month/year	Species (source)*	Spacing (m)	No. Planting Units (PU)	Dimensions for Plots (m)
Redfish	A	4/84	<i>Halodule wrightii</i> (nat. freq.)	1	100	1-4x19 grid
	A	4/84	<i>Syringodium filiforme</i> (term.)	1	100	1-4x19 grid
Treasure Island	B	4/84	<i>Halodule wrightii</i> (nat. freq.)	1	100	1-9x9 grid
	B	4/84	<i>Syringodium filiforme</i> (term.)	1	100	1-9x9 grid

*Planting stock collected based on the number of apical meristems on the rhizomes. Natural frequency "(nat. freq.)", indicates no culling was made while terminal: "(term)" indicates only rhizomes with apical meristems were used.

respectively). Each site was substantially different from the others, as described below. Dimensions of study plots are given in Table 2.

The northernmost Keys site (Rock Harbor) was located on Key Largo on a spit removal site in the northeast end of Lake Largo. An illegal road had been built through subtidal habitat, among red mangroves, for fill removal access. The road was removed in 1982 and the area was returned to local subtidal level 6 months prior to planting. The Channel Five site was located on the southward corner of Craig Key, where the channel passes under the Channel Five Bridge. During construction of the Keys bridges, vessels maneuvering barges to the site prop-washed several hundred square meters of carbonate sediment onto an adjacent *T. testudinum* meadow. Planting was done about 2 yr after the deposition. The Boog Powell Marina site was located on the south side of the causeway entering Stock Island from the east. Another fill access road was removed at this site 3 months prior to planting. The southernmost site was at the Stock Island power plant located approximately 1 km south of the Boog Powell site on a large flat directly across from King Point Marina. This area previously was a large *Thalassia* flat, but was altered by thermal effluent from the power plant (R. Lewis, Mangrove Systems, Inc., Tampa, Florida., personal communication). Several years prior to our planting, effluent had been diverted to a site offshore, leaving several acres of flat sparsely vegetated by macroalgae and seagrass.

Seagrasses were transplanted using the technique described by Fonseca et al. (1984). The seagrass transplanting stock was obtained from nearby, natural meadows and consisted of rhizome fragments with at least five short-shoots per rhizome. Some of the stock was culled and made

into planting units whose rhizomes had only intact apical meristems. Other planting units were prepared with stock containing the natural ratio of rhizome fragments with apicals to rhizome fragments without apicals. This was done to determine whether more rapid coverage would be obtained by selecting stock with rhizome apical meristems only. In all cases, the energy regime of the source stock beds was noted using the classification of Fonseca et al. (1983). Winter and spring plantings were done in the Keys, but not in north Florida, where only spring plantings were performed because a clear seasonal growth pattern was observed.

Population growth rate and areal coverage of the transplanted shoots were assessed consistently for all sites. At time 0 (planting day), planting units were arbitrarily selected and counted for numbers of short-shoots and numbers of apical meristems. Those same planting units also were measured for bottom area covered by averaging the width (d, in meters) of the planting on two perpendicular axes, and computing areas as:

$$\pi (d/2)^2 = \text{area (m}^2\text{)}. \quad (1)$$

The number of planting units surviving was counted regularly until establishment was apparent (within 90 days). At each successive survey, ten randomly selected planting units per treatment were counted for number of short-shoots, and the area covered was determined. Data presented are for planting units prior to their coalescence, because the phase of growth up to the point of coalescence is of immediate concern to the transplanter. Coalescence refers to the time when planting units have grown together so that the origins of all shoots cannot be traced to particular planting units.

In this paper, we have grouped all growth and coverage data by geographic location. The goal was

TABLE 2. PLANTING SITES, SOUTH FLORIDA.

