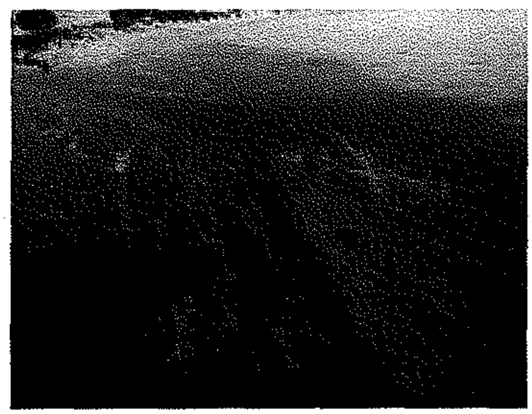
Science for Solutions

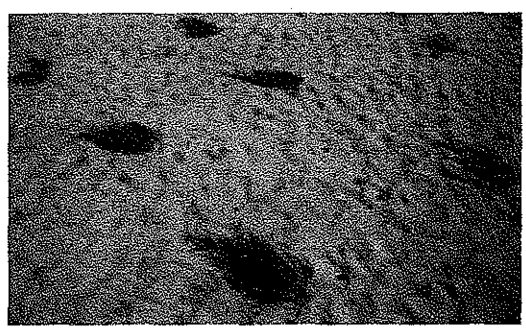
NOAA'S COASTAL OCEAN PROGRAM
Decision Analysis Series No. 12

Guidelines for the Conservation and Restoration of Seagrasses in the United States and Adjacent Waters

Mark S. Fonseca,
W. Judson Kenworthy, and
Gordon W. Thayer

November 1998





U.S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
Coastal Ocean Office

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Cover photographs (clockwise from top right): Propeller scarring in the Florida Keys National Marine Sanctuary (courtesy C. Kruer); 250 -day old eelgrass *(Zostera marina)* plantings on a wave-swept shoal near Beaufort, NC (M. Fonseca); Alaskan eelgrass bed adjacent to a restoration site (courtesy US Army Corps of Engineers); eelgrass planting in San Diego Bay, California (courtesy G. Robilliard).

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Note to Readers

Guidelines for the Conservation and Restoration of Seagrasses in the United States and Adjacent Waters was developed by Mark Fonseca of NOAA's Beaufort Laboratory, along with Jud Kenworthy and Gordon Thayer, with funding from NOAA's Coastal Ocean Program. The document presents an overview of the current state of seagrass conservation and restoration in the United States, discusses important issues that should be addressed in planning seagrass restoration projects, describes different planting methodologies, proposes monitoring criteria and means for evaluating success, and discusses issues faced by resource managers.

The Coastal Ocean Program (COP) provides a focal point through which NOAA, together with other organizations with responsibilities for the coastal environment and its resources, can make significant strides toward finding solutions to critical problems. By working together toward these solutions, we can ensure the sustainability of these coastal resources and allow for compatible economic development that will enhance the well-being of the Nation now and in future generations. The goals of the program parallel those of the NOAA Strategic Plan.

A specific objective of the COP is to provide the highest quality scientific information to coastal managers in time for critical decision making and in formats useful for these decisions. To help achieve this, the COP inaugurated a program of developing documents that would synthesize information on issues that were of high priority to coastal managers. As a contribution to the Decision Analysis Series, this report provides a critical synthesis of the methods for planning, planting, monitoring, and evaluating seagrass restoration projects. A list of available documents in the Decision Analysis Series can be found on the inside back cover.

As with all of its products, the COP is very interested in ascertaining the utility of the Decision Analysis Series, particularly in regard to its application to the management decision process. Therefore, we encourage you to write, fax, call, or E-mail us with your comments. Please be assured that we will appreciated these comments, either positive or negative, and that they will help us direct our future efforts. Our address and telephone and fax numbers are on the inside front cover. My Internet address is COASTALOCEAN@COP.NOAA.GOV.

David Johnson

David Johnson
Acting Director
NOAA Coastal Ocean Program

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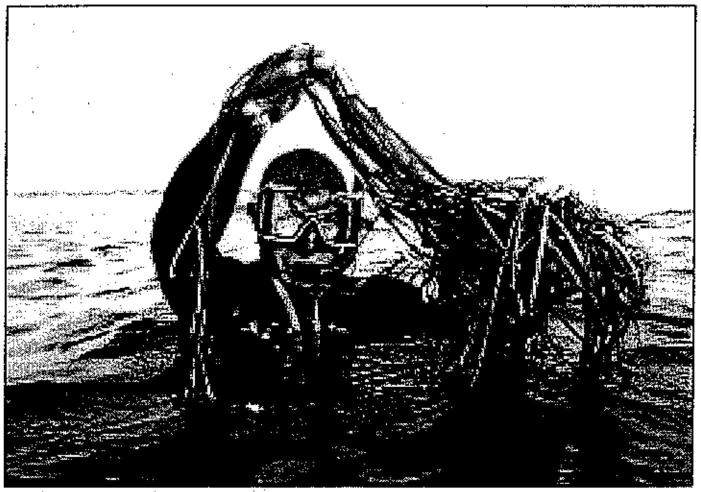
Executive Summary

- Seagrass ecosystems are protected under the federal "no-net-loss" policy for wetlands and form one of the most productive plant communities on the planet, performing important ecological functions.



- Seagrass beds have been recognized as a valuable resource critical to the health and function of coastal waters. Greater awareness and public education, however, is essential for conservation of this resource.

- Tremendous losses of this habitat have occurred as a result of development within the coastal zone. Disturbances usually kill seagrasses rapidly, and recovery is often comparatively slow.

- Mitigation to compensate for destruction of existing habitat usually follows when the agent of loss and responsible party are known. Compensation assumes that ecosystems can be made to order and, in essence, trades existing functional habitat for the *promise* of replacement habitat.

- While planting seagrass is not technically complex, there is no easy way to meet the goal of maintaining or increasing seagrass acreage. Rather, the entire process of planning, planting and monitoring requires attention to detail and does not lend itself to oversimplification.

- The success rate of permit-linked mitigation projects remains low overall, but this appears to result from failures in the planning process as much as any other cause. To prevent continued loss of habitat under compensatory mitigation, decisive action must be taken by placing emphasis on improving site selection, compliance, generating desired acreage, and maintaining a true baseline.

- Seagrass planting is no longer experimental, but planting will not succeed unless managers appreciate and emphasize the extreme importance of site selection, care in planting, and incorporation of plant demography into the planning and planting processes.

- Seagrass beds can be restored, but preservation is the most cost-effective course of action to sustain seagrass resources. Although techniques and protocols exist that produce persistent seagrass beds, they are often applied inconsistently, which has resulted in large-scale failures.

- A logical and ecologically defensible goal is to attain replacement of the lost seagrass species with an area of bottom coverage that compensates for interim lost resource services and a comparable shoot density. Seagrass plantings that persist and generate the target acreage have been shown to quickly provide many of the functional attributes of natural beds.

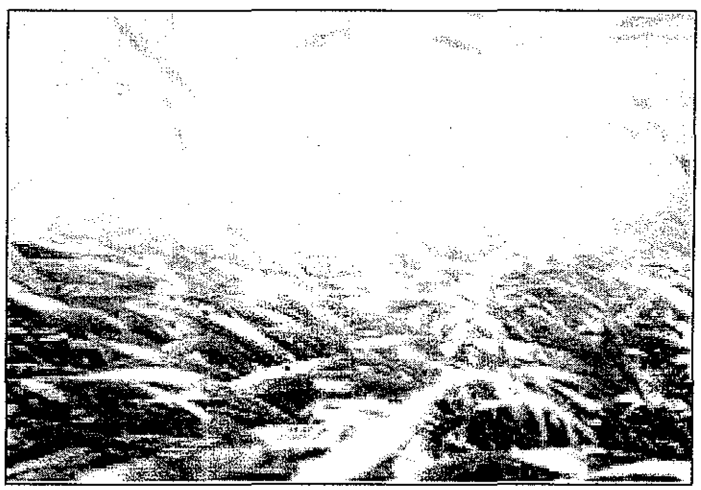
- When destruction of the impact site requires planting in another location (i.e., off-site) it is often difficult to find a site elsewhere with suitable biological and physical parameters required for seagrass growth and persistence.

- As more information is made available to managers regarding the function of seagrass ecosystems and the difficulties involved in mitigating for their loss, fewer permitted impacts are occurring with more emphasis placed on impact avoidance and minimization.

CHAPTER 1

Background

WHAT ARE SEAGRASSES?



Seagrasses are unique marine flowering plants of which there are approximately 60 species worldwide (den Hartog 1970, Phillips and Menez 1988). With the exception of some species that occur in the rocky intertidal zone, they grow in shallow, subtidal or intertidal unconsolidated sediments. Thus, they bind millions of acres of shallow water sediments in the coastal waters with their roots and rhizomes while simultaneously baffling waves and currents with their leafy canopy (Ginsberg and Lowenstam 1958, Taylor and Lewis 1970, den Hartog 1971, Fonseca et al. 1983, Fonseca 1996a). In this manner the canopy inhibits resuspension of fine particles and traps water-column-borne material (Ward et al. 1984, Short and Short 1984), clearing the water column. This cleansing effect extends to water column nutrients as well. Nutrient uptake by seagrass blades and their associated epiphytes and macroalgae as well as roots incorporate dissolved nutrients into plant biomass, which can improve water quality (Harlin and Thorne-Miller 1981). The baffling effect of the canopy on sediment stabilization is enhanced by the presence of a robust root and rhizome mat, although the relative contribution of the mat has not been isolated from canopy baffling in its role of sediment stabilization (Fonseca 1996a). The physical stability, reduced mixing and shelter provided by the complex seagrass structure provides the basis for a highly productive ecosystem (Wood et al. 1969). Overall the importance of seagrasses and their role in many coastal ecosystems has been extensively documented (see reviews by Thayer et al. 1975, Phillips 1982, Zieman 1982a, Thayer et al. 1984, Zieman and Zieman 1989) and the nature of their general function and high resource value are no longer an issue.

Seagrasses occur in all coastal states of the U.S. with the apparent exception of Georgia and South Carolina where freshwater inflow, high turbidity and tidal amplitude combine to prevent their occurrence. There are at a minimum thirteen species of seagrass currently recognized to occur in U.S. waters (Table 1.1). The presence of a fourteenth species, *Zostera asiatica* on the West Coast remains a subject of debate (Phillips and Wyllie-Echeverria 1990). We will not include in this discussion seagrass species occurring in U.S. possessions in the Pacific Ocean because little is known about their status; through NMFS Southwest Regional Office reports, we know that *Enhalus acoroides* and *Halodule uninervis* occur on Rota Island and Saipan Island in the Pacific Territories. Also, Phillips and Menez (1988) list *Halophila ovalis* and *Halophila minor* (fifteenth and sixteenth species) as species that occur in Hawaii. *Halophila hawaiiana* is also reportedly present on Hawaii (K. Bridges, Univ. Hawaii, pers. com.). Drawings of the major U.S. species are given in Figure 1.1. One species, *Halophila johnsonni* was only recently described as a separate species despite its occurrence in the heavily-studied region of southeast Florida. Because of its limited distribution, this species is currently under consideration for listing as a threatened species as defined by the Endangered Species Act. Another species, *Zostera japonica* was recently introduced to the Pacific Northwest . It is spreading and tends to colonize shallow intertidal flats, converting them from their historical ecological status as mudflats to intertidal eelgrass habitat (Harrison and Bigley 1982, Pawlak 1994).

Although recognized for their value where they occur, the distribution of seagrass is not as well known as it should be for proper management (Wyllie-Echeverria et al. 1994a). Moreover, knowledge of population-level temporal dynamics is only rudimentary at best. We know that at least 90 percent of the southeast United States seagrass acreage (~1.1 million hectares) exists in the Gulf of Mexico (Orth and Van Montfrans, 1990). But nationally, the distribution and abundance of two genera in particular have been overlooked. The full extent and function of the reported ~400,000 hectares of seasonal *Halophila* beds off the west coast of Florida (Iverson and Bittaker 1986) is unknown. Similarly the distribution of the Hawaiian *Halophila* is not reported. Also, very little is known about local distribution (distribution meaning localized, specific locations of beds, not the range of a species) of a unique West Coast dominant, the rocky intertidal *Phyllospadix* spp., although work has been done regarding its population ecology (Turner 1985, Turner and Lucas 1985). The distribution of seagrass on the West Coast, including both Alaska and Hawaii, has not been systematically compiled to the degree seagrasses have on the east and Gulf coasts

Table 1.1. List of seagrass by family, genus and species, and common names (if given) that are found in the United States and adjacent waters. Species marked with (?) are not fully documented as occurring in U.S. waters.

Family, Genus, and Species	Common Name[a]
Hydrocharitaceae	
Enhalus acoroides Royle	
Halophila decipiens Ostenfeld	paddle grass
Halophila engelmanni Ascherson	star grass
Halophila hawaiiana Doty and Stone	*Hawaiian* seagrass[a]
Halophila johnsonii Eiseman	Johnson's seagrass
Halophila minor (Zollinger) den Hartog?	unknown
Halophila ovalis (R. Brown) Hooker f.?	unknown
Thalassia testudinum Konig	turtlegrass
Potamogetonaceae	
Halodule wrightii Ascherson	shoalgrass
Halodule uninervis?	
Phyllospadix scouleri Hook	*Scouler's* seagrass
Phyllospadix torreyi S. Watson	*Torrey's* seagrass
Phyllospadix serrulatus Ruprecht et Ascherson	surfgrass
Ruppia maritima L.	widgeon grass
Syringodium filiforme Kutz	manatee grass
Zostera japonica Ascherson et Graebner	*Japanese* eelgrass
Zostera marina L.	eelgrass
Zostera asiatica?	*Asian* eelgrass

[a] Italics on common names indicate suggested common names; R. Phillips, Battelle Laboratories, Richland, Wa., pers. com.

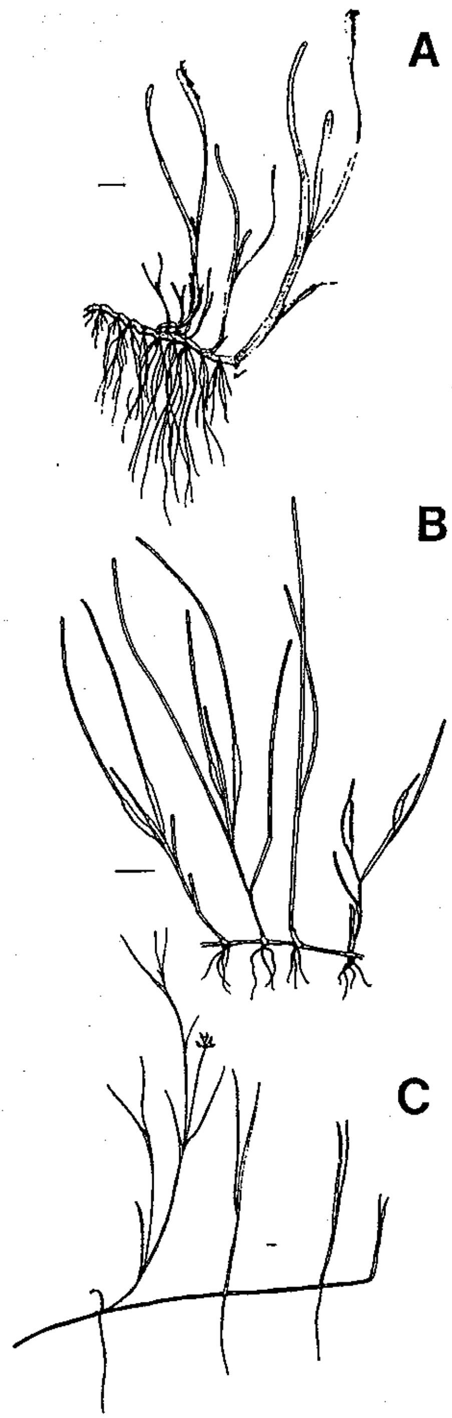
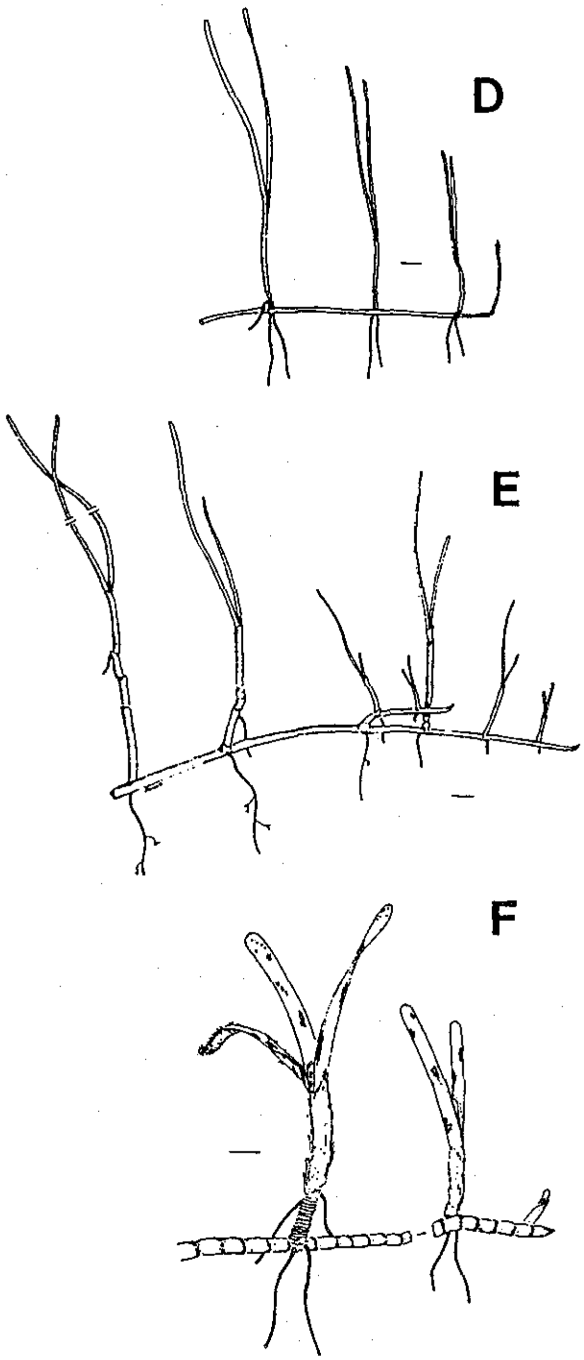


Figure 1.1. Drawings of most seagrasses found in U.S. waters (taken from Phillips and Menez 1988 and Fonseca 1994). All scale bars are set at 2cm and thus vary with seagrass species. A=Zostera marina; *B*=Zostera japonica; *C*=Ruppia maritima; *D*=Halodule wrightii; *E*=Syringodium filiforme; *F*=Thalassia testudinum; *G*=Halophila engelmanni; *H*=Halophila decipiens; *I*=Halophila johnsonni; *J*=Phyllospadix serrulatus; *K*=Phyllospadix torreyi; *L*=Phyllospadix scouleri.



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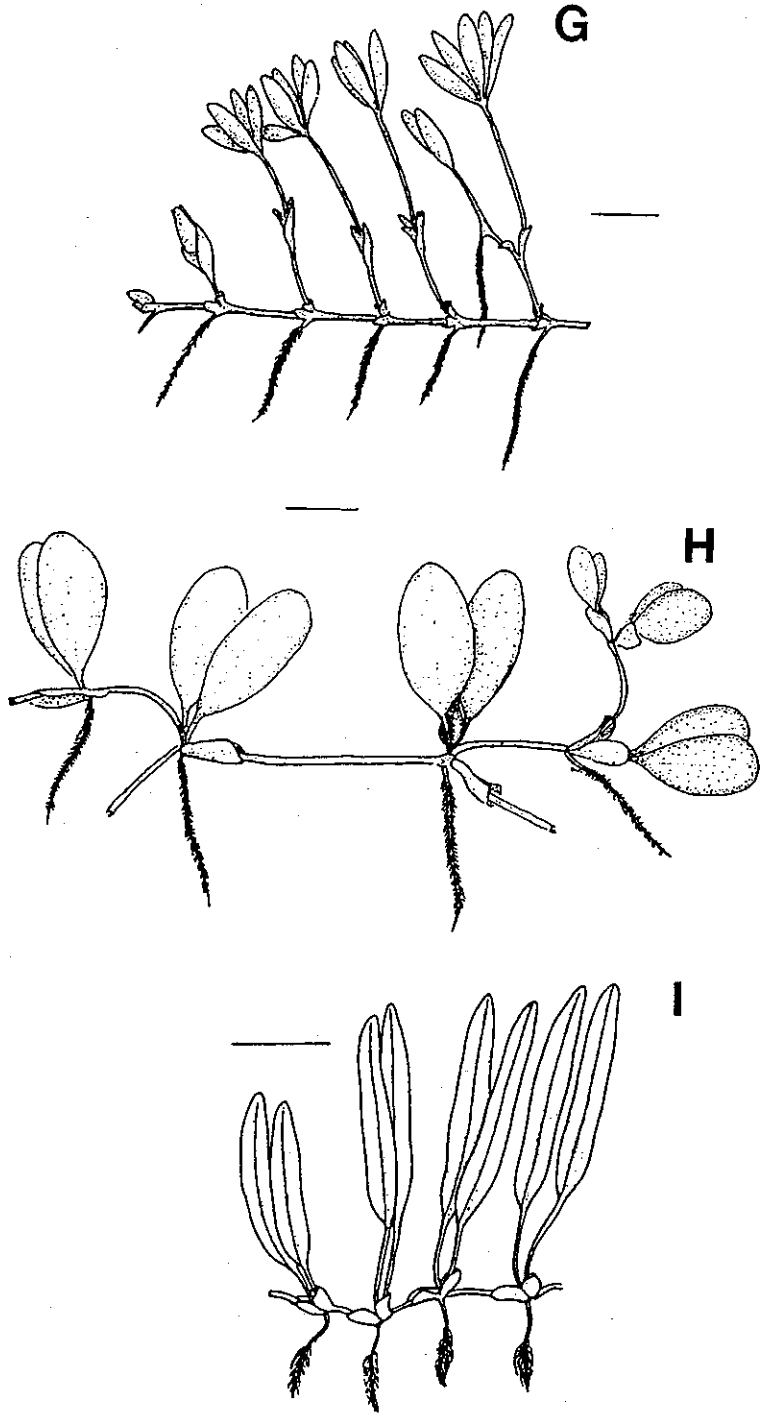

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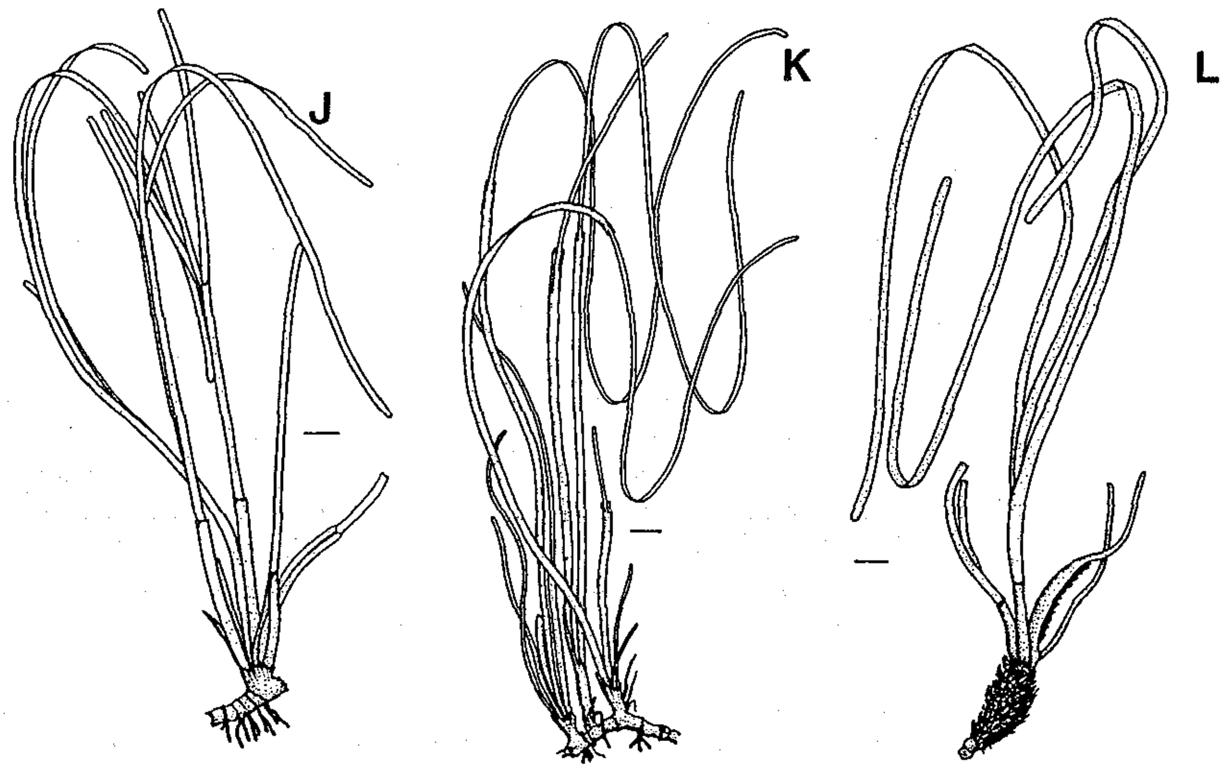


Figure 1.1. continued.

although the general range of species' distributions has been reported (Wyllie-Echeverria and Phillips 1994).

Historically, emphasis been placed on aspects of seagrass primary and, to a lesser degree, secondary production attributes (see descriptions in Zieman 1982a, Phillips 1984, Thayer et al. 1984). Extensive information is available regarding light and nutrient requirements of seagrasses (Kenworthy and Haunert 1991, Dennison et al. 1993, respectively). Seagrasses are flowering plants and much attention has been paid to the mechanics of pollination and seed dispersal (see review by Cox 1993 and references therein) but much less is known about the role of seeding in bed maintenance or colonization of new areas (Kenworthy et al. 1980, Harrison 1993, Orth et al. 1994). With the exception of some recent studies (Duarte et al. 1994, Durako 1994) and previous transplanting data sets (Fonseca et al. 1987c), demographic studies have been sorely neglected in this country yet this is a topic area where managers ask many questions: How quickly will a seagrass bed recover from a given impact? Is planting necessary? Given intrinsic recovery rates and transplanting success, how do we compute replacement ratios or estimate interim loss? Should we be concerned about genetic diversity of the population? These questions are only now being addressed.

Defining Seagrass Habitat

Seagrass beds exist in a wide variety of physical settings that lead to different coverage patterns. The problem is coming up with a consistent definition of what constitutes a seagrass bed. Although small patches may themselves have significant resource value, how does one assess the collection of patches and determine the boundaries of a seagrass habitat? Seagrasses exhibit a variety of growth strategies and coverage patterns which occur from rocky and soft-bottom intertidal habitats to depths of at least 40 meters. Some species can rely heavily on seeding to ensure year-to-year survival (e.g., *H. decipiens* and possibly *H. engelmanni*) meaning that surveys during winter months would need to include sediment seed bank assessments to accurately define the presence of a seagrass bed. Moreover, some species, such as *Z. marina*, can exist either as perennials or annuals, again requiring very different assessment strategies, varying between seed bank and vegetative material depending upon time of year. Clear knowledge of seagrass population ecology is a requirement for effective management and planting; that is, one-time snapshot inventories are a very, very poor basis upon which to delineate seagrass habitat.

Seagrass beds move. Depending on the species and the physical setting, the rate at which portions of the seafloor switch from vegetated to unvegetated may vary on the scale of days or decades, meaning that the amount of open seafloor required to maintain patchy seagrass beds is greater than the coverage by the seagrass itself at any one point in time (Figure 1.2), sometimes by a factor of two (i.e., over time, the movement of seagrass beds means that they will soon occupy at least twice the presently unvegetated bottom evident at any one survey time). Thus, if unvegetated areas among existing patches of seagrass are converted to channels, the long-term (within four years, unpubl. data) baseline acreage of seagrass in the vicinity of the converted habitat, will decline. Therefore, seagrass habitat must be recognized as including not only continuous cover beds, but chronically patchy habitat; a policy that requires considering the (presently) unvegetated spaces between seagrass patches as seagrass habitat as well. Management of seagrass resources therefore depends on understanding the spatial and temporal dynamics of seagrass coverage.

One of the biggest problems regarding delineation of seagrass habitat relates to the choice of sampling scale during the process of inventory, especially prior to a planned impact to a seagrass bed (see section, "Spatial Scale and its Role in Defining Seagrass Habitat," below). Scale is roughly defined here as the variation of pattern as a function of the range and resolution of examination. The scale at which assessments of seagrass coverage take place varies tremendously, depending on some covariate of acreage, interest and time available to conduct surveys. In contrast, after a planting is installed, monitoring of seagrass plantings is less prone to scale problems as direct count methods are usually employed and statistical sub-sampling protocols can be instituted to ensure adequate sampling intensity. However, assessment of existing natural seagrass and post-coalescent seagrass plantings takes place at many spatial scales and this leads to very different values of seagrass abundance. If aerial photographs are used, the altitude of the airplane, the camera lens, film, solar angle, water turbidity, and wind waves affect the ability to detect seagrass beds, particularly at the lower end of their depth distribution. Similarly, if one chooses to survey a potential impact site from the deck of a small boat then wavelets, reflectance, turbidity and an individual's search image all influence ability to assess seagrass abundance. Aerial photography such as that recommended by the NOAA Coastal Change Analysis Program (C-CAP) (Dobson et al. 1995), has a minimum mapping unit of 0.03 ha. At that resolution roughly 37 percent of the permits issued for alter-

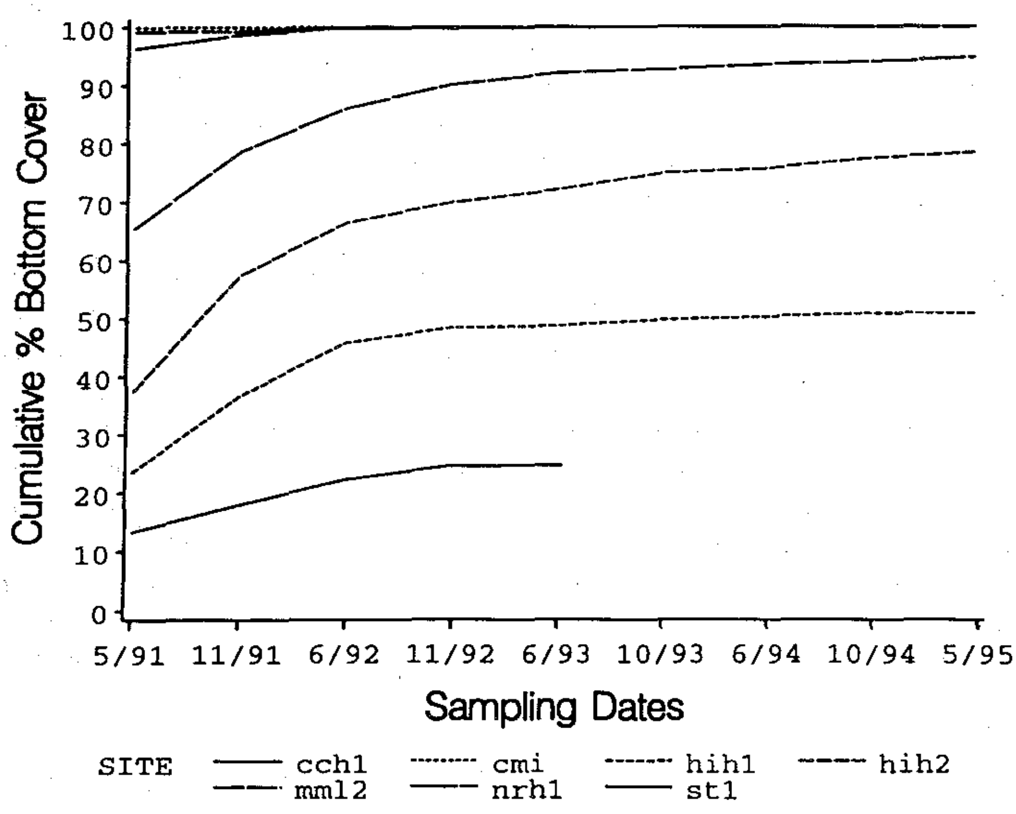


Figure 1.2. Plot of the cumulative area of bottom covered in 50 x 50m survey areas over time. Y-axis = cumulative cover assessed by adding new square meters of cover to that not previously covered in any survey. X-axis = sampling dates. Each line type represents a different 50 x 50m site.

ation of submerged aquatic vegetation habitats could not be detected. Fortunately, those that could be detected with 0.03 ha resolution accounted for ~99 percent of the acreage impacted (Rivera et al. 1992). Inherently patchy seagrass beds would be even more difficult to detect and quantify at a spatial resolution less than 0.03 ha using C-CAP techniques. These scales < 0.03 ha are spatial scales that questions of planting unit (PU) spacing and groupings of PU must be addressed (see section on "Spacing of Planting Units"), and persistent seagrass patches can be produced at these smaller scales.

Fonseca (1989a) suggested that at the 1:24,000 scale of aerial photography when the ratio of average seagrass patch diameter to the distance between patches exceeds 50:1, seagrass habitat continuity no longer fosters cognitive recognition by a viewer as constituting seagrass habitat. He suggested that above that ratio the area should no longer be considered continuous seagrass habitat. Clearly this ratio is scale dependent. If a ratio of 50 shoot widths to the distance between shoots were used, then many seagrass beds on the West Coast and in the northeast where individual plants are very large (> 2 m length) would no longer be considered seagrass habitat even though the unit area biomass might be comparable to other seagrass beds in the country. Unfortunately, we are not aware of any quantitative description of how bed boundaries are interpreted (i.e., when a bed is drawn as one large polygon or many small polygons). However, variation in seagrass bed form can easily be visually detected from low-level aerial reconnaissance (Figure 1.3), and appears to be correlated with exposure to waves and currents. Under wave and current conditions beds can take extreme forms; Molinier and Picard (1952) and Fonseca (1996a) described vertical walls of *Posidonia* and *Zostera*, respectively, revealing the extent to which seagrass could reduce erosion and enhance sediment accumulation. Seagrass patterns also change, revealing areas of seagrass coverage loss and gain at meter scales within short time periods (months) (Figure 1.4) attesting to the consistent ability of seagrasses to stabilize sediments. For at least 20-30 years after Molinier and Picard's work, little in the way of a quantitative association of seagrasses' effects on water motion and, conversely, the effect of water motion on seagrass bed development took place. During this time, interest in the physical processes occurring in seagrass beds was confined largely to qualitative descriptions of their geological role and, to a lesser degree, the implications of this geological stability on animal utilization.

It is unlikely that there will be a universal standard for defining seagrass habitat. Different seagrass species form beds that occupy too great a diversity of habitats and exhibit such a range of life history strategies that a universal definition would almost certainly be restrictive and unworkable. Further, published data on seagrass biomass, density, and structural complexity (e.g., surface area) have tended to be collected from

Figure 1.3. Aerial photograph of mixed Halodule wrightii, Ruppia maritima, Syringodium filiforme, *and* Thalassia testudinum *beds on the western margin of Tampa Bay, Florida. In the foreground at the bayward edge of the shoal are what appear to be wave-sculpted beds while further landward, in shallow water are more continuous cover bed. Reduction in wave energy from both the shelving shoal and the grass itself is thought to be responsible for the resultant seagrass bed landscape pattern. Taken from Fonseca (in press).*

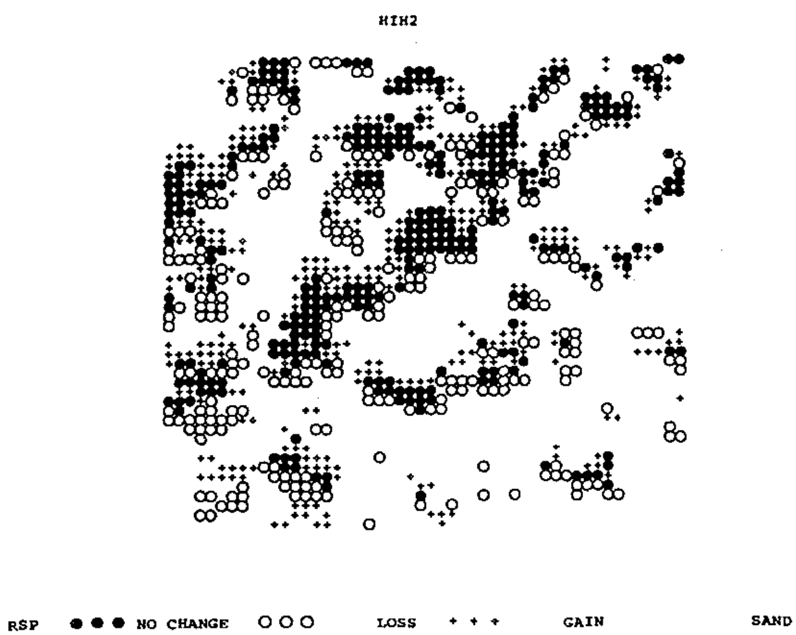


Figure 1.4. Change in seagrass bed cover in a wave-exposed, patchy seagrass bed near Beaufort, NC. Dark circle=m² areas with no change in cover (6-month period), + = areas of seagrass gain, ∘ = areas of loss and no symbols = areas of unchanged sand.

seagrass beds that form large unbroken meadows. Limited comparative information on bed spatial heterogeneity is available from the full range of habitats or landscape patterns that seagrasses form. Therefore, if we used published data to set boundary definitions of seagrass beds, it is quite likely they would tend to define only certain seagrass species (i.e., commonly studied species such as *Z. marina*, etc.) in certain settings (e.g., relatively wave-protected and low current speeds which yield extensive, non-patchy habitat). Further, because data collection has been historically biased toward beds in lower energy environments, the more fragmented, patchy nature of higher energy seagrass beds would be an element of seagrass bed structure that would not be captured in such a universal definition. On-site, direct surveys of local undisturbed seagrass beds in similar physical settings, or better, pre-impact surveys of the status of a seagrass bed remain over time the best guidelines for delineating seagrass habitats.

What we suggest is that managers must have some historical perspective. One-time surveys are completely inadequate data (i.e., see Figure 1.2) upon which to base management decisions that could have effects for years. Bed form migration (*sensu* Patriquin 1975, Marba et al. 1994, Marba and Duarte 1995), presence of seed banks, annual populations, recent nonpoint source anthropogenic impacts (e.g., decreased water clarity), and even deliberate removal of seagrasses all combine to cast doubt on the veracity of one-time surveys (i.e., see Figure 1.2). For evaluations of extant beds, even seemingly straightforward information such as shoot density can be misleading. Data such as shoot density are sometimes inversely related to shoot size, meaning that shoot densities of even less than one shoot m^{-2} may be significant, especially if that shoot is very large. Conversely, populations of *Halophila* spp., of which there may be in excess of half a million hectares in the Gulf of Mexico and Indian River Lagoon (Iverson and Bittaker 1986, Continental Shelf Assoc. 1991, Kenworthy 1992), return almost exclusively from seed every spring (Williams pers. com.). As with other species that rely heavily upon seeds for seasonal recovery, surveys taken during months where aboveground biomass is all but absent and that do not incorporate seed bank surveys would erroneously conclude the area did not support seagrass.

Spatial Scale and its Role in Defining Seagrass Habitat

If physical processes have the potential to affect habitat heterogeneity in seagrass communities then there is the potential for affecting associated fauna (Fonseca and Fisher 1986). Seagrass beds composed of isolated, dune-like patches of ~2 m in diameter can coalesce within several growing seasons upon elimination of waves and tidal currents (pers obs). Despite the clear relationship of water motion to seagrass bed form, we have only begun to evaluate their spatial (or temporal) organization (Virnstein 1995), otherwise seagrass beds have consistently been treated as a "black box" at the landscape scale. To build on information accumulated on ecosystems and apply this information to seagrass systems, research emphasis must include not only the normative 1 m scale study, but scales that are relevant to mechanisms that contribute to the formation, maintenance, and function of whole systems, such as sediment transport pathways or an organism's range.

If the pattern of distribution observed in seagrass beds is the result of physical processes whose effects vary with the spatial scale of examination, then it follows that the influence of bed pattern on such things as faunal abundance will, in turn, vary with spatial scale as well (*sensu* Bian and Walsh 1993, Fonseca 1996). Therefore, knowing the range of these scales is potentially valuable if, after gathering empirical evidence, one can infer structural attributes at other scales of interest, especially scales that may be less expensive to derive (e.g., aerial photography).

Resource managers must realize that a relationship between ecological phenomena and the spatial scale of a survey is real and sometimes intuitive. At the least, such relationships are a statistical reality that can strongly affect interpretation of field survey data (Rossi et al. 1992, Cao and Lam 1997). The notion that interactions at one scale (spatial or temporal) affect that which is expressed on another scale provides the basis for hypothesizing scale-dependent effects. Therefore, spatial and temporal patterns seen in seagrass ecosystems are the result of physical processes acting both on individual plants and the local population level (individual patch). Responses of individual plants to water motion and associated phenomena (e.g., sediment particle size) may be cumulative and affect seagrass landscape patterns perceived at coarser scales of resolution. To summarize, examples of the importance of deriving scale dependence in seagrass beds include identification of:

1. The scale at which samples taken in the landscape are independent of one another and improve sampling stratification,

2. Their effect on animal utilization and distribution, and

3. The relevant scales over which sedimentary processes are controlled providing a better prediction of alterations in current patterns, interception (or lack thereof) of wave energy, and sedimentary processes as the result of altering the seagrass landscape.

One result of recent research on seagrass landscape patterns is that there are ranges of spatial scales over which estimates of coverage vary as the result of the scale of sampling resolution chosen by the investigator (Fonseca 1996b). Moreover, for seagrass beds in North Carolina and Tampa Bay that experience relative wave exposure values (see "Constraints Imposed by Physical Setting on Planting Operations," below) greater than 3×10^6 (on a scale that runs from 0 to ~ 6×10^6) any estimate of seagrass coverage will differ depending on the size of the sampling unit and/or the distance separating those sample units at scales < 10m (Fonseca 1996b). This means that interpretation of any factors related to seagrass bed coverage sampled within this range of 1-10 m will be different among any studies that sampled at different spatial scales (i.e., samples taken 1 m apart versus, for example, 5 m apart). Therefore, comparisons among studies or surveys, even of the same bed, will differ to some degree simply because different size quadrats were used and not necessarily as the result of actual differences in the factor being compared. Of course, comparisons between studies can be different because different numbers of samples (which approximates statistical power) are taken. Finally, this has implications for the integrity of sampling schemes because any samples taken in this range of scale dependence will not be statistically independent, casting doubt on the validity of among-study or among-survey comparisons which were conducted at different spatial scales. This can create problems for interpretation of planting success.

Scale dependence in sampling has not only spatial but temporal considerations. We raise this caution regarding temporal scale dependence because in our section titled "Comparative Analysis of Seagrass Planting Efforts" we found that many projects changed assessment frequency during the course of the monitoring period. In fact, we too recommend a change in assessment protocol depending on whether it is being conducted before or after coalescence of planting units. Therefore, statistical comparisons should be made with caution between data collected from pre- and post-coalescence because such comparisons of one site over time likely violate rules of sample independence. Because many planting projects cannot escape problems with sample independence over time, the use of simple descriptive measures (such as area covered and persistence) as standard measurement protocols becomes very important to minimize problems with comparative analysis among studies or among dates within studies.

Another problem with spatially heterogenous (i.e., patchy as opposed to continuous) seagrass beds is the perception of their comparative ecological function. Spatially heterogenous seagrass environments in North Carolina have been classified as "scattered" (Carraway and Priddy 1983) versus continuous cover beds that are termed "dense." This unfortunate classification inferred a lower resource value despite the fact that the former landscape pattern covers many thousands of acres of estuarine seafloor in North Carolina, has shoot densities and primary production equivalent to continuous cover beds, has significantly higher below-ground biomass than continuous beds, and often supports equal densities of some economically valuable species such as pink shrimp (Murphey and Fonseca 1995).

Vulnerability and Susceptibility of Seagrass Ecosystems

Why are seagrasses so often impacted by human activity? One of the reasons is their location in the coastal zone. Because of their relatively high (compared to phytoplankton) light requirements (Kenworthy and Haunert 1991) they occur in shallow, nearshore waters, a situation that makes them extremely susceptible to damage by human activity such as nutrient loading (Short and Burdick 1996), light reduction (Dennsion et al. 1993, Kenworthy and Fonseca 1996), and propeller scarring (Sargent et al. 1995). As our utilization of the coastal zone grows so will the damage to seagrass ecosystems unless proactive steps are taken to avoid those impacts and successfully mitigate when impacts occur. Because they are now universally recognized to be valuable habitats, efforts to mitigate their losses have been underway for many years.