Site Name	Figure 2 Referenced Location	Date planted month/year	Species (source)*	Spacing (m)	No. Planting Units (PU)	Dimensions for Plots (m)
Rock Harbor	A	6/83	<i>Halodule wrightii</i> (nat. freq.)	1	100	9x9 grid
		6/83	<i>Halodule wrightii</i> (term.)	1	100	9x9 grid
		6/83	<i>Halodule wrightii</i> (term.)	1/2	290	9x9 grid
			<i>Syringodium filiforme</i> (term.)	1	100	9x9 grid
		6/83	<i>Syringodium filiforme</i> (term.)	1	100	9x9 grid
		4/84	<i>Thalassia testudinum</i> (seedlings)	1	40	(2)2x10 grids
Channel 5	B	2/83	<i>Halodule wrightii</i> (term.)	1	100	9x9 grid
		2/83	<i>Syringodium filiforme</i> (term.)	1	100	9x9 grid
		2/83	<i>Thalassia testudinum</i> (term.)	1	100	9x9 grid
		6/83	<i>Halodule wrightii</i> (term.)	1	50	2x24 grid
		6/83	<i>Syringodium filiforme</i> (term.)	1	50	2x24 grid
Boog Powell Marina	C	2/83	<i>Halodule wrightii</i> (term.)	1	100	9x9 grid
		2/83	<i>Syringodium filiforme</i> (term.)	1	100	9x9 grid
		2/83	<i>Thalassia testudinum</i> (term.)	1	100	9x9 grid
		6/83	<i>Halodule wrightii</i> (nat. freq.)	1	100	9x9 grid
		6/83	<i>Halodule wrightii</i> (term.)	1	100	9x9 grid
		6/83	<i>Halodule wrightii</i> (term.)	1/2	290	9x9 grid
		6/83	<i>Syringodium filiforme</i> (nat. freq.)	1	100	9x9 grid
		6/83	<i>Syringodium filiforme</i> (term.)	1	100	9x9 grid
		4/84	<i>Thalassia testudinum</i> (seedlings)	1	40	(2)2x10 grids
Stock Island power plant	D	2/83	<i>Halodule wrightii</i> (term.)	1	100	9x9 grid
		2/83	<i>Syringodium filiforme</i> (term.)	1	100	8x8 grid
		2/83	<i>Thalassia testudinum</i> (term.)	1	90	

*Planting stock was collected based on the number of apical meristems on the rhizomes. Natural frequency, "(nat. freq.)", indicates no culling was made while terminal, "(term.)" indicates rhizomes with apical meristems were used.

to replicate treatments across as many environmental conditions as possible within a geographic area. By grouping these independently assessed plantings, we provide representative population growth and areal coverage models for the area in question.

RESULTS

Shoot-generation-rate data are presented in Figure 3. At planting, *H. wrightii* had 17-22 short-shoots PU⁻¹ for all natural frequency plantings (approximately 3 of these short-shoots had apicals), and 7.1 short-shoots PU⁻¹ for apical plantings, for an overall average of 13.9 short-shoots PU⁻¹. *S. filiforme* had 5-10 short-shoots PU⁻¹ for all natural frequency plantings (approximately 4 of which had apicals) and 5 short-shoots PU⁻¹ for apical plantings,

with an overall average of 4.9 shoots PU⁻¹. *Thalassia testudinum* had an average of 2.8 short-shoots PU⁻¹. Each *Thalassia* planting unit was composed of a rhizome with an apical meristem; the shoots counted were still attached to the rhizome.

Stock selection made little difference in growth rates. Slopes of regression lines for treatments using only apical shoots of *H. wrightii* show that the transplants grew at almost the same rate as the treatments planted with stock composed of uncultured plants [natural frequency of apical meristems (Table 3)]. The use of natural frequency plantings instead of apical plantings made little difference in the short-shoot generation rate of *S. filiforme* (Table 3). If anything, natural frequency plantings for both species produced shoots at a slightly faster rate than did apicals alone, but this difference appeared marginal.

TABLE 3. REGRESSION EQUATIONS FOR ALL EXPERIMENTAL TRANSPLANT TREATMENTS.

Ln # Shoots PU ⁻¹	= Slope	(days)	+ y-intercept	r	Date Planted	Stock ^{**} Source	TREATMENT Planting Environment	LOCATION Reference Figure & Letter	REGRESSION Reference Figure 3 Letter	
SOUTH FLORIDA										
<i>H. wrightii</i>	= 0.00613	(d)	+	2.5297	0.4217	2/83	LE,T	LE	2 C	A
uncoalesced	0.02703	"	+	1.8819	0.8728	6/83	LE,T	LE	2 C	B
	0.03785	"	+	1.0988	0.8620	6/83	LE,nf	LE	2 C	C
	0.00202	"	+	2.5297	0.1915	2/83	LE,T	HE	2 B	D
	0.03030	"	+	1.7715	0.8721	6/83	LE,T	HE	2 B	E
	0.04657	"	+	1.5540	0.9410	6/83	LE,T	LE	2 A	F
	0.04336	"	+	1.2112	0.9086	6/83	LE,nf	LE	2 A	G
	0.00801	"	+	1.9224	0.5276	2/83	LE,T	LE	2 D	H
combined	0.02584	"	+	1.8776	0.7186					
<i>S. filiforme</i>	= 0.00787	(d)	+	2.2102	0.7069	2/83	LE,T	LE	2 C	A
uncoalesced	0.00754	"	+	1.8713	0.5407	6/83	LE,T	LE	2 C	B
	0.01547	"	+	1.2447	0.7728	6/83	LE,nf	LE	2 C	C
	0.01708	"	+	1.5674	0.7250	2/83	LE,T	HE	2 B	D
	0.00516	"	+	1.7824	0.6067	6/83	LE,T	HE	2 B	E
	0.01491	"	+	1.6599	0.7266	6/83	LE,T	LE	2 A	F
	0.02114	"	+	1.6715	0.7604	6/83	LE,nf	LE	2 A	G
	0.00434	"	+	1.5438	0.3925	2/83	LE,T	LE	2 D	H
combined	0.00766	"	+	1.8504	0.5271					
NORTHEAST GULF										
<i>H. wrightii</i>	= 0.00420	"	+	2.7295	0.2322	4/84	LE,nf	LE	1 A	A
	0.00353	"	+	2.4641	0.2133	4/84	LE,nf	LE	1 B	B
combined	0.00415	"	+	2.5691	0.2365					
<i>S. filiforme</i>	= 0.00315	"	+	1.7707	0.3788	4/84	LE,T	LE	1 B	B
	-0.00053	"	+	1.8154	-0.0695	4/84	LE,T	LE	1 A	A
combined	0.0015	"	+	1.7862	0.1830					
<i>T. testudinum</i> [*]	= 0.00271	"	+	1.0437	0.7490	2/83	LE,T	LE	2 C	A
	0.0	"	+	1.2348	0.0	2/83	LE,T	HE	2 B	B
	0.00241	"	+	0.9302	0.3418	2/83	LE,T	LE	2 D	C
	-0.00174	"	+	1.3863	-0.1356	4/84	LE,T	LE	1 B	D
	-0.00037	"	+	1.3863	-0.0169	4/84	LE,T	LE	1 A	E
combined	0.00274	"	+	1.0143	0.6985					

* *T. testudinum* is for all Florida sites (NE Gulf and S. Florida)

**LE = low energy site, HE = high energy site, nf = natural frequency of terminal meristem occurrence, T = only terminal meristems were used.