It is critical that one recognizes that seagrass mortality, whether mechanically induced, such as dredging, or physiologically induced from reduction in light (e.g., docks, turbidity), often happens rapidly; time scales for loss can range to as little as weeks or months. Recruitment, however, does not typically keep pace, yet if the site were capable of supporting continued cover, seagrass may recolonize within a few growing seasons (Kenworthy et al., 1980, Harrison 1987, Fonseca et al. 1990, Thayer et al. 1994). Recovery via natural recruitment is a demographic process with tremendous spatial and temporal variation (e.g., 0 to > 10,000 seeds m^{-1} for *Z. marina*) and is very difficult to predict. It is clear, however, that seed set and successful germination are often requisite for *rapidly* (1-2 growing seasons) balancing anthropogenically induced seagrass mortality. In contrast, vegetative encroachment may take many years (Johannson and Lewis 1992) or even longer, as is suggested by the lack of seagrass recovery in portions of the northeast U.S. from the "wasting disease" loss of the 1930's (*sensu* Short et al. 1993). The point here is that there are fundamentally different time scales involved in population-scale losses and their recovery. Only recently have investigations begun to assess the population-scale processes of seagrass bed formation and maintenance (Orth et al. 1994). In fact, scientists have no clear idea what constitutes a population for these plants or what population processes are at work (i.e., existence of metapopulations, *sensu* Orth et al. 1994). At a minimum, documentation of distribution together with elucidation of demographic process must be a research priority.

HISTORICAL IMPACTS AND LOSSES

We have mentioned environmental constraints to seagrass planting (see review by Phillips 1982), but there are many other management constraints that determine the effectiveness of seagrass planting. One is the degree of philosophical alignment among federal, state and local agencies whose jurisdictions include seagrass habitat. The U.S. Army Corps of Engineers, whose function includes issuance of dredge and fill permits, sometimes cannot follow recommendations from other agencies to conserve seagrass habitat (Mager and Thayer 1986). Conflicts between preservation of seagrass (and many other wetland habitats) and implementation of public-interest development projects must be balanced by resource agencies but often results in the loss of seagrass habitat (*sensu* Race and Fonseca 1996). The loss of seagrass habitat is sometimes addressed by proposing in-kind mitigation. In addition, maintenance dredging projects, particularly those associated with national security, are often considered exempt from mitigation requirements although in instances of very long dredging cycles (years to decades), mitigative actions are sometimes implemented to minimize immediate impacts. It has been our experience that as more information is presented to managers regarding the functions of seagrass ecosystems and the difficulties involved in mitigating for their loss, fewer permitted impacts are occurring in seagrass beds.

Although the loss of seagrasses due to dredging has been significant (Taylor and Saloman 1968, Onuf 1994), it is likely that the majority of seagrass habitat loss does not result directly from dredge-and-fill activities. More recently, direct impacts from mooring scars (F. Short, Jackson Est. Lab., Durham. NH, pers. com.), propeller scars (Sargent et al. 1995), jet skis (Kreuer pers. com.) and vessel wakes (pers. obs.) are emerging as a major source of seagrass habitat loss. For some species of seagrass such as *Thalassia* which is slow spreading (Fonseca et al. 1987c), physical damage is extremely long-lasting (Zieman 1976, Durako et al. 1992). Short et al. (1993) and the Chesapeake Bay Program (1995) recognized improvement of wastewater treatment, surface run-off, restrictions on certain fish and shellfish harvesting techniques, and regulation of boat traffic as key elements in protecting seagrass beds. Although scallop harvesting has been shown to damage seagrass beds (Fonseca et al. 1984) as has raking (Peterson et al. 1984) and prop-dredging for clams (Peterson et al. 1987), other fishery techniques such as trawling for bait-shrimp with specially-designed gear can have little apparent effect on seagrass although by-catch mortality is severe (Meyer et al. in review). Work by the Chesapeake Bay Program (1995) also lists (blue) crab dredging (scraping) as a significant impact on eelgrass beds. Fishing gear impacts to seagrass beds must be examined on a gear-by-gear basis.

Reduction in water quality, including water clarity, is another significant agent of seagrass loss (Dennison et al. 1993, Gallegos 1994, Onuf 1994, Gallegos and Kenworthy in press). Burkholder et al. 1992 and Dennison et al. (1993), like Batiuk et al. (1992), provided general guidance on maintaining water chemistry to support healthy seagrass beds. In doing so, Dennison et al. (1993) essentially determined the converse of health standards; they defined some critical water chemistry conditions at which harm would come to seagrass beds (Table 1.2). These data, and those promulgated by the Chesapeake Executive Council (1989) and the Chesapeake Bay Program (1995), are perhaps the only quantitative water chemistry information for managers to evaluate the health of seagrass environments at this time. They are likely useful for most temperate seagrass ecosystems and likely describe levels that would be too high for typically oligotrophic tropical and sub-tropical waters, particularly those dominated by carbonate sediments (*sensu* Fourqurean et al. 1995). However, the correlation between human development of the shoreline and seagrass decline is clear (Short and Burdick 1996).

Although seagrass beds are dynamic systems, with some beds persisting essentially unchanged for decades, others change with the season (den Hartog 1971, Zieman and Wood 1975, Phillips 1980a, Fonseca et al. 1983, Duarte and Sand-Jensen 1990). Some changes in seagrass communities can be attributed to the life histories of individual seagrass species (e.g., *Halophila* spp.). However, natural perturbations

Table 1.2. Chesapeake Bay submersed aquatic vegetation habitat requirements. For each parameter, the maximal growing season median value that correlated with plant survival is given for each salinity regime. Growing season defined as April-October, except for polyhaline (March-November). Salinity regimes are defined as tidal fresh = 0-0.5 o/oo, Oligohaline = 0.5-5 o/oo, Mesohaline = 5-18 o/oo, Polyhaline = more than 18 o/oo. (Taken from Dennison et al. 1993).

Salinity regime	Light attenuation coefficient ($K_d\,m^{-1}$)	Total suspended solids (mg/l)	Chlorophyll *a*(*u*g/l)	Dissolved inorganic nitrogen (*u*M)	Dissolved inorganic phosphorus (*u*M)
Tidal freshwater	2.0	15	15	-	0.67
Oligohaline	2.0	15	15	-	0.67
Mesohaline	1.5	15	15	10	0.33
Polyhaline	1.5	15	15	10	0.67

greatly influence the distribution of seagrass species. Disease has been widely implicated in the loss of seagrass beds since the pan-Atlantic decline in the 1930's (Rasmussen 1973, Short et al. 1987, Muehlstein 1989). Through this time, seagrass declines attributed to disease have added significantly to fluctuations in seagrass distribution. Physical disruption from storms and shifting channels redefine seagrass bed distribution and composition. Seasonal disturbances, such as low tides which expose and desiccate beds (Phillips 1980a, Thayer et al. 1984), and catastrophic events, such as hurricanes (Eleuterius and Miller 1976, Livingston 1987), can dramatically restructure seagrass beds both in terms of bed size and seagrass species composition. We have found that reductions in seagrass bed coverage as the result of storms is a positive function of how exposed to wind-generated waves a bed is prior to a storm; rapid loss of coverage can occur within a period of hours (unpubl. data), reiterating the fact that one-time surveys of seagrass coverage can be misleading as to the potential distribution of seagrass in a water body.

Biological disturbance of seagrass beds by a variety of organisms can also be extensive. Overgrazing by herbivores such as urchins has also affected spatial distribution and standing stock of seagrass beds (Camp et al. 1973). Ice scour (Robertson and Mann 1984) and extreme cold (Lalumiere et al. 1994) have been shown to control *Z. marina* distribution in the sub-Arctic. Also, excessive epiphytic load (Sand-Jensen 1977), burrowing shrimp (Suchanek 1983), vagile macrofauna (Valentine and Heck 1990, Valentine et al. 1994), green algae (den Hartog 1994a), and lugworms (Philippart 1994) have all been shown to limit seagrass distribution (but see Reusch et al. 1994; fertilizer enhancement of eelgrass by blue mussel biodeposition). Rays too have been implicated in many seagrass planting failures (Merkel 1988a, Mote Marine Laboratory and Mangrove Systems Inc. 1989, Fonseca et al. 1994) and may even contribute to the maintenance of natural bed patchiness (Townsend and Fonseca in 1998). These are, however, natural processes. Similarly, some dieoffs of seagrass such as the "wasting disease" of the eelgrass (*Z. marina*) in the North Atlantic during the 1930's (Short et al. 1988) and the current demise of *T. testudinum* in Florida Bay have been attributed to a pathogenic form of a marine slime mold, *Labyrinthula zosterae* (Robblee et al. 1991), among other factors. In nature, however, the outbreak of this fungi has not been easy to classify as a cause of seagrass decline as opposed to being a by-product of some other environmentally- or anthropogenically-derived decline in the quality of the seagrass habitat (*sensu* den Hartog 1996).

When human impacts are added to the natural stresses imposed on seagrass beds, disastrous losses of seagrass can occur. Such losses have been documented in Australia (Kirkman 1981, Cambridge and McComb 1984) and southeast Asia (Fortes 1988). In the U.S., large scale losses have been documented in the Chesapeake Bay (Orth and Moore 1981) and in the Gulf of Mexico (Livingston 1987). Significant impacts to seagrass beds in Tampa Bay were documented by Taylor and Saloman (1968), eventually reaching over 50 percent of the historical seagrass cover in Tampa Bay (Haddad 1989). Similarly, 35 percent of the seagrass acreage in Sarasota Bay has been lost as well as 29 percent of that in Charlotte Harbor, Florida, and 76 percent of that in Mississippi Sound (Eleuterius 1987). Pulich and White (1991) reported a loss of 90 percent in Galveston Bay, Texas. Thom and Hallum (1991) report similar ranges of losses from Puget Sound. Large losses of seagrass have also been reported from San Francisco and San Diego Bays (Kitting and Wyllie-Echeverria 1992), the Laguna Madre (brown tide, Onuf 1994), and large-scale damage from propeller scarring has been reported in Florida (Sargent et al. 1995).

Loss of seagrass cover leads to several undesirable and difficult-to-reverse conditions. First, the sediment binding and water motion baffling effects of the plants themselves are lost (Fonseca et al. 1983, Fonseca and Fisher 1986) allowing sediments to be more readily resuspended and moved (e.g., Florida Bay, Thayer et al. 1994). The physical ramifications include increased shoreline erosion and water column turbidity. Seagrass planted in areas with these conditions may not survive due to light limitation from the elevated turbidity. Loss of seagrass, of course, eliminates all important, associated habitat functions (Kikuchi 1980, Peterson 1982).

Much of the documented seagrass loss is due to human-induced reductions in water transparency (Kenworthy and Haunert 1991, Bulthuis 1994; these losses are often not included with other wetland or even seagrass loss statistics). Only in the last few years has it become clear that seagrasses typically require light intensities reaching the leaves of at least 15-25 percent of the light which has penetrated to just beneath the water surface (Dennison and Alberte 1986, Gallegos 1994, Gallegos and Kenworthy 1996). Moreover, the length of time over which a seagrass plant spends at photosynthetically-saturating light intensities too has been shown to be correlated with growth and survival (Dennison and Alberte 1985, 1986, Zimmerman et al. 1991). However, water transparency standards have historically been based on requirements of phytoplankton which may need only ~1 percent of incident light (Kenworthy and Haunert 1991), meaning that there is often no legal mandate for requiring improvement of water transparency to support seagrasses. This absence of technical and legal mandates makes the task of demonstrating the need for restoration of water quality to support seagrasses difficult.

There are many factors that act to reduce water column transparency (*sensu* Dennison 1987, Dennison et al. 1993, Gallegos 1994, Gallegos and Kenworthy 1996). Excess suspended solids and nutrients which enter the water column as the result of poor watershed management combine to reduce transmitted light below that of natural fluctuations, increasing vulnerability to local population extinctions. Suspended solids and water color changes reduce water transparency directly. Nutrient additions, such as from septic systems (Burkholder et al. 1992, Short and Burdick 1996), accelerate growth of light-absorbing algae in the water column as well as benthic macroalgae (den Hartog 1994a,b) and that growing epiphytically on seagrass blades (Sand-Jensen 1977), all of which combine to reduce light availability to seagrasses. Moreover, the seagrass canopy has intrinsic light attenuation effects through mutual shading (Dennison 1987, Enriquez et al. 1992) by the individual plants.

When losses have occurred due to decreased light availability, often only changes in watershed management (such as controlling storm water and sewage discharges) can reverse the trend of decline. Such a reversal in decline is rare but has occurred (Johansson and Lewis 1992). Transplanting into areas experiencing seagrass loss due to decreased water transparency without independent improvements in water quality will only result in the death of the transplants. This is especially problematic in areas where water turbidity may be due to sediment resuspension which arises as a result of seagrass already lost and is therefore not necessarily a current watershed management problem.

Reduction in water transparency is not the only anthropogenic source of seagrass loss (see Phillips 1982 for an early, detailed review). Thermal effluents from electric power plants have caused extensive losses such as those documented at the Turkey Point station in Biscayne Bay, Florida (Zieman and Wood 1975) as well as that associated with the Stock Island (Key West) station (pers. obs.). In the past, dredge-and-fill-associated losses were commonly associated with private sector development but more recently, many losses can be ascribed to public interest projects, such as the replacement of the Florida Keys Bridges (Mangrove Systems Inc. 1985a, Thayer et al. 1985). In addition, the rapidly increasing number of small boats in coastal waters has resulted in the aforementioned widespread damage from propeller scarring (Sargent et al. 1995). Because of the chronic nature of propeller scarring, hull impacts, and, more recently jet ski scour, such damage is likely very difficult to repair by planting (e.g., ferry boat landings in Puget Sound, R. Thom, Battelle Pacific Northwest Lab., Sequim, Wa.), Sargent et al. (1995) recommend a four-point plan to reduce scarring in moderately and severely scarred meadows (defined under their criterion) which includes (1) education of the public as to the nature and scope of scarring impacts, especially in the *Thalassia testudinum* beds which are very slow to recover from impacts, (2) installing channel markers as aids to navigation, (3) enforcing state and federal statutes that address propeller scarring and caused by propulsion systems dredging, and (4) establishment of limited-motoring zones in areas where, due to the extreme shallowness of beds, impacts from propulsion systems would be unavoidable.

A Short History of Seagrass Mitigation and Restoration

Addy's (1947) basic logic was to match planting and harvest site environments, and this remains a fundamental tenet in almost all seagrass planting today. Aside from early interest by Phillips (1960), almost 30 years elapsed before serious attention to planting seagrass developed. It was not until Eleuterius (1975), van Breedveld (1975), Thorhaug (1976), and Churchill et al. (1978) that documents again began to emerge presenting seagrass planting in a guideline format. But even though suitable planting methods have long existed, the track record for successful mitigation of impacts to seagrass beds remains variable (see review by Phillips 1982). Some spectacular failures of seagrass planting (Stein 1984) have created a lasting impression that restoration of seagrass beds is still an experimental management tool. Yet there have also been many successful plantings (e.g., Thayer et al. 1985). Seagrass beds have often been successfully planted and have come to perform much as naturally-propagated beds (Homziak et al. 1982, McLaughlin et al. 1983, Fonseca et al. 1996b). Still it has not been clear what factors are the most important to address to ensure planting success. We had previously thought that seagrass planting was, as Ronald Phillips put it, "a two-edged sword" (R. Phillips, Battelle Labs, Richmond, Wa., pers. comm.), providing a means of ameliorating habitat losses but perhaps encouraging habitat destruction through the mere existence of a possible remedial technique. In our opinion a more conservative trend has emerged. As resource managers and developers have become educated as to the value of seagrass systems and the realities of their costly repair, more emphasis appears to now be placed on impact avoidance and minimization.

Much emphasis was placed on technique development in the late 1970s and early 1980s (see reviews by Phillips 1980, 1982, Lewis 1987, Fonseca et al. 1988, Thom 1990), but relatively little attention was given to developing a management framework within which these techniques could be effectively implemented. As a result, most seagrass mitigation projects failed to achieve the goal of 1:1 habitat replacement (i.e., offset a net loss of seagrass habitat: *sensu* Fonseca et al. 1987c, Fonseca 1989a, but see Merkel 1988a,b), nor have they consistently addressed whether functional equivalency has been achieved (often a permit requirement).

Phillips (1980b) published seagrass planting guidelines that relied on elevation in the tidal zone, current speed, salinity, soil type (sandy, combination, or cohesive) and seagrass species. Decision keys for each coast of the U.S. were compiled. However, with additional research some of Phillips' (1980b) threshold criteria should be changed. He accepted current speeds up to 1.82 m s^{-1} whereas we would strongly caution against planting in current speeds exceeding 0.5 m s^{-1} (see below). Further, Phillips indicated that planting in sandy sediments was a cause for rejection of a planting site, but we have found excellent success in sandy sediments (Lewis 1987, Fonseca et al. 1987a,b,c). Zimmerman et al. (1991) argue that factors increasing root and rhizome anoxia such as cohesive soils recommended by Phillips put seagrass (at least when using bare-root planting methods) under severe physiological stress, a factor to be especially avoided during planting operations. Similarly, Merkel (1992) recommended planting on sandy sediments on the West Coast and avoiding consolidated clays and mudstones (although he [correctly] noted that rhizome extension is slower in coarse sediments). More recently, detailed information on habitat requirements for seagrass (and other submerged aquatic vegetation, SAV) has emerged, but only in well-studied areas. Notable is the work ongoing in the Chesapeake Bay. Batiuk et al. (1992; see also Dennison et al. 1993) provide a detailed synthesis of water quality requirements for SAV (Table 1.2). Based on experimentation and strong correlative evidence of these water quality parameters and SAV distribution, they also developed a series of target water quality conditions that would have to be met to expand SAV distribution by allowing it to colonize greater depths. This study should serve as a model approach to investigate seagrass restoration efforts in other areas. The applicability of these data to other areas is discussed in greater detail under the section entitled "Light Requirements for Transplanting."

Merkel (1992) has developed a field manual for planting eelgrass on the West Coast that includes planning protocols and detailed guidance on planting execution that is otherwise generally lacking in the literature. Aspects of Merkel's report will be reviewed throughout this document. Fonseca (1989a, 1992) published what were essentially Agency checklists for planning and evaluating seagrass plantings; the design of those checklists were the basis for the more comprehensive, yet regionally-specific guidelines published later (Fonseca 1994). The planning, planting, and monitoring sections of this document were adapted from Fonseca 1994: "A Guide to Planting Seagrasses in the Gulf of Mexico." Lockwood (1990) published criteria for placing marinas in eelgrass habitat that extolled impact minimization as the only guideline for mitigation. Based on case reviews of seagrass mitigation projects (Thayer et al. 1985), Thayer et al. (1990) published a preliminary decision matrix that incorporated site selection criteria as well as environmental conditions required for the growth of specific seagrass species.

In general, studies of seagrass restoration and management have only recently become a focus of attention (e.g., Chesapeake Executive Council 1989) and more recently, funding. NOAA's Coastal Ocean Program has focused on these issues for both seagrass and saltmarsh through its Estuarine Habitat Program, C-CAP, and Decision Analyses Series. In conducting our study, we have found the information base for seagrass management difficult to locate. For example, a survey of published literature since 1985 using BIOSIS™ revealed that there were 655 published works on seagrass. This search of the open literature reveals that over the last five years most of the focus in seagrass research has been on aspects of the plant's physiology. This is typical of seagrass research over the last quarter century where interest in plant physiology and seagrass bed-associated fauna have dominated the open literature. Cross-referencing "seagrass" with "restoration" found nine references while "mitigation" provided one reference. From the literature we accumulated directly from journals and solicitation of colleagues, we found that approximately half was found outside the open literature. The literature on the subject of seagrass bed restoration and mitigation is found in the grey literature and is often not subject to the rigors of peer review (but see Batiuk et al. 1992). Another large body of information lies in unpublished project reports, the quality of which are highly variable. We feel that the trend to generate information on seagrass restoration and mitigation for dissemination in forums other than the open literature has been one of the major reasons that seagrass restoration and mitigation is perceived as an experimental tool, when it could be an established management practice.

What are the problems managers face in restoring seagrass beds? Chief among these problems is the tendency to plant seagrass in areas where there is no prior history of their existence (Fredette et al. 1985; unless of course the site was created for the purposes of planting seagrass). The chronic absence of seagrass from a site, especially when there are propagule sources nearby, usually indicates that the site cannot consistently support seagrasses. Ensuring sufficient light, moderate nutrient loads (Batiuk et al. 1992, Dennison et al. 1993, Kenworthy and Fonseca 1996, Short and Burdick 1996) and protecting plantings from disturbance are major considerations for developing a persistent seagrass bed. Planting stock must be chosen so that there are sufficient young shoots and growing meristems to make up for mortality, a ratio that changes dramatically depending on what portion of a seagrass bed is examined, the species, as well as time of year. Most seagrasses are comparatively short-lived and have high natural mortality rates, and suitable growing conditions are needed to allow new shoot generation to compensate for this mortality. Thus, development and incorporation of seagrass demographic information into the management process is a high priority area for research. There are many other caveats that must be imposed to expect successful restoration of seagrass beds. These will be discussed later both in general terms and specifically by region around the country.

Having argued that seagrass mitigation is no longer experimental and should be considered an established management tool, why then place such a priority on conservation? The reason is that while techniques and protocols exist that can produce persistent seagrass beds, the history of the field shows that guidance and protocols are often inconsistently applied. This has resulted in spectacular large-scale planting failures (e.g., the aforementioned Port of Miami expansion project: a multi-million dollar ~200 acre seagrass mitigation which produced only a few acres; Stein 1984). The fact that much information on this subject is conveyed through the grey literature, which does not always circulate widely, has resulted in repetititve mistakes, such as selection of inappropriate planting sites.

MANAGEMENT CONTEXT FOR MITIGATION AND RESTORATION OF SEAGRASS ECOSYSTEMS

Scientists and managers are always faced with uncertainty in decisions regarding ecosystem management. As pointed out by Vitousek (1994) for global environmental change issues, scientists know with certainty that changes are occurring and that they are human-caused. What scientists cannot do is always predict the particular consequences of a given human activity on the environment. However, some trends are obvious and the consequences of inaction can be logically derived. It is irrefutable that extensive loss of seagrass resources have occurred in this country (see previous section), but what are the management options for halting and reversing this decline?

We have compiled this synthesis of seagrass restoration in an attempt to identify reasons for failures and successes which will then allow managers to improve the odds of success in restoring seagrass ecosystems. By acting to mitigate, restore and maintain these resources, managers can offset collateral decline of many ecological functions which we as a society hold important (erosion control, water filtration, fisheries production, and associated aesthetics). However, as the human population grows it is highly likely that losses of these unique plant communities will continue (e.g., Sargent et al. 1995). There are no ecological substitutes for their role in coastal ecosystems.

The critical role that seagrasses play in many coastal environments, coupled with their extensive losses, have created widespread support for their conservation and restoration. The "no-net-loss" policy promulgated by the Executive Branch provided an additional impetus to consider seagrass conservation and restoration. Meanwhile, numerous policy changes have occurred at the state and local levels over the last ten years to support no-net-loss of habitat. Therefore, as an information-based system of judging the value of seagrass ecosystems has emerged over the last decade, the question is no longer whether seagrasses should be protected, but how? When all avenues of protection have failed (e.g., sequencing; the US Army Corps of Engineers-EPA sequence of first seeking impact avoidance and minimization, and then compensatory mitigation, the latter being composed of some combination of enhancement, restoration, creation and under rare circumstances, simply preservation), then active planting may be the only option to avoid a permanent net loss of seagrass.

In order to proceed with discussions of management issues, some terminological clarification is needed. We will utilize the terminologies of Fonseca (1994) which are reprinted in amended form in Appendix A. Particularly, we wish to draw the reader's attention to the differentiation among the terms "restoration" and "mitigation." They are not interchangeable terms. Mitigation refers to activities related to permits (particularly sec. 404 of the Clean Water Act) and embodies a sequence of avoidance, minimization and ultimately, if needed, compensatory mitigation, whereas restoration is simply returning a site to a previous condition. Restoration as used here does not apply to permit-associated planting projects. We will also differentiate the terms "transplanting" and "planting." Transplanting is a subset of planting in that here it refers to harvesting of existing plants whereas planting can involve cultured plants, seeding, or any number of methods. The terms restoration and mitigation set very different constraints on the establishment of performance criteria and the evaluation of compliance (i.e., success). Lewis (1989) defines and differentiates restoration and mitigation as follows:

RESTORATION — "Returned from a disturbed or totally altered condition to a previously existing natural, or altered condition by some action. Restoration refers to the return of a pre-existing condition."

MITIGATION — "...the actual restoration, creation, or enhancement of (functionally equivalent, authors' note) wetlands to compensate for permitted wetland losses."

The term "mitigation" can be used without any modifiers but is often applied to situations more aptly termed "compensatory mitigation." Restoration is a term which generally applies only to planting activities which are not being counted against the destruction of existing habitat. Rather, restoration embraces the concept that anything we can do to right a past loss, a loss for which there may be no litigative recourse to seek damage recovery, is a plus to set against the Nation's balance sheet for no net loss, but not against that of a project with a pending permit to eliminate seagrass. From a management perspective, restoration for the sake of restoration only (properly planned and professionally executed), should be vigorously pursued because it will, if one utilizes the above definition, bring a community back toward previously existing conditions (i.e., it generally cannot make the situation worse).

Pitfalls in the Mitigation and Restoration Process

Compensatory mitigation is a process of questionable merit (Race and Fonseca 1996). Unlike restoration projects which are not necessarily initiated under the 404 permit process, the circumstances under which a compensatory mitigation are initiated have a large potential to make matters worse, because compensatory mitigation usually involves the destruction of existing habitat. The existing habitat is or has been traded for the promise of replacement habitat. With restoration, we are dealing with a past loss for which the responsible party may or may not be identifiable. With compensatory mitigation, the agent of loss and the responsible party are known and sometimes a decision (likely controversial) might be made to trade existing habitat for replacement habitat. Of course when a injury occurs to a seagrass bed outside of the permit process the loss of seagrass habitat occurred without a secure means of mitigating for its loss. However, whether an injury is deliberate or not, if existing habitat is lost, an often tangled negotiation process follows to determine the means by which compensation for that loss will be made. In many instances, the negotiation process can be prolonged, delaying restoration and resulting in larger impacts than might occur if restoration had begun sooner.

There are, however, a number of management decisions that can be made within the permit process to ameliorate a loss in habitat and better approaches the goal of no-net-habitat-loss. Mitigation in its broader definition typically also includes impact avoidance and minimization (the latter term unfortunately implying an acceptable net loss of acreage). In practice, avoidance and minimization are sometimes difficult to achieve. The existence of techniques to transplant seagrass has often been used to justify the destruction of existing, productive habitat (pers. obs.). But as pointed out earlier, this approach has consistently produced a net loss of habitat. This net loss of habitat occurs for a number of reasons, and the permit-associated activities that destroy seagrass beds in the first place typically are long lasting (i.e., creation of channels, bridges, bulkheads). Those activities also often do not allow enough area for on-site planting to offset the loss of habitat. If planting is considered at a location not on the original impact site (off-site restoration or mitigation), that site would preferably not be an area that itself has lost seagrass to some other impact. This is a subtle point that is often overlooked because of the often costly (in time and effort) site history data that must be obtained to make a quantitative evaluation of no-net-loss. The problem works like this: if one permits a loss of seagrass for some form of coastal development (e.g., -1 acre) and plants an equivalent area (+1 acre) onto a site which had previously lost seagrass (e.g., -1 acre) but was not associated with the project at hand, then the net change in habitat is: (-1 + -1) +1 = -1 acre. All that was accomplished was the repair of the original problem at the planting site, but it does not address the loss at the new, most recently impacted site. While there would be no net loss from immediate, present day acreage, the lack of consideration of past losses results in a net loss on a recent historical time scale. The critical question here is at what point in the past do we choose to represent baseline seagrass acreage? Moreover, what if a site chosen for planting does not currently support seagrass? In the absence of site history information, one must then ask why it does not presently support seagrass. This often indicates some inherent difficulty in colonization or persistence of seagrass. The events influencing the colonization process are sometimes difficult to document because they are often aperiodic, acute events (e.g., extreme low tides, storms, migrating rays excavating the bottom). Naturally unvegetated seafloor should not be substituted for vegetated bottom as this typically creates only a transient seagrass bed and alters, not necessarily improves, existing habitat functions. The take-home message is that if one contemplates off-site compensatory mitigation, there are usually few, if any sites available that: (a) can support seagrass growth, and if they do; (b) do not involve habitat substitution; or (c) do not satisfy the no-net-loss goal.This is not to say that previously damaged sites should never be used for mitigation or restoration, they just must be accurately represented in any no-net-loss accounting. As pointed out by Short (Jackson Estuarine Lab., Durham, N.H., pers. com.) in reference to the above description of trade-offs, if no mitigation is done on a previously damaged site, one ends with a -2 acre net loss of habitat instead of -1 acre of loss.

Regional Breakdown of Permit Activities Dealing with Seagrass Mitigation

Under a Memorandum of Understanding with the U. S. Army Corps of Engineers, the National Marine Fisheries Service, Office of Habitat Protection, comments on development permit requests under Section 404 of the Clean Water Act and Section 10 of the Rivers and Harbors Act. While seagrass restoration has been conducted on an experimental scale along all coasts and within all coastal regions of the U. S., actual mitigation of impacts resulting from Corps of Engineers-permitted activities has been relatively small, and has been greatest in the NMFS Southeast and Southwest Regions. A summary of NMFS-recommended and acted upon mitigation actions by NMFS Region, based on reports received from NMFS Regional Offices as of early 1996, is provided below (note that these regions do not match the ecoregions described later in the text).

Northeast (NE) Region (Maine through Virginia)

Seagrass mitigation in the Northeast Region of NMFS is in its infancy and, while permits have been reviewed which deal with seagrass habitat, few actions are ongoing. In 1991 NMFS began recommending seagrass mitigation for projects without practicable and feasible alternatives that would damage seagrass habitat. At the time of this report mitigation actions have been considered in New Jersey, Maine, and New Hampshire, but site selection and test planting for a 3-acre mitigation in the Piscataqua River (N.H.) is the only ongoing permit-related mitigation which NMFS has been involved in making recommendations. This has included not only transplanting but also consideration of alteration of bottom topography to achieve appropriate planting depths for eelgrass. Proposals are currently being discussed for a 10-30 acre eelgrass mitigation in the upper Penobscot Bay (Maine).

In addition to supporting the experimental transplanting work that is ongoing in each of the NE states, the NMFS Regional Office has taken a proactive approach to seagrass habitat protection. This has included involvement in the development of seagrass management policies, development of seagrass survey guidelines, encouragement for interagency mapping of seagrasses including involvement of the NOAA Coastal Change Analysis Program (C-CAP) mapping efforts, and the convening of information transfer and education meetings for state and federal agencies on seagrass ecology and transplanting technology.

Southeast (SE) Region (North Carolina through Texas including the U. S. Virgin Islands and Puerto Rico)

In the SE Region Habitat Protection Offices reported seagrass mitigation in North Carolina, Florida, and Texas. In each state the permits were obtained or under consideration primarily for channel maintenance or development related to onshore construction. In Texas, however, permit activities relate primarily to petroleum pipeline construction and mitigation of illegal prop-dredging activities. With the exception of Texas there has been little or no monitoring or follow-up to assess the degree of success of the projects. In addition to permit-related activities noted below, field offices of the NMFS have participated in similar management activities noted for the NE as a means of educating state and federal agencies and potential developers of the ecology and sensitivity of seagrass species and habitats.

Between 1985 and 1994 the Habitat Protection Field Office in Beaufort, North Carolina, recommended seagrass mitigation on 5 permits. The direct seagrass damage (i.e., removal of habitat) ranged from 0.23 to 2.0 acres, and the ratio of seagrass planted to that lost ranged from 1:1 to 3:1. However, in one case there was no on-site or in-kind alternative, and oyster reef creation was accepted as an alternative. The seagrasses involved were *Zostera marina* and *Halodule wrightii* primarily, but *Ruppia maritima* also was recommended to be transplanted in one instance. During the 9-year period between 1985 and 1994, a total of 3.25 acres of seagrass habitat were permitted to be destroyed with a requested mitigation of 4.74 acres of seagrass transplantation. Evaluation of the mitigation sites has been carried out in two cases, one demonstrating success and one demonstrating failure.

Florida has the largest extent of seagrasses in the contiguous U.S., followed by North Carolina and Texas (see earlier discussions). Between 1978 and 1994 a total of 167 acres of seagrass habitat have been requested for mitigation by the NMFS Habitat Protection Field Office in Panama City, Florida. These permit requests have generally been the result of new channel construction and port development and have ranged in mitigation acreage from 0.09 to ~200 acres. This latter was the result of a permit for additional development of the Port of Miami. *Thalassia testudinum, Halodule wrightii,* and *Halophila engelmanii* have been involved in the recommended mitigation. Based on reports from the Panama City Field Office, the degree of success of these permit-related mitigation has been generally poor and in many cases, unknown.

The Galveston, Texas Field Office of the NMFS Habitat Protection Division reported that there have been 6 major seagrass mitigation activities, almost all in the Laguna Madre, between 1985-1991. A total of 107 acres of seagrass habitat have either been recommended for creation or restoration. These have included filling of unused pipeline channels and associated re-contouring of the bathymetry to downgrading of dredge material islands. In some instances, natural recovery of the site(s) has been recommended while in others transplanting has occurred. The species involved in natural recovery have been *Halodule wrightii, Halophila engelmanni,* and *Ruppia maritima,* whereas *Halodule* has been the species of transplant choice. In most instances, oil companies have hired private concerns to monitor the mitigation sites or staff from the Galveston Field Office have had the opportunity to visit the mitigation sites. It appears that site selection and proper bathymetric contouring has occurred because the Field Office reports that with the exception of 22 acres, there has been a mitigation site coverage by seagrasses of between 40-99 percent within a 3 year period by either natural or transplanted methods. Some planted seagrass sites in Florida and Texas are currently being evaluated by National Marine Fisheries Service staff for seagrass and faunal recovery.

Northwest (NW) (Oregon, Washington) and Alaska

While research on experimental restoration approaches have been or are being carried out in these two NMFS Regions, both Regional Offices have been involved to only a very limited degree in seagrass mitigation and restoration. For a summary of eelgrass transplanting projects in the Pacific northwest see Thom (1990).

Southwest (SW) (California, Hawaii, and Pacific Territories)

Similar to other field and Regional Offices in the SE and NE, the Southwest Regional Office has participated in seagrass habitat management at both the permit as well as research and educational levels. They have held state-federal seminars involving the scientific community in discussions on the ecological value, sensitivity, and restoration of seagrasses. In 1991 an eelgrass mitigation policy was drawn up and adopted by NMFS, the U.S. Fish and Wildlife Service and the California Department of Fish and Game that includes recommended transplanting approaches, monitoring approaches, and measures of success that should be considered (see local evidence for seagrass function in Hoffman 1986).

From 1976 through 1993 the SW Region recommended eelgrass mitigation on 25 permits in California while 2 were recommended for *Enhalus acoroides* and *Halodule uninervis* on Rota Island and Saipan Island in the Pacific Territories. With the exception of 6 permits, most mitigation projects have not exceeded 0.1 hectare; the remaining 6 ranged from 0.8-3.8 hectares. Twenty sites have been visited where the mitigation activity had been completed, 11 of which are considered a success by Regional Office staff while 4 have shown a continued decrease in seagrass coverage and the remainder have shown no change in coverage. Overall, the success rate of seagrass planting in this region has been high (Hoffman pers. com.).

Comparative Analysis of Seagrass Planting Efforts

At this time we are not aware of any previous analysis of seagrass planting effort in the U.S. that used a comparative method. Therefore, we documented the status of seagrass planting projects from around the country by soliciting information on planting activities from many individuals of whom we were aware had conducted seagrass plantings. In addition, we requested that all National Marine Fisheries Service Regional Offices provide us with listings of all seagrass mitigation projects for which they had reviewed permits under their statutory authority. We also conducted site visits, especially on the West Coast where we were less familiar with planting activities in order to collect additional planting information. Finally we then compiled all references on this subject which we could acquire; this included review of all literature cited by reports and papers we collected.

This is not a complete survey and is complete only through 1995. We undoubtedly have missed some individuals and/or planting projects. Some persons did not respond to our queries. Again the absence of this work from the peer-reviewed literature made it difficult to find the information. The value of this survey then is heuristic, but addresses questions such as "where has effort generally been expended"? What data have been collected? What techniques have been used? How were sites selected and how was compliance and/or performance of plantings determined? How consistently has planting technology been applied?

We also broke down the survey by ecoregions which we have defined for the purpose of isolating practices and caveats peculiar to different parts of the country. Ecoregion is also the basis for the creation of modules where recommendations for planning, planting and monitoring are specifically discussed for each ecoregion. In addition, our original intent was to collect information on coverage rates and shoot addition of individual planting units (PU) from around the country. Any differences in species' coverage and shoot addition rates would aid in the definition of ecological regions for management. However, as our information collection progressed, it became clear that there were insufficient data from most parts of the country to conduct these coverage and shoot rate change analyses. Therefore, we have divided the coastal regions of the country based on our knowledge of growing season. The ecoregions for this report are as follows:

NORTHEAST — Maine through New Jersey: known species present = *Zostera marina* and *Ruppia maritima.*

MID-ATLANTIC — Delaware through North Carolina: known species present = *Halodule wrightii, Ruppia maritima* and *Zostera marina.*

GULF OF MEXICO AND THE FLORIDA EAST COAST — Mexico to Cape Sable and north of Jupiter Inlet to Cape Canaveral: known species present = *Halodule wrightii, Halophila decipiens, Halophila engelmanni, Halophila johnsonni, Ruppia maritima, Syringodium filiforme,* and *Thalassia testudinum.*

SOUTH FLORIDA AND THE CARIBBEAN — South of Jupiter Inlet to Cape Sable and P.R. and USVI: known species present = *Halodule wrightii, Halophila decipiens, Halophila engelmanni, Halophila johnsonni, Ruppia maritima, Syringodium filiforme,* and *Thalassia testudinum.*

CONTERMINOUS WEST COAST — California to Washington: known species present = *Phyllospadix scouleri, Phyllospadix serralatus, Phyllospadix torreyi, Ruppia maritima, Zostera japonica, Zostera marina.*

ALASKA — *Zostera marina* and *Phyllospadix* spp.(at least *P. serralatus*).

HAWAII AND PACIFIC TERRITORIES — known species present = *Halophila hawaiiana* (K. Bridges, pers. com.).

We compiled a collection of 138 documents ranging from published, peer-reviewed papers to project reports. Some of these documents were reviews or guidelines of how to transplant seagrass; some were feasibility studies; some were laboratory or mesocosm experiments directed at enhancing transplant technology. Each document was categorized several ways. We first determined where the document originated. Roughly 46 percent of the documents were found in the white literature, 29 percent were unpublished reports, 22 percent in grey literature and ~3 percent were theses (Table 1.3). All together, these papers reported on the fate of over 686,000 planting units of seagrass, totaling ~78 ha of field acreage, that have been monitored.

Over time the publication rate of documents concerning seagrass planting have increased. We found less than 1 percent of the documents published prior the 1960s. In the 1960s we found 2 percent of the documents; in the 70s 21 percent; in the 80s 46 percent; and so far in the 90s, 28 percent of the documents. At this rate the 1990s will produce the greatest amount of documents on the subject of seagrass planting. Some of this increase in publication rate may be that more recently created docu-

Table 1.3. Percent of documents on seagrass planting compiled by literature type.

Literature Type	Percent of Documents of this Type
White Literature	46
Report	29
Gray Literature	22
Theses	3

White literature = peer reviewed journal articles.
Report = not peer reviewed.
Gray literature = not in a library circulated journal, may or may not be peer reviewed.
Theses = masters thesis or doctoral dissertation.

ments are easier to locate, but it seems more likely that interest in the subject has grown.

The purpose of the documents varied widely. The largest group was field-research-oriented which comprised ~57 percent of the total. Next were review documents (29 percent), followed by laboratory experiments (~10 percent), and a miscellaneous group (6 percent), which included feasibility assessments, economic analyses, project summaries, and recovery assessments. In addition, there were three planting-associated theses. Most laboratory experiments and review documents were published in the peer-reviewed literature, while only half of the documents presenting new planting data were in the peer-reviewed literature.

Of the ecoregions we constructed, most documents originated from either the West Coast (~26 percent) or the Gulf of Mexico (also ~26 percent) (Table 1.4). South Florida and U.S. Caribbean territories produced ~19 percent, mid-Atlantic region ~18 percent, the northeast U.S. ~9 percent, and Alaska ~2 percent. Studies from other countries (Australia, France, Great Britain, Italy) were also reviewed but not utilized in computation of summary statistics.

The greatest number of planting units have been installed in the South Florida ecoregion (Table 1.5), followed by the northeast, West Coast, and mid-Atlantic states (the latter three regions being almost equal in number of planting units install), the Gulf of Mexico, and lastly, Alaska.

Table 1.4. Percentage of complied documents on seagrass planting presenting field transplanting studies, listed by ecoregion. Values are percent of total. Table does not include studies from outside the U.S., guidelines, reviews, or studies involving freshwater plantings. See section on "Regional Breakdown of Permit Activities Dealing with Seagrass Mitigation," above, for regional boundaries.

Region	**Percent of Documents Found in Region**
West	26
Gulf	25
South Florida	19
Mid-Atlantic	18
Northeast	9
Alaska	2

Table 1.5. Reported area of planted seagrass in square meters and number of planting units deployed in field studies by region and species.

Region	Species	Area M^2	No. PUs
ALASKA	*Zostera marina*	?	40
GULF	*Cymodosa manitorum*[a]	?	150
	Halodule beaudettei[b]	?	150
	Halodule wrightii	8,421	17,956
	Ruppia maritima	1	36
	Syringodium filiforme	591	2,336
	Thalassia testudinum	735	1,087
	Zostera marina	2,025	5,000
MID-ATLANTIC	*Halodule wrightii*	2,442	3,924
	Ruppia maritima	56	450
	Zostera marina	63,987	26,960
NORTHEAST	*Zostera marina*	18,449	82,560
SOUTH FLORIDA	*Halodule wrightii*	227,639	161,503
	Syringodium filiforme	17,417	20,364
	Thalassia testudinum	332,770	332,239
WEST	*Phyllospadix torreyi*	?	300
	Zostera marina	102,395	31,262

Alaska = entire coast of Alaska (Ak.)
Gulf Coast = Gulf of Mexico to Cape Sable, Fl. and the Florida East Coast North of Jupiter Inlet to Cape Canaveral (Tex., La., Miss., Ala., Fl.)
Mid- Atlantic = Delmarva Peninsula to North Carolina (Del. Va., Md., N.C.)
Northeast = Maine to New Jersey (Maine, R.I., N.H., Mass., Conn. N.Y., N.J.)
South Florida = South of Jupiter Inlet to Cape Sable, Puerto Rico and the U.S. Virgin Islands (Fl., P.R., U.S.V.I.)
West = Washington to California (Wa., Ore., Calif.).

[a]Probably *Syringodium filiforme.*
[b]Probably *Halodule wrightii.*

PU = planting units.
? = Insufficient data to calculate the area.

The use of different planting methods by ecoregion and seagrass species was also evaluated (Table 1.6). We constructed fourteen categories of planting methods, one of which was an "other" category that contained a number of methods not widely used and includes studies [a category of "unknown"] where the method of planting was not described. Of the fourteen categories, plugs or staples were the most common; ~40 percent of the plantings were done using one of these methods. The next most common was bare root-unanchored sprigs (15 percent), anchors of some sort (8 percent), followed by turfs (7 percent) and peatpots, biodegradable mesh, seedlings and seeds (all at ~5-6 percent each). Unusual or unknown methods accounted for [were employed in] ~2 percent of the plantings. Grids, seed tapes, bagged plants and attachment to boulders, with and without mesh grids, and passive seagrass fragment capture were used in the remaining ~4 percent of the plantings. The Gulf of Mexico ecoregion had the greatest number of planting categories (11), followed by the West Coast (10), south Florida (7), and the mid-Atlantic states (5).

We also compiled the frequency of planting methods used by seagrass species (Table 1.7). *Thalassia testudinum, Zostera marina* and *Phyllospadix* spp. have been transplanted mostly using techniques that involve removal of the native sediment from the root-rhizome matrix (in the case of *Phyllospadix*, there may have been no sediment to remove in the first place). The remaining three species listed in Table 1.7 have been transplanted using sediment-free and sediment-included methods in about equal proportion. Three species, *H. wrightii*, *T. testudinum*, and *Z. marina* accounted for 95 percent of the planting units put in the bottom (26, 21, and 48 percent, respectively). *S. filiforme* composed the remaining 3 percent of the PU while two other species composed ~0.00013 percent of the total number (one paper reported *Halodule beaudetti* and *Cymodocea nodosa* as occurring in the Gulf of Mexico but we suspect these were either *H. wrightii* and/or *Ruppia maritima).* Acreage of planting by species closely followed percentages for PU (Table 1.9). Some seagrass species that have broad distribution have received comparatively little attention to that given *Halodule, Thalassia* and *Zostera*. For example, few studies have been done regarding *Phyllospadix* spp. planting (Phillips et al. 1992), and these involve attachment to large rocks. Aside from Phillips et al. (1992), little else is known regarding *Phyllospadix* spp. planting techniques even though this species ranges along the entire U.S. West Coast (Phillips 1979, Wyllie-Echeverria and Phillips 1994). Turner (1985) provided important data regarding inherent stability and recovery of natural stands that have at least heuristic value for restoration in that the dynamic aspect of the community can be recognized and incorporated into planning (see Chapter 2, Planning). Similarly *Ruppia maritima*, which occurs in every ecoregion, and *Halophila*, of which there may be (based on an incomplete survey) half a million hectares off the West Coast of Florida alone (Iverson and Bittaker 1986), have received virtually no study as to their

Table 1.6. Percentage of all transplanting methods by ecoregion. Values are percent of total. This table does not include studies from outside the U.S., guidelines, reviews, or studies involving freshwater plantings.

Method	**Region**					
	Alaska	**Gulf**	**Mid-Atlantic**	**Northeast**	**South Florida**	**West**
Plug	25	29	20	43	25	12
Peatpot		6	15			8
Turf		19				
Mesh		6	15		3	
Grid		3				
Seedling		3			16	4
Seeds	25		5		13	4
Anchor	25	6			9	16
Sprig	25	11		29	19	20
Seed Tape				14		
Staple		6	45	14	16	24
Boulder						4
MBoulder						4
Other		3				4

Alaska = entire coast of Alaska (Ak.); Gulf Coast = Gulf of Mexico to Cape Sable, Fl. and the Florida East Coast North of Jupiter Inlet to Cape Canaveral (Tex., La., Miss, Ala., Fl.)
Mid-Atlantic = Delmarva Peninsula to North Carolina (Del.,Va., Md., N.C.)
Northeast = Maine to New Jersey (Maine, R.I., N.H., Mass., Conn., N.Y., N.J.
South Florida = South of Jupiter Inlet to Cape Sable, Puerto Rico and the U.S. Virgin Islands (Fl., P.R., U.S.V.I.)
West = Washington to California (Wa., Ore., Calif.).

Planting methods are defined as follows (categories are mutually exclusive):

Plug = tubes as coring devices are used to extract the plants with the sediment and rhizomes intact.
Staple = U-shaped metal staples with attached bare root (no sediment) planting units.
Sprig = bare root planting units (without staples or anchors).
Anchor = any structure used to keep the planting units in the sediment.
Turf = large square sods of seagrass that are ussually extracted with a shovel and planted as is.
Peatpot = a plug of seagrass that is transplanted into a biodegradable compressed peat container.
Biodegradable Mesh = seagrass sewn to a biodegradable mesh fabric and attached to the sediment surface as a planting unit.
Seedling = a newly sprouted seed with one short shoot.
Seed = seeds with no sign of shoots sprouting.
Plastic Mesh Grids = similar to biodegradable mesh except these are plastic (non-biodegradable).
Seed Tape = method of planting seeds using tape that has seeds sticking to it; the tape is then rolled out along the sediment surface.
Boulder = *Phyllospadix torreyi* is attached to boulders.
MBoulder = *P. torreyi* is attached to mesh and then attached to boulders.
Other = rarely used methods and includes studies where the method was not stated in the document.

Table 1.7. Percentage of transplanting methods by seagrass species. Dashed line separates methods that transport associated sediments (above line) from those that do not (below line).

Method	Species					
	Hw	Pt	Rm	Sf	Tt	Zm
Plug	32			42	11	21
Peatpot	9		25	8		6
Turf	7		25		3	2
Mesh	9		25		4	4
Grid	3					
Seedling					21	2
Seed					14	6
Anchor	6			17	7	11
Sprig	3			17	25	19
Seed Tape						2
Staple	23		25	17	7	23
Boulder		?				
MBoulder		?				
Unknown	3					2

Hw = *Halodule wrightii*
Pt = *Phyllospadix torreyi*
Rm = *Ruppia maritima*
Am = *Zostera marina*
Tt = *Thalassia testudinum.*
Sf = *Syringodium filiforme.*

Planting Methods are defined as follows (categories are mutually exclusive):

Plug = tubes as coring devices are used to extract the plants with the sediment and rhizomes intact.
Staple = U-shaped metal staples with attached bare root (no sediment) planting units.
Sprig = bare root planting units (without staples or anchors).
Anchor = any structure used to keep the planting units in the sediment.
Turf = large square sods of seagrass that are usually extracted with a shovel and planted as is.
Peatpot = a plug of seagrass that is transplanted into a biodegradable compressed peat container.
Biodegradable mesh = seagrass sewn to a biodegradable mesh fabric and attached to the sediment surface as a planting unit.
Seedling = a newly sprouted seed with one short shoot.
Seed = seeds with no sign of shoots sprouting.
Unknown = the method was not stated in the document.
Plastic mesh grid = similar to biodegradable mesh except these are plastic.
Seed Tape = method of planting seeds using tape that has seeds sticking to it; the tape is then rolled out along the sediment surface.
Boulders = *P. torreyi* is attached to boulders.
MBoulders =*P. torreyi* is attached to mesh and then attached to boulders.
? = insufficient data to calculate a percentage.

Table 1.8. List of experimental parameters and the percentage (in descending order) that were incorporated as a data collection or as independent variables in field transplant studies. Pre-survey = the site selected as a transplant site was surveyed prior to transplanting for its suitability to sustain a transplant.

Experimental Parameters	Percent of Documents Using this Parameter
Pre-survey of site	62
Planting method	45
Post-survey of site	27
Depth	26
Cost analysis	22
Fertilization type	21
Season	21
Faunal study	18
Planting unit spacing	17
Tidal zone	15
Energy regime	14
Donor survey	12
Sediment particle size	9
Enclosure	8
Shoot numbers	8
In vitro propagation	8
Genetics	6
Light intensity	5
Bioturbation	3
Burial recovery	3
Apicals	1
Salinity	1

Planting method = different methods of transplanting were tested for their effectiveness.
Post-survey of site = the effect of transplanting on the site location was evaluated.
Depth = effects of different depths on transplanting success was determined.
Cost-analysis = the total cost of the transplanting was determined.
Fertilization type = effects of fertilizers on transplanting was evaluated.
Season =effects of time of year on transplanting was evaluated.
Faunal study = fauna was sampled in transplanted beds.
Planting unit spacing = the effects of different spacing of planting units was evaluated.
Tidal zone = effects of different tidal zones on transplanting was examined.
Energy regime = effects of energy regime on transplanting.
Donor survey = there was a study conducted on the recovery of the transplant donor bed.
Sediment particle size = effects of different sediment size on transplanting.
Enclosure = effects of enclosure devices on transplanting.
Shoot numbers = effects of different planting unit shoot numbers on transplanting success.
In vitro propagation = growing seagrass in the laboratory to be transplanted.
Genetics = genetic experiments on transplanted seagrass were conducted.
Light intensity = effects of various light levels on transplanting success.
Bioturbation = bioturbation effects on transplanted seagrass.
Burial recovery = effects of sediment burial on transplanted seagrass.
Apicals = effects of the presence, absence, or different numbers of apicals in planting units.
Salinity = effects of different salinity on transplanting.

Table 1.9. List of ten most common parameters recorded in monitoring of transplant studies. Some studies considered more than one parameter.

Monitoring Parameter	Percent of Studies with this Parameter
Irregular frequency monitoring	74
Percent survival (PU)	65
Shoot counts	55
Shoot density	53
Percent cover	47
Leaf length	29
Leaf width	12
Rhizome length	6
Directm mapping	3
Biomass	3

Irregular Frequency Monitoring = irregular time intervals were chosen for follow-up monitoring of a transplant site.
Percent Survival = percent of planting units (PUs) that survived were monitored.
Shoot Counts = direct counts of planting unit shoots was conducted.
Shoot Density = density of the planting units was monitored.
Percent Cover = time zero area was known and considered 100 percent cover so that future areal coverage could be compared as a percent of that original coverage.
Leaf Length = leaf lengths were measured directly.
Leaf Width = leaf widths were measured directly.
Rhizome Length = total length of living rhizome.
Direct Mapping = actual mapping of the planting units for the area covered.
Biomass = weight of a given area of seagrass.

ecological role in the coastal zone. Although there has been little in the way of focused attention on development of planting techniques for these latter two species, we expect that existing methods such as plugs or peatpots may have promise (see Chapter 2, Planting).

There are also some incidental plantings of which we are aware. We know that *Halophila decipiens* was transplanted in 15 meters of water on St. Croix, U.S.V.I. in 1986 (authors unpubl. data) using mini-staples constructed of 130-pound test wire leader; plantings spread and apparently persisted to the end of the normal growing season. Harrison (1990) has also transplanted *Z. marina* in British Columbia using unattached shoots, cores, and by attaching shoots to re-bar (*sensu* Kelly et al. 1971). *Phyllospadix* was planted in the Monterey Bay Aquarium, California. Indoor small tank exhibits had poor survival but plantings lodged under rocks and experiencing mild simulated wave conditions in the aviary persisted for several years (Monterey Bay staff, pers. com.). Similarly, *Thalassia* has been grown in the coral reef exhibit at the Smithsonian Institution in Washington, D.C. for several years but with great logistic cost.

Not only did purposes of these papers vary widely, so did the design parameters of the studies. Table 1.8 describes the various parameters that were manipulated for documents reporting new field planting data (66 documents total: reviews and lab experiments excluded). Twenty-four different parameters were examined. Preliminary surveys of some environmental conditions at the planting site was the most common design feature: ~62 percent of the papers performed some pre-planting evaluation of the site. Slightly less than half of the papers tested planting methods. Only ~25 percent of the papers continued to survey some environmental conditions after plantings were installed, making it very difficult to establish any linkage between plant performance and episodic events. Planting depth (a rough surrogate for light availability, but also potentially related to frequency of emersion) was at least noted, if not a factor tested for influence on plantings in approximately 30 percent of the papers. Tidal zone (as opposed to some sea level-normalized depth measure) was also noted in 17 percent of the papers, but these data were not as specific as depth data. Together, however, water depth and tidal zone considerations were in 47 percent of the papers. Cost analyses, comparisons among planting season, and fertilizer effects were aspects of project design in ~20 percent of the papers. Comparative faunal assessments, effects of PU spacing, physical energy on the site, and recovery of plants at the donor site were parts of project designs in 12-18 percent of all studies. An additional 12 parameters were examined in the papers we reviewed but were never included in more than 10 percent of the papers.

What is interesting here is not so much what was either manipulated or noted but the proportions of what was not; that is, data that were considered relevant varied tremendously among studies. Thirty-eight percent of the papers did not consider or at least did not report what information was used to choose a planting site. Of those reporting, 33 percent simply used the criteria of no vegetation present which when used alone has been previously described as an unacceptable criteria (Fredette et al. 1985, Fonseca et al. 1987c, Fonseca 1989a, 1992, 1994) because selecting unvegetated areas with no known history of seagrass cover disregards the fact that any one of several mechanisms may be at work maintaining that level of patchiness (e.g., waves, currents, bioturbation). There is a rich body of literature on the role of habitat heterogeneity on ecosystem function that would have to be ignored to recommend converting naturally unvegetated areas to vegetated. Thus, in addition to being a high-risk planting area, planting in such an environment temporarily substitutes one habitat type for another. Therefore, based on our survey, 24 percent used what we consider to be an appropriate site selection criteria (the site having been previously vegetated but was now barren, although there are caveats to this criteria; see Planning chapter). Approximately 19 percent of the plantings were on dredged material while 20 percent were on unvegetated spaces adjacent to existing seagrass.

Few other factors were consistently integrated into design plans. Only 17 percent surveyed fauna after planting. Twelve percent considered the impact of harvesting on donor seagrass beds (i.e., monitored recovery of donor site). Less than 10 percent of the studies manipulated time 0 shoot number in a planting unit (generally as an attempt to determine optimal planting unit size).

Interestingly, two parameters besides percent PU survival that we have long recommended as being critical baseline monitoring data (Fonseca et al. 1982, Fonseca 1989a,1992,1994), number of shoots PU^{-1} and percent cover of the bottom, were only in ~53 and ~47 percent of the papers, respectively. We have recommended these forms of data collection because, when combined they describe many aspects of planting viability. In contrast, shoot density, a parameter over which there is little control, was used as a performance criteria in ~51 percent of the studies. Recent findings (Fonseca et al. 1996b) also suggest that macroepibenthic faunal abundance in planted seagrass beds asymptotes at comparatively (to natural beds) low shoot densities (as little as one third of natural beds), indicating that it might not be relevant to require shoot density in a planted bed to equal that of natural beds to support faunal densities equivalent to natural (but see performance criteria suggested by Short (1993 p. 51). Although some lower-than-ambient threshold shoot density may be suitable for generating faunal equivalence, lower shoot densities may not provide a sufficient buffer to population fluctuations of the seagrasses themselves. Thus, the issue of demographic status of the seagrasses of restored vs. natural beds is only beginning to be evaluated.

Most disturbing was that less than 7 percent of the papers actually provided quantitative data on two of the most critical limiting parameters known for seagrass planting success, light regime and bioturbation. Although depth and tide zone were frequently recognized as important factors, the absence of direct measurements of light means that depth and tide zone data are not easily extrapolated because we do not know the transparency of the water column. We can look up information on tidal amplitude and periodicity, but the interaction of light and tides on seagrass growth is only now being modeled (Zimmerman et al. 1994, Dennison and Kirkman 1996, Koch and Beer 1996), although these papers suggest the interaction of tidal amplitude and light availability to accurately predict site suitability based on transmissivity data.

Apart from those parameters that were monitored and/or manipulated, a total of ten parameters were actually utilized as measures of planting performance and/or success (Table 1.9). The percentage of PUs surviving was the most common criteria but was reported in only ~66 percent of the papers. However, ~74 percent of the papers varied the frequency of monitoring after planting over the course of their respective investigations (e.g., contrast fixed interval monitoring with a study that conducts monthly sampling for the first year then shifts to biannual monitoring for following years). The duration of monitoring in the papers we reviewed ranged from zero to eight years with a mean and median of ~1.5 years. During these monitoring periods the frequency of monitoring was also highly variable, again ranging from zero to an equivalent of 30 times y^{-1}. The average frequency of monitoring was 4.6 times y^{-1}.

Fonseca (1989, 1994) has recommended that early, frequent (usually quarterly) monitoring be performed for the first year after planting followed by less frequent (e.g., biannual) monitoring. Despite problems with changing temporal scales in analysis (see section "Scale and its Role in Defining Seagrass Habitat"), we continue this recommendation because many, but not all (particularly plantings with high initial loss of PUs) of our successful experimental plantings followed a sigmoidal population growth curve; initially high, exponential growth with low mortality followed by a balancing of natality and mortality of shoots which leads to an asymptote of plant density. Past recommendations for this monitoring strategy (Fonseca 1989a, Fonseca 1992, Fonseca 1994) actually agree well with at least the mean monitoring time values of the papers reviewed. Similar frequencies of monitoring were recommended by Merkel (1992), of time 0, 3, 6, 12, 24, and 36 months but with an additional recommended survey at 60 months. Choice of 3 years for monitoring resulted largely from compromise in that permit monitoring is rare (Race and Fonseca 1996) and shorter monitoring periods increase the possibility of acquiring monitoring compliance. So, for a given planting, how long should monitoring proceed in order to judge planting performance? Taken together with the average monitoring period of 4.3 y, and the fact that only 10 percent of the papers we surveyed achieved an ideal 100 percent cover, indicates that previous suggestions of 3-year monitoring by Fonseca (1989a, 1992, 1994) may be a serious underestimate of the time required to document project success; times in excess of 5 years may be more appropriate.

What is probably the most documented parameter in natural beds, seagrass biomass was only measured in ~3 percent of the papers, perhaps because it is a destructive sampling technique. Also, several measures of the plant's morphology were used frequently to determine planting performance (Table 1.9). We view these criteria suspiciously; seagrasses are often phenotypically plastic, and variation in plant shape and size is only loosely linked to functional attributes of seagrass beds at this time (Bell et al. 1991, Fonseca et al. 1996a) although morphology has been linked to significant genetic differences (Fain et al. 1992). We find it disturbing that simple parameters such as survival and coverage were not more universally recorded. From the low replication of criteria among studies, it is no wonder that quantitative performance and compliance thresholds, when they appear in mitigation plans, vary so tremendously (Thayer et al. 1985). Moreover, some papers used irreproducible units such as "scoopfuls" and "bucketfuls" to describe sampling units. Such vague planning criteria should not be used by resource managers.

The results of monitoring efforts have revealed some unexpected trends regarding success. To analyze this, we chose two categories, the final reported percent PU survival and the percent of the target area covered that was reported at the conclusion of a paper. Of 53 papers that reported percent PU survival, the median percent PU survival was 35 percent; mean 42 percent; standard deviation = 29.9; coefficient of variation = 70 with a distribution heavily skewed to lower percent survival (Sk = 0.35), suggesting adoption [use] of the median value (Figure 1.5). Roughly 5 percent of the plantings reported 100 percent PU survival. We found 27 papers that reported percent of the target area covered. The median percent area covered was 40 percent and was closer to the mean percent area covered of 42 percent; a standard deviation = 31.2; but still with a high coefficient of variation = 75 and a distribution again skewed to lower coverage amounts (Sk = 0.41) (Figure 1.6). We should point out that some of the variance in the data also results from areas such as southern California enjoying generally very high success rates (approaching 100 percent). The reasons for that success rate may have to do with quiescent settings for planting, high experience level and perhaps, comparatively low bioturbation levels. However, on a national scale, only approximately 10 percent of the plantings achieved 100 percent cover within the monitoring period. Thus, these data indicate that replanting is a consistent requirement of seagrass operations unless substantial initial overplanting is conducted to compensate for anticipated losses. Moreover, low initial survival rates may explain why seagrass plantings often produce less acreage than originally planned, suggesting that initial PU survival levels should be held to high standards to help ensure achieving target acreage.

An extreme interpretation of these findings would be that based on the median survival (a planting should have an overplanting ratio of approximately 3.0). In other words, if you wished to ensure that 100 planting units will survive, 300 should be

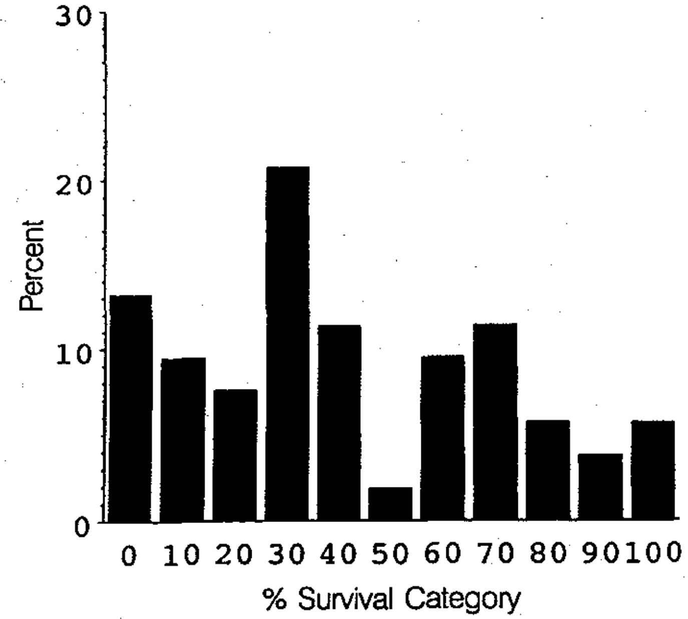


Figure 1.5. Frequency distribution of percent planting unit survival from the documents surveyed nationally. Y-axis = percentage of the survival values falling in the percent survival categories on the X-axis (10% increments).

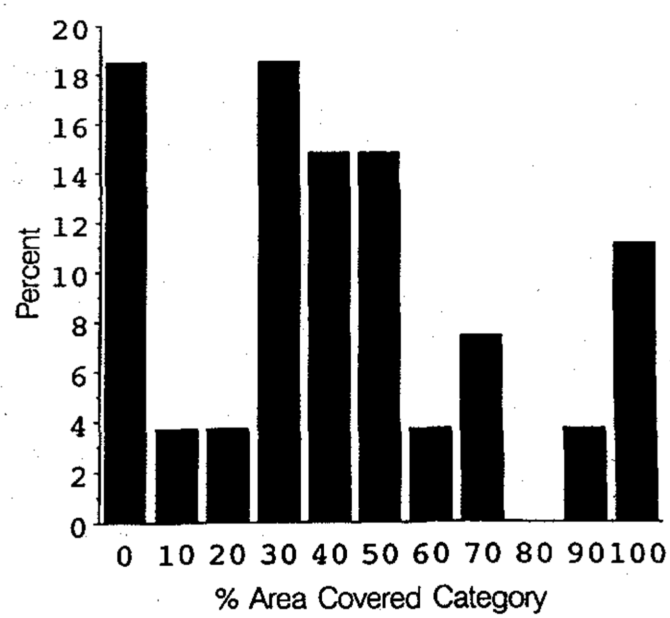


Figure 1.6. Frequency distribution of percent area covered by plantings from the documents surveyed nationally. Y-axis = percentage of the area covered values falling in the percent area cover categories on the X-axis (10% increments).

planted. Similarly, based on this national average, to ensure the required area of seagrass bed to be generated, a replacement ratio of ~2.5 units of area planted to 1 unit of area lost is needed to meet a no-net-loss criteria (i.e., 1:1 replacement ratio). We conclude that this is an extreme interpretation because many plantings used to compile these statistics were conducted on sites that would have been expected to produce patchy seagrass beds in any event. Also, many sites were chosen that violated recommended site selection criteria which would skew the distribution toward low survival and coverage. If site selection criteria are employed as described later, it is possible that these replacement ratios could be made much lower.

Managers must be cognizant of the different sources of planting failures and judge planting proposals under strict criteria. The practice of seagrass bed mitigation should not be questioned based on a failure in judgment on the part of someone who performed a planting. Such human failures must be separated from failures of the approach as a whole in order to responsibly assess seagrass planting as a mitigative tool (Fonseca et al. 1994). The key is to determine what made some plantings so successful and others so marginal.

Monitoring as recommended in the past (e.g., Fonseca 1989a, 1992, 1994) does not lend itself to determination of agents of planting loss. Only sophisticated monitoring equipment with high frequency recording capacity could hope to detect environmentally-induced losses. Acute and capricious events such as bioturbation and vandalism are even more difficult to determine with complete certainty (although use of exclosure cages may go far in suggesting the influence of bioturbation, Merkel 1988a, Fonseca et al. 1994). Therefore, the agents of loss among these studies cannot accurately be presented as a ranked set. However, based on our observations in the field, one might speculate that most failures occur from improper site selection (see criteria for site selection, below) and execution. From our experience and conversations with others (not to mention some published findings: Mote Marine Lab. & Mangrove Systems Inc. 1989; Merkel 1988a,b; Fonseca et al. 1994), we conclude that once a site has been appropriately selected under the criteria described below (e.g., previous history of seagrass cover, etc.) the primary agents of loss vary between bioturbation, acute storm events, algal smothering, and vandalism.

These compilations indicate that most of the planting experience is centered in the southern and western parts of the U.S. Also only a few species are regularly utilized in mitigation projects. Given the widespread impacts to seagrass ecosystems, concern that the absence of these other species from the literature indicates that impacts to those species goes unnoticed. Either that or these plant communities may not be receiving sufficient protection under current management practices. These survey data also indicate that there has been extensive experimentation with planting methodologies, and it appears that only a few are consistently employed and, again, only for a few species. The concentration of planting effort in Florida and the West Coast may be due to comparatively high development pressures in these areas. Although high habitat loss rates also occur in the mid-Atlantic states ecoregion, the proximity of research laboratories that have historically focused on seagrass to those estuaries may also explain the concentration of work in that ecoregion.

Are Planted Seagrass Beds Functionally Equivalent to Naturally-Occurring Beds?

What is "functional equivalency"? In a general sense, this means that a restored or mitigated system attains functions the same as those of an unimpacted system in a similar setting. Seagrass beds have many functions (*sensu* Wood et al. 1969), some of which may be more difficult to restore than others. As is the case with much of biology, the answer to the question of functional equivalency is both "yes" and "no." We tend to take the stance that if an area has recovered equal or greater acreage than that which was lost, and that area persists with the same seagrass species, a planted seagrass bed can become equivalent, but not identical to a natural, unimpacted bed. Our stance is not universally accepted. Equivalent means "equal to" but is sometimes taken to mean "identical." However, since no two samples of any natural ecosystem are ever truly identical, some subjectivity comes into play, both in terms of the degree of equivalence and the appropriate functions to measure. The problem then is what drives the subjectivity? A developer may interpret functional equivalency of their mitigation project in far more general terms than a trained biologist. What then are the relevant parameters by which to document equivalency?

According to our comparative analysis of the literature, thirty-three different parameters were used to describe success. This indicates the broad definition of functional equivalent — practitioners obviously target many different factors and differ in their opinions when ranking importance of these factors. Moreover, there is conflicting guidance from the literature regarding the rate at which planted beds take on attributes of natural, undisturbed beds. Brown-Peterson (1993) and Montagna (1993) conclude that attributes of planted seagrass beds were still not equivalent to natural ones after 31 and 14-17 years, respectively. Similarly, Smith et al. (1988a) found that planted beds did not provide equivalent bay scallop habitat over a growing season. Hoffman (1988) concluded that one-year old *Z. marina* plantings in San Diego did not support some fauna at levels exactly equal to that of natural beds, although some of differences were small. In contrast, Nessmith (1980), Homziak et al. (1982), Fonseca et al. (1990, 1996 a,b), and Wyllie-Echeverria et al. (1994b) found that faunal abundance and composition in planted beds approached that of natural beds within 2-3 years.

Much of this discrepancy among studies may be the result of intrinsic differences among natural reference sites and planted areas, the organisms chosen to evaluate recovery, and different worker's interpretation of what constitutes a difference. Because of the tremendous variability among natural beds, we question the efficacy of precise numerical comparisons in an interpretation of planting success; comparisons that include estimates of variance might be more appropriate. For example, distance and/or isolation of planted sites from natural beds will cause some differences (Bell et al. 1988, 1992). Brown-Peterson (1993) compared fish communities among planted and reference sites but the sites were located on opposite sides of a barrier island lagoon. Montagna (1993) compared beds established both by natural recolonization and planting in scraped-down dredged material islands with relatively open areas. Thus, there is some question as to whether differences among planted and natural treatments were the result of planting or of innate differences due to the physical setting. However, Montagna (1993) points out that most studies suggesting faunal equivalency have focused on more vagile macrofauna such as fish whereas certain infauna (e.g., clams) may not colonize as quickly. Kenworthy et al. (1980) and Homziak et al. (1982) found rapid colonization of a planted *Z. marina* site by scallops and meiofauna as did Wyllie-Echeverria et al. (1994b) for salmon prey (largely meiofauna). McLaughlin et al. (1983) concluded that recolonization by a wide variety of macrofauna occurred in planted *Thalassia* beds within only a few years. Similarly, Fonseca et al. (1990) found that after experiencing widespread failure of a planted area, the same site then naturally colonized by seed and supported a macrofaunal community not statistically different from adjacent planted sites within six-months of the onset of seed germination.

More recently, Fonseca et al. (1996 a,b) found that *H. wrightii* and *S. filiforme* beds planted on 0.5 m centers in Tampa Bay developed fish, shrimp and crab density and composition statistically indistinguishable from nearby natural sites within three years. One interesting aspect of that work was the relation of animal density to plant density (Figure 1.7). The seagrass density at which animal density in planted beds equaled ($p < 0.05$) that of natural beds was only approximately one-third of the mean natural bed shoot density. That density can be obtained within one year. They found that although linear models could account for approximately 65 percent of the variance of animal density as a function of plant density over time, a non-linear, asymptotic relationship between natural-log transformed animal density and seagrass areal shoot density was apparent (Figure 1.7). Although transformation of a straight line

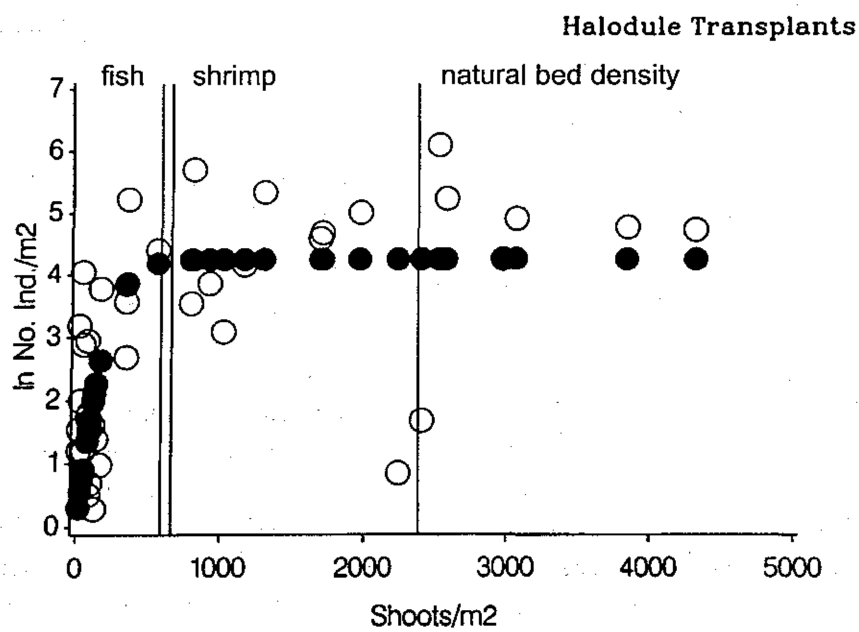


Figure 1.7. Natural-log transformed faunal density plotted against areal shoot density. Open circles=observed points. Closed circles=predicted points using an asymptotic function. Vertical line (VL) "shrimp" or "fish"=areal shoot density at which densities of these animals first became not significantly different among planted and natural versions of that seagrass species; "natural bed density"=the study-wide average areal shoot density of natural beds of that seagrass species. Taken from Fonseca et al. (1996b).

will yield some asymptotic tendencies, they are not enough to account for the pattern observed. The fact that animal density had an asymptotic relationship with shoot density implies that monitoring shoot density over time may be an inexpensive diagnostic parameter for determining a threshold planting success in terms of fauna. Short (1993) found similarly rapid colonization of *Z. marina* plantings in New Hampshire by a wide variety of fauna. Therefore, monitoring shoot density over time would be much less costly than direct measures of faunal communities. In order to justify use of only plant data in assessing some aspects of planting success, however, the temporal relationship among shoot density and faunal community structure must be collected from planted beds across a broad geographic range.

Although some ecological attributes may return quickly after planting seagrass, there is still a measurable period of time until the system has attained full function. The loss of ecosystem production in the time between when a seagrass bed is damaged and functions are restored has long been an issue of concern to managers. This loss of production has been termed "interim loss." The manner in which this loss has been calculated varies widely, as will be discussed later under sections on planting. We raise the point here because in many instances a reckoning of functional equivalency among planted and reference sites is made in an attempt to recoup interim loss of various living resources.

Seagrass species substitution is another issue that has bearing on the question of functional equivalency among planted and reference beds. Although much of the temperate U.S. is dominated by one seagrass species, *Z. marina*, subtropical areas and more recently the Pacific Northwest must contend with the functional ramifications of substituting one seagrass species for another (Pawlak 1994). There are few data to guide this decision. Temporary species substitution has been suggested for subtropical species (Fonseca et al. 1987c) where faster-spreading species such as *H. wrightii* and *S. filiforme* are planted as predecessors to recover areas previously dominated by the much slower-spreading *T. testudinum*. In that circumstance the reasoning was that interim loss of ecosystem functions could be minimized by establishing any form of seagrass coverage. Based on their work in Tampa Bay, Fonseca et al. (1996b) have suggested that differences in macroepibenthic faunal communities among these three subtropical species may not be as great as previously inferred (Stoner 1983). One reason for the differences among studies could be that Fonseca et al. (1996b) focused on unit area of seagrass bed-based surveys while others had compared animal populations among seagrass species weighted by attributes of habitat complexity, such as leaf surface area. While acknowledging that such faunal assessments should not be construed as indicators of all ecosystem functions (e.g., nutrient cycling, bed stability), the findings of Fonseca et al. (1996b) support the notion that seagrass species substitution to ameliorate interim losses of some ecological attributes may be a legitimate means to an end where that end is eventual replacement of the seagrass species that was damaged.

There is another tack to take in assessing whether planted seagrass beds provide resource functions equivalent to those they are intended to replace. On a much simpler level, we are not aware of any study that suggests that a seagrass bed is *not* a highly productive habitat. Therefore, we may infer that if one can produce a desired amount of seagrass habitat which persists over time that many ecosystem functions eventually will be restored. Whether a planted bed is the exact replacement for another seems to us to be an inappropriate question. It will depend upon what one considers to be an important ecosystem function and how much of it must be replaced to be considered "equivalent." Utilizing generally accepted significance levels (i.e., $p < 0.05$) to detect differences among two unique portions of the ecosystem is the most objective and scientifically acceptable means of testing differences. Of course, such differences are certainly affected by decisions of sampling gear, its size, the bias of that gear towards certain fauna and size classes, and temporal and spatial density of sampling. For example, the differences reported by Brown-Peterson (1993) and Montagna (1993) may well be within the range of year-to-year and site-to-site variation if longer periods of time were sampled and across wider areas (the choice of sampling scale being a powerful determinate of our perception) (e.g., crab densities: Fonseca et al. 1996b). Given the spatial and temporal variance in animal numbers, we feel it very difficult to justify additional planting based on relatively small differences in faunal attributes as compared to unplanted areas; using less stringent significance levels may be acceptable as well (i.e., $p < 0.10$).

In general, we believe that planting is a success if the acreage is planted, persists, and eventually (if not immediately) leads to replacement of the same resource functions of the seagrass species that were damaged. By success we imply functionally equivalent to natural beds. Again, we stress that the use of acreage and persistence as diagnostic features of planting success needs additional geographic replication.

As mentioned earlier, this view of assessment of planting success is, of course, not universally held. Short (1993) established other criteria for mitigation success for a project in New Hampshire. There, based on the recovery observed in his planted beds, he suggested that for eelgrass plantings to be considered initially successful, they should cover 30 percent of the planted area in one year. In addition, he stipulated that 40 percent of the following parameters be attained by plantings within one year: seagrass primary production, shoot density, leaf area, percent cover, and continuity (i.e., meaning that 40 percent of the plantings have coalesced). Moreover he stipulated that fish and infaunal assemblages constitute 25 percent of the following ecological parameters: presence of dominant species and total numerical abundance as compared to nearby natural beds. The values put forth by Short (1993) appear to coincide with findings for Tampa Bay (Fonseca et al. 1996b) in that faunal recovery will be closely linked with planting success and persistence.

Another reason for using plants rather than fauna as a metric of planting success is that one can envision scenarios where faunal recruitment to a bed could be inhibited by recent natural events such as storms or local pollution sources that may or may not equally affect a reference site. Conversely, we know of no evidence where an otherwise unimpacted natural seagrass bed has not supported high faunal density and diversity. Thus, if the faunal/seagrass relationship is not necessarily reciprocal, use of fauna alone may not be as easily tracked as the plants themselves, which may be suitable to infer the development of collateral faunal resource functions. At the very least, following plants alone is arguably less expensive than faunal data collection and may represent the most realistic data collection effort except in heavily subsidized restoration efforts. A pragmatic reason for accepting simple measures of acreage and persistence is that few plantings are well-monitored and enforcement of non-compliance with permit conditions is sporadic (*sensu* Race and Fonseca 1996). The parallel between seagrass presence and fauna we feel is strong enough to accept some metric of seagrass alone as a viable indicator of functional recovery. Although additional research must be conducted to strengthen the seagrass-faunal function link on a broader geographic basis, our research suggests that monitoring the seagrass itself is a very useful option for assessing restoration/mitigation success.

Combinations of parameters have also been suggested as an appropriate metric for gauging success of seagrass plantings (F. Short, Jackson Estuarine Lab., Durham, NH., pers com.). Combining factors that are known to affect faunal abundance, such as shoot size, density and water depth (taken together as a measure of habitat complexity), may be a way to provide a better means of indirectly comparing functions among planting, impact, and reference sites, especially when they occur in different geographic and/or physical settings. Short (Jackson Estuarine Lab., Durham, N.H. pers. com.) has proposed canopy volume:

(shoot density * canopy height; [m/m^2= #shoots/m^2 * m/shoot])

because it is a value that should change more slowly than shoot density alone and thus be more tightly coupled to a greater range of ecosystem attributes beyond faunal development (e.g., current speed reduction, change in sediment composition, nutrient cycling). Moreover, the canopy volume metric can be obtained with non-destructive methods and, unlike shoot density, would likely not exhibit overcompensation responses with some seagrass species (e.g., *Syringodium filiforme*: Williams 1990, Fonseca et al. 1994) which do not accurately reflect long-term recovery from injury to a seagrass bed.

Another factor in assessing functional equivalency is habitat size. We do not know if there is any relationship between the size or shape of a seagrass bed and its functional attributes; this is true for both planted beds and natural beds. This is an area of study needing much additional work. From our experience, however, even very small patches 1-2 m^2 of seagrass in the Beaufort, N.C. area has significantly greater numbers of fish, shrimp, and crabs than found in adjacent sand areas (unpubl. data). Moreover, Murphey and Fonseca (1995) found that on a unit area seagrass basis, even beds in the range of 30-40 percent cover had penaeid shrimp densities virtually indistinguishable from that of continuous cover, 100 percent beds. These data taken together with years of personal observation of seagrass beds (both planted and natural) have left us with the impression that even these very small, isolated patches provide resource functions comprising much of that observed in more extensive, unbroken coverage beds on a unit area basis. Thus, even very small patches of seagrass deserve protection.

CHAPTER 2

Planning

PRE-PROJECT PLANNING CONSIDERATIONS: IMPORTANCE OF GENETIC DIVERSITY IN SEAGRASS POPULATIONS

As habitat loss, fragmentation and geographic isolation of relict habitat has increased worldwide, scientists and resource managers, have become justifiably alarmed at the rapid loss of species and genetic diversity within remaining populations. Seagrass beds are no exception (*sensu* Ruckelshaus 1994a, Williams et al. 1996). In recent years the genetic status of seagrass beds has begun to be examined and the impact of human encroachment on its genetic diversity questioned (Alberte 1993, Williams and Davis 1993). However, as is the case in most wild populations, quantitative information regarding the genetics of individual plants, let alone populations, is scanty; and management decisions at the population level are those that most resource managers are likely to make. The question remains, however, does reduced genetic diversity actually matter in terms of population recovery trajectories and, thus, persistence of seagrass populations? Moreover, do differences in genetic diversity among planted and natural beds signal the disintegration of gene complexes specifically adapted to local conditions (S. Williams, Univ. California, San Diego, CA., pers. com.)? Because quantitative phylogenetic analysis of seagrasses is only beginning (Procaccini and Mazzella 1996, Waycott and Les 1996), it is currently impossible to infer adaptive value for any attributes of seagrasses, including gene complexes. Genetic diversity is generally assumed to be critical to the survival of restored populations, but before genetic screening can become a management tool, much more research is needed to clarify the consequences of any changes in genetic structure of seagrass populations as the result of habitat destruction and planting projects.

Some information is available regarding the role of genetic vs. environmental controls of seagrass. Backman (1991) concluded that genetic variation accounted for 14 percent of morphologic variation, environmental setting 32 percent, and interaction of genetic and environmental factors 35 percent. Based on these findings, Backman (1991) also suggested differentiating *Z. marina* into five varieties. However, employing similar techniques, Dennison and Alberte (1986) conducted reciprocal transplants of a *Z. marina* population in Massachusetts and found that growth responses were largely environmental and not genetic.

But for management purposes, maintenance of seagrass populations must be based on more than correlative inference of adaptive capabilities. Questions regarding population maintenance and genetic structure therefore must include some comprehension of how connected seagrass populations are along coastal areas. This includes some assessment of gene flow, genetic drift, influence of founder effects, existence of heterozygote advantages, identification of selection pressures, and determining the existence of metapopulations (*sensu* Orth et al. 1994).

The few extant publications differ somewhat in their conclusions regarding genetic variation in seagrass beds. Laushman (1993) found that genetic variance of *Zostera* populations was less within bays than among bays. Alberte et al. (1994) asked a different question and that is how genetic relatedness is associated with geographic separation; like most spatially dependent data, they found that the closer the eelgrass patches were to each other the more alike they were. Ruckelshaus (1994b) suggested that the role of local extinction as the result of disturbance (e.g., sedimentation, storm-induced scour) and subsequent recolonization of such areas may be an important source of genetic diversity. Because spatial and temporal variation in disturbance have long been known to have dramatic impacts on genetic diversity (e.g., founder effect and subsequent genetic drift: Futuyma 1986), historical contingency is often the initial basis for differences in genetic makeup among geographically isolated populations. Without some idea of the historical context of a population, however, it may be difficult to determine what management strategy would best serve as a response to detection of lowered genetic diversity, especially in anthropogenically-disturbed areas.

These population level questions are difficult to answer even in terrestrial environments where direct observation is much easier than in seagrass beds. For example, gene flow among geographically-separate populations is generally considered to be significant if only one individual exchanges genetic information with another individual in a separate population once each generation. In the case of clonal plants such as seagrass, it is not clear what a generation time might be. An individual ramet can live for days (e.g., *Halophila* spp.) or years (e.g., *Thalassia*). However it reproduces both by seed which is the result of genetic recombination (which can incorporate genetic information from an individual from another population: gene flow); and by vegetative branching which involves no external genetic input and merely produces a second ramet that has the same genetic makeup as its first (i.e., a clone, excluding potential somatic mutations). Therefore, in a genetic sense, populations that employ a consistent, season-to-season and year-to-year component of vegetative (i.e., asexual) reproduction, a single generation may be composed of changing individual ramets but one genetic constitution. Thus, a single generation could conceivably last for decades or longer and for these populations, exchange of genetic material (gene flow) might not be needed among geographically isolated populations for many years. However, we do not know the appropriate time scales over which to evaluate the relationship between genetic diversity and gene flow, one reason being because we do not know how long a generation is for these clonal plants. On the other hand, most evidence points to interdigitation of genets on small (< 1 m) spatial scales, thus gene flow should not be limited across small spatial scales (Ruckelshaus 1995). Local gradients in genetic structure might then be assumed to be the product of locally abrupt selection gradients (i.e., water depth and light availability, *sensu* Fain et al. 1992).

It is also difficult to determine the size of a population that is interbreeding (effective population size; but see Ruckelshaus 1994b). In seagrass beds, Orth et al. (1994) reported that seeds typically disperse near their source although many avenues for long distance dispersal exist, but others have found evidence for both long- and short-distance dispersal (Ruckelshaus 1995). Once a location is colonized (or re-colonized) the degree of subsequent isolation raises questions of founder effects (the initial reduction in local genetic diversity given that the few founders do not represent the genetic richness of the parent population) which can result in locally distinct genetic structure in a group of plants. A similar scenario of limited gene flow was inferred by Alberte et al. (1994), a phenomenon that supports the metapopulation theory of Orth et al. (1994). Subsequent long-term reproductive isolation can also enhance genetic differences among groups, especially in the aftermath of a founder effect. Another means of enhancing local genetic diversity was described by

Ruckelshaus (1994b) where *Zostera* presence in an area of the Pacific Northwest experienced repeated local extinctions and recolonization which enhanced genetic diversity.

Other potential (though not insurmountable) problems exist in the utilization of these data; our knowledge of the appropriate genetic indicators is limited, which imposes potentially severe technological limitations. Estimations of population size using genetic probes typically requires that the alleles being targeted are not acted on by natural selection. Because we do not know what aspects of a seagrass plant's biology are influenced by a given allele, it is conceivable that natural selection could be simultaneously altering allele frequencies being tested and, thus, biasing our interpretation of recent selection events. This is particularly a controversy with allozyme techniques (Futuyma 1986; p. 98), although these problems can be resolved if appropriate testing of probes is conducted and reported (S. Williams, Biology Dept., San Diego State Univ., San Diego, CA., M. Ruckelshaus, National Marine Fisheries Service, Seattle, WA., pers. com.).