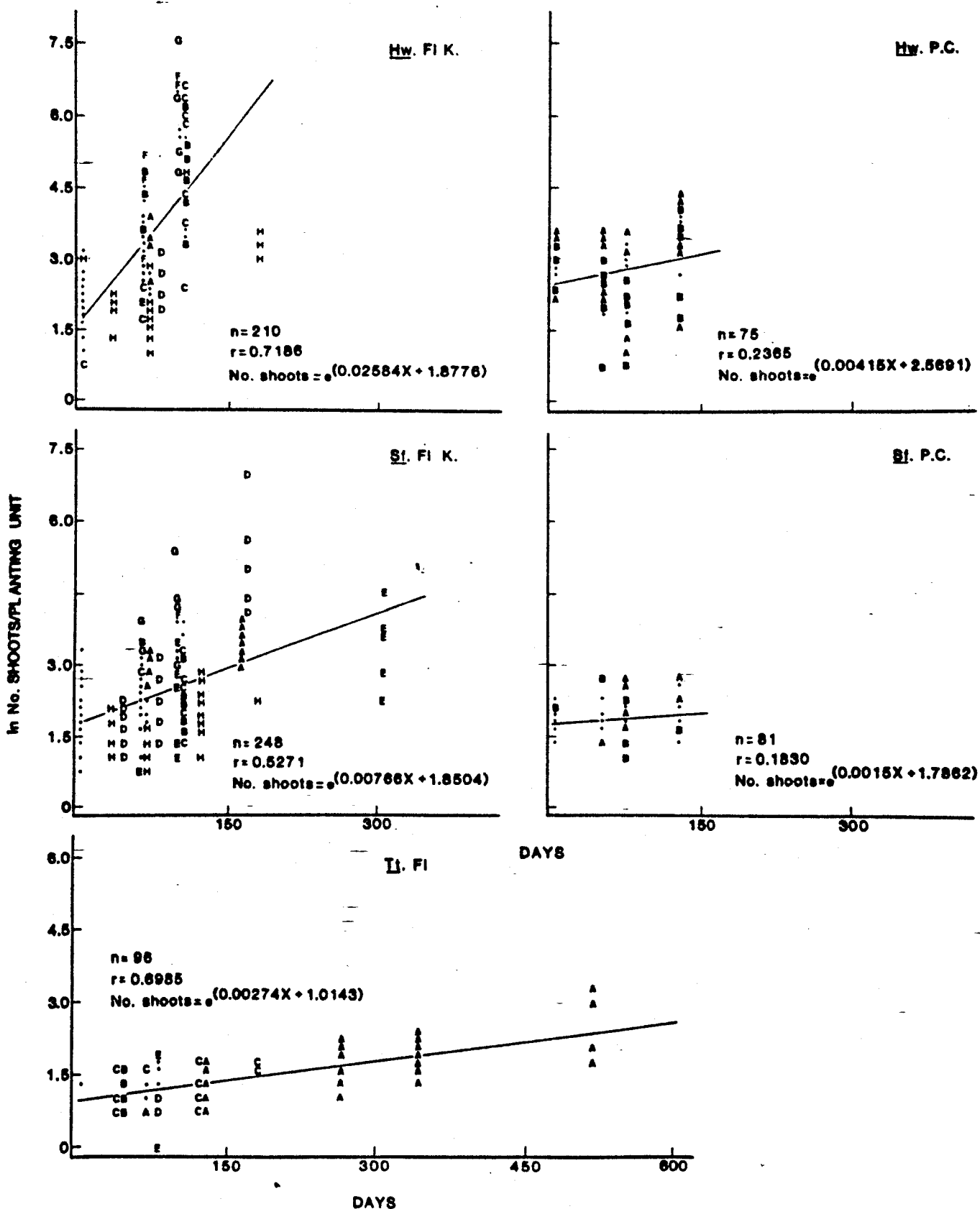


Figure 3. Natural log of number of shoots per planting unit regressed on time. Fl K. = Florida Keys site; Fl = all Florida sites combined; P.C. = Panama City or northern Gulf of Mexico sites. Letters refer to individual planting units at different sites (cross-referenced with Table 3). Dots indicate several overlapping values. Regression equations are for lines shown.

Planting season appeared to have significant impact on short-shoot generation rate for *H. wrightii* and *S. filiforme* in south Florida. Although plantings were successfully performed in both February and June, the slopes of the lines for February plantings were considerably lower than those for June (*H. wrightii*: February mean slope = 0.005386, June mean slope = 0.0366; *S. filiforme*: February mean slope = 0.009763, June mean slope = 0.012844) (Table 3). Because seasonal effects on growth are noticeable even at these latitudes, more northerly sites, which experience a stronger continental influence on climate, should be considered suitable only for spring plantings of these species.