The point here is that without knowledge of how often genes are exchanged among populations, the boundaries of populations, the existence or role of metapopulations, the duration of a generation, and the importance of seeds (a product of genetic recombination) in maintaining populations (and for the moment ignoring mutation), it is very difficult to direct specific management actions (i.e., site-specific) based on static surveys of genetic diversity. However, some guidance can be given even with preliminary data. If genetic variation were found to be partitioned more among sites of known geographic separation (therefore, populations are loosely defined as different sampling sites), rather than within sites, it would be advisable to equally protect geographically isolated sites (populations), and treat them as a larger, continuous resource rather then self-sufficient, isolated populations.

Following the findings of Ruckelshaus (1994a,b), we suggest that the environmental and geographic context under which surveys of genetic structure and diversity are undertaken are as relevant as the detailed information regarding polymorphic loci that emerge from any given study (*sensu* Ruckelshaus 1994b). For example, eelgrass beds on the West Coast of the United States exist in a geographically fragmented distribution among various water bodies and have, in the last century, experienced high localized losses. The situation on the East Coast is somewhat different. Although localized losses have occurred on the East Coast too, *Z. marina* populations went through a bottleneck with the wasting disease of the 1930's. These historical contingencies represent events known to affect the genetic structure of populations (geographic isolation and bottlenecks). It may be that the lack of historical information on many of these populations and the absence of data on effective population size and gene flow (particularly human-mediated gene flow such as seed transport, may severely bias generalizations regarding erosion of genetic diversity). In the Pacific Northwest, such a process was an effective mechanism for the introduction of an entire species, *Zostera japonica*, in the last quarter century.

It is critical to separate naturally low genetic diversity from an anthropogenically-imposed loss of diversity when attempting to set management standards for protection and maintenance of genetic structure (i.e., one may be setting baseline conditions too low). Another reason for differentiating naturally-low genetic diversity from human-induced declines is that some clonal plants with low genetic diversity can persist for millennia (Cook 1985). Laushman (1993) suggests that hydrophilic taxa (including seagrass) have intrinsically lower heterozygosity, polymorphic loci and alleles per locus than non-hydrophytes, meaning that simple identification of low genetic diversity may not signal a management dilemma (but see Alberte et al. 1994). Thus, it is the detection of the erosion of genetic structure, a measure that requires evaluation over appropriate temporal and spatial scales, that will determine the response by management (*sensu* S. Williams, Biology Dept., San Diego State Univ., San Diego, CA., pers. com.).

Even if genetic differences are seen among populations we still do not know if those differences are important to maintaining seagrass populations in areas under existing conditions. However, Alberte (1993) has found preliminary evidence of reduced genetic diversity of *Zostera* in impacted, as opposed to unimpacted, areas. Similarly, Williams and Davis (1993) have found evidence for reductions in counts of polymorphic loci among recently transplanted beds when compared to older, more persistent beds in San Diego Bay from which the transplants originated. More Williams et al. (1996) demonstrated "...that genetic diversity (percentage of polymorphic loci, allele richness, expected and observed heterozygosity, and proportion of genetically unique individuals) was significantly reduced in transplanted eelgrass beds." Williams et al. (1996) attributed this reduction in genetic diversity to small sizes of planting projects, limited geographic range in donor stock collection, and failed sexual reproduction, all inferring founder effects. Geographic variability has since been shown to contribute greatly to differences in genetic composition among planted and natural beds (Williams and Orth 1998). This is a potentially serious trend, even though very old (non-seagrass) plant clones of low genetic diversity have been shown to exist in nature. The response of decreased genetic diversity in recently transplanted beds observed by Williams and her co-workers (1996) is exactly what might be expected of a disintegrating population; one potentially becoming less able to respond to the vagaries of environmental variation produced by the comparatively rapid human alteration (as opposed to evolutionary time) of the nearshore environment. Localized extinctions might be expected under these conditions. However, reduced genetic diversity in transplanting might also be expected as it could mimic a natural founder effect, and with time, genetic diversity may increase in these planted beds especially as sexual reproduction (seeding) contributes more to these beds over time. Thus, it is possible that the small-scale variation in genetic structure observed in natural eelgrass beds (Williams et al. 1996) is the result not only of tillering and branching, but deposition of seeds (the product of genetic recombination). Thus, we are not yet sure whether reduced genetic structure and diversity are long-term problems. If these planted beds are genetically deficient compared to their parent beds, they may certainly pose short-term problems if managers rely on planted beds as donor stock areas to mitigate for losses of natural beds. Planted beds may not have the genetic makeup to deal with stressed conditions. Basic research on the phylogenetics of seagrasses is greatly needed. Recent advances by Waycott and Les (1996) and Uchiyama (1996) provides guidance on the comparative status of breeding systems which in turn allows us to begin to understand the role of vegetative vs. sexual reproduction in species' maintenance (*sensu* Procaccini and Mazzella 1996).

All of the above questions aim at resolving the issue as to whether differences in genetic diversity, or even its loss, means anything to the short- or long-term survival of seagrass populations under planting operations? Do changes in diversity influence maintenance of increasingly impacted and fragmented seagrass habitats? How long do these differences persist? Can we collect information on genetic structure over spatial and temporal ranges and with sufficient resolution to formulate management directives? Without specific, continued funding to support this work, the answer is unfortunately, "no." Therefore the question remains as to whether seagrass populations, particularly those currently becoming fragmented, will have the resilience to deal with the environmental changes brought on by human encroachment in the coastal zone. It seems to be an extraordinarily risky gamble to assume that present rates of habitat loss do not constitute a threat to the genetically-based resilience of some seagrass populations. We concur with Alberte et al. (1994) that "Studies that examine genetic structure of populations over time in disturbed and undisturbed habitats are needed so that the impacts of chronic habitat deterioration on genetic stability and resilience of (*Zostera marina*) can be ascertained." We recommend that scientists and managers continue to investigate and take seriously the threat of diminished genetic diversity and population isolation (especially when combined with degradation of environmental conditions critical for seagrass growth, such as light) that is anthropogenically imposed (*sensu* Williams et al. 1996). Thus, conservation of existing stocks and avoidance of population fragmentation and isolation would be a rational approach until a decision process based on data is available.

At present it is our opinion that except for highly impacted estuaries, the major short-term problems of maintaining seagrass beds will be providing suitable water clarity, appropriate nutrient levels (which also influence water clarity), and minimizing direct physical disturbance (e.g., dredging). Problems of restoring seagrass beds are largely ones of appropriate site selection and subsequent bioturbation. No gene complex can provide protection against grossly insufficient light, excessive nutrient loading, or the depredations of bioturbating organisms in a recently planted bed. Following Williams et al. (1996), we suggest that in practice, interim concerns regarding genetic diversity should be met by selecting planting stock from beds throughout the water body which is closely connected with the planting site. Stock selection (dealt with more completely, below) thus follows recommendations very similar to that of Addy (1947), especially in light of the fact that Fain et al. (1992) and Ruckelshaus (1994b) have both found greater genetic similarity within seagrass populations at similar tidal elevations than among populations. Any proposals to conduct surveys of genetic diversity of seagrass beds must contain provisions for periodic resampling to assess the influence of periodic disturbance and seedling colonization (Laushman 1993, Ruckelshaus 1994b) on genetic makeup, realizing of course that such sampling may have to be conducted for many years to detect the temporal sequence of disturbance that actually influences local genetic diversity (*sensu* Ruckelshaus 1994b).

More Pre-Project Planning Considerations: Seagrass Bed Spatial Requirements and Planting Site Surveys

Acquiring pre-impact data on seagrass distribution and environmental conditions at a site is vital to good planning, but there are constraints in obtaining those kinds of data. One problem is that site surveys are often done at a single point in time. The problems with one-time surveys of impacted sites have been manifested in many ways. One way is when decisions are made to place channels among patchy seagrass beds. Typically, a survey such as an aerial photograph will be employed to pick the alignment that will minimize impacts to present-day seagrass patches. However, because seagrass beds are spatially dynamic in time (i.e., they move: Orth 1977, Marba et al. 1994, Marba and Duarte 1995), such an alignment will almost certainly result in a decision to effect a long-term reduction in seagrass abundance because seagrass patches require that today's unvegetated space be available for them to occupy in the future (Figure 1.2). Thus, if a portion of the unvegetated space is removed from among patchy seagrass beds, it is unavailable for colonization. Therefore, when the space occupied by present-day seagrass is vacated (via bed migration or mortality), there will be insufficient space for colonization and a local net decrease in overall seagrass abundance, *even though no seagrass was immediately impacted* (*sensu*, Figure 1.2). This is the kind of impact that has been mitigated for in the past (Short 1993).

To demonstrate this effect, we have plotted the cumulative amount of bottom area near Beaufort, North Carolina, covered by seagrass in several 50 x 50 m plots which we have monitored for almost four and one-half years (Figure 1.2). These plots were mapped repeatedly for the presence/absence of seagrass cover with 1 m^2 resolution. Thus, over time, we counted the number of new square meters of bottom space occupied by seagrass each subsequent survey time, but excluding any m^2 locations that had ever had seagrass since the initial survey time (i.e., once a m^2 location is scored as having seagrass, it will never be added to this cumulative value again in the study period; only locations that had not been previously observed to have seagrass can be added). An important point is that the average percent cover at each site was very stable over the survey period. What this graph reveals is that sites that have nearly 100 percent cover to begin with, of course, remain at that level over time. However, sites that have lower percent coverage (y-intercept value)at any one point in time (here in North Carolina, largely the result of wave and tidal current effects), have had seagrass occupy twice the number of m^2 areas in that 4.5 year period than were observed at any one point in time.This cumulative coverage (over time) represents the spatial requirements needed to maintain a representative seagrass bed in a given physical setting over time; here at least twice the area of the standing seagrass coverage is required to sustain the patchy seagrass cover over a 4.5 year period.

With enough time, one would expect that ultimately all possible locations would eventually have supported seagrass cover; this is evidenced by the "hih2" site where cumulative coverage was still increasing at the May 1995 survey time. The asymptote of the cumulative coverage lines results at least in part for the tendency for m^2 areas near to existing patches of seagrass to be lost and recolonized more frequently than m^2 areas distant from a patch. What this graph clarifies is that a decision to remove a portion of the unvegetated space among seagrass patches through conversion to a channel or some non-seagrass habitat will, in many instances, result in the additional loss of seagrass acreage within a four-year period. Also, we hypothesize that depending on the alignment of a channel and the direction of bed migration, the channel may act as an interceptor, creating a large zone of low seagrass abundance in the down-migration direction, much as a snow fence or sand jetty accumulates material leaving the down-flow direction starved for that material. Thus, a knowledge of the spatial dynamics of seagrass beds over time is critical to maintaining present-day levels of seagrass acreage, information that is probably critical for the *Halophila* genera in particular.

Planning for Execution of a Planting Project

In order to prepare for a seagrass planting project, several factors must be considered. As in Fonseca (1994), the heading for each sub-section below can serve as an abbreviated checklist of information needs and subsequent actions which should be anticipated. Common to all these considerations is the need for early coordination with State and federal resource agencies. Since many States have a management system set up for federal agency review of such plans, early coordination can resolve regulatory problems before they become costly. In addition, we present a decision flow diagram (Figure 2.1) as a summary of this section. We suggest that readers familiarize themselves with this section before attempting to apply the decision process in Figure 2.1. When needed, they should also consider the guidance on how elevation affects seagrass survival through the interaction of tides and light regime presented by Dennison and Kirkman (1996; see section on Emersion Effects, below).

Identification of Project Goals

Although there are many methods to plant seagrass, frequently the goals of a project are not defined. Is the project for restoration or compensatory mitigation? Although the differences in project goals have little to do with the execution of the planting technique, it is important to recognize that planting in exchange for permitted losses may elicit different responses from resource agencies than planting for the sake of restoration only. For a review which touches on agency concerns, the reader should refer to Fonseca (1989a, 1992, 1994). Project goals should identify the species of plants that are to be used. Eventually attaining the same seagrass species as what was lost with an equal area of bottom covered is a logical, ecologically defensible goal.

Pre-Construction Planning

Because population growth rate varies with geographic location (and planting spacing), the timetable for meeting project goals will vary. For example, at ~ 1 m spacing, it will typically require two years to reach coalescence of planted areas for a shoalgrass bed in the Florida panhandle whereas it may take only six months to reach the same level of coverage for the same species in the Florida Keys (Fonseca et al. 1987c). This time lag should be anticipated for any planting and varies both by ecoregion and initial spacing of plantings (Fonseca et al. 1996a).

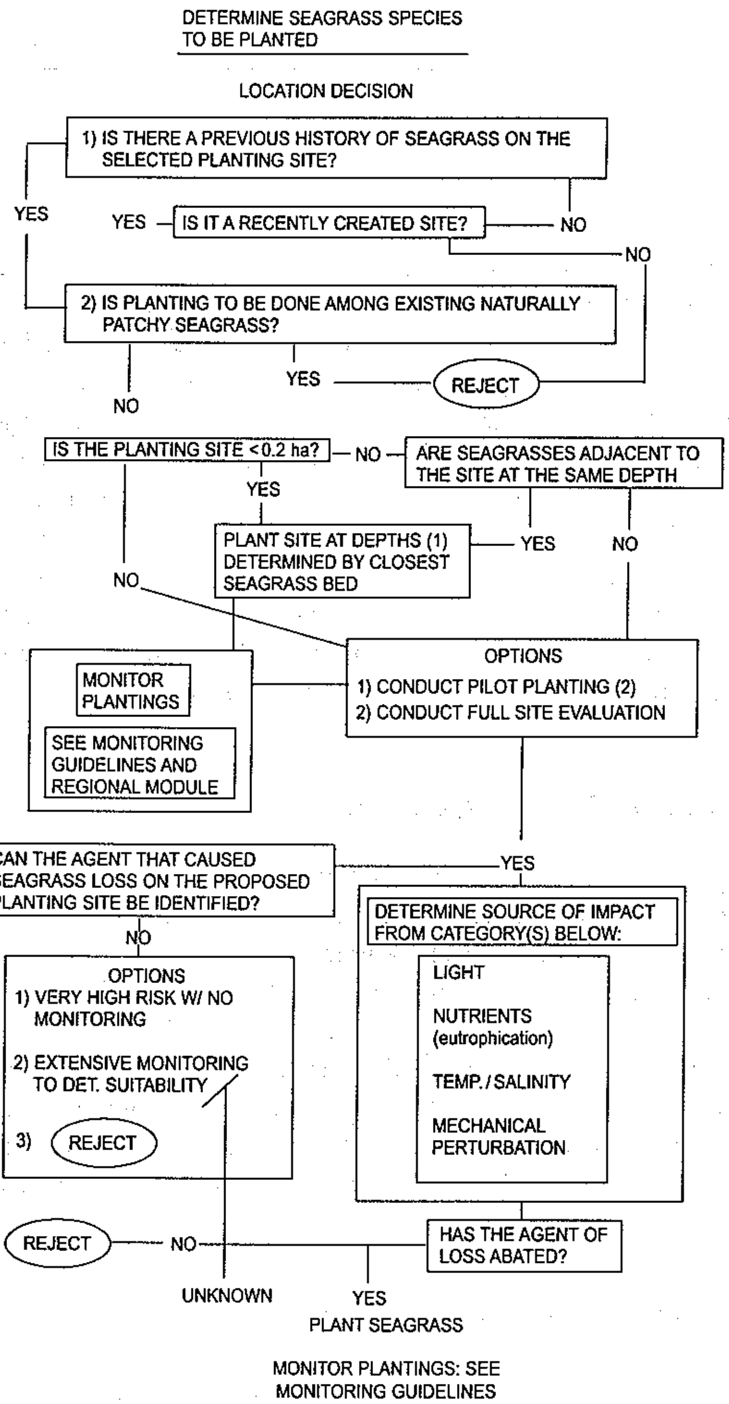


Figure 2.1. Decision flow diagram for seagrass planting with an emphasis on a mitigation scenario.

Early coordination with permitting and commenting agencies is critical. Because there are typically several agencies with overlapping jurisdictions, several permits may all be required for a single planting (e.g., municipal, county, aquatic preserves, state parks, State and federal agencies). This process should certainly be started months, and preferably a year in advance of the time planting is anticipated. Because many states prohibit harvest and/or planting without permits, failure to comply with permitting requirements can result in civil penalties.

Several coordination protocols have been developed. One was designed specifically for siting marinas (Lockwood 1990) while another was designed to assist agencies collect resource information prior to commenting on a permit request (E. Nelson, NMFS, Northeast Region, pers. com.). In Nelson's protocol, two tiers of information are requested. Tier one relies on the discovery of extant information regarding the distribution, quality (e.g., biomass, density), and function of the beds (e.g., fishery surveys) in question and directs the applicant to a host of potential information sources. Should tier one efforts not provide enough resource information to proceed with processing a permit request, then a new, relatively detailed on-site assessment is triggered (tier two). The idea of a standardized protocol is essential to accurately convey the scope of the potential resource injury to the public stewards and to simultaneously treat applicants in a consistent and fair manner.

There are a number of other policies and/or activities that have been developed to aid in seagrass protection and management. Hershman and Lind (1994) have summarized the variety of programs in the Pacific Northwest which exist and placed them into six categories:

1. project review occurring at all levels of government;

2. water quality policies many of which can impact directly on seagrasses;

3. public land management policies of state, federal and tribal agencies owning submerged lands;

4. restoration/habitat development policies implemented by all levels of government;

5. damage reduction policies; and

6. inventory and mapping programs which should include, but most frequently do not include seagrasses.

They note that there is little coordination among these processes and that, in fact, many lack the necessary geographic scope to effectively protect seagrasses. A major finding is that, although federal and state jurisdiction exists over seagrasses, no specific policies for managing seagrasses exist in the Northwest. Although policies exist for other portions of the coastal zone of the United States, they often suffer from lack of coordination both among and within agencies, as noted by Hershman and Lind (1994) for the Northwest.

Such coordination protocols would be better employed if some broader, state-level policy regarding seagrasses were in place. To our knowledge there are a limited number of published policies that specifically mention or are designed to address the protection and/or mitigation of damage to seagrass habitats (Stephan et al. 1997). Here we review the Southern California eelgrass mitigation policy, EPA's Chesapeake Bay Program, and the State of Connecticut's Coastal Zone Management Program.

Southern California

The Southern California eelgrass mitigation policy was adopted in July 1991 after having been developed by federal and state resource agencies (National Marine Fisheries Service, U. S. Fish and Wildlife Service, and the California Department of Fish and Game). This policy recognizes the ecological value of seagrasses, specifically eelgrass, and lays out procedures to be used for on-site mitigation performed to compensate for adverse impacts caused by projects addressed in Section 404 permits. The recommendations for site selection, transplanting techniques, and monitoring measures are largely based on published articles of the authors of this study. The Southern California policy also recommends criteria for success which are limited to the plant component and not to the system as a whole, a point supported by Fonseca et al. (1996b).

Chesapeake Bay

The EPA Chesapeake Bay Program has developed an awareness of the value of seagrasses in the ecology of the Bay, and in July 1989 developed an Agreement Commitment Report signed by the states of Virginia, Maryland, and Pennsylvania, the District of Columbia and the Environmental Protection Agency entitled "Submerged Aquatic Vegetation Policy for the Chesapeake Bay and Tidal Tributaries." This agreement states that the signers will work together to implement four major areas relative to seagrasses. These include assessment of the distribution and abundance of the resource, development of protection and restoration guidelines, and implement an education component to increase public awareness of the value of the resource. The educational component recognizes the need for scientific research to improve our knowledge and understanding of submerged aquatic vegetation to ensure that efforts to protect and restore this resource continue to be effective. It is refreshing to see that recognized in a ecosystem or watershed management approach such as is being conducted in the Chesapeake Bay.

The Submerged Aquatic Vegetation Workgroup of the Chesapeake Bay Program developed a guidance document for protecting submerged aquatic vegetation from physical disruption (Chesapeake Executive Committee 1995). As part of this guidelines document, the workgroup has summarized policies and activities of the states of Maryland and Virginia, the District of Columbia, and four federal agencies that directly impact the health of submerged aquatic vegetation in the Chesapeake Bay (U. S. Army Corps of Engineers, U. S. Environmental Protection Agency; U. S. Fish and Wildlife Service, and the National Marine Fisheries Service) principally as they pertain to permit applications under the Clean Water and Rivers and Harbors Acts. The reader is referred to this document for specific actions taken by these states and federal agencies.

Connecticut

The General Statutes of Connecticut, for Sections 22a-90 through 22a-112 for the Connecticut Coastal Management Act (revised January 1, 1993), presents legislative goals and policies which include insuring that the development, preservation or use of the land and water resources proceeds in a manner consistent with the ability of these resources to support development, preservation, or use without significantly disrupting either the natural environment or sound economic growth. This Act also recognizes the need to conduct and sponsor research to improve the information base upon which these decisions are made. The Act states that policies include managing estuarine embayments to "protect, enhance and allow natural restoration of eelgrass flats except in special limited cases; notably shellfish management..." While this is the only direct statement related to seagrasses, the document states that adverse impacts include "...degrading or destroying essential wildlife, finfish or shellfish habitat through significant alteration of the composition, migration patterns, distribution, breeding or other population characteristics of the natural species or significant alteration of the natural components of the habitat..."

Although many other states have developed policies relating to seagrasses, the few reviewed here differ from the other to some extent. However, they are all consistent in that they place an unquestionably high value on the maintenance of seagrass (or SAV) ecosystems, a position that signals the intent of the resource agencies to deal seriously with the resource and that allows potential developers a better understanding of the serious nature of an injury to these ecosystems.

Assessment of Interim Losses

Another goal of many seagrass plantings is an attempt to recoup *interim loss of ecosystem functions.* This was mentioned earlier as an attribute of functional equivalency. Because the concept of success and functional equivalency are so closely tied, planning for successful restoration and/or mitigation requires early incorporation of interim loss considerations. The manner in which interim loss has been addressed historically has been through adjusting replacement ratios (how much acreage to plant per unit acreage lost). However, the manner in which interim ecosystem losses have been computed has not been consistent. Replacement ratios of less than 1:1 to as high as 5:1 have been proposed (Fonseca et al. in press), based on a number of criteria, but that ratio is usually inversely proportional to the degree which a project was in the public interest.

To compute losses though, requires some assessment of not only acreage lost but also of how long a time the functions of that acreage were lost to the ecosystem at large before it was returned to pre- or un-impacted levels. Depending on how long one wishes to amortize a loss will influence how much replanting must be done. In theory, if one hectare of seagrass were destroyed today and three hectares were replanted tomorrow and reached standards of equivalency in three years, then after those three years the planting would have largely compensated for the total loss of production; the net loss of production over this three year period would be very low. However, things rarely work this way. First, it is very difficult to consistently locate and successfully create new seagrass habitat that meets our site selection criteria (which precludes simply substituting naturally unvegetated bottom for vegetated bottom). Finding large acreage for planting in close proximity to the impacted area is rare; this means that planting is often done at a site physically removed from the impact area and any functions affected by spatial elements of ecosystem linkages (i.e., geographic setting) are lost. Second, the production that was lost was removed from a specific point in time; ecosystem functions were disrupted and those specific resources are not replaced, such as that year's spawn of herring (e.g. as in the Pacific Northwest). Further, if there was a greater hiatus between the time of impact and recovery, then one could argue that plantings conducted longer after an impact or further away from an impact have less value than ones conducted sooner or nearer. This realization is the basis for new approaches by NOAA to quantitatively standardize the interim loss problem (Fonseca et al. in press).

The assessment strategy to calculate interim loss is based on four steps of analysis:

1. documentation and quantification of the injury,

2. identification and evaluation of restoration options,

3. scaling of the restoration project to compensate for the injury over time, and

4. determine the appropriate means of compensation (e.g., monetary or planting).

The scaling aspect is the portion of the process that helps standardize the way in which interim losses are computed, irrespective of the habitat type involved. Interim lost services can be considered to be the integral of service lost from some baseline level over time (Figure 2.2). To compare services lost with those recovered by some remedial action (such as planting seagrass), the product:

square m of habitat lost x time = square m–years

is set against square m–years of services provided by the planting project, but discounted as a function of time since the initial injury (Figure 2.3). Discounting is a accepted economic principle, used to transform monetary or service flows over time into present value terms for purposes of comparison. Plantings that occur longer after an impact are discounted more than plantings conducted shortly after an impact and therefore more planting must be done as more time elapses. The NOAA Damage Assessment and Restoration Program is currently applying this procedure to seagrasses and it now has been upheld in court (U.S. District Court, 92-10027-CIV-DAVIS). Initial results appear promising but require an empirical assessment of rates of recovery by seagrass. These rates are poorly known and experiments have been implemented to provide additional data. Population growth and coverage rate information has been useful (*sensu* Fonseca et al. 1987a,c) and supports previous claims for the need to collect these kinds of data as part of any monitoring of seagrass plantings (Fonseca 1989a, 1992, 1994)

Pre-Impact and Pre-Planting Surveys: Identifying Presence, Absence and Reasons for Absence of Seagrass Coverage

Lockwood (1991) provided guidelines for surveying sites prior to impact and how to interpret these data to plan subsequent plantings. Essentially, any quantitative survey method will work, such as line transects or grid sampling, but some basic quantitative standards such as presence/absence of a seagrass species over known areas must be met. Sampling for the presence/absence of seagrass should encompass the entire impact site on the closest spacing practicable and clearly specify the range over

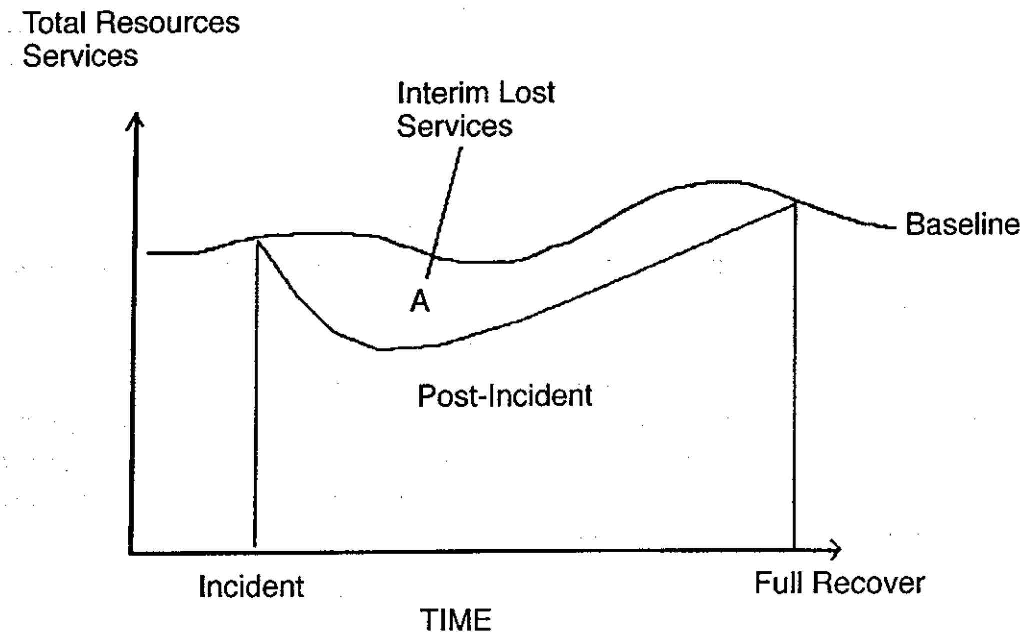


Figure 2.2. Diagrammatic representation of the computational process used for assessing interim loss of seagrass habitat functions (from B. Julius, DARP/NOAA).

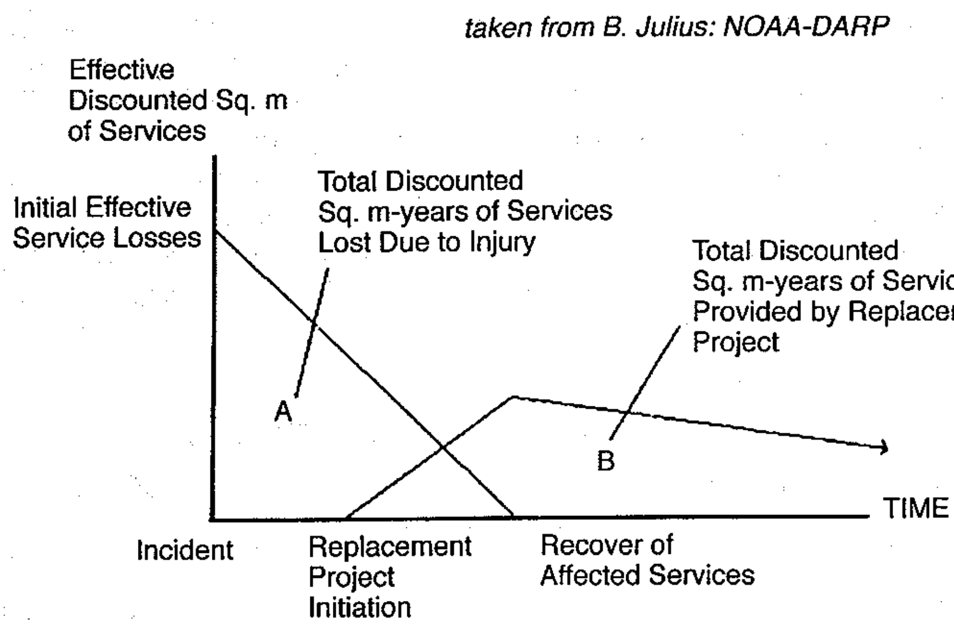


Figure 2.3. Diagrammatic representation of the discounting process used to compute replacement ratio for planting seagrass under a mitigation scenario (from B. Julius, DARP/NOAA).

which sampling is to be done and the resolution with which samples are to be taken. For example, the following description would be considered adequate for such a report:

> **On (dates) [representing spring and summer growing conditions], five 100-m transects spaced 20 meters apart were swum by divers over the site who then recorded the average percent cover seagrass [percent of 16, 25 x 25 cm subunits within a 1 m^2 quadrat that contained a minimum of 1 seagrass shoot] where 1 m^2 quadrats were assessed on 10-m intervals along each transect). Alternatively, a video camera may be towed over a transect line marked at 1-m increments and the cover estimated using a point-grid method (e.g., Braun-Blanquet 1965, Virnstein 1995).**

The depth distribution and coverage of each species present should be recorded. If seagrass occurs as small isolated patches of grass among open, unvegetated areas, then the sum total area occupied by the patches (portion of the sea floor where rhizomes overlap) could be recorded per unit seafloor (where the unit area of seafloor is at least a minimum of 100 m^2). If plants are very large and separated from each other then the aforementioned quadrat method will be more appropriate for assessing coverage. Data on species composition should be used to guide the selection of species for later planting. Further, these data can be used to determine the amount of seagrass that can be salvaged for planting other sites or potentially stored for replanting onto the original site if the disturbance is short-lived.

Aerial photographs of appropriate resolution can provide useful information for evaluating existing seagrass beds. A time series of aerial photographs, (preferably ~1:20,000 scale) if available, can be particularly useful in determining the dynamic nature of a site. Photographs should be used only if taken during the peak biomass season for the seagrass in question. Moreover, it is sometimes difficult to accurately determine the lower depth limit of seagrass on a site from aerial photographs. Lower depth limits of seagrass distribution should be verified by on-site inspection, especially if bottom features which are clearly deeper than the apparent lower limit of the seagrass in the picture cannot be discerned in the photographs. However:

> **If aerial photographs taken over a ten-year period indicate no history of seagrass cover, then the potential planting site should be regarded as marginal, or better, rejected.**

Episodic seagrass cover on a potential planting site, either among years or seasons (as might be the case with seed recruitment) would suggest that planting there would only pulse the system and not provide sustained habitat replacement. As these caveats imply, unlike many other wetlands where site engineering is often an option, it is very difficult to locate a planting site that will provide self-sustaining seagrass habitat. Also, as discussed under the section titled "Spatial Scale and its Role in Defining Seagrass Habitat," aerial photography must be taken with sufficient resolution to detect the smallest patches of seagrass in the target area; this may require fairly low overflights (< 1000 feet) which may present significant problems in certain areas where Federal Aviation Administration rules prohibit low flights. S. Nixon and B. Kopp (Graduate School of Oceanogr., Univ. Rhode Island, Narragansett, RI., pers. com.) have employed Geographic Information Service technology to overlay water column light transmissivity, water depth, wave exposure and other factors in a site selection process that may become a model approach for regional restoration planning.

Minimum Size to Justify Pre-Planting Monitoring

Environmental data or pilot test planting results should be collected to provide an indication of planting success prior to the commitment of the entire project's resources. However, personal observations suggest that some plantings may be sufficiently small (~500-1000 planting units) so the cost of collecting environmental data and performing pilot plantings are equivalent to the cost of planting the entire site itself.

Planting Site Selection and Off-Site Vs. On-site planting

Selecting an appropriate planting site is perhaps the single most important step in the entire process. It is also the step that is the most difficult to objectively verify. This is because the circumstances contributing to the presence or absence of seagrass at a given site vary tremendously (see "Pre-Impact and Pre-Planting Surveys," above and criteria in Appendix E, p. 211). Planting areas may be classified as either on- or off-site. When an off-site planting area must be selected, whether it be for restoration or mitigation, it must pass a simple, but exacting, test: "If seagrass does not currently exist at the (chosen) site, what makes you believe it can be successfully established?" (Fredette et al. 1985).

The absence of seagrass on what may appear to be an otherwise suitable site often indicates some inherent difficulty in colonization or a temporally dynamic site (e.g., as the result of disease, F. Short, Jackson Estuarine Lab., Durham, NH, pers. com.). In the case of disease-induced loss of cover, planting may be considered similar to planting among existing seagrass patches in that temporarily freed-up bottom space would be used as an inappropriate planting site. Planting among patches of existing natural seagrass should also be rejected because this too will only pulse the system and not create any long-term increase in seagrass acreage and this space is soon required by the spatially dynamic seagrass patches (see sections on "Defining Seagrass Habitat" in Chapter 1 and "Constraints Imposed by Physical Setting on Planting Operations," below). The take-home message is that if one contemplates off-site compensatory mitigation, there are usually few, if any sites available that: a) can support seagrass growth and, if they do, b) do not involve habitat substitution, or c) do not satisfy the no-net-loss rule.

One form of off-site planting that meets the criteria discussed above is grading down uplands to elevations suitable for planting. Although this entails the trade-off of upland habitat for seagrass, if that upland is zoned for development, then its conversion to seagrass habitat type is warranted. Other off-site options include filling of dredged areas or areas that have experienced an improvement in water quality (e.g., transparency, temperature, etc.). These latter two choices, however, may include areas which historically supported seagrasses and thus may not be effective to offset seagrass loss in compensatory mitigation (see section "Pitfalls in the Mitigation and Restoration Process"). In the case of on-site planting associated with a particular project (i.e., planting back into a portion of the site which suffered a loss of seagrass), the activity which originally caused the loss of seagrass must have ceased.

ENGINEERED SITES

In many instances seagrass planting takes place on sites that have met the criteria of past seagrass presence, identifiable (and human-induced) agents of loss, and the termination of those loss agents (Thayer et al. 1985). Less frequent, however, is planting on sites that have been specifically engineered to accommodate seagrass planting. Those that have been contoured to appropriate elevations have had good success; in the Laguna Madre, the interiors of dredged material islands were returned to subtidal elevations suitable for seagrass growth and connected to the adjacent sound; both plantings and natural recolonization were successful (Montagna 1993). Work in San Diego Bay in the early 1980's featured the creation of a submarine dike that allowed placement of dredge material shoreward of the dike, raising the bottom to elevations (~ -5 m) with suitable light for seagrass growth (pers. obs.). Short (1993) conducted a similar planting in Great Bay, NH. Both projects created viable seagrass habitats that are currently supporting extensive seagrass cover, even though portions of the New Hampshire site was susceptible to winter ice shear. The drawback to this approach is that naturally unvegetated subtidal seafloor was converted to seagrass habitat; such habitat substitution may not be an acceptable mitigation tradeoff as implied by recent symposia on the important ecological roles performed by unvegetated estuarine sediments (Marine and Estuarine Shallow Water Science and Management in the Mid-Atlantic Region, Atlantic City, NJ).

Other engineered sites include those near Beaufort, NC, where seagrass and salt-marsh habitats were created on old dredged material islands for the purposes of both stabilizing eroding shorelines and experimentally investigating recovery trajectories and linkages among these two habitat types (Fonseca et al. in press.). In this case sites that did not receive subsequent dredged material successfully supported seagrass plantings. Thus, sites have been engineered to support seagrass mitigation and have enjoyed good planting success. Such attention to site preparation is encouraged although the costs in performing such site engineering may often place it outside the realm of possibility for small mitigation projects.

Channel Plantings, Effects of Structures, and Other Human Activities

On-site planting often entails planting into permanently modified areas, such as in the case of channel dredging, and typically cannot accommodate a replanting ratio (planted seagrass area/lost seagrass area) above 1:1. Planting along banks of artificially-created channels is logical if the depth of planting does not exceed that at which the plants occurred prior to dredging even though a larger potential planting area may be created. Channel margins, however, are highly susceptible to subsequent grounding events by vessels which will limit replanting options. In addition, many channel bottoms at navigational depths will not support seagrass due to insufficient light or severe scour from propwash. Channels, being deeper than the surrounding seafloor often act as areas of enhanced deposition, especially of organic material. As a consequence, highly reducing sediments and prolonged periods of anoxia may be found in channels which are typically highly stressful, if not lethal conditions for seagrasses. As a general rule, planting in and around channels is very risky.

As an alternative to dredging access channels, docks are often built to access vessels from land. Docks, however, have their own suite of potential impacts to seagrass beds. Besides actual impacts where dock pilings are installed, unless a dock is narrow and high above the water, it will create a substantial shadow that will reduce seagrass density and biomass, as seen in Massachusetts (Burdick and Short 1995). At the deeper end of the seagrass bed, plants will be killed as the shadow pushes them below compensation irradiance (see section on "Light Requirements for Transplanting," below). Relationships derived by Burdick and Short (1995) in New Hampshire reveal that at those latitudes and tidal regimes, a 1-m wide dock had to be nearly 5 m high to maintain eelgrass (*Zostera marina*) bed quality. Burdick and Short (1995) also found that plank spacing and width were far less important than height of the dock and its compass alignment (north-south docks had less impact than east-west docks because their shadow moved further across the bed, thus mitigating the shading effect). Burdick and Short (1995) recommend that T-shaped docks, with a floating dock beyond the outer edge of the seagrass, minimizes collateral disturbance from propwash and boats settling onto or perpetually shading beds when they are moored at the dock. Similar findings were reported by Loflin (1995) in Florida. While existence of collateral disturbance (e.g., dispersing of fish and wildlife) are not well-documented in seagrass systems, there is also no reason to expect that such disturbances would not occur. These kinds of empirical estimates are greatly needed by resource managers nationwide but must be developed on a regional basis.

Other collateral direct impacts to seagrass beds caused by human activity include propeller scarring and mooring chain scars. Sargent et al. (1995) have found thousands of acres of seagrass beds in Florida either destroyed or significantly degraded as the result of vessel scarring, which includes not only vessels with propellers but jet skis as well (pers. obs.). The source of injuries to seagrass beds is varied, but collateral impacts from otherwise seemingly benign activities such as foot traffic or mooring a vessel to those actually designed to minimize or avoid impacts to seagrass beds (docks) can sometimes result in significant damage. Care must be taken to avoid these impacts.

Constraints Imposed by Physical Setting on Planting Operations

If significant physical alteration such as dredging has occurred at a site, on-site plantings often cannot provide sufficient acreage to prevent a net loss of habitat. Another common, but less obvious physical alteration occurs when bulkheads are installed. Many bulkheads are designed as walls to efficiently reflect waves. This wave reflection effectively doubles the wave energy seaward of the wall, often eroding existing offshore beds and creating a situation where they likely cannot be replanted. When physical alterations are subsequently ameliorated, on-site planting is appropriate and offers one of the few circumstances where substantial acreage can be generated, even though historical levels may not be attainable.

The physical setting will dictate the amount of seagrass coverage per unit seafloor as well as the pattern of that coverage. The organization of the coverage into patches is correlated with wave and current climate (Fonseca et al. 1983). Application of wave and current data into site layout is given below in the section, "Spacing of Planting Units." Physical setting is correlated with sediment stability which can severely limit planting success (Fonseca et al. 1985). In general, sites with high wave exposure and tidal current speeds are difficult to restore because of frequent disturbance.

A precise survey of the physical conditions at the planting site will assist in determining the amount of plant material required later. In the case of mitigation projects, a similarly precise survey of conditions prior to any proposed impact is required to obtain an accurate estimate of the seagrass habitat to be lost. This will allow accurate computation of planting ratios (mitigated acreage vs. impacted acreage) to be computed. For example, if 0.5 acres of continuous cover seagrass bed (i.e., low-energy site) were lost to a project and a high-energy planting site was chosen which would typically support patchy beds one would have to:

1. anticipate planting an acre of bottom to achieve 0.5 acres of cover, and

2. budget for substantial replanting (as much as 50 percent of the original planting) because planting failures increase with higher currents (Fonseca et al. 1985).

We focus on wave and tidal current effects on seagrasses in unconsolidated sediments because we have no quantitative information regarding seagrass growing on consolidated sediments (e.g., *Phyllospadix* spp.) to guide selection of a physical setting. We draw the reader's attention though to work by Turner and Lucas (1985) who outlined spatial and temporal dynamics of a rocky intertidal seagrass community and work by Sousa (1979) who noted an inverse relationship between boulder size and frequency of disturbance, an important factor to recognize if plantings are attached to boulders.

For seagrass growing on unconsolidated sediments, we have some data from southern Core Sound, North Carolina, and Tampa Bay, Florida, to indicate a strong relationship between wave exposure (described below) and tidal current speed. To obtain these relationships in North Carolina, seagrass coverage data was determined by mapping 18, subjectively chosen seagrass study sites in Core and Back Sounds, Carteret Co. (latitude 34.40–34.50 N, longitude 76.20–76.40 W). These sites were chosen based on examination of aerial photographs and ground-truthing to represent the full range of seagrass coverage that could be locally identified. These sites are also colonized by a mixture of *Zostera marina* and *Halodule wrightii*. Carteret Co. represents the primary area of overlap of these two seagrasses on the east coast of the United States. The occurrence here is marked by different seasonal peaks of abundance (Thayer et al. 1984).

Exposure to waves was calculated for each site using methods of the Shore Protection Manual (1977) and Keddy (1982) as reported by Murphey and Fonseca (1995):

$$\text{Exposure} = \sum_{i=1}^{8} (V_i \times P_i \times F_i)$$

where:

i = i^{th} compass heading (1-8)
V = average monthly maximum wind speed in m s^{-1}
P = percent frequency which wind occurred from the i^{th} direction
F = effective fetch

To analyze the effect of both forms of water motion (waves and currents) in North Carolina, exposure values (based on mean monthly maximum wind speeds), and tidal current speed (peak free-stream speed over seagrass at the lunar maxima) were plotted against percent coverage using all the sites from 4 sampling times covering 2 years. The percent coverage for each site was calculated by dividing the total number of one meter square (pixel) observations that had seagrass by the total area of the survey plot (2500 m^2). Site maps were produced from the survey data and each pixel registered as containing seagrass plotted as a square.

These surveys produced significant relationships as seen in Figure 2.4 (Fonseca and Bell in press):

Percent Cover = Exposure index (−0.0135) + 92.525
$r^2 = 0.45$

Percent Cover = Maximum monthly tidal current speed in
cm s^{-1} (−2.644) + 111.044
$r^2 = 0.60$

We are not sure how these data may be applied to other areas because to our knowledge, seagrass plantings have not been conducted to account for anticipated

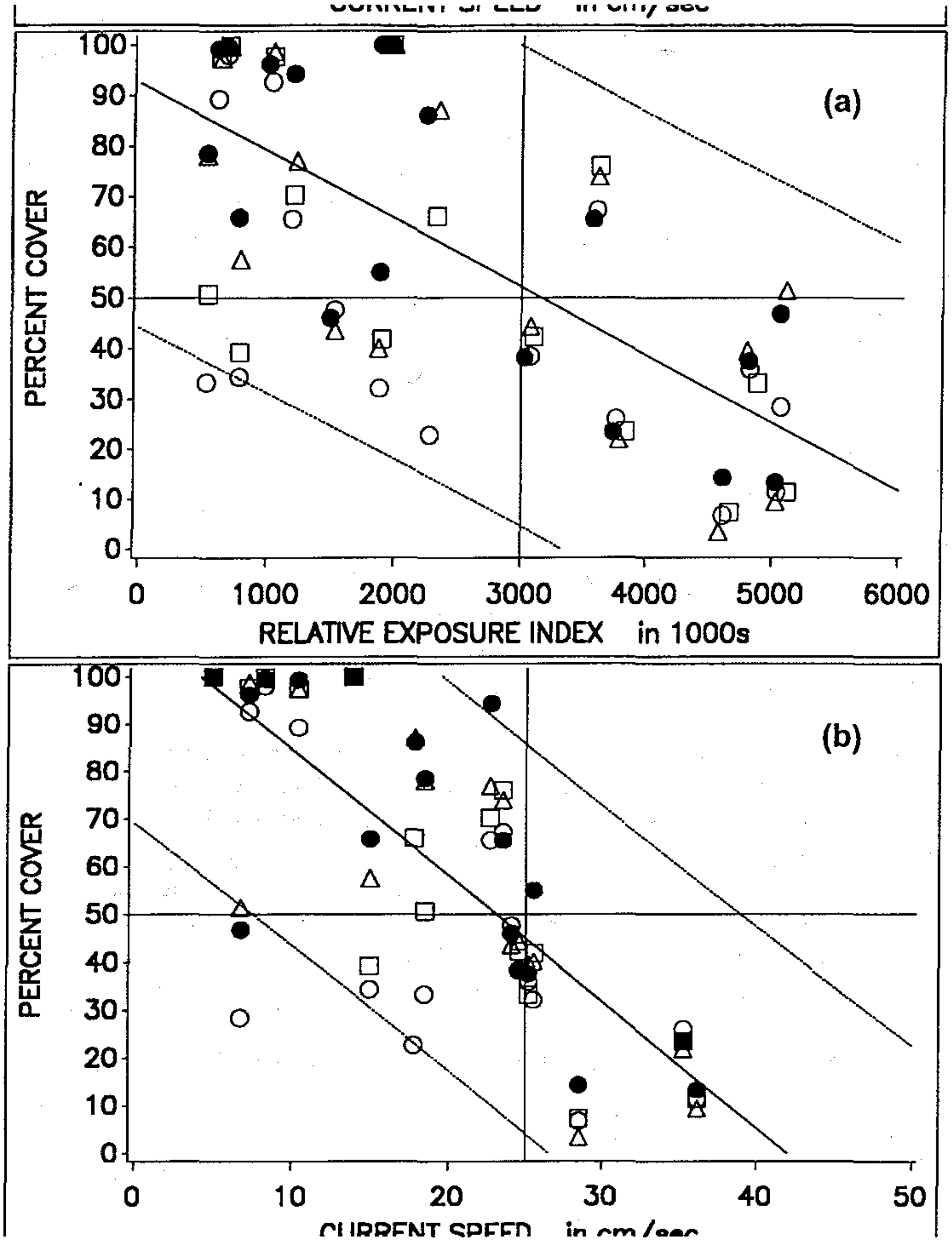


Figure 2.4. Relationship between seagrass cover and physical setting for seagrass beds near Beaufort, NC. (a) Relationship between seagrass cover and exposure index (eq. 1 in text) from eighteen, 50 x 50m plots surveyed with 1m resolution. (b) Relationship between seagrass cover and tidal current speed from eighteen, 50 x 50m plots surveyed with 1m resolution.

landscape patterns. We suggest that in areas where currents speeds are less than 15 cm s^{-1}, only wave exposure be considered in predicting coverage. For lack of better guidance, but taking into account observations of dune formation even in subtropical areas (Fonseca 1996a), we suggest that the tidal current speed model hold for seagrasses in all settings when currents are greater than 15 and less than 50 cm sec^{-1} (tidal current speed > 50 cm sec^{-1} would indicate rejection of the site, Figure 2.5). It should be noted that application of these data are extremely experimental but may represent an important aspect of planning that might be considered in planting operations.

EMERSION EFFECTS

Despite the long-standing recognition of seagrass emersion as a factor in its ecology (Johnson and York 1915, cited in Harrison 1982), until recently little work has been done to document direct effect of emersion on the plants. The effects of emersion vary widely around the U.S.; in the southeast and Gulf states, emersion can cause significant mortality. In northern states eelgrass beds may be regularly exposed at low tide but avoid serious desiccation due to local micrometerological factors, such as fog, cool air temperatures, and high local humidity in the immediate vicinity of the seagrass canopy. Moreover, seagrass beds can also trap water in their canopy as the blades lay over at low tide by making the path for water drainage extremely long by causing flow to wend through the leaves and stems. Powell and Schaffner (1991) reported this phenomena in Florida Bay and it has also been observed in New Hampshire (F. Short, pers. com.) and elsewhere (authors' pers. obs.). In addition, Bulthuis et al. (1984) demonstrated that seagrass beds retain both fine sediments and nutrients while trapping water. Limited work by Harrison (1982) on comparative emersion effects among *Zostera* spp., and more detailed studies by Cooper and McRoy (1988), Cooper (1989) on isotopic variation with emersion, and Perez-Llorens and Niell (1993) on *Zostera* spp. may constitute the entire body of quantitative work on the subject. Perez-Llorens and Niell (1993) describes perhaps the only work on non-rocky intertidal seagrass where an experiment was designed specifically for the effects of temperature and emersion. In that study, photosynthetic rates were significantly higher in water than in air, but a narrow-leaved morphotype of *Zostera noltii* displayed higher photosynthetic rates than a broader-leaved morph. Increased temperature decreased photosynthesis during the 2-h incubation period for both morphs, although the narrow-leaved morph was more resistant. However, emersion time was not varied and recovery of photosynthetic capacity was not measured. Adams and Bate (1994) removed individual blades for desiccation effects and measured chlorophyll fluorescence. They found that the wider-bladed *Zostera* was significantly more

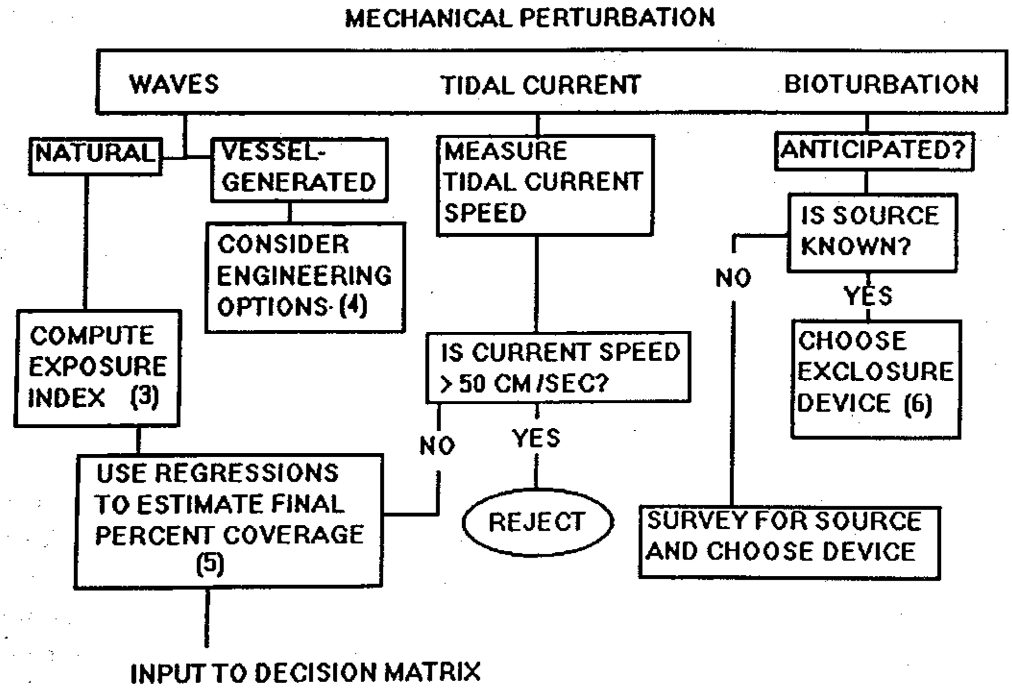


Figure 2.5. Site selection process using physical setting.

resistant to desiccation than narrower blades of *Ruppia*, although tests were conducted on blades in isolation and thus did not account for the possible mitigative effects of a canopy or moisture content of the sediment on desiccation.

The findings of Perez-Llorens and Niell (1993) have implications in other regions. For example, in the Beaufort, NC area, two seagrass species occur at the edge of their distributions — the temperate species, *Z. marina* and the subtropical species, *H. wrightii*. *Zostera marina* leaves are approximately 3-5 mm wide whereas *H. wrightii* leaves are 1-2 mm wide. The narrower-leaved species can exist in slightly shallower water which receives more frequent emersion (*sensu* Perez-Llorens and Niell 1993). Thus, the interaction of species and canopy morphology with tidal regime may be a significant zonation factor as has been documented for macroalgae. If this is the case, then the ability of the narrow-leaved form of *Zostera noltii* to maintain higher photosynthetic capacity at higher temperatures might result from an adaptive advantage for narrow leaves in regulating leaf temperature. These findings are contradicted by those of Adams and Bate (1994) where narrow-leaved species were less resistant to desiccation. Clearly experimental designs that simulate natural settings are needed to generate quantitative measures of desiccation tolerance of seagrasses. Knowledge of desiccation tolerance will facilitate selection of appropriate seagrass species for a site in an ecoregion where such choices may be possible.

Despite contradictory experimental findings, exposure to air is often detrimental to most seagrasses. Human alteration can render many sites too shallow for seagrasses. For example, reduced water depth from dredge material deposition may result in sites where transplants would be exposed at low tides, causing them to desiccate. Sufficient water depth must be maintained to cover the plants even at lowest tides. Very short term events (2-3 hours) may substantially alter seagrass abundance and distribution. For example, desiccation caused by an extreme low tide at mid-day in the summer can determine the upper limit of seagrass distribution in an entire bay for the following year (Beaufort, NC, pers. obs). Areas with high turbidity and tidal amplitudes are extremely difficult to plant given that desiccation at low tide and light extinction at the lower depth limit must both be avoided (E. Koch, Horn Point, MD, pers. com., *sensu* Koch and Beer 1996). Because plantings are generally more stressed than established beds, we suggest (as we will for light) that emersion should be avoided. Arranging planting depth for minimum emersion is best planned by surveying elevations of nearby beds.

Dennison and Kirkman (1996) suggest a balance of tidal elevations for seagrass survival based on the premise that Secchi depth is equal to the compensation point for some seagrasses, while considering tidal range and type. For seagrasses existing intertidally (with the possible exception of *Phyllospadix* spp.) mean range of astronomical tides (MTRA) must be greater than the mean range of barometrically-driven tides (MTRA > MTRB) for their survival. Also, Secchi depth (Zd) must be greater than MTRA (Zd > MTRA). For subtidal seagrasses, Zd > MTRA and Zd > maximum depth of seagrass distribution (Z; Zd > Z) for seagrass survival. This model has not, to our knowledge, been tested in North America.

BIOTURBATION

Prior to rooting and coalescence of plantings, seagrasses are especially vulnerable to bioturbation (Fonseca et al. 1994). Bioturbation is widespread and has been shown to limit distribution of natural beds as well. Ogden et al. (1973) documented the effect of the echinoid *Diadema antillarum* on the formation of grazing halos in seagrass beds associated with West Indian patch reefs. Similarly, Camp et al. (1973) and Valentine and Heck (1991) demonstrated the role *Lytechinus variegatus* in producing unvegetated areas from seagrass beds in the northern Gulf of Mexico. Orth (1975) attributed the destruction of large areas of eelgrass (*Zostera marina*) in the

Chesapeake Bay to the feeding activity of the cownose ray (*Rhinoptera bonasus*). Suchanek (1983) and Harrison (1987) demonstrated the negative impacts of the burrowing shrimp *Callianassa* on the seagrasses *Thalassia testudinum* and *Zostera* spp., respectively. Valentine et al. (1994), however, found that even large animals such as rays were apparently unable to create unvegetated patches within existing *Thalassia testudinum* beds, and that only very large rays were capable of producing pits at the bed-sand margin that resulted in damage to these seagrasses' rhizomes. They also found that sand dollars (*Mellita quiquiesperforata*) did not disturb these edges whereas stone crab (*Menippe* spp.) burrows were disruptive. They point out that the deep rhizome layer of *T. testudinum* (as compared to *Z. marina* or *H. wrightii* as found in the Beaufort, NC area) may insulate these plants from ray and sand dollar disturbance. Various waterfowl can graze down seagrasses (e.g., Black Brant, redhead ducks, mallards, etc.) (Thayer et al. 1984) and can destroy early stage plantings (Beaufort, NC, pers. obs.). If waterfowl grazing is anticipated, then exclosures should be covered on top and not just on the sides. Bioturbation has been linked to maintenance of fragmented seagrass landscapes (Townsend and Fonseca 1998).

Bioturbation is a factor that can require a substantial replanting budget; thus some kind of exclusion device is often needed. For example, Fonseca et al. (1994) found that in areas of Tampa Bay where currents did not exceed 13 cm/sec, a greater than 50 percent loss of planting units occurred due to sediment disturbance, apparently by rays. Fonseca et al. (1994) found that caging of *Halodule* plantings with one inch mesh galvanized chicken wire cages (sides and tops) in Tampa Bay made a difference of < 1 percent survival with no cages versus 60 percent survival with cages. Merkel (1988a) also found extensive disturbance of seagrass transplants in San Diego Bay and used stakes, fencing, and erosion matting in an attempt to improve planting survival. Short (1993) constructed gill-net cages with no tops which excluded horseshoe crabs and green crabs and preserved eelgrass plantings in a New Hampshire estuary. However, Short (pers. comm.) has also reported that certain polychaete worms (e.g., *Nereis*) will pull the blades of early stage eelgrass plantings down into their burrows to feed on epiphytes. This lays the short shoot along the sediment surface where it is then subject to other attacks and burial. Short found that decreasing planting density from 0.5 m on center to 0.1 m centers resulted in slower, but still complete incorporation of blades into burrows. We are not aware of a remedy for this source of bioturbation.

Bioturbation events can occur quickly. We have experienced 100 percent loss of *Halodule* and *Syringodium* planting units within 24 hours of planting (Florida Keys backreef area) due to grazing where chicken wire cages were not used. The lack of good pre-project information on bioturbation potential will usually cost one more in remedial planting than will be saved from planting a minimum of material for a test of bioturbation at the onset.

SEDIMENT THICKNESS

Insufficient sediment thickness (e.g., bedrock too near the surface) has been shown to be limiting to the distribution of some seagrasses (Zieman 1982b), particularly *Thalassia testudinum*. Although not documented for any other seagrass, the potential for exposure of bedrock by currents due to shallow unconsolidated sediments should be considered when choosing a site. In relatively quiescent areas, we have successfully established *H. wrightii* and *S. filiforme* beds on sites with as little as 15 cm of loose carbonate sand over bedrock (Fonseca et al. 1987a). Generally, species with shallow root and rhizome systems (e.g., *Halophila, Halodule, Zostera*) may not be inhibited by thin veneers of sediment.

SEDIMENT STABILITY: EROSION AND BURIAL OF SEAGRASS SHOOTS

Generally, sediment stability is going to be correlated with wave exposure and tidal current speed. The relation between exposure and/or currents and sediment stability is difficult to predict given variation in sediment grain size (therefore different erosion thresholds) among sites and the episodic nature of wind events. The apparent threshold responses evident in Figure 2.4 suggest, however, that some erosion threshold may be represented by exposure indices near 3×10^6 and tidal current speeds of 25 cm sec^{-1}. Merkel (1992) suggests that erosion rates of 0.5 mm day^{-1} and burial rates of 0.3 mm day^{-1} are limits for *Z. marina* survival on the West Coast. These data compare favorably with the sediment fluctuation limit of ~1.0 mm day^{-1} found on the East Coast for both *Z. marina* and *H. wrightii* (Fonseca et al. 1985).

Conversely, there are few data to indicate critical burial depths; those depths likely vary among species. However, our preliminary data for *H. wrightii* (unpubl. data) indicate that when 25 percent of the shoot is buried, 75 percent of the plants survived, but when 75 percent of the shoot was buried only 5 percent survived. This response suggests an exponential decline of survival with percent burial. We are aware of no data on North American species that gives guidance on the duration of burial time after which recovery of the plants would be expected (but see Marba and Duarte 1995, Terrados et al. 1998).

Prior to planting, measurement of sediment fluctuation relative to numerous (minimum of five) fixed datum established on-site is recommended (*sensu* Fonseca et al. 1985, Merkel 1992). The number of datum should be increased to as many as practicable, covering all areas of a planting site where differences in sediment erosion and accumulation are anticipated. Readings should be taken daily if possible for at least one lunar cycle at the same time of day to allow measurements at as many tidal stages during a lunar cycle as feasible. Merkel (1992) recommended placing a one foot long 1/2-inch diameter PVC pipe halfway into the sediment as a datum and measuring sediment elevation relative to the top of the pipe. In areas of moderate waves and/or currents, we suggest that the pipe be longer and buried deeper into the sediment, up to 50 cm. We also suggest that an inverted T-shaped device for measuring sediment height be used. The cap of the T should be a 25 cm long segment of a wooden yard or meter stick bolted through the leg of the T so it can pivot when placed on the bottom. In this way the effect of local scour around the pipe on computation of sediment elevation change is minimized. These methods are designed for detection of chronic conditions; the reader should keep in mind that extreme, aperiodic events often determine limits to distribution of seagrass (*sensu* Gaines and Denny 1993).

Possibility of Natural Recolonization

One question that repeatedly arises is the potential for natural recolonization and, thus, avoidance of the cost of planting. The ability of seagrass to recolonize a site is very difficult to predict. Rapid recolonization by *Z. marina* has been observed in North Carolina (Kenworthy et al. 1980, Fonseca et al. 1990) and British Columbia (Harrison 1987), as has *H. wrightii* in the Florida Keys (Thayer et al. 1994) and *Halophila* spp. (Kenworthy 1992). Annual populations of *Z. marina* in Nova Scotia, the Gulf of California, and San Francisco Bay require seeding for year-to-year persistence (in fact, planting of vegetative stock using the techniques designed to capitalize on persistent vegetative growth is wholly inappropriate for these annual populations; Fredette et al. 1985). However, it has been our observation that seedling recruitment success in some seagrasses (e.g., *Z. marina* and *T. testudinum*) have some years with extraordinary seed and seedling production. In the interim, seedling success appears to be minimal. Of course this observation could be confounded by grazing, sediment disturbance, etc., and seed production and seedling germination could exhibit little interannual variation. However, we do know that seedling recruitment to unvegetated areas varies with current regime; high current areas can have very low- to nonexistent seedling recruitment while relatively quiescent areas have heavy seed sets (Fonseca and Kenworthy 1987). Thus, there is evidence to suggest that site conditions alone can influence recolonization potential (Orth et al. 1994).

Recolonization can also occur vegetatively (rhizome extension) from any adjacent seagrass. The proximity of plants and the geometry of the open area combine to produce variable recovery scenarios that are currently unquantified. However, plants close to a small (several meters wide) open area will, in the absence of seeding or fragment colonization (*sensu* Cambridge et al. 1983), colonize that area more rapidly than a larger area, a process very much like that described for patches opened in coral reefs and rocky intertidal ecosystems (Connell and Keough 1985). Bioturbation can arrest recovery of open areas (see "Bioturbation" section, above). Therefore, natural recolonization of even small-to-moderately-sized open areas can potentially be arrested, but conversely may be stimulated by the use of bioturbation exclusion devices. It is critical though, that the reason for lack of recolonization be determined. Propagule limitation and inappropriate environmental conditions (i.e., periodic exposure at low tide) can yield similar results: no coverage. We conclude that natural recolonization is almost always such a chance occurrence that is strongly influenced by disturbance events, that management practices should not, in the absence of some pilot data (e.g., monitoring of a site with planting held in abeyance or prior local quantitative observations of recolonization on similar sites), rely on natural recolonization to restore coverage.

Nutrient Requirements for Transplanting

When considering the nutrient requirements of seagrass transplants there are three important site-specific questions that require attention. First, are there sufficient nutrients to support the growth and reproduction of transplants? Second, are nutrients present in excess of what the seagrasses can utilize, thereby available to stimulate epiphytes, phytoplankton, and macroalgal growth? Finally, are nutrient concentrations toxic to the plants? Recent evidence suggests that NO_3 may be toxic to some seagrass species (Burkholder et al. 1992). Uptake by the plants of this form of nitrogen apparently cannot be controlled, possibly leading to a loss of flexural stiffness which makes the plants lay over (leading to the same problems encountered by F. Short with *Nereis* bioturbation). In general though, there is very little additional information suggesting that any of the major macronutrients (N, P, K) occur in high enough concentration to negatively affect seagrasses. This also includes a consideration of organic herbicides in surface and ground waters, which do appear to have a negative effect on seagrasses in concentrations observed in the field (Schwarzschild et al. 1994).

There is a large body of evidence indicating that mature, well established seagrass beds can be nutrient limited under certain conditions, despite large reservoirs of nutrients in the sediments (Short and McRoy 1984, Dennison et al. 1987, Short 1987). These conditions include periods of time when optimum temperature and light regimes coincide to allow very high rates of primary production by the seagrasses (Perez et al. 1991). During these periods seagrasses extract nutrients from the available reservoirs faster than they can be regenerated by biogeochemical processes, eventually exhausting their resources (Short et al. 1985). The strongest and most consistent evidence for nutrient limitation has been demonstrated for seagrasses growing on sediments with sufficient amounts of biogenically derived carbonate to tightly adsorb phosphorus (Powell et al. 1989, Short et al.1990, Perez et al. 1991), although they may also be limited by phosphorus on siliceous sediments (Murray et al. 1992).

The largest reservoir of nutrients available for seagrasses are the sediments (Kenworthy et al. 1982, Short 1987, Fourqurean et al. 1992a). With the exception of one genus, *Phyllospadix*, all seagrasses grow rooted in soft sediments. Results from a wide range of studies, including comparisons of porewater and water column nutrient concentrations, sediment organic content, functional anatomy, and physiological ecology, all suggest that seagrasses can derive their nutrition from both the sediments and the water column (for a review, see Short 1987). However, because of relatively higher concentrations of dissolved inorganic and organic nutrients in the interstitial water, seagrasses obtain most of their macronutrients from the sediments. Fertilization experiments, which have added nutrients to the sediments, confirmed this for inorganic nitrogen and phosphorus with several North American species including *Z. marina*, *H. wrightii*, *S. filiforme*, *T. testudinum*, and *R. maritima* (Orth 1977, Orth and Moore 1982, Short 1987, Powell et. al. 1989, Short et al. 1990, Murray et al. 1992), as well as for other species such as *Cymodocea nodosa* (Perez et al. 1991) and *Heterozostera tasmanica* (Bulthuis and Woelkerling 1981). For *Phyllospadix* spp. most, if not all, of their nutritional requirements must be met by water column constituents. Because of very low nutrient concentrations in the water column, *Phyllospadix* species depend on the flux of nutrients generated by water movement and may be nutrient limited more frequently than other seagrasses (but see Koch 1993).

The nutrient requirements of some genera have received minor attention. Very little is known about the nutritional requirements of the three *Halophila* species living in the southeastern United States and Caribbean region. All three, *H. decipiens*, *H. engelmanni*, and *H. johnsonni*, grow on soft sediments, but the roots only penetrate a few centimeters into the substrate. Because rooting depths are much shallower than any of the other species, reduced access to the larger sediment reservoir may result in nutrient limitation. Likewise, *Halophila hawaiiana*, a seagrass endemic to the Hawaiian Islands, and *Halophila ovalis*, another species common in the Pacific territories, have a morphology and rooting depth similar to the other three *Halophila* species and should also have limited access to the sediment nutrient reservoir.

Sediment reservoirs may also become depleted for species which depend on the substrate for their nutrition. This occurs when regeneration rates cannot keep up with plant demands, leading to nutrient limitation (Short 1983). Since established beds of clonally integrated plants can exhibit nutrient limitation, it is expected that young, independently (i.e., not physiologically integrated with a larger clonal unit) developing shoots of transplanted seagrasses can experience more severe limitation. Young, transplanted shoots have immature root systems, and support by translocation of nutrients from neighboring plants in the clone may be interrupted if (when) the rhizome if broken during transplanting. This is exacerbated by the fact that the rhizosphere is usually disturbed during planting. Disturbance may be more severe with bare root planting techniques than if cores are utilized. Cores usually retain the sediments intact so that the substrate is planted together with the seagrasses, minimizing disturbance to the biogeochemical recycling processes. Considering the previous discussion and a very large body of evidence indicating that seagrasses are nutrient limited, we expect that the survival and growth of seagrass transplants can be improved by addition of nitrogen and phosphorus to sediments (Kenworthy and Fonseca 1992).

Despite the evidence suggesting that seagrasses can be nutrient limited, the few studies examining nutrient fertilization of seagrass transplants have demonstrated inconsistent results (Orth and Moore 1982, Kenworthy and Fonseca 1992). Reasons for this variation include the possibility that the delivery of nitrogen and phosphorus is altered by the presence of flooded, anaerobic sediments. Most commercial fertilizers were developed for terrestrial sediments with a much smaller fraction of water to solubilize the fertilizer in proportion to a very large fraction of soil surface to adsorb the inorganic ions as they are released. Granulated fertilizers leach nutrients rapidly in flooded soils, possibly much faster than can be adsorbed or utilized by the plants. Granulated fertilizers are also difficult to deploy in a flooded sediment without a means of containing the granules. Slow release fertilizers show the most promise, although past evidence has shown variable release rate characteristics with fertilizers containing both nitrogen and phosphorus (Kenworthy and Fonseca 1992). Unfortunately, many studies using the slow release forms in field and greenhouse experiments did not directly test the release characteristics of the fertilizers nor were many of these experiments done on transplants (Pulich 1985, Short et al. 1990, Erftemeijer et al.1994). The majority of the studies were done with established or patchy seagrass beds which may not be indicative of the response of transplants. Slow release forms of inorganic and urea nitrogen (38 percent N) and phosphorus (41 percent P), which are encapsulated separately, seem to perform more consistently and show some promise (e.g., Osmocote™). Slow release forms are more appealing because they are designed to deliver nutrients over an extended period of time in a steady dose during the most important and vulnerable time for developing transplants. Research has demonstrated the utility of slow release forms with the delivery of nitrogen and phosphorus in agar-nutrient mixtures (Perez et al. 1991, Murray et al. 1992) and improved forms of encapsulated commercial fertilizers (Fred Short, Jackson Estuarine Lab., Durham, NH, pers. com.). These studies have suggested that slow release forms will stimulate the growth and reproduction of seagrasses as well as the nutrient content of their tissue, which is a strong indication that the plants are nutrient limited and utilizing the supplemental nutrients provided by the fertilizers (Fourqurean et al. 1992b).

Originally, Kenworthy and Fonseca (1992) thought that by encapsulating the nitrogen and phosphorus separately, problems originally encountered with the lack of phosphorus release would be overcome. Subsequent work (Fonseca et al. 1994) has shown that even phosphorus alone does not always release on schedule although nitrogen pellets appear to follow manufacturers' specifications. The point here is that fertilizer additions have not always performed as anticipated based on terrestrial applications.

Kenworthy and Fonseca (1992) and Fonseca (1994) recommend that no reduction in planting effort should be enacted in anticipation of fertilizer benefits. However, they also note that no negative effects have been reported meaning fertilizer use is either neutral (considering the small additional cost per planting) or positive (accelerating new shoot formation). When employed, fertilizers should be added to the sediments at the time of planting and during the growing season of the particular species. If peat pot methods are used, a measured dose of fertilizer can be placed in the pot prior to installing the seagrass plug. If a coring technique is used, the fertilizer can be installed into the hole where the plants are to be placed after extracting the plug of sediment, or placed directly into the core that is to be planted (see "Methods" section in Chapter 3). In both cases the fertilizer may be installed into some type of porous container such as tissue paper (Bulthuis and Woelkerling 1981) or fine mesh screen (Kenworthy and Fonseca 1992). This helps avoid the problem experienced by Bulthuis et al. (1992) where bioturbation redistributed buried fertilizers onto the sediment surface, essentially terminating their effectiveness as a source of nutrition for the seagrasses. Preferably, the material containing the fertilizer should be biodegradable. Pre-weighed fertilizer can be packaged inside small paper envelopes, although these can disintegrate in water during handling. The envelope is stapled shut and installed with the planting units. Recommended application rates are between 5 and 10 g of balanced N-P slow release fertilizer (including the capsule) per planting unit (Kenworthy and Fonseca 1992).

For best results it may be necessary to reapply fertilizer within the prescribed release period, as long as the time is within the window of the species' growing season. The best way to determine the need for reapplication is to set aside a representative sample of fertilizer packets as controls and periodically recover replicate samples to determine the residual fertilizer, and thus, the release rate (see Kenworthy and Fonseca 1992); or simply re-fertilize in accordance with the release schedule given for the fertilizer at ambient temperatures.

Nutrients may also be added with commercially available fertilizer stakes which can be purchased at garden and hardware stores (Williams 1990). Stakes are easier to work with because the fertilizer is compacted into one single unit rather than several pellets, eliminating the need for a container. These stakes are another form of slow release but their specific delivery-rate characteristics in marine and estuarine sediments are unknown. When planting in terrigenous sediments both nitrogen and phosphorus fertilizers should be applied. In pure carbonate sediments it has appeared as though fertilization only with phosphorus (thereby avoiding the stimulation of any other macrophytes or microalgae by added nitrogen) was needed. However, Duarte et al. (1995) has shown that iron limitation may play an important role in *Thalassia* and *Syringodium* growth in carbonate sediments. We speculate that use of iron staples as anchors in planting, as recommended later, may inadvertently contribute to overcoming that limitation.

An important consideration before planting is the status of water column nutrients at a restoration or mitigation site. An excess of nutrients (from outside the planting, not from fertilizers installed in the sediment as part of the planting process) can lead to the overabundance of chlorophyll in the water column and, eventually, to severe light limitation for the seagrasses (Twilley et al. 1985, Dennison et al 1993). Excess nutrients can also lead to the growth of nuisance macroalgae which compete with seagrasses for space and light. Blooms of macroalgae may actually overgrow seagrasses and smother them (Harlin and Thorne-Miller 1981, Walker and McComb 1992, Short and Burdick in press), even the large climax species like *T. testudinum* (Tomasko and Lapointe 1991). The smaller *Halophila* species, seedlings of larger species, and young developing transplants of all seagrasses are especially vulnerable to overgrowth and displacement by macroalgae.

Nutrient enrichment in the water column and the general degradation of water quality are also responsible for stimulating the overabundance of epiphytic algae which grow on seagrass leaves (Sand Jensen and Borum 1983, Borum 1985, Silberstein et al 1986). Excessive amounts of epiphytes will shade out light and diminish the productivity and growth of seagrasses (Bulthuis and Woelkerling 1983, Wetzel and Neckles 1986, Neckles et al. 1993). Young, newly-established transplants adjusting to the shock of planting are particularly vulnerable to overgrowth of epiphytes; therefore, the nutrient status of the water column is an important consideration when selecting a planting site.

Generally, it is not practical to measure water column nutrients at the precision and frequency necessary to determine if there is a statistically significant excess present. Because inorganic nutrients in the water column are utilized and turned over so fast, an excess may not be detected by sampling the dissolved forms of these nutrients in the water column (Tomasko and Lapointe 1991). This is especially pertinent for inorganic phosphorus in subtropical-tropical systems of south Florida, Puerto Rico, the Virgin Islands and possibly the Pacific territories (although effects of volcanic soils on the sediment chemistry and seagrass growth is poorly understood) where carbonate sediments are the primary substrate (Fourqurean et al. 1992a).

Alternatively, much more practical indicators of nutrient enrichment or limitation are the organisms themselves. There are three reliable indicators that can be used as semi-quantitative descriptors of the site-specific nutrient regime. The first is the amount of phytoplankton chlorophyll which reflects the concentration of nutrients in the water column (Smith et al. 1981, Valiela et al. 1990). Sustained water column chlorophyll concentrations in excess of 10-15 mg l^{-1} are usually indicative of nutrient enrichment and a general degradation of water quality (Batiuk et al. 1992, Dennison et al. 1993, Table 1.2). At these concentrations chlorophyll can make a significant contribution to water column light attenuation (McPherson and Miller 1987, Gallegos 1994) and be detrimental to seagrass transplants.

If long-term data for dissolved inorganic nitrogen and phosphorus are available for potential planting sites located in the mid-Atlantic, Northeast, Pacific West Coast and Pacific Northwest, then the nutrient criteria provided in Dennison et al. (1993) can be used to accept or reject a location. These criteria would apply mainly to sites with terrigenous sediments and not with biogenically derived carbonates. Dennison et al. (1993) showed that polyhaline regions of the Chesapeake Bay supported *Z. marina* where dissolved inorganic nitrogen and phosphorus were < 0.10 and < 0.67 uM, respectively. These are median values derived from data obtained over several years, but only for the growing season of the plants (March to November). At these same locations median values for chlorophyll a were < 15 ug l^{-1}. Keep in mind that these criteria were developed using established seagrass beds that persisted over multiple growing seasons and define upper threshold values.

The success of newly developing plantings could potentially be improved by avoiding sites with similar maximum concentrations as those described by Dennison et al. (1993). We suggest that, as a first guess, locations with concentration values ~25 percent lower than the predicted maximum constitute a reasonable starting point to introduce plantings. These criteria (Table 1.2) are probably reasonable for planting sites in the northern Gulf of Mexico and Texas, but are probably much too high for comparatively oligotrophic conditions of southeast Florida, Florida Bay, Florida Keys, Puerto Rico, the Virgin Islands, and the Pacific Territories. In these oligotrophic waters with carbonate sediments, larger amounts of nitrogen and phosphorus are normally tied up in plant biomass and sediments. Water column concentrations of nitrogen and phosphorus in the ranges reported by Dennison et al. (1993) would result in the relaxation of nutrient limitation for phytoplankton and benthic macroalgae, likely leading to eutrophication (Tomasko and Lapointe 1991). Unfortunately, there are no comprehensive studies that define nutrient criteria as specific as Dennison et al. (1993) for the aforementioned carbonate environments. We recommend that inorganic phosphorus criteria for suitable planting sites in areas of carbonate sediments be established at concentrations an order of magnitude less than reported by Dennison et al. (1993) and Tomasko and Lapointe (1991).

A second indicator of nutrient effects is the tissue Redfield Ratio or C-N-P content of the seagrasses themselves (Atkinson and Smith 1983, Short 1987, Duarte 1990, Fourqurean et al. 1992b, Perez et al. 1994). If seagrasses are present in the vicinity of a planting site, evaluation of their tissue nutrient composition can be used to determine the necessity for fertilization or provide some indication that there are excess nutrients. In nutrient enriched sites seagrasses have higher than average tissue concentrations for the nutrient occurring in extra abundance relative to the composition of plants at sites isolated from enrichment (controls). Deviation in the tissue concentration of a particular element can provide a clue as to which nutrient is either limiting or in excess. Duarte (1990) suggested that seagrass leaves with median nutrient levels of <1.8 percent N and <0.2 percent P are strongly nutrient limited. If seagrasses in the area at or near the planting site display similar levels, fertilization of transplants could be helpful. Likewise, deviations upwards from these values would indicate either adequate nutrients or, possibly, nutrient enrichment at the site. The advantage of using the plant tissue is that the seagrasses act as a barometer for continuous longer-term monitoring of their environment and reveal the conditions without sampling error (Dennison et al. 1993).

A third indicator of nutrient enrichment is the presence of large amounts of macroalgae, especially the faster growing species of green algae like *Ulva* spp. or *Enteromorpha* spp. These non-vascular plants utilize nutrients at a much faster rate and have higher turnover than seagrasses, allowing them to out-compete vascular plants for essential nutrients (Harlin and Thorne-Miller 1981, Lapointe and Clark 1992, Walker and McComb 1992). There are no quantitative criteria defining a threshold amount of macroalgae that is detrimental to seagrasses, but if quadrat sampling indicates macroalgal cover in excess of 50 percent of the bottom it is likely that seagrasses will be negatively impacted.

Light Requirements for Transplanting

All seagrasses in the United States with the exception of one genus, *Phyllospadix*, grow in flooded and chronically anoxic sediments. This section focuses on those seagrasses in unconsolidated sediments given to anoxia. Light requirements for the intertidal *Phyllospadix* spp. are unknown, but their distribution is likely more tied to emersion limitations (see "Emersion Effects" above).

To survive and grow in anaerobic sediments, seagrasses require photosynthetically-produced O_2 to support the aerobic metabolism of non-photosynthetic root and rhizome tissues and the dark respiration of leaves (Smith et al. 1988b). In the absence of available O_2, less efficient anaerobic fermentation leads to a demand for carbohydrate reserves which may be provided by stored material in the rhizome or translocated from healthy adjoining shoots in the clone (Harrison 1978, Dawes et al. 1987, Libes and Boudouresque 1987, Tomasko and Dawes 1989). Transplanting may disrupt the physiological integration between rhizome and adjoining shoots, limiting the ability of healthy shoots to support stressed short shoots (ramets), depending on the amount of physiological integration among shoots in the first place (a phenomenon that is now only poorly understood). In mature, well established meadows, neighboring shoots attached by the same rhizome may contribute to the survival and growth of other shoots by translocating carbon and nutrients which can be utilized during periods of time when resources are depleted. This relationship can be especially critical for young, developing plants that have not yet produced enough photosynthetic tissue to be independent of clonal support. Young developing shoots are very important when transplanting because they are the basis for survival and expansion of the planting. Therefore, if young shoots are physiologically dependent on adjacent, older shoots, the light requirements of transplanted seagrasses require special attention because they are likely to be higher than the requirements determined from established meadows. Moreover, there are cases in terrestrial systems where young ramets are sacrificed by older ramets should those young ones venture into physiologically unsuitable areas. Even if this were the case in seagrass, most planting stock consists of fragments of the root-shoot-rhizome complex and the integration of shoots is broken in any event. Therefore, even if young shoots were to be sacrificed by older ones, the feedback mechanism would not exist when rhizomes are fragmented during planting, therefore light requirements should still be higher for the young shoots.

When transplanting with mature shoots, either completely removed from the sediment (bare root technique) or left intact in a sediment core (plug), the rhizomes are severed. Presumably, the plug method would be less stressful because of minimal disturbance to the rhizosphere. However, in both cases transplant survival and new growth depend on the formation of vegetatively reproduced shoots which may have less support available than shoots growing in a dense, clonally-integrated meadow.

Seagrass minimum light requirements have been determined by three approaches: (1) photosynthetic measurements, (2) whole plant carbon balance models, and (3) correspondence (correlation analysis) between light availability and maximum depth of seagrass growth. Photosynthetic measurements alone are inadequate and have severely underestimated seagrass light requirements (Drew 1979). Carbon balance models have improved our understanding of light requirements because they account for the additional carbon requirements of non-photosynthetic tissue (Zimmerman et al. 1989, Fourqurean and Zieman 1991, Zimmermen et al. 1994). Correspondence analysis between some statistical average light level (mean or median) and the maximum depth to which seagrasses grow uses a long term response by the plants to record the requirement (Dennison 1987, Duarte 1991, Kenworthy and Haunert 1991, Dennison et al. 1993, Onuf 1994). The third approach is particularly useful because it depends on the plants interacting with their environment to reveal their actual response, which has usually indicated a higher light requirement than either of the first two approaches (Kenworthy and Haunert 1991). Even though a correspondence analysis seems more appropriate to estimate light requirements, it still must be adjusted upward for estimating transplanting requirements. The deep edges of the beds used as the barometer in the correlation between light and depth of growth are formed from well-established stands and are probably maintained by support from adjoining shoots. Moreover, the level of resolution in a correspondence analysis does not contain enough information about the possible influence of short-term departures from average light levels (Zimmerman et al. 1994). The survival of individual shoots in a transplanting unit is thus potentially vulnerable to short term fluctuations in light levels whereas mature beds are buffeted by reserves within the clone. For these reasons we recommend that the light environment necessary for the initial survival and growth of transplanted seagrasses exceeds minimum requirements for established meadows.

Another important consideration is the fact that light requirements are not the same for all species of seagrass. Large differences occur between genera that are based on growth form of individual shoots, clonal architecture, and physiology. In temperate regions, both in the Northeast and on the West Coast, the dominant seagrass, *Z. marina*, has a much different growth architecture than any of the five species found in the southeastern United States. Initially, on *Z. marina*, vegetatively reproduced short shoots are arranged morphologically and physiologically close to the parent shoot from which they are formed. This relationship is only temporary because both the parent shoot and its offspring grow away from each other quite rapidly. Horizontal rhizome growth separates the mature plant from the younger shoot and the metabolic activity of the rhizome diminishes rapidly as the older nodes age, severing the physiological coupling between ramets in the clone (Kraemer and Alberte 1993). *R. maritima*, *Z. noltii*, and *Z. japonica* all have a growth architecture and morphology similar to *Z. marina* and should be as sensitive to light-limiting conditions immediately following planting.

In the southeastern ecoregion, the potential for integrating shoots is greater for the three larger and most common subtropical genera, *T. testudinum*, *S. filiforme*, and *H. wrightii*. For these species horizontal internode distance is deterministic and adjoining short shoots remain the same distance from one another, usually throughout their entire life span. For all these species, adjoining shoots in a clone have far greater potential for sharing resources and supporting one another than is available for *Z. marina*. In *Zostera*, horizontal rhizome growth and new shoot formation are not as closely coupled and, therefore, clonal integration does not likely make as great a contribution to survival. The individual *Zostera* shoot in the initial stages of planting is independent of the clone sooner and may be more vulnerable to physiological stress than either of the three larger subtropical species.

The fourth tropical genus and the smallest of all seagrasses, *Halophila*, appears to have the lowest light requirement. *Halophila decipiens* is usually found growing in the deepest and most turbid water and is almost never observed in the canopy of the larger species; these are attributes that make its detection by conventional remote-sensing methods unlikely (Dobson et al. 1995). Because it has thin leaves and relatively lower root rhizome biomass it was once believed to have been better equipped to survive in low light environments (Josselyn et al. 1986). But this conclusion was drawn without full consideration of the life history of the plants and the seasonal variation in light. Many coastal environments have regular seasonal fluctuations in light which include periods when light is well above and well below the average. Part of the explanation for growing in low light environments is that *H. decipiens* reproduces by seed and avoids the stress of the lowest light periods (usually winter) in a temporary seed bank (Kenworthy 1992). *Halophila engelmanni* has a similar life history strategy, but unlike *H. decipiens*, this species may be found beneath the canopy of the larger seagrasses. Very little is known about the life history or physiology of *H. johnsonii*, however, it does grow in high light environments in the intertidal zone (Kenworthy 1992). Also, little is known about how these three *Halophila* species would respond to transplanting. However, we have successfully transplanted *H. decipiens* among existing beds of the same species in 15 m of water on St. Croix, USVI, but we are unaware of any other plantings with this genera.

The ability to avoid the stress of low light periods by surviving in a seed bank may be important for the persistence of other tropical genera (McMillan and Phillips 1979) and temperate species *Zostera marina*, *Z. noltii*, and *Ruppia maritima* as well (Harrison 1982, Orth and Moore 1983, Hootsmans et al. 1987). Like *H. decipiens*, some temperate species may have annual life history strategies completely dependent on seed reproduction for their survival (Keddy and Patriquin 1978, McMillan 1983). In these annual populations the plants do not experience a large portion of the seasonal light regime. Even though correspondence analysis has demonstrated higher light requirements for seagrasses in general, the averaging processes may include periods of low light that are not actually critical to the survival of the plants. In tropical, subtropical, and temperate regions seagrasses reduce their growth rates significantly during the colder temperatures in winter, and therefore, the light regime during this period may be much less important to the plants.

Light requirements should be determined based on the time period when the plants are responding to solar insolation and not when metabolism is slowed or when the population is residing in the sediments as seeds. This suggests that seagrass minimum light requirements determined from correspondence analysis (Dennison 1987, Duarte 1991) exceed the frequently cited value of 10 percent surface light. Recent evidence suggests that more realistic values for *H. wrightii* and *S. filiforme* are in excess of 15–20 percent (Kenworthy 1992, Kenworthy and Fonseca 1996), and for planting may even be higher (~25 percent). Moreover, light requirements of an individual species may vary as a function of the optical water quality indicating a very site specific component (Kenworthy 1992, Gallegos 1994, Gallegos and Kenworthy 1996).