The overall short-shoot generation rate for *T. testudinum* is presented in Figure 3 and for each site in Table 3. Data on *Thalassia* for Florida Keys and Panama City were combined. Because of the low

rate of short-shoot production, no area models were developed for this species.

Areal coverage rates were compiled for un-coalesced *H. wrightii* and *S. filiforme* data only (Figure 4). This was done because, after the planting units began to coalesce and reached 100% coverage in some areas, we were no longer able to document areal spread of rhizomes from individual planting units. Thus, overlap was a problem in areal coverage estimates, but was less so in short-shoot generation estimates, where changes in short-shoot density would have been indicative of a continuing rate of new short-shoot production. The short-shoot generation rate apparently decreased, suggesting that areal spread of rhizomes at coalesced sites may have slowed (Fonseca, unpublished data).

During the monitoring period, not all plantings in south Florida coalesced. Those at Channel 5 and

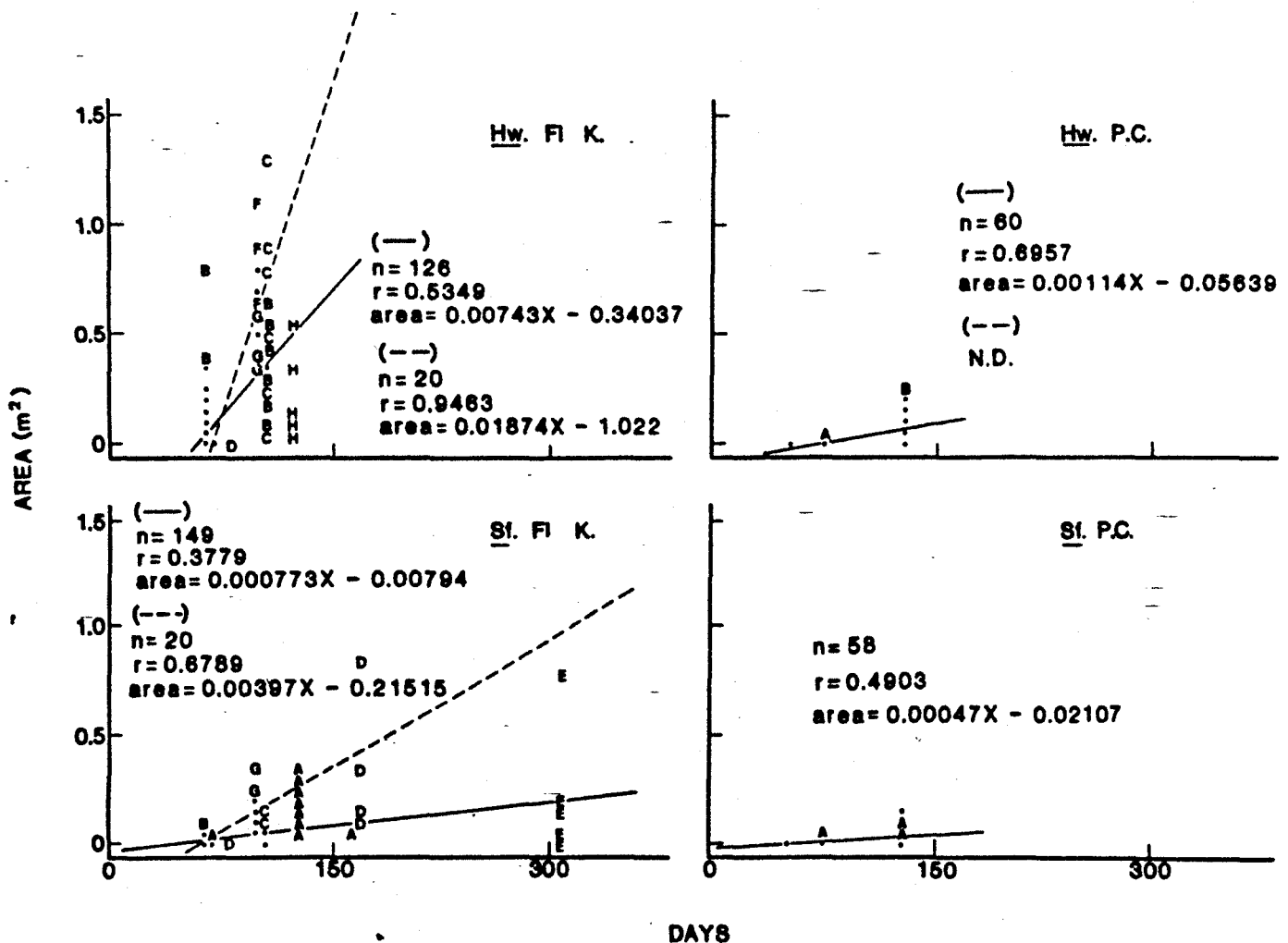


Figure 4. Area in m² planting unit regressed on time. Fl K. = Florida Keys sites; P.C. = Panama City or northern Gulf of Mexico sites. Letters refer to individual planting units at different sites (cross-referenced with Table 3). Dots indicate several overlapping values. Solid regression lines are for all data. Dashed regression lines represent selected, best-growth treatments.

Stock Island did not, while those at Rock Harbor and Boog Powell did.

Syringodium filiforme had a coverage rate proportionally lower than that of *H. wrightii* at the Panama City and south Florida sites. The areal coverage rate for *H. wrightii* at the Panama City site was similar to that observed in North Carolina sites (Fonseca et al., 1985). Since *S. filiforme* does not occur as far north as Beaufort, no comparison could be made.

DISCUSSION

We will discuss short-shoot generation rate in order to directly compare growth responses between species and geographic areas. Areal coverage rates are a product not only of short-shoot generation rate, but also of the interaction of environmental conditions that control rhizome elongation and short-shoot density. Due to this interaction, direct comparisons of areal coverage models are less useful in examining actual population responses than are short-shoot numbers alone. Areal coverage data are presented as the basis for planting arrangements described later.