Pre-restoration monitoring and site selection criteria should incorporate elements of the preceding discussion to improve the likelihood of planting success. Most sites will not have long-term data bases from which to characterize the light environment. If they do, then only data collected during the growing season should be used to calculate the appropriate statistic for a light level parameter. Regional growing seasons (see below: Planting Contingencies by Ecological Region) can be determined from the literature and planting should occur as early as possible in the season to take advantage of the optimum light regime.

Without light data taken over a growing season, the next best parameter is to utilize the maximum depth to which seagrasses grow in seagrass beds located in the area around the planting site. These reference beds can only be used to establish a maximum depth for transplanting rather than a specific light level. The reference site should have as similar a fetch (unobstructed over-water distance over which the wind blows), sediment composition, and tidal velocities as the planting site to assure that suspended sediments and light attenuation by turbidity at the two sites will be similar. Also, there should be as little difference as possible in freshwater discharge and water column chlorophyll concentration so that there are no gross differences in water color which could mean a different level of light attenuation. The reader should keep in mind that only light recordings performed at high frequency during growing seasons can yield prediction strength.

The depth to which local seagrasses grow would represent the minimum for an established bed and would likely overestimate the depth that transplants could survive. Beers Law describes the decline of photosynthetically-active radiation (PAR: the wavelengths of the light spectrum that activate chlorophyll) with depth in the water column by the following equation:

$$Iz = I_o * e^{-kz}$$

where:

I_o = Incident light just beneath the surface
Iz = Percent incident light at depth
e = base e
$-k$ = diffuse light attenuation coefficient
z = water depth

Therefore, we can estimate the percent light reaching a predetermined depth. If seagrasses in the area grow to a depth of 2.0 m and the average diffuse light attenuation coefficient is 0.75 (similar to the Indian River Lagoon; Kenworthy 1992), this indicates seagrasses require 22 percent surface light at their maximum depth of growth. At this location the light reaching shallower depths of 1.75 or 1.55 m is 26 and 32 percent, respectively. Transplanting at these two shallower depths would increase the relative amount of light available by 18 and 45 percent, respectively (+4 and 10 percent, respectively, of the amount of light available at the lower limit). Because the decay of light in water is exponential in the upper portion of the water column, a small change in depth yields a proportionally larger change in percent surface light reaching that depth. Reducing the depth of planting by 12.5 percent increases the amount of light by 18 percent whereas a reduction in depth by 25 percent adds 45 percent more light. This also works in the other direction whereby planting at slightly deeper depth would yield considerably less light. Paying close attention to depth and its relationship to available light could make the difference between success and failure of a planting.

In some cases planting sites may be isolated from established reference beds, especially in large-scale restoration efforts where entire lagoons or large portions of estuaries are involved. In these instances reference sites with established seagrass beds must be carefully matched with the restoration site for sediment composition, fetch, and tidal velocities. Short-term between site-paired comparisons of optical conditions can be made *in lieu* of a direct correspondence analysis or the availability of any long-term water quality data in order to grossly infer the lower depth limit. A paired comparison between sites would involve the acquisition of either light or water quality data over the same time period and under the same environmental conditions at each site (e.g., no localized storms affecting one site and not the other). Measuring equipment and methods must be intercalibrated to avoid detecting differences that cannot attributed to the sites.

One of the best parameters for inter-site comparison is the diffuse attenuation coefficient for photosynthetically active radiation (k_d PAR). If available, this parameter can be used to calculate percent surface light reaching a predetermined depth in the equation for Beers Law (p. 94) so that light available at depth can be used to help select planting depth. If at all possible, the technique for estimating k_dPAR should utilize quantum sensors instead of a Secchi disk (Kenworthy 1992). A Secchi disk severely underestimates light attenuation in estuarine water where there are dissolved organics (influencing color). Two types of quantum sensors are available, the cosine-corrected flat type sensor that measures downwelling PAR and the spherical quantum sensor that measures light from all directions (scalar PAR). The spherical quantum sensor is the preferred equipment although under most conditions the two different sensors will yield similar attenuation coefficients.

In a short-term paired-site comparison the sensors can be deployed in either a continuous or profiling configuration. In a continuous mode, at each site where one intends to plant seagrass, two sensors should be placed in the water column at two fixed depths at least 20-50 cm apart and PAR recorded continuously using a data logger (Zimmerman et al. 1994). A calculation of k_dPAR at each site over the same time period should be made. Attenuation coefficients should be calculated for time periods between 10 a.m. and 2 p.m. to avoid the errors associated with solar angle and path length (Miller and McPherson 1995). If the equipment is available, this method is appropriate for small sites, but for larger sites, where intra-site variation may be a problem, a profiling method is necessary. This method uses two sensors deployed at fixed depths from a mobile vessel so that more stations can be sampled to examine spatial variation. The same calculation algorithm is used as described above, however, for inter-site comparison one must be certain that profiles are obtained at the same times for accurate comparisons. Ideally, both the continuous recording and the profiling methods should use a base station with a sensor measuring incident radiation to correct for errors from local cloud conditions (Morris and Tomasko 1993).

Another method to compare sites is to use a properly calibrated optical water quality model (Morris and Tomasko 1993, Gallegos 1994, Gallegos and Kenworthy 1996). This method estimates light attenuation by summing the additive properties of scattering and absorption due to three commonly measured water quality constituents: turbidity (NTU), color (Pt-Co units), and chlorophyll (chl in mg l^{-1}). This kind of modeling approach is convenient because once the calibration is completed the model uses the water quality constituents that are inherent optical properties and not subject to errors of solar angle (i.e., time of day). Thus, measurements can be taken at any time of day and the information tells you what particular water quality constituent is having the greatest effect on light attenuation. Model calibration requires a minimum of 30 profiles of these 3 constituents and light attenuation for a water body. Calibrations have been successfully completed in the Chesapeake Bay region (Gallegos 1994) and the Indian River Lagoon (Gallegos and Kenworthy 1996).

SALINITY AND TEMPERATURE REQUIREMENTS FOR TRANSPLANTING

Salinity and temperature tolerances of seagrass species must be considered when selecting off-site planting locations. Seagrasses exhibit a wide range of tolerances to salinity but the effect of periodicity and duration of extremes in salinity on seagrass survival are poorly documented (see review by Zieman and Zieman 1989). Matching salinity regimes between the planting site and donor site is therefore strongly recommended. Temperature regimes should be similar as well. Temperature extremes may be problematic if a planting site has been constructed with restricted circulation allowing water temperatures to rise above levels found in natural beds.

Known temperature and salinity tolerances and optimum ranges for seagrasses growing in the continental United States are presented in Table 2.1. It is important to realize that the stress effects of these variables may be synergistic, but the effects of any such synergism on planting survival are poorly known in nature and likely vary widely depending on circulation, tidal zone, and geographic location within a species' distribution.

As seen in the data presented in Table 2.1, the tropical seagrasses are more stenothermic than temperate seagrass species. *Z. marina* has the widest temperature range (-6.0 to 40.5° C), and is found growing in a variety of temperate to sub-arctic habitats on both the east and west coasts of the United States. The range of reported optimal temperatures for this species is also greater than that reported for any other species (Bieble and McRoy 1971, Phillips 1984, Thayer et al. 1984, Evans et al. 1986, Bulthuis 1987). Data concerning the introduced species, *Z. japonica*, is less readily available, but its distribution suggests that its temperature tolerances are close to, if not greater than, that of *Z. marina* (Baldwin and Lavvon 1994). *R. maritima* and *H. wrightii* also have broad temperature tolerances as witnessed by their distribution patterns along the east Coast. Both are wide ranging species, often found in estuarine conditions where temperature conditions vary greatly with water depth, circulation patterns and exposure to periods of desiccation. While these species are often found growing in mixed beds with *Z. marina*, they have higher optimum temperatures (Evans et al. 1986, Orth and Nowak 1990, Zieman 1982a, Zieman and Zieman 1989).

Seagrasses growing in the more stenothermal subtropical and tropical marine conditions have narrower temperature tolerances than the estuarine species listed above (e.g., *T. testudinum*, *S. filiforme*, *Halophila* spp.). Information regarding members of the *Phyllospadix* genus is sparse. The distributional patterns of the three United States representatives of this genus suggest that they have similar temperature ranges, with a low end temperature of 5 and a maximum around 25° C (Drysdale and Barbour 1975). Tropical seagrass species are the least temperature tolerant, with maximum ranges of 15 degrees and optimum temperatures close to 30° C (McMilllan and Phillips 1979, McMillan 1984, Zieman 1982a, Zieman and Zieman 1989).

Salinity tolerances of seagrasses follow a similar pattern with *R. maritima* and *Z. marina* having the broadest tolerances. The coastal and marine species, which live in less euryhaline conditions, are more sensitive to changes in salinity. As with other physico-chemical requirements, comparisons of temperature and salinity with existing beds provide the best local guidance for placement of planting areas. In the absence of nearby vegetation, a minimum of weekly monitoring during both ebb and flooding tides should be conducted on a projected planting site to determine if values fall within those proscribed in Table 2.1. More intense sampling should be conducted (daily) just after several rainfall events, hopefully of differing rainfall amounts, to determine the response of a site to freshwater inflow. Although the tolerance ranges are known, the effects of dosing periodicity of either salinity or temperature extremes on planting survival (or natural beds for that matter) are unknown to us. For want of better guidance, we suggest that persistence of borderline values for more than five days should be cause for concern about site suitability.

Table 2.1. Temperature and salinity ranges of seagrass species occuring in the continental United States, optimal ranges are in parantheses.

Species	**Temperature Range**	**Salinity Range**
	(° C)	(ppt)
Halodule wrightii	9 - 37 (20-30) i,l,m	3.5 - 44 (20-35) i,l,m
Halophila decipiens	25 - 30 (20-36) d,j	30 - 35 (?) d,j
engelmannii	15 - 30 (24-28) c,j	15 - 35 (?) c,j
johnsonii	15 - 30 (?) d	20 - 30 (?) d
Phyllospadix torreyi	5 - 25 (11-21) e	2.9 - 29 (25-30) e
scouleri	?	?
Ruppia maritima	7 - 35 (20-30) f,j	0 - 32 (0-15) g,j
Syringodium filiforme	20 - 35 (30) i,j	20 - 35 (25-35) j
Thalassia testudinum	20 - 35 (25-32) b,h,i,l	3.5 - 60 (25-35) l,m
Zostera marina	0 - 40 (5-20) b,f,k	0 - 35 (?) k
japonica	5 - 30 (?) a	10 - 30 (26) a,e

a. Baldwin and Lavvon 1994
b. Bulthuis 1987
c. Dawes et al. 1987
d. Dawes et al. 1989
e. Drysdale and Barbour 1975
f. Evans et al. 1986
g. Mayer and Iow 1970
h. McMillan and Phillips 1979
i. McMillan 1984
j. Phillips 1960
k. Thayer et al. 1984
l. Zieman 1982a
m. Zieman and Zieman 1989

Micropropagation and Laboratory Culture of Seagrass for Planting

Laboratory culture of plant fragments (micropropagation) for large-scale field plantings is an active research area (Durako and Moffler 1981, 1984, Lewis 1987, 1990, Ailstock et al. 1991, Koch and Durako 1991, Durako et al. 1993, Bird et al. 1994, DeLeon et al. in press.). In the future, large-scale plantings may rely on laboratory-reared plants as there are many potential advantages to using this approach. However, there are many practical problems that must be overcome first as well as questions regarding the general efficacy of the approach. Below we contrast some of the argued benefits with what we feel are limitations to the technology. In summary, the advantages of relying on laboratory-cultured plants are as follows:

1. **Donor bed damage reduction:** Reduce damage to donor bed by making small field collections and geometrically expanding the numbers of plants in the lab to provide planting stock;

2. **Genetic stock improvement:** Improve genetic mix of stocks (avoidance of founder effects, see section "Pre-Project Planning Considerations," above) to go into the field;

3. **Disease/stress resistance:** Select for disease-resistant or stress-tolerant strains of plants;

4. **Cost reduction:** Reduce project costs through mass production of planting units;

5. **Stock availability:** Maintain donor stocks to meet the sporadic demands of disjunct planting projects;

6. **Bioassay tool:** Develop genetically consistent stocks that can be deployed and thus used as a bioassay standard of water quality, and potentially, planting-site suitability for subsequent restoration projects.

Taken together, the above arguments would seem to logically place culture techniques at the forefront of the seagrass mitigation/restoration effort. However, there are serious questions regarding many of the proposed benefits of micropropagation. The disadvantagess to the above claims are as follows (response by corresponding number):

1. **Donor bed damage reduction:** Sustained injury to donor sites from one-time impacts have been demonstrated to be an issue for only one of the ~13 North American seagrass species, *Thalassia testudinum* (e.g., Zieman 1976, Durako et al. 1992; see following section). Donor bed injury recovery can be rapid (Fonseca et al. 1994) especially if a minimum attempt is made to disperse the collection effort. However, some States (e.g., Florida) have placed restrictions on harvest of wild seagrass stock for planting projects; this appears to fuel the donor-bed impact argument.

2. **Genetic stock improvement:** "Improvement" of genetic stocks is currently speculative. A broad knowledge of the existing genetic structure of seagrass populations and the factors influencing genetic stock (especially site-specific; *sensu* Ruckelshaus 1994) would be requisite to form a comparative basis for "improvement," which requires stricter definition. Moreover, while strain selection is applicable to terrestrial crops where gene flow and population interactions are not an issue, the potential for improvement of genetic structure is untested in the management of wild, clonal plant communities.

3. **Disease/stress resistance:** The disease/stress resistance of micropropagated plants has not yet been demonstrated, nor has the consequences of trading off natural genetic diversity for selection of, and introduction to nature, of a few strains designed to meet specific environmental problems. Moreover, if a bed were developed in a stressed area through selection of resistant stock, would such a bed become functionally equivalent to natural beds? Introduction of manipulated genotypes remains a significant issue in population ecology, and is many years away from resolution.

4. **Cost reduction:** While large-scale production costs would be less on a planting unit basis, only DeLeon et al. (in press) has, to our knowledge, considered a quantitative review of the entire planting cost using micropropagation techniques that includes the overhead and amortization of culturing facilities, materials, and labor. Break-even points (i.e., number of planting units needed) have not been determined. Moreover, culturing does not eliminate out-planting costs which may be similar among cultured or wild-harvested plantings.

5. **Stock availability:** Like (4) above, we are not aware of published data on cost break-even points, which are strongly influenced by required frequency and magnitudes of requests for planting stock needed to support such a facility.

6. **Bioassay tool:** This aspect of seagrass micropropagation offers one of the most powerful applications of this technology. Having a "standard plant," coupled with plant health assessment techniques such as variable fluorescence techniques (*sensu* Adams and Bate 1994) could yield rapid and consistent water quality and planting site suitability indices, as evidenced by Durako et al. (in press).

At this time, there are only two seagrass species that have been successfully cultured: *Ruppia maritima* and *Halophila decipiens* (M. Durako, University of North Carolina, Wilmington, pers. com.), two species that have very high sexual and asexual reproduction and thus, high colonization (and likely, donor bed recovery) rates. The technology is not currently available to the only species for which there are demonstrable, long-term donor bed impacts, *T. testudinum*.

Wild Stock Selection, Availability, and Performance

The choice of species is often dictated by project goals, such as the desire to replace in kind the seagrass species that was lost. In subtropical areas where several species co-occur, it is sometimes appropriate to temporarily substitute faster-covering species in order to stabilize a planting site (*sensu* "compressed succession," attributed to M. Moffler). The specific choices of available species are covered under each ecoregional below. In general however, the early recommendations of Addy (1947) still hold where matching conditions at donor and planting sites were recommended.

For seagrass planting projects to eventually be successful, it is critical that they consider physiological requirements and life histories when selecting a planting site (see sections dealing with growth requirements, e.g., light, nutrients, temperature, salinity, above). For example, species with a slow coverage rate (i.e., *T. testudinum*) are very difficult to restore in the time frame often allotted projects. It can take decades for a bed to re-create the dense root system, organic-rich substrate, and nutrient cycling capabilities of turtlegrass beds (*sensu* Zieman 1976). *Halophila* spp. have very different strategies. This species often colonizes disturbed areas rapidly and requires relatively little light to grow (Josselyn et al. 1986). Interestingly, although the spatial distribution of *Halophila* spp. indicates an ability to colonize low light environments relative to other seagrasses, it only has growth during times of the year when light levels are at their highest. This signals a potentially complex interaction between light and temperature in effecting *Halophila* distribution. An individual leaf pair of *H.*

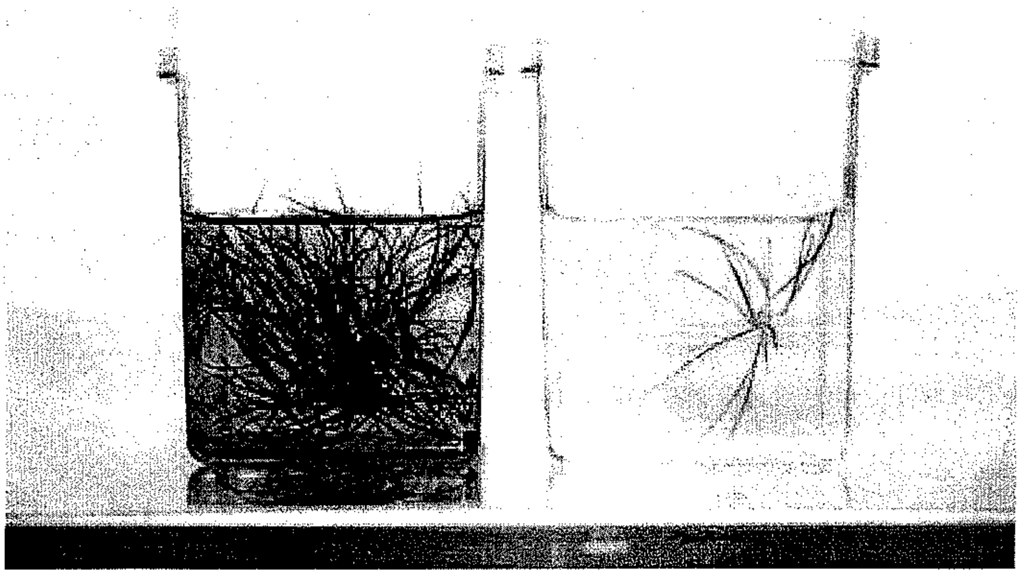


Figure 2.6. Ruppia maritima *in culture. Courtesy M. Durako.*

decipiens may live for only six weeks, and produce many seeds, a strategy typical of a species living in marginal environments. Its shallow root system, however, makes it vulnerable to disturbance. *Ruppia maritima* (widgeon grass) has a wide tolerance of salinities and grows in fresh water, brackish water, or among other seagrasses in full strength seawater. Like *Halophila*, this species has a very high seed production and covers the bottom quickly. Finally, *H. wrightii, S. filiforme, Zostera marina* and, likely, *Z. japonica* have comparatively moderate coverage rates. Little is known regarding coverage rates of other seagrass species in the United States, such as *Phyllospadix* spp., under planting conditions. For the subtropical species, the comparative coverage rates are important both for predicting recovery rates (e.g., Fonseca et al. 1987a,c, Lewis 1987) and choosing a fast-covering species with which to initiate plantings (i.e., "compressed succession": begins with shoalgrass, allow coalescence, and then add the slower-growing target species if it were initially different from shoalgrass). For temperate areas where the pioneering species is the same as the climax species (i.e., begin with eelgrass and end with eelgrass), coverage rates are useful for recovery rate estimation. Thus, the different growth strategies of seagrasses implicitly define the anticipated performance, monitoring, planting schedule and, ultimately, function of the restored system.

Although some data exist to select planting stock by ecotypes (Backman 1991), costs of collection are also important. Collection of plants from areas of high density, such as sandy shoals, is often more cost-effective when plants are small. Larger individual plants, such as *Zostera*, that are sometimes planted singly or with only a few in a planting unit (Davis and Short 1997), are sometimes removed more easily from softer sediments. Some seagrasses, such as *Halodule* and *Syringodium*, sometimes have ramets extending, rootless, into the water column, and these make excellent planting stock without having to extract material from the donor bed rhizosphere (Fonseca et al. 1987a,c, Lewis 1987). *Thalassia*, with its deeper rhizome system, is most easily collected at bed margins where the rhizome development is nearest the sediment surface.

At this time virtually all planting material must be obtained from existing, wild vegetative stocks. However, the collection of vegetative material from existing beds is rigorously managed in many states. Collection without appropriate permits may result in fines. Because of the evolving nature of this field of restoration science, it is imperative that anyone planning seagrass planting carefully coordinate their actions with state and local government. It is not uncommon to find permits required from not only the federal government, but also numerous state agencies. Individual counties and municipalities too may require consultation. Because of the increasing volume of permit requests reaching permitting agencies, obtaining a permit could take months. Such a delay must be anticipated in order to collect plants (if approved) and plant at the desired time of the year.

Once planting is permitted, wild stocks are usually used. *Zostera marina*, *H. wrightii* and *S. filiforme* can be harvested from wild stands with no long-term (> 1 year) impact to the donor site (Williams 1990, Fonseca et al. 1994) (Figure 2.7). However, **unless specifically created as a donor site, repeated harvest of donor sites within a calendar year should not be permitted.** It should also be noted that wild stock harvesting will cause some interim loss of habitat functions and productivity. Therefore, we recommend that harvesting impacts be composed of numerous, individual small collections rather than opening large holes in the seagrass cover.

Although not currently documented, it is highly probable that *Ruppia*, *Halophila* spp., and other *Zostera* spp. would recolonize small harvest patches quickly (< 0.25 m^2 patches returning to normal density within 1 year) because of their high population growth rate and seed production. Harvest from high current areas (> ~30 cm/sec) however, could initiate the development of an erosion scarp which would spread and damage the donor bed irreparably (*sensu* Patriquin 1975). *Thalassia* can be transplanted with good survival but slow population growth (Fonseca et al. 1987a,c, 1989a, Lewis 1987, Tomasko et al. 1991), but harvest damage to those donor beds may last for years (Zieman 1976, Fonseca et al. 1987c), and harvest of vegetative

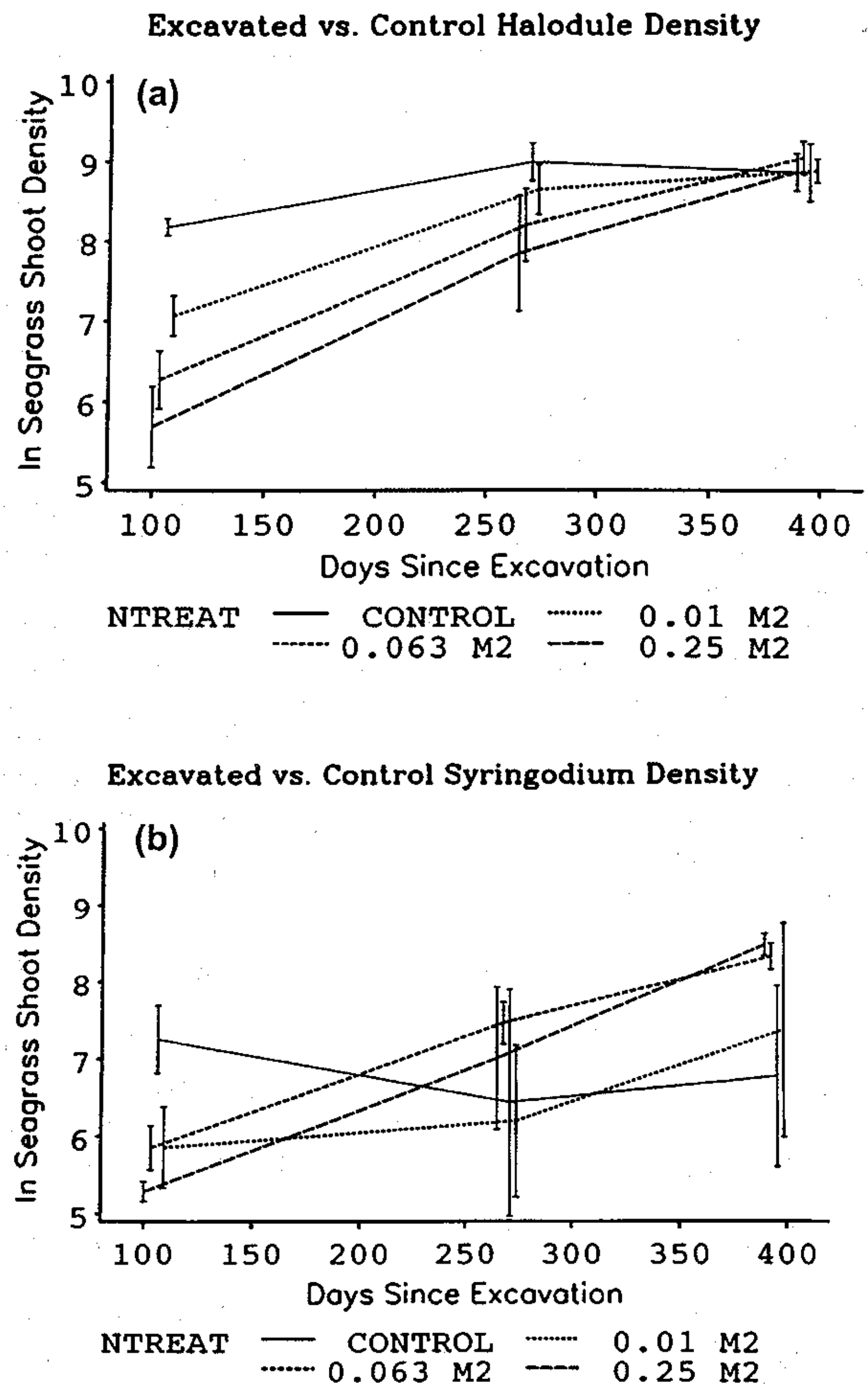


Figure 2.7. Recovery of donor beds harvested for planting Halodule wrightii *(a) and* Syringodium filiforme *(b). Shoot density* m^{-2} *by original excavation size and time since excavation (days). C=un-excavated control. NTREAT=treatment, or size of excavation (M2 =* m^2*); control=no excavation. Taken from Fonseca et al. 1994.*

Thalassia stock should be from bed margins (to minimize rhizosphere disturbance) if a salvage operation (from planned or permitted disturbance) is not available. Another means of acquiring *Thalassia* planting stock is to harvest its seeds which wash up on shore (Lewis and Phillips 1980), which has no negative influence on existing beds. The impact of harvest of seeds or seedlings of any species from within existing beds or colonizing areas currently has an unknown effect on the maintenance of seagrass in those areas. However, given that seed harvest probably gathers only a small percentage of the seed production, we expect the impact to be small. The impact of donor bed harvesting on *Phyllospadix* spp. is largely unknown.

In general, planting stock should be selected from a site with conditions as similar as possible to the planting site, as near the planting site as possible, and at similar or equal water depths, salinity, and sediment type. The concept of choosing plants of the same size as those lost, perhaps accounting for potential races of seagrass, was suggested 45 years ago (Addy 1947). Little data have emerged to suggest changing this practice although concerns have been voiced regarding the maintenance of genetic diversity in "Pre-Project Planning Considerations," above). Until more is known about the genetic structure of seagrass ecosystems, harvesting of plants from as wide a geographic range as feasible is recommended. Similarly, matching sediment types of the donor site with the planting site (percent silt and clay, and percent organic matter content of the sediment) is thought to facilitate transplant success.

Planting material may become available as salvage prior to the imposition of a project. Utilization of salvaged material requires good up-front organization so that a planting site is available before the plants are destroyed (e.g., turtlegrass, Lewis 1987). Long-term storage of salvaged plant material to use for future plantings has not be scientifically evaluated, but has been accomplished for at least a week (pers. obs.). Longer term storage may be possible but may significantly increase handling costs.

In summary then, by use of environmental monitoring data, the most prudent way to select planting stock is to match conditions at a donor site with the planting site. The temperature, salinity, surface sediment (top 3 cm) particle size and organic content, tidal current speeds and wave exposure of the planting should be as similar as possible to that of a donor bed of the same seagrass species.

Long-Term Management

Creation of transplanted beds for the sole purpose of providing donor material to subsequent operations would be prudent. This would alleviate the problems of storage costs, relieve some of the time constraints and permitting problems that accompany most projects, and prevent damage to native seagrass beds. However, once these beds are planted, they fall under the permit jurisdiction of resource agencies as would any seagrass bed. However, experimental beds, such as those we have created, total many acres and were not created to offset any particular loss. These should be made available as mitigation/restoration donor site. In any event, planting of beds for future donor material needs to be organized early and in coordination with permitting agencies. Moreover, to avoid net losses in baseline acreage, we recommend that planted beds be given special status and be protected from any subsequent consideration for permitted impacts. Given that early stage plantings have been found to have lowered genetic diversity (see above section), institution of these donor sites should be linked to an evaluation of their genetic structure to avoid embedding additional lowered genetic diversity into planted populations.

Planting Contingencies by Ecological Region

Northeast Region — Maine through New Jersey: known species present = *Zostera marina* and *Ruppia maritima*.

Compared to many other parts of the country, the growing season is shorter here. Water temperatures are also comparatively cold and will be a cost factor in planting operations especially when using divers. Planting may begin as early as the waters are ice-free, but to obtain robust planting stock one usually will have to wait until April or May, and sometimes as late as June. The drawback to early planting is that shoots of *Zostera* will not yet have flowered. Planting flowering shoots of this species will potentially add seed stock, but because the shoots die after flowering very little vegetative spreading will result from planting flowering shoots. Thus, as much as 30-40 percent more plants might need to be installed when planting early in the year to make up for this loss, unless flowers are reliably culled from plantings.

Ice shearing is a significant problem in many locales, depending, of course, upon the severity of the winter (*sensu* Robertson and Mann 1984 and F. Short, Jackson Estuarine Lab., Durham, NH, pers. com.). Coarse, cobble sediment can become especially destructive to planted seagrass when it moves during storm events. Grazing by waterfowl (Thayer et al. 1984) and green crab, horseshoe crabs, and various fishes are significant sources of bioturbation (Nereid worms have also been reported to injure plantings; F. Short, Jackson Estuarine Lab., Durham, NH, pers. com.). Also, wide tidal ranges, up to several meters, will force planting to either be done in narrow low-tide windows of opportunity or will require divers. In steep catchments, periodic rainfall events can become especially concentrated, rapidly driving salinities down and elevating turbidity. These aperiodic events, as with storms, may be worthy of attention when selecting sites.

Mid-Atlantic Region — Delaware through North Carolina: known species present = *Halodule wrightii, Ruppia maritima* and *Zostera marina.*

Planting strategies differ markedly for the species in this region. *Halodule* is a subtropical species, and like the temperate *Zostera*, is at the edge of its distribution in this region (i.e., the northern and southern limits of these two species overlap in North Carolina). For *Zostera* planting may be done from April through November, although the farther south one goes in this region, planting later in the year (September-November) gets plants into the bottom for the longest time possible before the next period of low growth (heat stress in July-August). Spring plantings of *Zostera* would still have the flowering problem described for the Northeast Region. Fall planting is also the best strategy for *Zostera* in the Chesapeake Bay (Moore and Orth 1982). For *Halodule* spring plantings are best. *Ruppia* is probably best planted in early spring as well (Bird et al. 1994). Sources of bioturbation are much the same as elsewhere in the country — rays, crabs, and horseshoe crabs.

Gulf of Mexico and the Florida East Coast — Mexico to Cape Sable and north of Jupiter Inlet to Cape Canaveral: known species present = *Halodule wrightii, Halophila decipiens, Halophila engelmanni, Halophila johnsonni, Ruppia maritima, Syringodium filiforme,* and *Thalassia testudinum.*

Of the seagrass species in this region, the three most commonly used species (*Halodule wrightii, Syringodium filiforme,* and *Thalassia testudinum*) have very different intrinsic coverage rates (Fonseca et al. 1987c). Planting should be done in the spring although plantings will survive (but spreading at lower annual rates) if initiated at other times of the year. Fonseca et al. (1994), however, found that a fall planting was more successful in high bioturbation areas (no cages used) because it apparently avoided the peak of the yearly bioturbation activity. Bioturbation has been reported by urchins, sand dollars, rays and crabs.

Coverage rates of the common species are: *Halodule wrightii* > *Syringodium filiforme* > *Thalassia testudinum.* Any of these species can be planted alone, but *H. wrightii* is considered a pioneering species and should be used to quickly establish cover. This may also be planted in alternating rows with the other species. Although *T. testudinum* may be planted alone, its very slow population growth and coverage rates under transplant conditions make it susceptible to interim erosion. The prolonged lack of cover would also likely extend the period of interim loss of fishery resources. If turtle grass is the target species, it should be added once faster-growing species (e.g., *H. wrightii*) have stabilized the bottom.

Ruppia maritima performs much as shoalgrass when transplanted (Stout and Heck 1991, Durako et al. 1993). Its high density of rhizome apicals allows the same planting techniques to be employed as used with shoalgrass (Stout and Heck 1991). In some areas widgeon grass has reportedly been pinned to the bottom in mats, after being intertwined in a biodegradable mat or allowed to grow over mats placed in natural beds after which the mats and intertwined shoots are removed for planting elsewhere. Durako et al. (1993a) used cotton mesh bags with *Ruppia* fragments and a small stone inside for planting; bags were thrown overboard and allowed to root on the bottom by growing through the cotton bag.

The *Halophila* species (paddle grass and star grass) are extremely fragile, but can significantly reduce currents and wave scour (Fonseca 1989b). Because of their growth strategy, with only 3 or 4 leaf pairs on a rhizome in close proximity to the rhizome apical, these species would likely be suitable for transplanting using the peat pot method described in Chapter 3, although we have not tested this method at depths > 1-2 m. We have successfully transplanted *H. decipiens* bare root sprigs in 15 m of water using 60 lb. test wire fishing leader bent into a U-shape as a staple to hold the plants to the bottom until they rooted. While few cases of *Halophila* spp. transplanting have been documented, their pioneering growth strategy and small size make them likely candidates for effective use in planting projects.

South Florida and the Caribbean — South of Jupiter Inlet to Cape Sable and Puerto Rico and the U.S. Virgin Islands: known species present = *Halodule wrightii, Halophila decipiens, Halophila engelmanni, Halophila johnsonni, Ruppia maritima, Syringodium filiforme,* and *Thalassia testudinum.*

The same guidance should be used here as for the above section on the Gulf of Mexico and Florida east coast. The only difference is that planting can be performed at any time of the year with little difference in expected response.

Conterminous West Coast — California to Washington: known species present = *Phyllospadix scouleri, Phyllospadix serralatus, Phyllospadix torreyi, Ruppia maritima, Zostera japonica, Zostera marina,* and potentially, *Zostera asiatica.*

Planting should be performed during the springtime (April, May, early June). Some specimens can be very large and careful handling is required. Planting of *Phyllospadix* spp. has been practiced very little, and we can offer little guidance except to review Phillips et al.'s (1992) report (also see Chapter 1, "Comparative Analysis of Seagrass Planting Efforts"). Cold water may, as in the Northeast, contribute to high planting costs. In some areas such as San Francisco Bay, there are extensive annual populations of *Zostera* which are problematic for transplanting. Transplanting of annual *Zostera* means there will be little vegetative spread (the shoots die after flowering). Seed deposition is the only mechanism that would sustain the transplant and this is very risky unless a site is extremely quiescent; even then seed predation remains a potential problem. Experimentation with seeding techniques would be appropriate in such settings. Vertical zonation of *Phyllospadix* spp. in the rocky intertidal must be recognized and matched with local distributions. Bioturbation sources include crabs, rays, some of these fishes, and sand dollars. Bioturbation in the rocky intertidal has not, to our knowledge, been documented for planted seagrass.

Alaska — known species present = *Zostera marina* and assumed some *Phyllospadix* spp.

Little is known about planting requirements here except that spring plantings are logically better and that cold water and ice shearing may be particularly problematic.

Hawaii and Pacific Territories — known species present = *Halophila hawaiiana*, *Halophila minor*, and *Halophila ovalis*.

As with Alaska, little is known here but we are aware of no reports of seagrass planting in this region. Planting guidance must be extrapolated from elsewhere.

CHAPTER 3

Planting

METHODS

Common to all planting methods are some fundamental constraints. As mentioned earlier, the basic premise is to adjust the ratio of births and deaths of shoots so as to effect net population growth. To achieve this, it is important to ensure the presence of growing rhizome apical meristems in individual planting units (PUs) as these provide a source of new shoots and horizontal growth; one means of colonizing of new areas (as opposed to seeds). Visually inspecting arbitrarily selected planting units for an absolute minimum of one apical shoot per PU is requisite for asexual reproduction; more than one apical is highly recommended. The number of short shoots on a long shoot should be maximized whenever possible so as to derive benefits from the clonal nature of the plants. Fonseca et al. (1987a) used an average of 2.6 short shoots per long shoot (horizontal rhizome with several short shoots) with turtle grass but Tomasko et al. (1991) found higher rates of new short shoot production when the short shoot/long shoot ratio increased (up to a ratio of 4). Davis and Short (1997) use only two *Zostera marina* shoots per planting unit in a modified staple method. It is also recommended that whenever possible, plants should be collected and planted on the same day. Any number of incidents may further shock the plants and inhibit their photosynthetic capacity for prolonged periods after planting. Seagrasses are inherently fragile, having evolved in a fluid medium that provides support for their structure. When out of the water, they are very susceptible to physical damage. To ensure transplanting success, it is critical that seagrasses are kept wet and handled gently. Moreover, seagrasses have very little resistance to desiccation. On a breezy, sunny day, plants left out of the water in the open can experience permanent leaf damange within minutes and protracted loss of photosynthetic efficiency within 1-2 hours (author's unpubl. data). Plants must be kept in ambient temperature and salinity water at all times! They may be covered with seawater-soaked cloth for short periods if transportation is necessary. Stacking of the plants on one another should be minimized. Although they appear and even feel robust, they are easily bruised and broken.

Numerous methods have been shown to successfully establish seagrass; however, familiarity with handling and planting methods as well as the ability to work in or under the water are requisite. The familiarity of an individual with these plant communities is inversely proportional to the difficulty encountered in executing a planting. Candidates for planting projects should be able to identify the species involved and, if needed, have the ability to snorkel or SCUBA dive. Planting inexperience is one of the most common causes of problems (and added cost) in a project failure.

Planting strategies can be divided into SCUBA and non-SCUBA assisted operations. In either case, once the required area for planting is selected, the planting area should be clearly marked off so its boundaries are visible from the surface (e.g., poles, buoys). Experienced boat operators and SCUBA divers may be required. The decision to utilize SCUBA does not necessarily mean that depths are over one's head. Where the water is deep enough to prevent a snorkeling diver from reaching the bottom without breath-holding, a person walking and either handing planting units (PUs) to the diver or pre-placing them for installation can greatly reduce physical exertion. Various combinations of planting and providing PUs to the planter will work effectively. Experimentation will typically improve efficiency by best utilizing the skills of the personnel involved. However, when SCUBA is required for planting, many logistical and safety problems are introduced (*sensu* Merkel 1992). At the least, higher wages associated with diving significantly increases planting costs sometimes by an order of magnitude.

Merkel (1992) gave careful consideration to the role of personnel and the use of volunteer labor. For intertidal bare-root (e.g., staple technique) planting he suggested a minimum of 7 persons (1 project coordinator and 6 staff); for subtidal bare-root planting he suggested a minimum of 9 persons (1 coordinator, 4 staff on shore, and 4 divers). Slightly fewer people were recommended for plug planting. As for volunteers, he points out that after the relatively brief learning curve for executing seagrass plantings, they often lose interest as the work is tedious and repetitive. Paid staff are often more cost-effective.

Plug Methods

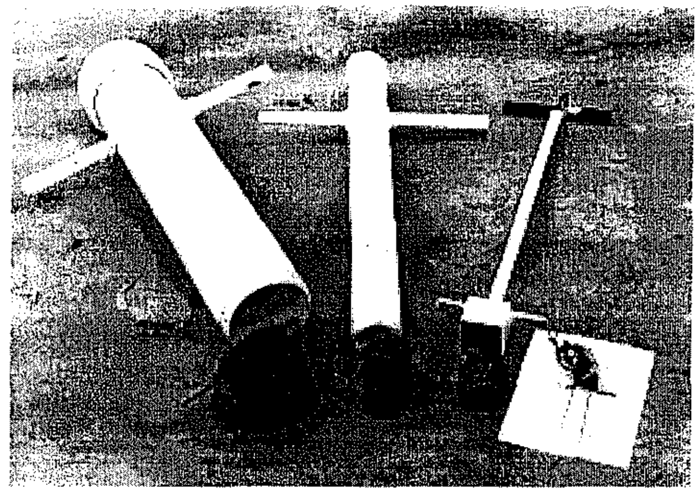


Figure 3.1. Comparison from left to right of 6" diameter plug or core, 3" diameter core, 9" square peat pot plug and staple unit. Note differences in size of collection apparatus and mass of material to be handled among techniques; this is a large part of the basis for differences in logistic burdens among techniques.

Plugs of seagrass with the associated sediment can be harvested using a core tube. Core tubes (Figure 3.1) are used to extract plugs from the donor bed and transport them in the tube to the planting site. The tube (usually 4-6 inch diameter PVC) is inserted into the sediment and capped, creating a vacuum so that when the tube is pulled from the sediment the small plug of seagrass with associated sediment is carried inside. Another cap is placed over the bottom to avoid losing the plug in transport. Another hole must be made at the planting site to accommodate the plug. This can be accomplished either by removing another core or by softening the bottom using a wedge. Fonseca (1994) described using tree planting bars of the kind employed in forestry practices for this purpose. To plant the plug, the bottom cap is removed from the core tube, and then the core tube is inserted into the new hole. The top cap is then removed, letting the plug slide out of the tube into the substrate. This method requires handling the caps and core tubes between planting and the next harvesting. Because of this handling time, the core tube planting was the most expensive (3.53 work-minutes per planting unit) tested by Fonseca et al. (1994). Costs for all methods included only work time to harvest, fabricate planting units, and plant those units. No transportation time, lodging, capital expenditure for equipment, boats, overhead or profit was included. Basic cost may then be computed by multiplying the number of planting units (PU) needed by time per PU and then by hourly wage. However, this method has been used extensively and for most species with good results.

Use of plugs requires that the sediment-root mass be sufficiently cohesive so that it remains in the tube when the plug is pulled from the bottom. The ability to retain a plug in the core tube varies inversely with particle size and core diameter, but positively with depth of the plug (filling more of the core tube with sediment; unfortunately with concomitant increase in mass) or root mat thickness.

Staple Method

The staple method has been used widely since its development in the late 1970's (Derrenbacker and Lewis 1982, Fonseca et al. 1982). Plants are dug up using shovels, the sediment is shaken from the roots and rhizomes in the process, and the plants with the roots and rhizomes are placed in flowing seawater tanks (or floating pens) for holding until made into PUs. Surprisingly, this handling results in no measurable loss in photosynthetic efficiency by at least some seagrass (*Zostera marina*), even after repeated insertion into the sediment. Groups of plants are attached to staples by inserting the root-rhizome portion of the group under the bridge of the staple and securing the plants with a paper-coated metal twist-tie (Figure 3.2). The twist tie is secured around the plants at the basal meristem so that the leaves will extend from under the staple up into the water column when planted. A small strip of paper has been used to protect the rhizomes from the twist-tie by wrapping the group of plants with the paper and securing the twist-tie over the paper strip. The staples are then inserted into the sediment so that the roots and rhizomes are buried nearly parallel to the sediment surface, as they occur in nature. (Fonseca et al. 1982, 1984). Loosening the sediment with a utensil such as a dive knife facilitates placing the roots into the sediment. One person may lay out the planting units beforehand at the appropriate spacing, while a second person follows and installs them.

This planting method takes less time than the core tubes, but the intermediary step of attaching plants to staples is time-consuming (see below). In calm areas, groups of plants may be stapled to the bottom without attaching them to the staples beforehand. When attached to the staples, these plantings have successfully withstood tidal velocities of up to ~50 cm/sec (Fonseca et al. 1985). The staple method required 1.91-2.07 work minutes per PU in a test by Fonseca et al. (1994). The relatively low cost and widely tested applicability make this one of the most useful methods available at this time.

Some criticism has been leveled at the use of metal staples, because the bridge of the staples will oxidize before the legs which are deeper in the typically anaerobic sediment, leaving two potentially sharp pieces of metal in the bottom (Merkel 1988b). However, we have deployed thousands of these PUs and, despite repeated visits to the sites, have not yet experienced an injury. The use of metal staples described here is emphasized for its sediment-free approach, reducing the burden of carrying associated sediment. Any degradable anchor may be substituted if shown to provide similar stabilization of the plantings until they root. Two variations of this method are described below.

Merkel (1988a) utilized a popsicle-stick technique where shoots were tethered on a short cotton string to a popsicle stick and inserted into the sediment (Figure 3.3). The stick would then rotate to a horizontal position deep in the sediment and resist dislodgement. The bundle of shoots, attached on their lead to the stick would then be resistant to erosion. Although we have not tested this technique, Merkel has used it extensively (pers. com.). It would seem that a fine sediment would facilitate deployment and lead holes as with peat pots or regular staples would be sufficient to install the PU. Also as with any PU, the depth of insertion of the anchor requires attention so as not to allow the plants to float out of the bottom or be held too deep in the sediment, covering the leaves. Information on fabrication and deployment costs are not available for comparison with other methods.

Another variation on the staple method is the use of a biodegradable anchor. Davis and Short (1997) have used bamboo "shish kabob" sticks in place of metal anchors (Figure 3.4). The sticks are soaked to enhance their flexibility and bent in half. The fibrous nature of the bamboo usually prevents complete breakage, thereby forming an inverted "V" or U-shaped staple, much like the prefabricated metal staples. Costs are expected to follow that developed for metal staples although when purchased in bulk, skewers can cost as little as $0.006 apiece as compared with $0.01 apiece for metal staples. These sticks lack the weight of a metal staple but are easily moved about if placed on the bottom (i.e., just prior to planting by a diver, pers. obs.) and alleviate concerns regarding potential injury from corroded metal staples. Also, Davis and Short (1997) did not attach plants to the staples beforehand: plants were pinned to the bottom by the diver who carried both staples and plants. They claim a substantial cost saving using this approach.

Peat Pot Method

Fonseca et al. (1994) recently modified the method of Robilliard and Porter (1976a). Peat pot plantings have been found to have the lowest cost per planting unit (1.21-1.49 work minutes per PU), despite the fact that substantial amounts of sediment are moved with the plants (Fonseca 1994). As with the coring method, shearing of blades may impair growth of larger plants. Shoalgrass and potentially widgeon grass and paddle grass (or any *Halophila* species) may be most suitable for this method, given their relatively high density and generally shorter blade lengths than manatee grass. The peat pots used by Fonseca et al. (1994) were 3 inches on a side and are readily available. The 3 x 3-inch sod plugger (Figure 3.5 a-g) used in that study can usually be purchased locally. The sod plugger is used to cut plugs from existing beds. The plug should then be extruded immediately into a peat pot and placed in a hold-

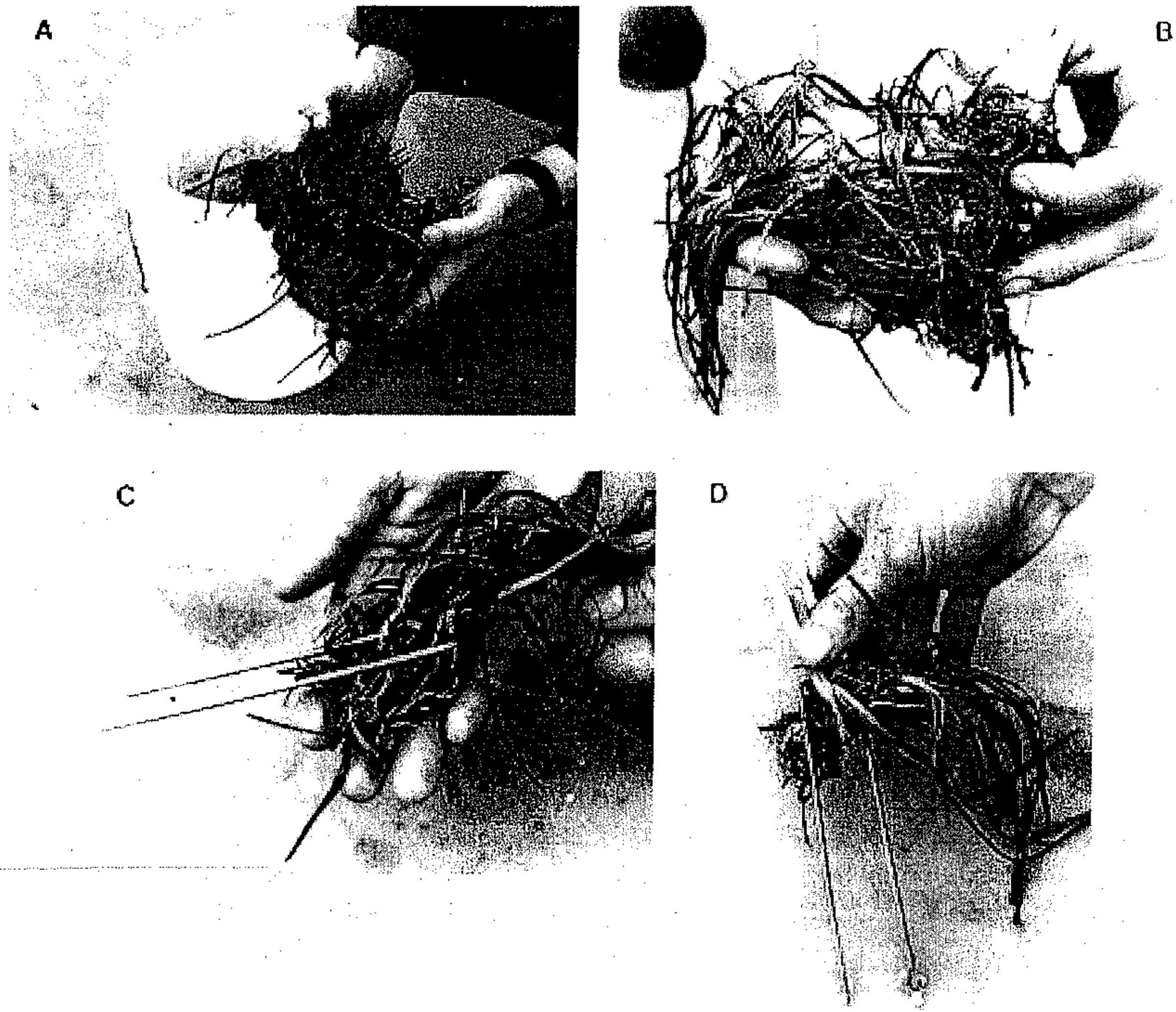


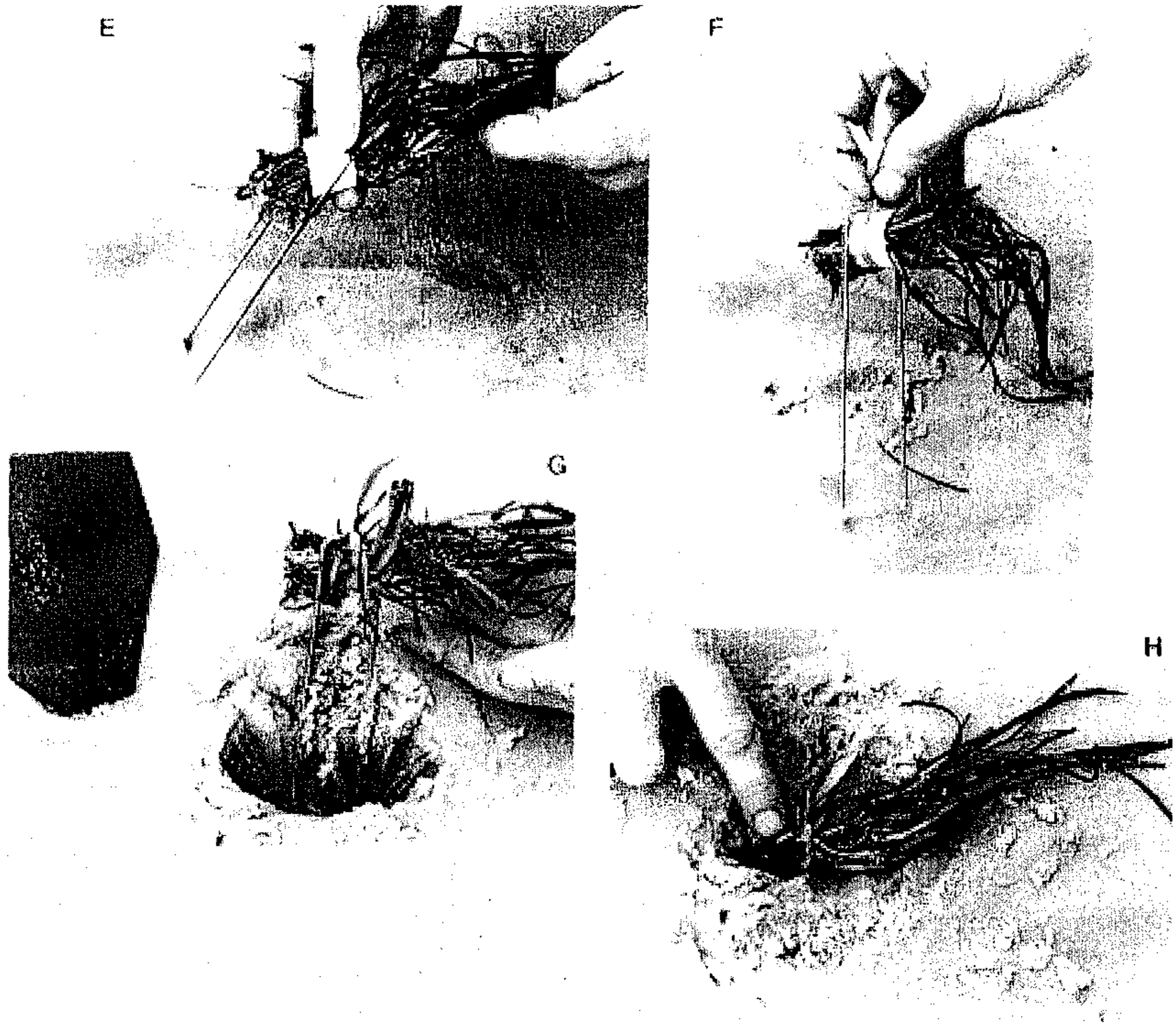
Figure 3.2. Staple Method. (All photographs staged on land and in air for demonstration purposes only; eelgrass Zostera marina *used in this demonstration.)*

(a) Typical shovel-sized sod that is excavated for collection of planting stock.

(b) Seagrass shoots remaining after rinsing sediment from sod.

(c) Depending on size of shoots from one (very large, ~1m long plants) to fifteen (small, ~10-20 cm long plants) are separated from the sod. A staple is then placed over the plants where the leafy shoots are attached to the rhizomes, separating what will become the above and below ground portions of the planting unit.

Note: In quiescent settings it may not be necessary to attach shoots to the staples; plants can be separated from sods and stapled into the bottom all in one act. Practitioners of this method work either in pairs where one person separates out planting stock and hands it to and second person who manipulates the staples and inserts the planting unit into the bottom or else planters work alone and develop ways to secure and carry staples. For example, Velcro™ straps have been fabricated to be placed along the forearm of a wetsuit under which many staples can be secured yet easily slid out from under the strap as planting progresses. If a planting does not require pre-planting attachment to staples, then skip the next three steps.



(d) (Opposite page) A paper-coated (not plastic) wire twist-tie™ is then placed under the shoot bundle and over the bridge of the staple and twisted snugly. Care must be taken not to crush the rhizomes and/or shoots when tightening the twist tie. The planting unit is then ready for transport to the planting site.

(e) An optional tactic that is sometimes useful with larger plants whose rhizomes are more brittle is the addition of a paper collar around the shoots prior to the addition of the twist-tie.

(f) The twist-tie may then be attached. The planting unit is then ready for transport to the planting site.

(g) A softened (fluidized) spot is usually created in the sediment unless the sediment is already soft. Because the staple unit is small, this can be accomplished with a dive knife although tree bars work very well. The points of the staple are inserted into the bottom, sometimes at an angle, instead of perpendicular to the sediment surface.

h) The staple is inserted into the sediment to the point where the bridge of the staple is just covered, burying the rhizomes taking care to make sure the plant bundle remains under the bridge of the staple. Also leaves must not be caught under the staple and should be free to extend up into the water column.

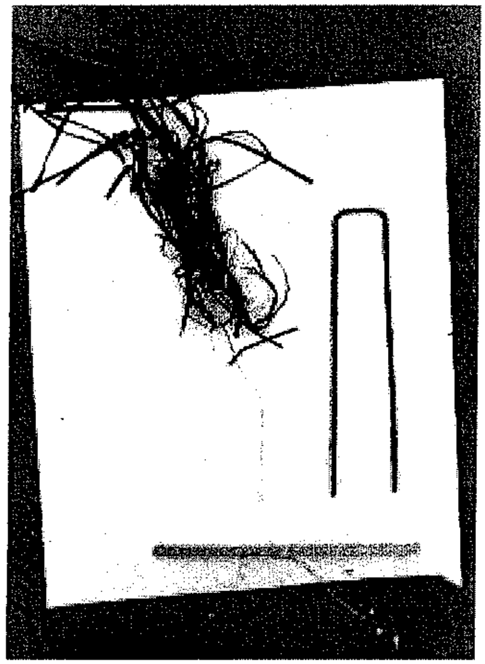


Figure 3.3. A technique developed by K. Merkel (Merkel & Associates, San Diego, CA) as an alternative to staples.. Bundles of seagrass are tethered to wooden anchors (e.g., popsicle sticks or tongue depressors), and inserted into the sediment. The stick remains horizontal in the sediment and holds the seagrass in place until rooting occurs.

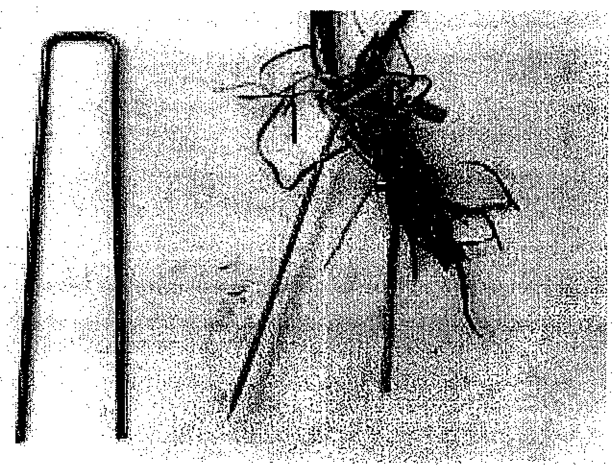


Figure 3.4. A technique developed by R. Davis and F. Short 1997 which substitutes bamboo barbecue skewers (right) for the metal staple (left). The skewers may be soaked overnight to increase their flexibility, and then broken in half. The fibers of the bamboo prevent the two ends of the skewer from separating, forming an inverted V-shaped staple which is used just as the metal staple. Advantage: biodegradable without formation of sharp points as sometimes occurs with metal staples when the bridge portion (nearer the oxygenated sediment surface) rusts away. Disadvantage: sometimes not negatively buoyant and pre-placement of planting units is not always possible.

ing tray. Typically, one person cuts plugs while a second person holds out the peat pots and arranges them in a floating tray. As the trays fill up, they may be sunk to the bottom until moved to the planting site. With either method of handling, all air must be squeezed out of the peat pots prior to submergence or the pots will capsize in the tray. The tray can be stabilized easily by placing a layer of wet burlap over the plants with an aluminum grid laid on top for ballast. The trays should be of a size to facilitate handling (~30 pots per tray). Planting can be accomplished in a number of ways. As with most of these methods, the PUs may be laid out by one person while others follow to plant them. One person loosens the sediment with a tree planting bar while the other person installs the peat pot in the bottom. Once in the bottom, the sides of the peat pot should be ripped down to allow rhizome spread. The rhizomes will not penetrate the peat pot wall. Despite their low cost, use of peat pots must be evaluated over a wide range of conditions before this technique is universally recommended. One such evaluation was provided by a worker in the mid-Atlantic region. Adapted from: "Utilization of Peat Pots in Transplanting Eelgrass" Ben Anderson (Delaware Department of Natural Resources and Environmental Control):

"This describes the methodology being used for the past two years in a program of eelgrass (*Zostera marina*) restoration in the Inland Bays of Delaware. From an operational perspective staple planting would work, but a major draw back was that it was extremely labor intensive. Various methods were tried to reduce the time and expense of the original three day process. The most successful and the method preferred to date was a variation on the above process. Three inch peat pots were used to transport and transplant the eelgrass. This method eliminated a whole day in the process, the most labor and personnel intensive-that of sorting bundling and anchoring the eelgrass planting units. This new process also shortened the time period that the plants were out of their environment and thus eliminated a level of stress that in all probability enhanced the success of the transplant and the well-being of the plants.

"The plants were harvested from the donor beds as before with a long-handled round nosed garden spade. The shovel passed through the sediments just beneath the eelgrass root zone, approximately three inches for these beds. The spade was then lifted to the water's surface and the eelgrass, with its intact sediment load, was transferred to a floating mesh box which was used as a sorting platform. The eelgrass/sediment matrix was gently broken apart into units that contained between 15 and 20 eelgrass shoots. The sediment volume for this size unit was too large to fit into the 3 inch peat pot; the rhizome sediment volume was reduced by gently cradling the roots and sediment in one's hands with fingers slightly spread and lightly massaging the sediment mass while allowing the sediment to fall between the fingers until the

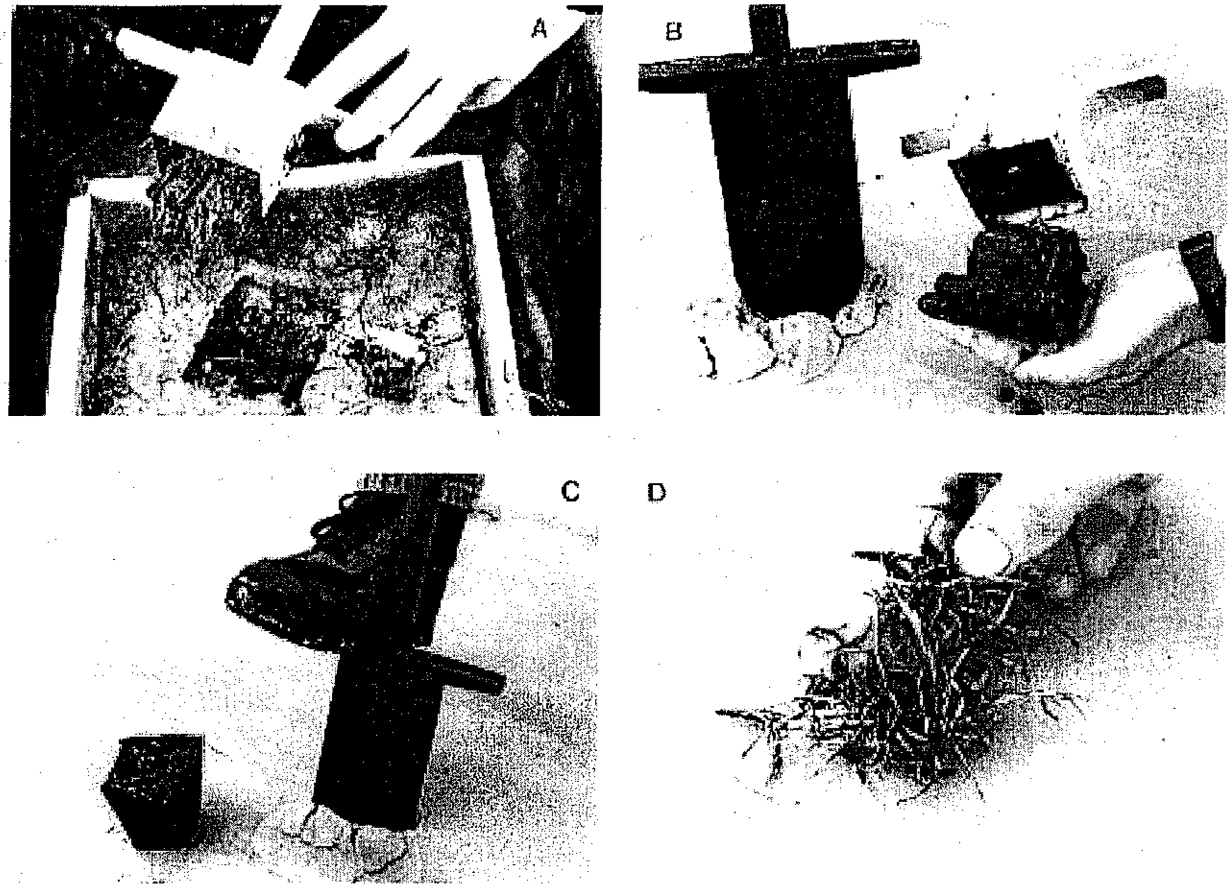


Figure 3.5. Peat Pot. (All photographs staged on land and in air for demonstration purposes only; Halodule wrightii *used in this demonstration.)*

(a) A sod plugger is used to take 3 x 3 inch plugs from existing seagrass beds.

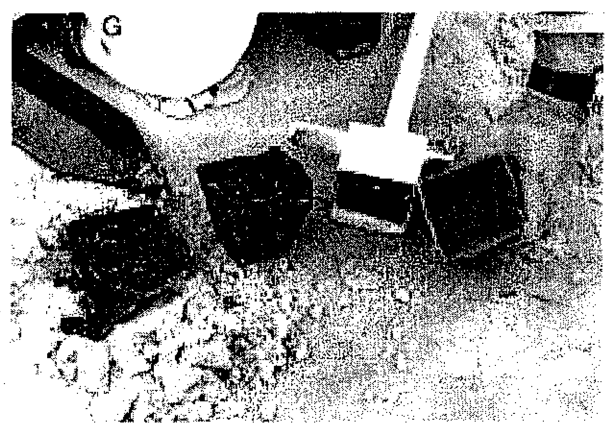
(b) Sod extruded to show size; note extruder plate inside plugger has been depressed to eject the plug. On left is a tree planting bar that is used to soften the sediment (by mixing the sediment with the overlying water; fluidizing) to insert the peat pot.

(c) Tree planting bar softening sediment. When underwater, a hole does not form but a fluidized zone forms in the sand, allowing the peat pot to be inserted easily into the sediment. Taking extra time to form a large fluidized area greatly facilitates peat pot installation. Use of the bar should just precede the installation of the peat pot else the sediment may de-fluidize and harden, preventing installation.

*(d) Peat pot planting material after washing away sediment; note small amount of planting material (*Halodule wrightii *in this case) used in a peat pot unit.*

(e) (Opposite page) Installing peat pot in the sediment.

(f) (Opposite page) A critical step after the peat pot is installed in the sediment: the peat pot walls must be torn down to allow the rhizomes to spread. Seagrass rhizomes often do not penetrate peat pot walls. This should be done immediately after inserting the peat pot into the bottom while the sediment is still fluidized.



(g) Summary of peat pot planting process. From right to left; a peat pot with fertilizer pellets prior to receiving a sod; sod cutter with extruder plate down; peat pot with sod, and installed peat pot with near wall torn down.

volume of the root/sediment mass would fit into the peat pot. This sediment reduction was best done in the water with hands just below surface. This allows a substantial degree of control in shaping and "feeling" the loss of the sediments until the dimension of the sediment/root volume approximates that of the peat pot. It was also important to overfill the peat pot so that a dome was created with sediment and roots in order to allow the rhizomes to creep over the peat pot as they grow and not allow a point of attachment for algae growth on the peat pot which would thus compete with the eelgrass. The potted eelgrass units are then placed into 9.5 gallon stackable Rubbermaid™ plastic storage trays approximately 24 x 16 x 8 inches for transport. Approximately 45 to 55 potted eelgrass units fit each tray. Our experience has shown that about 700 potted units can be collected by a three person team in about three hours. The team consists of one person on the shovel gathering the plants and two persons sorting and packing the peat pots.

"The trays were kept out of direct sunlight until planting and may be placed submersed in a shallow protected cove for a few days without any noticeable harm done to the plants. The trays have sufficient weight from the contained sediments to remain submerged and can be further stabilized by stakes and rope should conditions dictate. After a planting grid was placed on the planting site, peat pots were planted by divers, using a small hand garden shovel, at each intersect node in the grid matrix. The peat pot was totally buried in the sediments with the "dome" of the eelgrass and sediments flush with the existing sediment surface. Any portion of a peat pot exposed to the water will invite colonization of algae and thus may compete with the eelgrass for resources. When inserting the peat pot into the sediment it may be advantageous to "crack" the peat pot sides gently and lightly before planting the pot into the hole. This will allow the newly formed roots an easy path into the sediments and allow for faster root colonization in its new location. The planting grid was carefully removed, being careful not to disturb or damage the transplants."

Other Methods

A wide variety of methods are reviewed by Phillips (1982), Fonseca et al. (1988), and Harrison (1990). These include the use of whole sods, plastic pots, iron rods, concrete rings, wire mesh, plastic bags, attachment to construction re-bar, nails, and seeds. But of all non-whole plant methods, the use of seedlings is currently receiving widespread attention yet has had little actual application. Sowing seeds of seagrass has been studied for a temperate species (R. Orth, Virginia Institute of Marine Science, Gloucester Point, VA, pers. com.) and large areas (acres) have been established in the Chesapeake Bay by this approach. Granger et al. (1996) has experimented with pelletization of seeds as well as embedding seeds in biodegradable mesh; these experiments are in progress at this time and appear promising. Seeding techniques currently hold what we consider the highest promise for large-scale restoration of some damaged seagrass species. However, seed predation and stabilization (hydrodynamic regime) are two important issues to consider in use of seeds. Areas of high seed predation or high currents/waves may be problematic in application of this seed technique, although Granger et al.'s (1996) pelletization method may help overcome these problems. With *Zostera*, seed collection must be performed months in advance of a planned project. Given the lead time required for many planting projects, though, this should not be an impediment. However, this approach is now feasible for only one (*Z. marina*) and perhaps two other species (*R. maritima* and *T. testudinum*). Thorhaug (1974) introduced *Thalassia* seedling planting, and Fonseca et al. (1985) and others have all used *Thalassia* seedlings and a patented turtle grass seedling grow-out method has been registered by Lewis (1987). These methods appear to work, but are ultimately dependent on wild stock harvest of seeds and may be better suited for quiescent areas.

Meinesz et al. (1992) have successfully used plastic meshes with attached shoots, similar to the employment of degradable erosion control fabric by Fonseca et al. (1979). However because of their plastic components, both methods are now illegal in this country. Further, Fonseca et al.'s mesh was actually not bio- but photodegradable; a feature that was severely compromised in the estuarine sediment. Others have attempted planting of freshwater and brackish water species using biodegradable mesh bags containing PUs dropped overboard (Durako et al. 1993). There has been mixed success and these methods have only been tested in small-scale experiments. Stout and Heck (1991) found no survival of bagged *Vallisneria* tubers while staple units had ~75 percent survival. Coconut fiber erosion control mats have been tested with micropropagated *Ruppia maritima* (5 planting units per 20x20 cm mat; M. Durako, University of North Carolina at Wilmington, Wilmington, NC, pers com.). Planting units were pinned to the mat with hair pins and mats attached to the bottom with erosion-control staples (see Staple Method, above). One advantage to this method is that shoots are held erect and are less susceptible to burial.

Another technique using manila line or twine has been mentioned to us independently by K. Merkel (Merkel and Associates, San Diego, CA) and the late K. Bird (Univ. North Carolina, Wilmington, NC), working on the West and East Coasts, respectively. The line method simply involves untwisting the line (or twine) itself which is a 3-ply, using a very loose lay, and inserting shoots between the open lays of the line (Figure 3.6). The line has enough resilience to close again, holding the shoot in place. Coils of line with inserted plants can then be quickly fabricated and payed out on a planting site and stapled to the bottom. Planting times are not available but this could be a very promising technique, especially in quiescent areas. It may be prudent to cut the line periodically so that after it is installed an errant propeller does not wrap up large portions of the planting area.

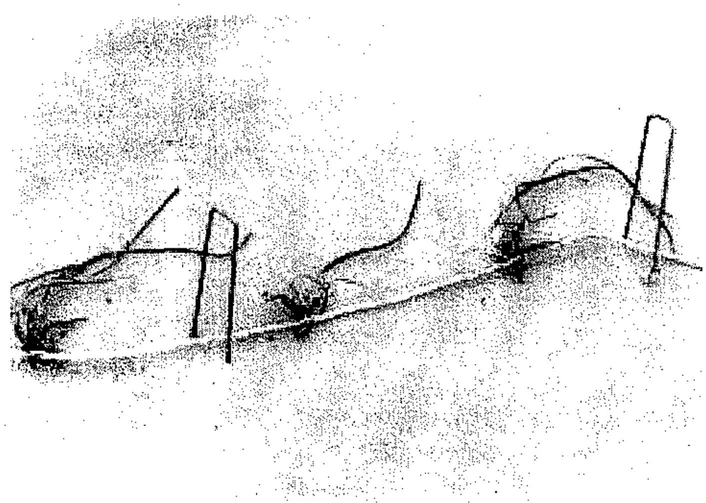


Figure 3.6. Line planting technique (suggested by several contributors). Seagrass shoots are inserted through natural fiber string, such as baling twine and pinned to the bottom (shown here with metal staples). String can be coiled in water-filled tubs and paid out over the side of a boat or floating tubs allowing rapid installation. We suggest that periodic cuts be made in the line after installation so that any failures (such as propeller entanglement) are not transmitted to large numbers of plants.

Another planting method has been developed by (F. Short, Univ. New Hampshire, pers. com.) that, like the cotton mesh bag method of Durako et al. (1993), is designed to avoid the cost of divers. The TERFS (transplanting eelgrass remotely with frame systems; Figure 3.7) may overcome several other potential planting problems. One is that this technique is suitable for deployment in contaminated areas (e.g., PCBs) that might otherwise not affect plant growth but would make diving operations extremely costly, not to mention hazardous. Thus, this method may be

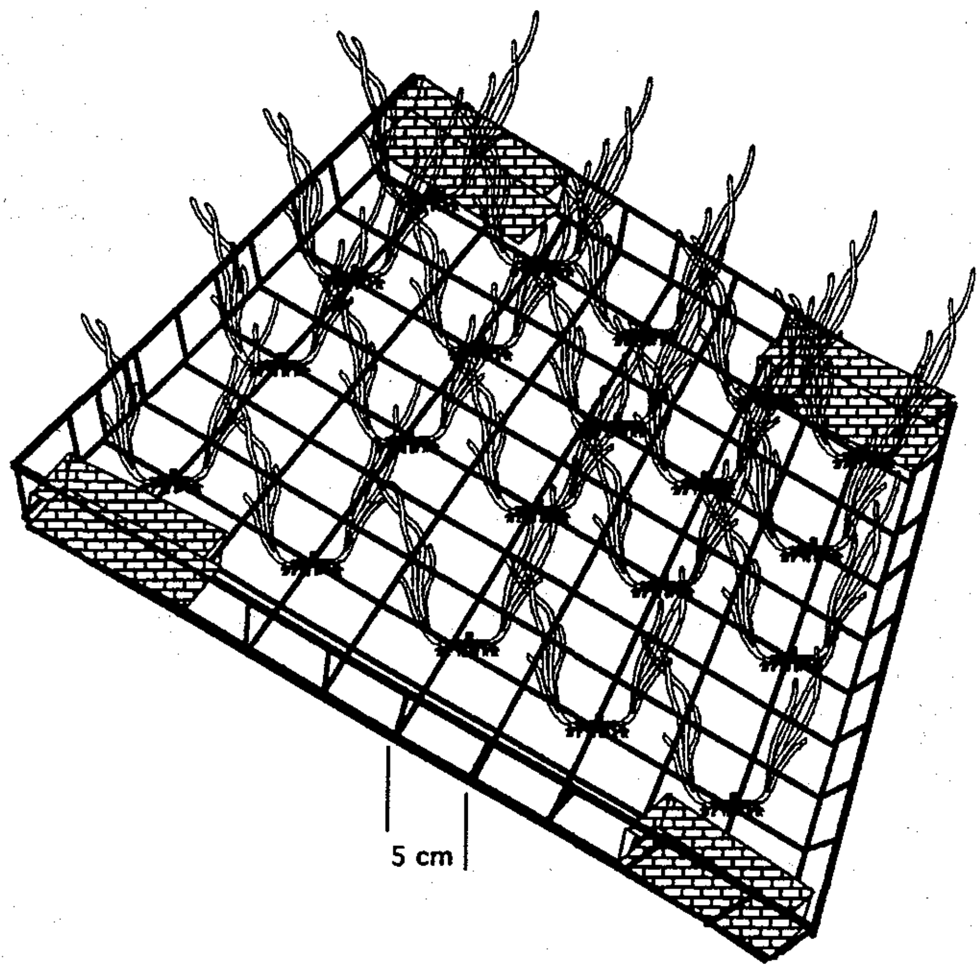


Figure 3.7. Remote planting method (TERFS: Transplanting Eelgrass Remotely with Frame Systems, developed by F. Short, Univ. New Hampshire), designed to avoid the cost of divers and for deployment in contaminated areas (e.g., PCBs) that might otherwise not affect plant growth but cause diving operations to be costly and hazardous. This method may also be applied as a bioassay tool. Plants are attached to the frame with paper ties, the frame can be retrieved after a suitable rooting time, leaving the plants in the sediment.

applied as a bioassay tool. The frame also may minimize some bioturbation hazards. Because the plants are attached to the frame with paper ties, the frame can be retrieved after a suitable rooting time, leaving the plants in the sediment. Strict cost comparisons are not yet available, but these remote techniques are promising for specialized applications.

For hydrodynamically rigorous settings, plantings with large sods may be appropriate. This approach is only now being applied (E. Paling, Murdoch Univ., Perth, Western Australia, pers. com.). Massive sods with their intact rhizospheres may possess sufficient integrity to allow establishment in areas where small cores or bare-root plantings are quickly eroded or exposed during sediment migration. Care would have to be taken to fully install sods into the sediment. A sod extending into the water column would be highly vulnerable to tidal current-induced erosion or acceleration reaction and lift forces under waves. Costs associated with moving large sods are unknown, but may prove cost-effective as compared with other methods.

Fertilizer Effects

A potential advantage to the peat pot method over staples is that slow release fertilizer may be added to the pots and installed with the plantings at little additional handling cost (Figure 3.5 g). An innovative technique is needed to add fertilizer to the sediment with other planting methods (see section on "Nutrient Requirements for Transplanting," above, to aid in guiding choices on application). J. Anderson (Ruskin, FL, pers. com.) has developed a pontoon boat system for injecting plant hormones and liquid fertilizer into the margins of prop scars but no data are currently available to assess its effectiveness. Previous work by Orth (1977), Pulich (1985), Fonseca et al. (1987b), Williams (1990), and Kenworthy and Fonseca (1992) has met with mixed results, due at least in part to suspicious performance of the fertilizer. Fonseca et al. (1994) did find slow-release pellets to be empty after the prescribed 70-day release period, with all their fertilizer apparently solubilized. They observed a significant increase in shoalgrass population growth in sediments which contain approximately 1.2 percent carbonate but only in association with phosphorus addition. Additions of nitrogen alone or in combination with phosphorus had little or no apparent effect. However, when these experiments were repeated, only nitrogen additions had any significant effect. These results are similar to those found by Short et al. (1985) and Powell et al. (1989) who found phosphorus-linked stimulation of seagrass productivity in carbonate sediments. Fonseca et al. (1994) and Kenworthy and Fonseca (1992) recommend that peat pot plantings of *H. wrightii* in sediments containing > 1.0 percent carbonate may benefit substantially from initial

additions of slow-release phosphorus fertilizer. The recent findings of Duarte et al. (1995) suggest that iron limitation in carbonate sediments may also be significant, implying the need for iron additions to plantings in carbonate sediments.

Spacing of Planting Units

Quiescent Settings

Much attention has been given to row spacing of plantings (Fonseca et al. 1982, 1984, 1985, 1987b,c, Merkel 1988b). The reader is directed to those references for a detailed study of the derivation of appropriate spacing. In practice, PU spacing typically ranges from 0.5 to 2.0 m on center. More rapid coalescence of bottom coverage is logically achieved with higher planting density. The benefit of increased rate of coalescence is offset by substantially higher costs due to the number of PUs involved. For example, a 100 m X 100 m (1 hectare) planting area planted on 2.0, 1.0, or 0.5 m centers would require 2,500, 10,000, or 40,000 PUs, respectively.

Wave-Exposed or High Current Speed Settings

In areas with currents over 30 cm/sec, or with long fetches (over 1 km), one may anticipate that the seagrass beds do not naturally cover the bottom completely (Patriquin 1975, Fonseca et al. 1983) (Figures 1.3,1.6, 2.4). In these instances, planting at high densities such as 0.5 m centers, in groups of plantings 5 to 10 m on a side will probably improve the chance of survival. Experimentation is needed for planting in high energy settings. As a result we offered a generalized planting modification scheme (Figure 2.5). Because percent cover by seagrass decreases nearly linearly with both wave exposure and current speed, we devised a decision matrix for calculating row spacing and grouping of plantings based on these models (Figure 3.8). These models are based on seagrass beds (mixed *H. wrightii* and *Z. marina*) in North Carolina (Figure 2.4); we have evidence to suggest that these models will not predict seagrass coverage as well in areas that do not have strong tidal currents. We are confident, however, that the general approach of modifying the arrangement of PUs to accelerate bed form development toward expected patchy, rather than continuous cover is appropriate. We urge users to modify this approach as might seem appropriate given the wide range of conditions that constitute high-energy settings.

1) Compare % cover estimate from equations on p. 75

2) Use smaller % cover value of those found using those equations

3) Compute required area of seafloor to be planted:

Impacted are (1M) * (1/% cover) = mitigation area
Ex: 1M = 5,000 m^2 Predicted % cover = 30%

note: 1M is based not only on area directly impacted but additional acreage computed for interim loss

5,000 * (1/0.3) = 16.667 m^2 of seafloor must be planted in this wave and/or current climate to achieve the total 1M acreage

4) Compute a nominal planting unit (PU) density (this is not the total number for the project)

Based on nominal 1 m spacing of PU under quiescent conditions...

$\sqrt{1M}$ = x then, $(x + 1)^2$ = Nominal PU density: save for later computations

$\sqrt{5,000}$ = 71. $(71 + 1)^2$ = 5.184 PU

Make observations of local patch sizes:
Ex: local patch sizes appear to be -10 x 10 m

5) Take the total area of seafloor to be planted (here 16,667 m^2) and divide it into subunits of the local patch size and multiply by predicted percent cover e.g., 30% cover:

(16,667 / 100) * 0.30 = 50 subunits (where 50 * 100 = original 5,000 m^2 to plant)

6) Take the square root of that area:

$\sqrt{5.000}$ = 71

7) Divide by the square root of the number of PU:

$\sqrt{16,667}$ = 130 which is: 71 / 130 = 0.55 m spacing

8) Multiply the reciprocal of spacing * square root of patch area, square that value and then multiply by the number of subunits to compute TOTAL NUMBER OF PU FOR THE PROJECT

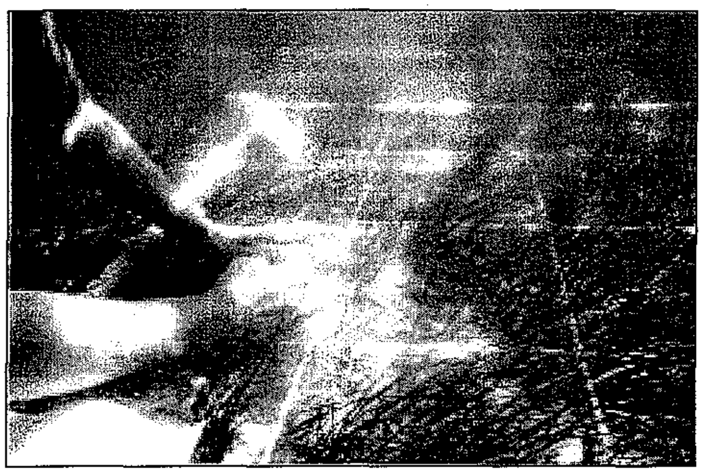
$\left((1 / 0.55) * \sqrt{100} \right)^2 * 50$ = 16,529 PU (note the correspondence between this number and the actual area of seafloor to be planted)

Figure 3.8. Computing row spacing with strong effects of waves and/or tidal currents.

CHAPTER 4

Monitoring and Evaluating Success

TERMINOLOGY



In order to promote effective restoration and mitigation, one must have an unambiguous definition of success. As seen in the comparative analysis (above), there are many criteria that have been used for evaluating planting success (Table 1.9). It is our opinion that simple measures of seagrass coverage and persistence are at this time: (1) the most parsimonious indicators of many other seagrass bed functional attributes, and (2) constitute the most pragmatic choices of monitoring parameters under the present system of resource management.

Through seagrass planting, one is attempting to establish a viable plant community that performs habitat functions equivalent to ones that were lost. The evaluation of all seagrass ecosystem functions (e.g., sediment stabilization, biomass production, nutrient cycling, secondary production) is almost always beyond the resources of any project. However, we have been conducting research with the goal of identifying diagnostic parameters which can be inexpensively monitored so as to infer with reasonable certainty that specific functional attributes have been restored. Many habitat functions (e.g., animal abundance, taxonomic composition, complexity of the seagrass canopy, macroalgal abundance) appear to relate simply to coverage and persistence of that coverage; parameters that are inexpensively monitored (Fonseca et al. 1990, Meyer et al. 1990, Fonseca et al. 1996a,b). Although these findings are limited to studies in Tampa Bay, FL, and southern Core Sound, NC, we feel that they provide sufficient basis to guide resource managers in some decisions regarding planting success. However, we stress that validation of the following generalizations should be an ongoing process with an emphasis on extending geographic replication. Therefore, we define seagrass planting success as:

> **the unassisted persistence of the required acreage of seagrass coverage for a prescribed period of time (suggested minimum of five years).**

The required acreage is a result of the replanting ratio (habitat acreage restored / habitat acreage lost) which is in turn a function of agency policy and the nature of the planting site. Fonseca (1989a, 1992, 1994) described what to monitor, how to perform the monitoring, and how to interpret the results. The following are modified excerpts from those publications.

Monitoring Specifications

No one data type can stand alone in a monitoring program (Fonseca et al. 1987c, Fonseca 1989a). Several factors must be considered and these lead to an ecologically valid characterization of seagrass planting success. Sufficient monitoring should be conducted to ensure that any contracted work was performed to specifications. However, in any situation, monitoring of planting performance using standard methods provides the basis for mid-course corrections (Fonseca 1989a) and improved planning of subsequent projects.

Survival

The number of PUs that survive should be recorded. This may be expressed as a percentage of the original number, but the actual whole number is critical as well. If a planting site is sufficiently small (~500-1000 PU), all PUs should be surveyed for presence or absence (survival survey). The existence of a single short shoot on a PU indicates its survival (hopefully that shoot is associated with a rhizome meristem, otherwise subsequent vegetative growth will not occur). If a site is large, then randomly (not arbitrarily) selected rows or subsections (area in m^2) should be sampled. Since each row or subsection is actually the level of replication, at least 10 replicate rows or subsections should be performed at the level which one wishes to generalize their findings (e.g., over the whole planting site). At the very least, stabilization of the running mean of survival should be obtained as a measure of statistical adequacy.

Areal Coverage

A **random** (as opposed to arbitrary) sample of area covered (m^2) per planting unit should be recorded until coalescence (the point where individual PUs grow together and the PU origin of individual shoots cannot be readily observed). The area covered by a PU may be measured by recording the average of two perpendicular width measurements (in meters) of the PU over the bottom. These numbers are averaged, divided by 2, squared, and multiplied by π (i.e., $\pi^* r^2$) to compute the area of a circle, and in this case, the PU. This procedure tends to give a higher value than use of a quadrat with 25 cm^2 resolution, particularly when PU expansion is not uniformly circular. With a quadrat survey, a 50 x 50 cm quadrat, divided into 5 x 5 cm grids (string on 5 cm centers across the quadrat frame) is laid over the PU and filled grids counted. In this case, the number of 5 x 5 cm grids (or half grids if there are only 1 or 2 shoots in the 5 x 5 cm grid) that have seagrass shoots are totaled and converted to meters square of cover for the PU. The quadrat method is more appropriate for seagrasses that propagate by long runners (e.g. shoalgrass), and do not form a clearly radial growth pattern (e.g. eelgrass). The number of surviving PUs may then be multiplied by the average area per PU to determine the area covered on the planting site. After coalescence, the area of bottom covered should be surveyed using randomized grid samples at the 1 m^2 scale width (Fonseca et al. 1985). These data may be used to assess persistence of the planting as well as total seagrass coverage.

For very large seagrass plants whose rhizome mats do not significantly interdigitate (e.g., *Zostera marina* on the West Coast of the U.S., R. Thom, Battelle Pacific Northwest Lab., Sequim, WA, pers. com.; pers. obs.), post-coalescence techniques may actually be more appropriate with the 0.0625 m^2 resolution being perhaps better scaled to this plant size and spacing.