The growth models suggest the need for extensive planting efforts at some sites that might be recognized as highly suitable for transplanting. This is because suboptimal treatments were included in the regression line (e.g., Stock Island). Highly suitable sites may be dredged sites specifically engineered to suit seagrass growth, or sites that have been damaged from short-term impacts and need assistance in rapid revegetation. We also have plotted the best-case treatment in our presentation of areal coverage results. When a best-case line is absent, we have not found sufficient qualitative differences in areal coverage between treatments to warrant separate lines.

These "best-case" lines should be used with great caution. For example, where a site appears suitable for planting, and adjacent meadows exist, the "best-case" line might be used. This means fewer planting units are needed and overall costs are reduced, because optimum population growth is expected. If the transplant grows optimally, the transplanter has achieved a substantial cost savings. However, if for some reason the transplant does not perform as expected, the transplanter will have far fewer planting units in the field than necessary to achieve coverage. Since there is no way to guarantee

vegetation, these decisions should be made in consultation with experienced personnel.

Population growth data for *T. testudinum* (Figure 3) indicate that even after 600 days, the average number of short-shoots per planting unit would be only 14. This is corroborated by other studies (Derrenbacker and Lewis, 1982) and by our observations of natural seedling growth. *Thalassia testudinum* is clearly not the primary species to use in transplanting operations. The slow growth means that destruction of *T. testudinum* systems will take years to mitigate, while *H. wrightii* and *S. filiforme* could coalesce from plantings on 1 m² centers in less than 200 days. If, using these regression models, we compute the number of short-shoots produced PU⁻¹ at the northern Gulf of Mexico sites in one growing season (250 days), we could expect *H. wrightii* to have > 1000 while *S. filiforme* would have 160.

Area-coverage-rate data suggest that natural meadows of *H. wrightii* and *S. filiforme* in the northern Gulf of Mexico could take substantially longer to recover naturally than areas farther south, such as the Florida Keys. Each of the northern Gulf sites has only one regression line because no distinct "best-case" or higher coverage rate treatment occurred: all treatments spread at similar rates. Growth of *H. wrightii* in the northern Gulf of Mexico is similar to that observed for Beaufort, North Carolina, sites (Fonseca et al., 1985). This is likely due to the similar climatic conditions of the two areas. Several factors—proximity to warm water, latitude, and influence of continental air masses—cause populations to have dramatic summer growth, a shortened overall growing season, negligible winter growth, and sometimes complete foliar die-back to the rhizomes (personal observation). The seasonal response and data from Carangelo et al. (1979) suggest that Gulf of Mexico plantings of *H. wrightii* and probably other marine angiosperms must be initiated in the spring and cannot be planted year-round, as was stated by Phillips (1980). Spring plantings should be done north of Tampa and west to the Rio Grande (at least). Plantings south of this latitude can be executed any time of the year, but our data suggest that even south Florida plantings do better if started in the early spring.

The data on population growth rates show several other sources of variation that contributed to low *r* values for the combined regression models (Table 3, Figure 3). The sites in the Keys were

during different seasons of the year. However, even with the degree of variation observed, clear overall differences in population growth and coverage rates exist among the three species. The question posed earlier as to whether an objective assessment of restoration can be performed, may now be addressed.

Our data allow standardization of the approach to determining performance and compliance. With standardization, restoration projects involving seagrass transplants may be objectively assessed. To begin with, these data provide conclusive evidence that the primary candidate species are *Halodule wrightii* and *Syringodium filiforme*, and not *Thalassia testudinum*. Although percent-survival of *Thalassia* transplants may be quite high, their slow coverage rate is unacceptable for a primary species in habitat restoration. If the goal of a mitigation project is to restore habitat function, then that should be done as quickly as possible. Again, the slow coverage rate of *Thalassia* disqualifies it from rapidly restoring habitat function unless it is transplanted in extremely dense arrangements with accordingly high costs.

Our data indicate that to achieve complete cover of *T. testudinum*, at an ambient density of 300 short-shoots m^{-2} (which is sparse), plantings on 1 m centers would require 4.0-4.7 years of growth. A similar planting of *H. wrightii* or *S. filiforme* may take as little as 160-250 days to achieve ambient densities. As representatives of an agency charged with managing marine fisheries, we would much prefer that any seagrass cover be established as quickly as possible. We recognize, however, that not all seagrass species are ecological equivalents. We suggest endorsement of the approach used by Derrenbacker and Lewis (1982) where the faster-growing species are established first, followed by the seeding or sprigging of *Thalassia* after the meadow has coalesced. This second step is intended to accelerate the successional sequence from *Halodule/Syringodium* to *Thalassia* (*sensu* Derrenbacker and Lewis, 1982).

Some areas will not be suitable for supporting the original seagrass species after habitat alteration has occurred. This was the case in the area of the Keys Bridges, where significant acreage was rendered unsuitable for restoring *T. testudinum* because of shallow sediment depths. Some areas could not be planted with any seagrass due to bridge shading or absence of unconsolidated substrate (Mangrove Systems, Inc., 1985). We suggest that, in these cases, planting of any seagrass that can be established (species substitution) will have more fishery

waiting for natural recolonization, which also may take years.

The population growth data presented here may be used to accurately project the date at which the performance standard should be achieved. The performance standard is really the replacement ratio. A minimum of 1:1 (ratio habitat acreage created/acreage destroyed) is recommended, but a 1.5:1 or 2:1 ratio may help offset low transplant survival and lost productivity during the period between destruction and restoration of habitat functions. Transplants do not begin to function as a natural meadow at the planting date, but, more likely, when the replacement ratio has been achieved.

The population-growth-rate data show that, in the southern part of Florida, a performance standard can be achieved more quickly and with less cost (less plantings needed) than in the northern Gulf. In the northern Gulf of Mexico, *H. wrightii* and *S. filiforme* would take approximately 2 years (2 growing seasons) to obtain optimum cover, while in the virtually continuous growing season of south Florida, they would only require approximately 150 days, with all plantings placed at 1 m intervals in sheltered areas. What this means is that, depending on the latitudinal location of the mitigation site, more or less time or planting units will be required to achieve compliance. Concomitant development plans should be staged accordingly.

Development plans can be more efficiently organized if planting tables are based on these types of data (Fonseca et al., 1982b, 1984, 1985). Area covered per planting unit can be computed from the regression lines given in Figure 4. The number of planting units needed for the site can be computed by dividing the area of the planting site (m^2) by that individual area at time t . Spacing between planting units is the square root of site area divided by the number of planting units. If the costs associated with planting that amount of material are too high (Fonseca et al., 1984), then a longer time should be selected so fewer planting units will be required. Associated time-dependent plans then can be adjusted accordingly. These planting computations also allow accurate planning of logistics to implement the actual transplanting operation, enhancing efficiency and reducing operation costs.

Another feature of using these growth models is that a quantitative monitoring program to determine compliance can be implemented. This monitoring is the responsibility of the permit applicant and should be included in the permit condition.

agencies to gain control of the mitigation process, which, if unmonitored, could develop costly problems. Monitoring population growth and coverage can be done inexpensively with the appropriate statistical approach. This process allows the applicant to gauge the progress of the planting towards compliance. For example, if many transplants fail to survive initially, due to erosion (a common problem), replanting could ensure that sufficient numbers survive to generate the required coverage in the time allowed. This may be simply extrapolated by multiplying the average area of individual planting units at the time when compliance coverage was to be achieved (computed from area regression, Figure 4) by the number surviving at any earlier time. Even given the variation of the regression lines, this provides an objective indication as to whether or not vegetation is sufficient to meet compliance in the prescribed time. Also, the earlier a supplemental planting occurs, the less densely it will have to be planted.

Monitoring also provides early and continuous feedback into the restoration process. Our experience has been that when the time came to approve a mitigation, plant coverage was insufficient for approval. This and numerous other aspects of a mitigation project's development require immediate action for economic reasons. Replanting is costly in both time and money. These expensive conflicts could be avoided by requiring a monitoring program including periodic reports on all seagrass mitigations.

The monitoring phase also provides the final and most crucial piece of data: the area covered when compliance is achieved. This information provides the only means by which the permitting agency can make an objective, defensible decision as to whether the mitigation effort should be approved. This seems to us to be little to ask, since, presently, no data show that seagrass mitigations actually constitute restoration of the system. The entire mitigation concept is based on the premise that equal restoration of vegetation means equal restoration of functional fishery habitat—a logical, but unsupported, precept.

Finally, these data argue for another form of mitigation instead of restoration, and that is conservation. This is a standpoint that most regulatory agencies embrace, but are required to compromise under specific circumstances (e.g., public interest projects). Our data suggest two important points to consider when permitting destruction of existing seagrass meadows. First, *T. testudinum* meadows

data clearly demonstrate that restoration of *Thalassia* meadows is a long-term affair, probably a stepwise process that is inherently costly. Applicants must be made aware of the problems involved. Second, *Thalassia* meadows are often very old systems—hundreds of years old in some cases. To trade off a long-standing, highly productive habitat for a restoration process that has a tenuous track record with this target species, seems to be a dubious management practice. Further, in the northern Gulf of Mexico, restoration of even *H. wrightii* and *S. filiforme* meadows may take at least two years to fully establish ambient cover. Disruption of the production of fishery organisms is likely to have downstream effects long after required cover is achieved. These points must be addressed with greater resolve if agencies are to refrain from seagrass meadow disruption and to engage in serious research of the functional values of restored seagrass systems.

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