Number of Shoots

Random samples should be collected to measure the number of shoots per PU. The data from pre-coalescence surveys may be used to compare performance relative to other, local plantings by plotting the average number of shoots per PU as well as shoot density (number of shoots PU^{-1}/area PU^{-1}) over time. The comparison may be statistical or visual on a graph (which often suffices to detect grossly different population growth rates). Early stage PUs that are still associated with the anchor are artificially clustered by the nature of the PU show an artificially high m^{-2} shoot density, sometimes ten times higher than a reference site. When shoot density of a PU is essentially equal to that of reference sites, this indicates that the plants have spread to a point where they are occupying area in a way consistent with long-term establishment and that successful colonization has occurred. This is an important indicator of planting performance and environmental suitability of a site. Shoot number is recommended in addition to areal coverage because shoot addition is a more accurate means of assessing the asexual reproductive vigor of the plantings. Also, areal coverage varies with the environmental setting of the planting. For example, in areas of high current shoots grow more densely. Without shoot number data, the patchy pattern and low areal coverage in high current environments could be erroneously ascribed to poor planting performance instead of a natural pattern of growth. Moreover, the size of the naturally-occurring patches gives an idea of the expected form of planted patches (see section on "Spacing of Planting Units," above).

MONITORING FREQUENCY

Survival, areal coverage, and number of short shoots per PU are straightforward measures, although they usually require snorkeling or SCUBA diving (a factor that is surprisingly not considered, or equipped for, by many attempting these data collections). An individual can be trained to count in a few hours, and can count individual PUs in 5-10 minutes or less at early stages of post-planting growth.

Monitoring of shoot numbers, area covered per PU, and shoot density should, at a minimum, be done quarterly for the first year after planting and biannually thereafter for a minimum of four more years (a minimum total of five years). After PUs begin to coalesce and the PU from which shoots originated can no longer be discerned, areal coverage and shoot density data should be recorded and counts on a PU basis suspended. However, any replanting after the initial planting resets the monitoring clock and monitoring frequency to time zero.

However, as suggested by Fonseca (1989a), mistakes may be made in selecting a site. Repeated plantings may fail. There needs to be some decision sequence to break the cycle of replanting in perpetuity before managers lose control of the process. We propose the following sequence. For *Zostera* spp, *Halodule*, *Syringodium*, *Ruppia*, and *Halophila* spp., monitoring should continue for a minimum of three years (Figure 4.1). For *Thalassia* and *Phyllospadix*, longer periods of time may be required. While the time course for monitoring *Phyllospadix* spp. has not been determined, its slow vegetative growth suggests that 5-10 years may be an appropriate length to begin with, with comparatively quiescent areas monitored for 5 years while more wave-exposed sites monitored for 10 years. For *Thalassia*, restoration of the rhizome mat could conceivably take a century. Because most permits cannot be followed for anywhere close to that time, we suggest that 5-10 years may also be appropriate. Duration of monitoring may also have to be extended and/or intensified for sites susceptible to subsequent human impacts such as propeller scarring.

If at any time a loss of PUs occurs, then replanting (remedial planting) should be done. Because this tends to happen with greater frequency soon after planting, heavy arrows are used (Figure 4.1) to indicate conveyance to Replanting #1; less

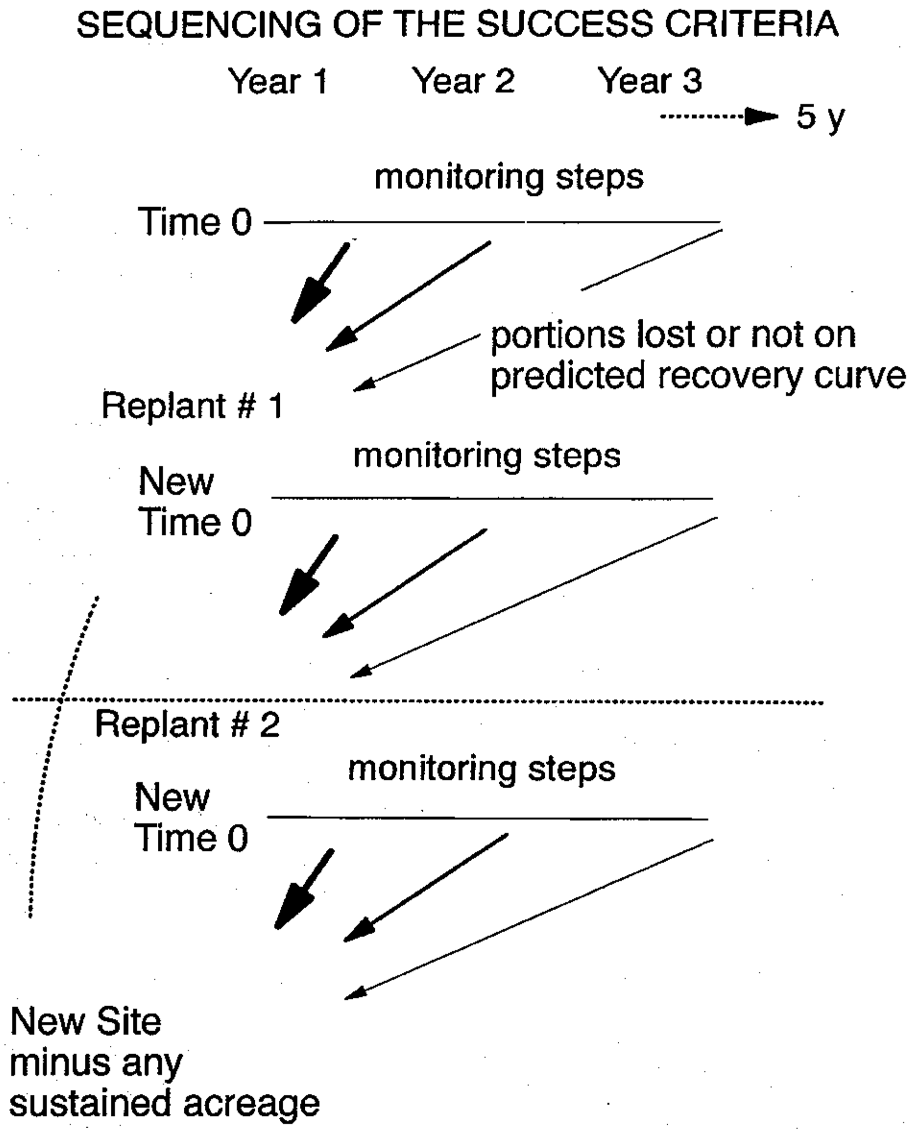


Figure 4.1. Sequencing of the seagrass planting success criteria. Initial planting is done at Time 0. Monitoring is to continue for a minimum of five years. If at any time a significant loss of planting units (PUs) occurs, then replanting (remedial planting) should be done. Because this tends to happen with greater frequency soon after planting, heavy arrows are used to indicate conveyance to Replanting #1; less heavy arrows are used to indicate the lowered expected frequency of remedial planting as time progresses. Any remedial planting on the original planting site itself means that for those PU, the clock is reset to zero. This prevents chronic replanting right up to the end of the five year monitoring period. If PU losses are again experienced, then a second remedial planting may be called for. At this point the dashed lines indicate that if some clear and overriding problem is evident with the planting site (e.g., repeated large scale losses due to unfavorable environmental conditions), then a decision may be made to select an altogether new site and start over (minus any acreage sustained at the original planting site). However if losses are minimal and actions can be taken to ameloriate the agent of loss (e.g., adding bioturbation exclusion devices), then continued remedial planting would be allowed. Only rarely would any additional replantings be allowed on the original site after two remedial tries. At that point an altogether new site should be considered.

heavy arrows are used to indicate the lowered expected frequency of remedial planting as time progresses.

Any remedial planting on the original planting site means that for those PUs, the clock is reset to zero. This prevents chronic replanting of an unsuitable right up to the end of the three-year monitoring period. If PU losses are again experienced, then a second remedial planting may be called for. At this point the dashed lines (Figure 4.1) indicate that if some clear and overriding problem is evident with the planting site (e.g., repeated large-scale losses due to unfavorable environmental conditions), then a decision may be made to select an altogether new site and start over (minus any acreage sustained at the original planting site). However if losses are minimal and actions can be taken to ameliorate the agent of loss (e.g., adding bioturbation exclusion devices), then continued remedial planting would be allowed. Only rarely should any additional replanting be allowed on the original site after two remedial tries. At that point a new site should be considered. Ideally, an alternate site would be selected in the initial site-selection process.

INTERPRETATION OF MONITORING DATA

The computations described above allow a direct comparison on a unit area basis of planted versus lost acreage. Success may then be based on whether the targeted amount of coverage has been generated. This is a quantitative measure which is assumed to be diagnostic of ecological function. If the planting project is for mitigation, then compliance may thereby be interpreted as both acreage generated and the unassisted persistence of that acreage over time (the three year period). The persistence issue is also critical. If the planting does not persist, then the ecosystem has experienced a net loss and the project has not been successful. The population growth and coverage data may be compared periodically with published values (dependent on species and ecoregion) as a relative indicator of performance.

Although these data collections may seem involved, they represent some of the simplest and least expensive metrics we found in our survey of planting projects. Moreover, without them cost projections cannot be made and cost overruns can follow. In computing costs, it should be recognized that little or no additional care may be required once the plants are established. Natural disturbance (rays, storms) and seasonal peaks and troughs in growth are to be expected.

As an example of how these various monitoring conditions we have added an example plan (Appendix E). There, we give specific language for planting site selection criteria, monitoring, replacement ratio computation surveying with the Braun-Blanquet (1965) method (also see Appendix E, p. 220). The plan was designed with both propeller scars and mooring scars in mind. However, unlike the specialized prop scar portion of the plan, the mooring scar part can be generalized to many plantings that involve broad, open areas that are not in high energy settings. To use the mooring scar part in high water motion areas, apply the decision sequences given in Figures 2.1 and then revise that by applying the decision processes from Figures 2.5, and 3.8.

Real Costs of Seagrass Transplanting

One of the most-asked questions is "how much does seagrass planting cost?" Given the wide variety of seagrass growth strategies and environmental settings under which seagrass beds occur, this is not an easy question to answer. Prices vary widely. Besides the direct influence of project size on cost, the following are some factors that we have seen to generally constitute grounds for increased costs (in an approximate decreasing order of importance):

1. inappropriate site selection,
2. inexperience (inefficiency, poor technique),
3. high disturbance, e.g., bioturbation (actual losses and therefore costs of re-planting or exclusion devices which are inherently costly to construct and deploy),
4. water depths that require use of SCUBA divers,
5. low visibility,
6. soft sediments (especially when wading or walking on the site is required),
7. rough seas,
8. cold water planting,
9. capitalization (purchasing equipment: e.g., boats, motors),
10. wide profit margins,
11. amount of site preparation (e.g., creation of subtidal dikes)
12. excessive frequency of monitoring, and
13. overly detailed parameters chosen for monitoring (blade width, length, faunal assessment).

Over the years there is a general trend for seagrass planting costs to be similar to those for salt marsh planting. A recent review by King and Bohlen (1994) found that submerged aquatic plant restoration typically ran between \$19,000 and \$20,000 acre^{-1} (~\$47–49,000 ha^{-1}; albeit based almost exclusively on freshwater work). Thorhaug and

Austin (1976) reported the direct cost of planting seagrass to be ~\$25,000 ha^{-1} and Fonseca et al. (1987a) reported collection, fabrication and planting costs to total an estimated \$19,000 $acre^{-1}$ (~\$46,940 ha^{-1}). The Port of Miami planting project reportedly paid almost \$3 million for a +200 acre planting (~\$37,000 ha^{-1}; Stein 1984). Plantings in Tampa Bay in the late 1980's were priced at approximately \$2.50 PU^{-1} (~\$25,000 ha^{-1}) (M.O. Hall, Florida Department of Environmental Protection, St. Petersburg, FL, pers. com.) and more recently we have seen planting projects with monitoring included at ~\$49,000 ha^{-1} (K. Fitzpatrick, Sebastian Inlet Tax District Commission, Indiatlantic, FL, pers. com.). Thus, even without corrections for inflation, the price of seagrass planting appeared to have been consistent for nearly the last 20 years, varying with species and the density of plantings, and other extenuating circumstances, but averaging near \$37,000 ha^{-1}. There are large cost discrepancies though. For example, if one takes the costs quoted by Zieman (1982) and correct them to 1997 dollars, the cost per acre is in the neighborhood of ~\$316,000 ha^{-1}. Therefore, we feel that most of these projects did not accurately compile the real costs associated with launching a restoration or mitigation project. To cost out the project plan described in Appendix E in 1997 US dollars, we used a 12,580 PU, *Halodule wrightii* peat pot planting that requires boat access. The costs included securing aerial photographs (~\$5,465), preparing maps and groundtruthing (~\$14,314), collecting, preparing and installing (~\$64,846), and monitoring with re-fertilization and report writing (~\$205,650), and contractor profit of 10% (~\$20,028), the total cost was ~\$206,000 $acre^{-1}$, or ~\$510,000 per ha^{-1} (a value closer to Zieman's 1982 estimate than more recent estimates from other projects). Our estimate did not include costs incurred by any Government oversight. Although there may be many ways to reduce cost, the discrepancy between ~\$37,000 ha^{-1} and over \$500,000 ha^{-1} means that large variation in costs can be expected and that it is prudent to conduct a detailed costing before allowing the loss of a seagrass bed.

More recently, some effort has been made to guarantee planting success. However we caution against such guarantees without specific caveats regarding remedial plantings (incidently, terrestrial crops are not guaranteed and there is much more experience in their production). With a planting guarantee, there are three possible outcomes:

1. A very successful planting that requires little or no additional planting is heavily over-compensated, e.g. \$100,000 per acre as opposed to the average of ~\$15 K $acre^{-1}$ (above: \$37 K ha^{-1}).

2. If extensive replanting is repeatedly required up to the end of the guarantee period (if a guarantee limit was set), then no effective mitigation has occurred.

3. If there is no limit and replanting must continue for some indefinite period, then the criteria of success are not met.

Because planting conditions do vary so widely it is sometimes inappropriate to judge each project based on an average cost (see reasons listed above, some of which are difficult to control). Rather, we have historically focused on providing managers with some estimation power by evaluating the costs to harvest plants, fabricate PUs (when a method called for this), and installation of the PUs (Fonseca et al. 1984, 1985, 1987c, 1994). These numbers have been challenged as being too optimistic both by consultants and independent investigators (pers. com.).

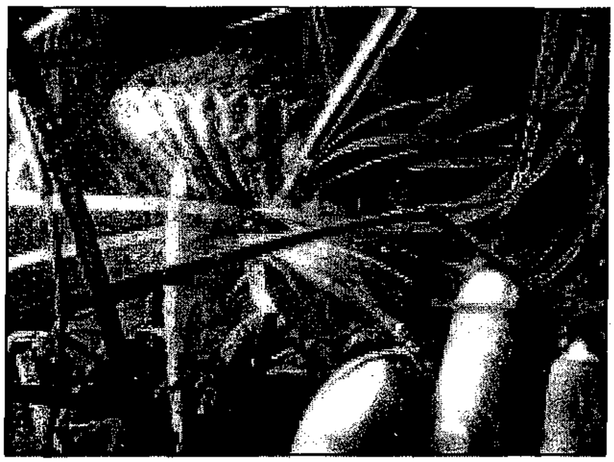
The reader must be absolutely clear that the data we have reported should only be used as a guide from which managers may roughly assess costs and from which they can ask questions about what factors are actually contributing to the cost discrepancy they present versus those reported by us in the literature that emphasize only planting costs. These other costs must then be added on, such as break times, travel, training, mobilization and demobilization, materials (usually negligible unless exclusion cages are employed, etc.) and scrutinized for their reasonableness. Because he planting itself cannot be conducted for much less cost than reported for planting alone, any reductions in project cost must be found in other components of the project (i.e., capitalization costs, profit, etc.). We emphasize the heuristic value of these data, recognizing they embody only certain aspects of a larger planting program and can only hope that resource managers utilize these data in that manner.

Our plantings costs in work-minutes for the three categories of effort listed above — collection, fabrication (if appropriate) and deployment of stock — have ranged between ~2 and 3.5 work-minutes PU^{-1} (Fonseca et al. 1994). For a 1-hectare planting on 1-m centers, this means between approximately 340 and 595 work hours of labor. At a given cost (e.g., \$10.00 h^{-1}) this gives a fundamental cost of ranging between \$3,400 and \$5,950 ha^{-1} or an average of ~\$1,900 $acre^{-1}$, similar to our previously published data (Fonseca et al. 1984, 1985).

CHAPTER 5

Manager's Summary

NO-NET-LOSS OF WETLANDS



The national "no-net-loss" policy for wetlands was adopted to counter tremendous losses of these valuable natural resources, with mitigation playing a central role in its implementation (White House Office on Environmental Policy 1993; Zedler 1996). For the purposes of this document, mitigation refers to activities related to permitted habitat conversions and includes a sequence of avoiding damage, minimizing damage, and finally, if needed, planting to compensate for damage. Compensatory mitigation usually follows the destruction of existing habitat when the agent of loss and responsible party are known. Compensation assumes that ecosystems can be made to order and, in essence, trades existing functional habitat for the promise of replacement habitat. In addition, the "no-net-loss" policy recommends increasing the quality and quantity of wetland resources through restoration of historically degraded habitats. Here the term, "restoration" does not apply to permit-associated projects, although planting techniques and assessment used may be identical.

SEAGRASS ECOSYSTEMS

Wetland resources include subtidal seagrass beds and their associated interspersed unvegetated bottom which perform a number of important ecological functions and are among the most productive ecosystems on the planet. There are at least 13 species of seagrasses in U.S. waters, with seagrasses occurring in all coastal states, with the possible exception of Georgia and South Carolina. Conservation, mitigation, and restoration attempts have been underway for many years, and despite the wide-scale distribution and ecological importance of seagrasses, surprisingly little is known regarding some aspects of their distribution, population biology, resistance to various disturbances, and rates of recovery following disturbance.

VALUE AND FUNCTION OF SEAGRASS HABITAT

Seagrasses occur almost exclusively in shallow, soft-substrate habitats where their roots bind sediments and their canopies baffle waves and currents. Seagrasses and their associated epiphytes are highly productive, produce a structural matrix on which many other species depend, improve water quality, and stabilize sediments.

Because of their requirements for high light levels, seagrasses are restricted to shallow coastal areas where anthropogenic disturbances that damage or kill them are common. Unfortunately, once seagrasses die, the sediments they helped stabilize may be resuspended into the water column, potentially lowering light levels to intensities that may not allow seagrasses to recover in this site unless the entire watershed is managed to improve water clarity.

LOSS OF SEAGRASS HABITAT

As human population concentrates along our coastlines, anthropogenic impacts to seagrass habitats increase through nutrient loading from runoff, light reduction from increased turbidity due to phytoplankton blooms, increased boat traffic, and more direct vessel impacts such as propeller scarring.

In recent years, seagrass losses of 30% to 90% have been reported from the Chesapeake Bay and coastal areas of Texas, Florida, Washington, and California. In some cases, historic losses from disturbance and disease appear to have been even greater. Disturbances kill seagrasses rapidly while recovery is usually very slow. If the resource services that seagrass beds provide are to be maintained, lost beds need to be restored and processes harming present-day beds need to be minimized or restored through improvement of environmental conditions that facilitate recovery.

MITIGATION AS A MANAGEMENT TOOL

Recent legeslation embodied in the Manguson-Stevens Fisheries Management and Conservation Act of 1996 recognizes that the long-term viability of living marine resources depends on protection of their habitat, and requires that each of the Fishery Management Councils describe and identify essential fish habitat in fishery management plans and avoid or minimize adverse impacts to such habitat. It also requires that the Secretary of the Department of Commerce initiate and maintain research to identify essential fish habitat, the impact of wetland and estuarine degredation, and other factors affecting the abundance and availability of fish. Also required are recommendations on research needed to develop restoration techniques for these habitats. Because productivity and recruitment success may be determined at different life history stages of a fishery species, the Plans are required to describe each of these stages and their connectivity to habitats. The first amendments are due to Congress for evaluation and approval in October 1998.

Seagrasses have been recognized as one of the many habitats that are essential to conservation agencies and organizations around all coasts. Many of the management organizations have formal and/or informal policies on aspects of management of seagrass habitats. Recently the Atlantic States Marine Fisheries Commission (ASMFC), which assists in managing and conserving shared coastal fishery resources of the 15 Atlantic coastal states from Maine to Florida, established a "Submerged Aquatic Vegetation Policy". The promulgation of this policy was based on the recognition that many of the ASMFC managed species are directly dependend upon SAV for refuge, attachment, spawning, food, or prey location. Coupled with the Essential Fish Habitat component of the Magunson-Stevens Act of 1996, seagrass meadows along the Atlantic coast and elsewhere should receive more conservation, protection and enhancement measures.

In the past, mitigation was perceived as an experimental tool rather than an established management practice. Given the documented success of mitigation, this perception is no longer appropriate. Seagrass planting is now a proven management tool. However, planting will not succeed unless managers appreciate and emphasize the extreme importance of site selection, care in planting, and incorporation of plant demography into the planting and planning process. Many planting failures have resulted from poor site selection or poor planting procedures rather than basic limitations of planting technology. When appropriate procedures are followed, planting has been relatively successful (e.g., Southern California sites). Planting of different seagrass species has been employed in a variety of habitats, using a wide range of procedures. The relative success of seagrass plantings when using different techniques, seagrass species, or habitats has often been difficult to judge rigorously because of the absence of standard assessment techniques following planting a problem common to habitat restoration in general (Mager and Thayer 1986, Race and Fonseca 1996). However, seagrass plantings that persist and generate the target acreage have been shown to quickly provide many of the functional attributes of natural beds.

Special Planning Considerations

Whether a project focuses on restoration or compensatory mitigation of an injured site, careful and thoughtful planning is crucial to project success. Managers need to determine if a seagrass system has been injured, how much area has been disturbed, and what constitutes adequate remediation. These decisions may not be as straightforward as they might seem. For seagrass beds that establish seasonally from seed banks, beds that are lush in the summer may appear as bare sand in the winter. Thus, if there is no historical or at least seasonal perspective, a manager could look at the site in the winter and conclude that no seagrasses are present and no mitigation is needed. A similar error could occur when monitoring a planted bed. If the responsible party planted a large bed that did well, set seed, and died back (as might be natural for this location and species of seagrass), mitigation might be judged to be successful if the bed were checked in the summer of establishment or during the next summer when the seed bank had germinated. If the site were checked in the winter, however, mitigation at this same site might be judged to have failed completely because only bare sand would be visible. Such instances call for more comprehensive site surveys such as coring for seeds and/or living rhizome and shoot meristems. Moreover, some beds migrate over time, meaning larger areas of the seafloor must be set aside to maintain the patchy population.

Methods of seagrass transplantation that are efficient and cost effective in one geographic region may be ineffective in another. Managers should consider the life history characteristics of local species, and how these species vary geographically, seasonally, and as a consequence of various physical (e.g., temperature) or biological (e.g., bioturbators) regimes.

Preserving Genetic Diversity

Continued research is needed to determine how anthropogenic actions may isolate small populations and erode genetic diversity. Managers should strive to potentially maximize genetic diversity by selecting planting stock from a variety of widely distributed seagrass beds. Collection of all planting units from one localized bed, even if that bed appears robust, may result in a high degree of relatedness among transplants; this lack of genetic diversity could depress sexual reproduction or make planted beds more uniformly susceptible to diseases or other disturbances.

Although populations are difficult to define, managers should also strive to conserve existing stock and minimize geographic isolation of seagrass beds as a long-term management goal to maintain genetic structure of local seagrass systems. But no gene complex can provide protection against insufficient light, excessive nutrient loading, or the depredations of bioturbating organisms in a recently planted bed.

Site Surveys Prior to Impact

It is important to obtain information about seagrass distribution and the environmental conditions at a site before that site is allowed to be disturbed. If sites are illegally injured prior to being assessed, extent of damage is especially difficult to assess. Site surveys are a recommended tool but they provide inadequate information if sites are surveyed at only one point in time. This is especially true when dealing with patchy seagrass beds. It is important that managers realize that bare areas among patchy seagrass beds are a natural characteristic of these beds and that over time seagrasses will move and alternately colonize and vacate these areas. If channels are placed in these beds in such a way that they intercept bed migration, unanticipated and persistent losses of seagrass habitat may occur. If possible, present-day beds should be evaluated over a sufficient period of time and with appropriate spatial resolution to reveal seagrass movement into bare areas and identify currently unvegetated areas that should be protected from negative impacts.

Identify Project Goals

Early in the planning phase, the project manager must determine whether the project will be for compensatory mitigation or for restoration. These projects could have different goals and may be evaluated according to different performance criteria by resource agencies. In any case, attaining the same seagrass species as what was lost with a comparable shoot density and equal or greater area of bottom covered (depending on time since injury and recovery potential) that compensates for interim lost services is a logical and ecologically defensible goal.

Permit Coordination Protocols

Because different agencies at the state, federal, county, and municipal levels may have jurisdiction over projects affecting wetlands or seagrass beds, delays can be avoided by addressing all permitting requirements as far ahead of planting as possible. Coordination protocols developed for Southern California, the Chesapeake Bay, and Connecticut provide guidance. A standardized protocol is essential to accurately convey the scope of the potential injury to the public stewards and to simultaneously treat applicants in a consistent and fair manner.

Interim Loss Assessment

It is essential to profile the injury area and determine the interim loss of ecosystem functions. This determination considers how much acreage will be lost and how long it will take to replace the ecosystem services that this area provides. In the past, interim loss assessment has been inconsistent, with highly variable replacement ratios. NOAA's Damage Assessment and Restoration Program is now utilizing an economically-based model to standardize interim loss computations using discounting methods and acre-years of lost services as a metric. A qualitative description of the model is provided (p. 66 and Appendix E).

Site Surveys

Guidelines for pre-injury and pre-planting surveys allow managers to quantitatively profile seagrass habitat. Surveys can identify species composition, distribution, and availability of seagrass to salvage and which could then be set aside for planting to other sites or replanting to the original site in the case of short-term disturbance. Aerial photographs can establish the historical perspective on the persistence and distribution of coverage on the site. Pitfalls to seagrass habitat replacement over the long term include transplanting into unsuitable areas, or into bare areas between established seagrass patches. If aerial photographs or other surveys indicate no history of seagrass cover over a ten-year period, then the planting site should be rejected as unsuitable, unless some specific actions are taken to improve the site, or, unless mitigating factors such as recently improved water quality can be demonstrated.

Site Selection

Site selection is the single most important step in the seagrass restoration and mitigation process. Important aspects of site selection and seagrass physiology include the following: emersion and desiccation effects; bioturbation; sediment thickness; sediment stability; natural recolonization; nutrient limitation or overload; light requirements and light attenuation characteristics of the site; salinity and temperature tolerances; and waves and current speed (see site selection criteria in Appendix E).

Planting areas are classified as either on-site or off-site. In some cases grading down of upland areas or engineering subtidal areas to create suitable sites may be possible. When destruction of the site requires planting in another location it is often *very difficult to find a suitable off-impact site location.* The seemingly simplistic question that must first be asked is "If seagrass does not grow there now what makes you believe it can be successfully established?" (Fredette et al. 1985).

Obtaining Transplant Stock

Most planting projects currently utilize wild planting stock which almost always requires a permit to collect. Managers are cautioned against repeated harvests from donor sites. Matching the environmental conditions of the donor site to the planting site remains, after 50 years (Addy 1947), the best rule of thumb for donor stock selection. In terms of long-term management, some planted beds should be created solely to provide donor stock and experimental beds no longer being studied could be made available for harvest.

Successful planting of seagrasses demands that: (1) planting units have intact meristems so that they can spread vegetatively, (2) they have enough short shoots per long shoot to facilitate growth following planting, and (3) minimization of stress to planting units so that they are healthy when planted. For seagrasses to undergo vegetative spread, they must have at least one apical meristem on a rhizome in each planting unit. Greater numbers of rhizome meristems is preferable. Spread of planting units will also be enhanced if they have several short shoots per long shoot. Minimizing stresses experienced by planting units will reduce the possibility of rhizome meristems being killed, and will facilitate more rapid establishment of transplants. To achieve this, plants need to be collected and planted soon thereafter, preferably on the same day, kept in seawater that is of ambient temperature and salinity, and not crowded or piled on each other in ways that cause bruising or breakage.

A common cause of planting failure is inexperience of persons involved in the project. Persons involved in the project need to be able to identify the species to be planted, be familiar with the handling and planting methods, and, in some cases, be comfortable snorkeling or SCUBA diving. Planting starts with selecting an appropriate area and marking it with poles or buoys so that its boundaries are visible. Waders, snorkelers, or SCUBA divers then begin planting, unless remote methods are used. As diving often increases costs considerably, it may be advisable to have workers pre-place planting units so that underwater time can be used most effectively. Previous efforts have shown that volunteers often lose interest in planting because it becomes tedious and repetitive following the brief learning period; paid staff may be more cost effective, but close attention to providing challenge and diversity in tasks is recommended.

Planting Methods

Planting can be conducted using any of several methods. The *plug method* involves driving 4-6 inch diameter PVC tubes into established seagrass beds, capping these tubes to create a vacuum that allows removal of the tube and its contents, and then transplanting this plug of seagrass, rhizomes, and sediment into a new habitat. Although this method tends to be more expensive than others, it has been extensively used with numerous species with good results. The *staple method* involves digging up plants and their associated rhizomes using a shovel, shaking sediment from the rhizomes, using twist-ties to attach rhizomes to metal, bamboo, or wooden staples, and planting these seagrasses by pushing the staples into sediments so that blades protrude upward and rhizomes are buried in the sediments. In calm areas, staples can be placed over groups of plants without securing the plants with twist-ties. This method is cost effective, widely used, and generally successful. The *peat pot method* has been used less than the above methods, but shows promise. A sod plugger is used to extract 3x3" plugs from an existing seagrass bed. These are immediately extruded into similar sized peat pots and the peat pots then transplanted into areas that are to be established. Once in the bottom, the sides of the peat pots are ripped to facilitate spread of the rhizomes. This method currently has the lowest cost per planting unit. Plants can also be collected with a shovel and plants with sediment shaped by hand into a peatpot sized mass and put into the pot for planting. *Other methods*, including sowing seeds, have also been tried. Some of these show promise and may be desirable for particular habitats or situations; however, most other methods have been used less extensively and are less well tested. Several investigators have attempted to improve planting success by adding fertilizers. These efforts have produced mixed results. At present, it appears that fertilization, and potentially, hormone treatment, cannot hurt and may improve planting success, especially phosphorus fertilization in carbonate sediments.

There is a considerable literature on how the spacing of transplants affects coalescence rates, potential disturbance in habitats subjected to different flow regimes, etc. In general, a balance will have to be achieved between desired coverage, rate of coverage, and the cost of planting at different densities or using different arrangements. Determining spacing requires knowledge of the natural history and physiology of the seagrasses being planted and an understanding of the hydrodynamics affecting the planting site. However, decreasing spacing may reduce bioturbation.

Evaluating Project Success

Seagrasses are planted in hopes of restoring all aspects of ecosystem function (sediment stabilization, nutrient cycling, etc.) that were lost when natural beds were injured. However, management resources are rarely available for monitoring planted beds to be sure that they each recover these functions. Although numerous criteria have been used for evaluating planting success, studies available to date indicate that simple measures of seagrass coverage and persistence are the most parsimonious indicators of a functioning seagrass bed, and are the measures that should be favored by resource managers. Therefore, successful seagrass establishment should be defined as beds that persist, unaided, at, or above, the desired acreage with comparable shoot density for a period of five years following planting, or in the case of slow-spreading species, on a trajectory for reaching the target acreage in a specified time. Use of Habitat Equivalency Analysis is strongly recommended to help identify and utlize realistic recovery horizons.

Monitoring Planted Beds

Monitoring planted beds is necessary to: (1) ensure that contracted work was performed to specifications, (2) allow for mid-course corrections, and (3) improve planning of subsequent projects. Adequate monitoring will involve determination of percent survival of planting units, the areal coverage of each planting unit, and the number of shoots per planting unit. For small plantings, these measures may be taken on each planting unit. For larger plantings, monitoring will need to be conducted using numerous randomly (as opposed to arbitrarily) located sites within the area that was planted. Specific recommendations on making these measurements, converting measurements into the most useful form, appropriate sizes of quadrats, etc. are provided. Monitoring should occur at least quarterly during the first year following planting, and biannually for at least four years after this (i.e., for a minimum of five years). If replanting is necessary, this sets the five-year-clock back to zero for the area that is replanted. This five-year rule may need to be extended in situations where seagrasses spread very slowly. If two replantings following the initial planting fail to establish a successful grass bed, then managers should abandon these failed sites or portions thereof and find areas more suitable.

Interpreting Results

The bottom-line is that the *target acreage* must *persist* with a comparable shoot density for an adequate period of time to assure that the planted seagrasses are well established and likely to provide the desired ecosystem functions. Although various percent survivorship criteria previously have been used to define planting success, these criteria may miss the point that it is not percent survivorship alone, but coverage and persistence that are the critical components of establishing adequate seagrass systems, especially since the ultimate metric of success is generating acre-years of seagrass service (i.e., to offset interim lost resource services).

Cost Estimates

Costs of successfully establishing seagrass beds vary from a few thousand to many thousands of dollars per hectare depending on site selection, experience of workers and managers, extent and rate of subsequent disturbance, water clarity and depth (i.e., light availability and quality), temperature, extent of monitoring following the plantings, and numerous similar factors. Published costs for planting range from about $25,000 to $50,000 per hectare, with an average of about $37,000. These general estimates likely greatly underestimate the cost of a particular project because of the particular concerns associated with each site, the size of the site, the coverage that needs to be achieved, the species of seagrass involved, special logistic costs, monitoring, and profit; the latter frequently overlooked! Values in the range of $200K per *acre* may be more reasonable over the life of the entire project.

Conservation, Mitigation, and Restoration

Despite proven techniques, the success rate of permit-linked mitigation projects remains low overall. There is continuing difficulty in translating mitigation concepts into legal principles, regulatory standards, and permit conditions that are scientifically defensible and sound. To prevent continued loss of seagrass habitat under compensatory mitigation, decisive action must be taken by placing emphasis on improving compliance, generating desired acreages, and maintaining a true baseline.

Seagrass planting is not an experimental technique. Seagrass beds can be restored but preservation is the most cost-effective course of action to sustain seagrass resources. Planting for mitigation should be treated as the last practicable alternative. There must be communication and coordination of efforts between agencies and those that would alter seagrass habitat. Seagrass beds have been recognized as a valuable resource essential to the health and function of coastal waters, and greater awareness and public education is necessary for conservation of this resource. The problems of restoring seagrass beds are largely those of appropriate site selection, plant demography, care in planting, and subsequent disturbance.

Seagrass habitat conservation must become a national focus because loss can occur rapidly when conditions are altered and because recovery occurs at a much slower rate. If an area is already stressed due to diminished water quality from point and non-point source runoff, addition of a new channel or increased boat traffic to a new marina may push the nearby seagrass population beyond its physiological limits. Once the habitat is lost, turbidity from nonstabilized sediments may make restoration impossible, with concomitant additional reduction in water quality.

As more information is made available to managers regarding the function of seagrass ecosystems and the costs involved in mitigating for their loss, fewer permitted impacts are occurring with more emphasis placed on impact avoidance and minimization. Our ability to wisely manage, conserve, or restore these productive ecosystems is limited due to our fragmentary understanding of seagrass ecology and distribution and inconsistent application of available technology. Managing seagrass systems requires that managers understand some basics of seagrass ecology and having historical perspective regarding the particular seagrass beds being affected by their management decisions. Why place such a priority on conservation if mitigation is no longer experimental? Although techniques and protocols exist that produce persistent seagrass beds, they are applied inconsistently, and have resulted in large-scale failures. Key issues to protect existing seagrass habitat include improved wastewater treatment, surface run-off control (i.e., watershed management), restrictions on certain shellfish and fish harvest methods, control of boat traffic, and public education.

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List of Appendices

APPENDIX A
Glossary

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allele(s): any of the different forms of a gene that occupy the same locus on homologous chromosomes (the latter being the chromosomes that pair during meiosis)

allozyme: one of several forms of an enzyme which is coded for by different alleles at a given locus on a chromosome; contrast with isozyme

apical: as in apical meristem; the distal unit of a seagrass plant that gives rise to new plant compenents, leading to vegetative expansion

arbitrary sample: a sample taken without regard to potential bias in the location, quality or quantity of the variable being sampled

areal coverage: coverage of the sea floor by seagrass expressed on a unit area basis

asymptote(s): portion of a graphed curve that levels out with respect to the independent variable

baseline acreage: the amount of habitat acreage at some past time that is used as a reference for computing subsequent changes in habitat abundance

Beer's Law: a mathematical expression that describes the exponential decay of light quantity with increasing water depth as a function of turbidity

bioturbation: the physical disruption of the sea floor and/or seagrass bed by the activity of any number of animals (e.g., rays, crabs, fish).

clone: here, an assemblage of seagrass shoots formed through asexual reproduction, arising from a sexually-reproduced individual

compensatory mitigation: the establishment of a wetland area for the purposes of offsetting a permitted loss of a like wetland

compliance: the degree to which stated project goals are attained

continuous cover: a seagrass bed with little or no open areas of unvegetated sea floor

coverage rate: the rate at which planting units colonize the sea floor expressed on a unit area basis per unit time

creation: in reference to wetlands, the conversion of persistent non-wetland area into a wetland, contingent upon the status of the non-wetland area having been persistent through 100–200 years

cultivated seagrass: seagrass plants that are generated under any one of several anthropogenically-mediated techniques (e.g., tissue culture, micropropagation)

donor bed: an existing seagrass bed from which transplant material is harvested for planting elsewhere

dredge and fill: the act of dredging or filling of a habitat, particularly in reference to the management of this activity under sec. 404 of the Clean Water Act and Section 10 of the Rivers and Harbors Act

enhancement: the increase in one or more values of all or a portion of an existing wetland by man's activities, often with the accompanying decline in other wetland values

erosion scarp: a point of erosion in a seagrass bed that results in vertical relief and often exposes seagrass roots and rhizomes.

genet: a group (here seagrass shoots) derived by asexual reproduction from a single original zygote such as a seedling or isolated shoot (usually with a rhizome apical where this is required for clonal expansion)

genotype: the heritable genetic constitution of an individual

growth strategies: the rate at which individual species of seagrass reproduce by either sexual (seed production) or asexual means (tillering of rhizomes across the bottom accompanied by vegetative production of new short shoots)

habitat: an unspecified spatial scale which has physical, chemical and biological attributes conducive to the maintenance and propagation of biota

habitat functions: services provided to the ecosystem by a given habitat type (e.g., shelter, stability, refuge, nursery)

impact avoidance: avoidance of any alteration of an existing wetland

impact site: a site containing jurisdictional wetlands which is being, or is going to be altered by anthropogenic actions

in-kind: planting a wetland species that is the same as the one that was damaged

isozyme: one of several forms of an enzyme that is produced by different (nonallelic) loci; contrast with allozyme.

jurisdictional wetlands: wetlands under the management jurisdiction of a regulatory agency

long shoot: a collection of short shoots physically located on the same rhizome

lower depth limit: the depth to which seagrass can grow which is determined by the amount and possibly the quality of available light; i.e., compensation depth

meristem: as in apical meristem; the portion of a plant that contains tissue which divides and gives rise to similar cells and/or plant structures (e.g., tissues, organs, rhizome, roots, leaves)

metapopulation: a group of conspecific populations co-occurring in time, but not in space

minimization: minimizing the degree of alteration of an existing wetland by modification of a project plan

mitigation: the actual restoration, creation, or enhancement of wetlands to compensate for permitted wetland losses

monitoring: collection of habitat attributes (e.g. depth, cover, species composition or planted seagrass growth) relative to assessment of site conditions, planting site suitability, or planting performance.

no net loss: a quantitative evaluation which compares habitat area replaced or conserved with the habitat area lost

off-site: planting of a wetland as some form of mitigation at a location not in immediate proximity to the physical location of the damaged wetland for which it is to compensate

on-site: planting a wetland on an area which has suffered a loss of a wetland habitat

out-of-kind: planting a wetland species that is not the same species as the one that was damaged or lost

PAR (photosynthetically active radiation): wavelengths of light between approximately 400 to 700 nanometers

patchy distribution: seagrass beds (areas where rhizomes overlap) and associated unvegetated bottom; either distinct, isolated patches of seagrass in a predominantly unvegetated sea floor or meandering patterns of unvegetated bottom in a predominantly vegetated area

peat pot: in reference to a seagrass transplanting technique where plugs of seagrass are removed and placed into commercially available, small cups constructed of compressed peat; the plug and peat pot container are then planted in the sea floor

permitting agency: a resource management agency (e.g., state, federal) that has statutory authority for issuing or commenting on permits dealing with wetlands modifications

phenotypic plasticity: the capacity for marked changes in a phenotype as the result of environmental influences upon the genotype during development

pioneering species: a species of seagrass with a growth strategy than enables it to rapidly colonize unvegetated sea floor

planting performance: attributes of a planted area which can be used as indicators of project success; e.g., planting unit survival, planting unit population growth and coverage rate.

planting ratio: the ratio of planted, and eventually, persistent seagrass acreage to the amount of acreage lost in a given project

planting unit(s): an individual core, plug, staple, peat pot, sod, etc., and the associated plant material used in a planting operation

plugs: in reference to a seagrass planting technique where hollow tubes are used as a coring device into a seagrass bed, thereby harvesting the sediment "plug" in the tube with the associated seagrass

propagule: any portion of a seagrass plant that is capable of colonizing a new site after becoming detached from an existing plant (e.g., seeds, plant fragment with leaves and roots)

propeller scarring: typically a long, linear furrow excavated in the bottom as the result of operating vessels in water depths shallower than the draft of the drive unit (propeller); results in excavation of seagrass

PU: planting unit

ramet: a member of a clone, assumed to be of identical genetic composition as the rest of the clone, and that will continue to survive if separated from the clone

random sample: a sample taken such that each sample unit has an equal (unbiased) probability of being selected

restoration: returned from a disturbed or totally altered condition to a previously existing natural, or altered condition by some action of man; refers to the return of a pre-existing condition

rhizome: sensu Websters' Collegiate Dictionary: a somewhat elongated, usually horizontal plant stem which produces shoots above and roots below and is distinguished from roots in possessing buds, nodes and scale like leaves

rhizome apical: the meristematic region at the terminus of a long shoot that gives rise to further rhizome growth and differentiates to give rise to short shoots

rhizosphere: portions of the sediment occupied by the roots and rhizomes of seagrass

salvage operation: transplanting seagrass from an area where activities are planned which will destroy that seagrass

Section 404 permit process: references section 404 of the Clean Water Act which provides the statutory authority to specific federal agencies regarding dredge and fill activities in the waters of the United States

sediment resuspension: the transfer of sediment from a resting position on the sea floor to the water column as the result of some external action such as wind waves, tidal currents, or a vessels' propeller(s)

short shoot: an individual meristem located on a long shoot which produces leaves and roots

site survey: a quantitative assessment of the amount of plant material to be disturbed, its distribution and the environmental conditions at impact and planting sites prior to initiating a project (see also monitoring)

slow release fertilizer: a fertilizer which releases nutrients over ime at a given temperature

staple: in reference to a seagrass planting technique where plants are washed free of sediment and typically are attached to a U-shaped metal bar (staple) which is then inserted points down into the sediment, pinning the seagrass to the sea floor

success (planting): although the definition of this term may be changed with the goals of the project at hand, a broadly applicable form is as follows: the unassisted persistence of designated seagrass coverage for a prescribed period of time (suggested minimum of 5 years)

turbidity: the degree of opacity of the water column as a result of dissolved and suspended material in the water column

unassisted persistence: seagrass beds maintained by natural recruitment and have not been assisted by any deliberate anthropogenic manipulation

unvegetated sea floor: the portion of the estuarine floor which is not colonized by rooted submerged aquatic vegetation

water transparency: a measure of the degree of water clarity

wild stock (stands): naturally-occurring seagrass beds

APPENDIX B

Seagrass Species Characteristics

(Alphabetical order by scientific name; modified from Fonseca 1994; for greater taxonomic detail see den Hartog 1970, Phillips and Menez 1988)

Halophila decipiens: An extremely fragile seagrass that is often confused with some rhizophytic algae, such as *Caulerpa* spp. The leaf blades are 10 to 25 mm long and ~5 mm wide and occur in pairs at each rhizome internode (one root per node). The blade margin is finely serrated and the end of the blade is rounded. Blades are ~1 cell thick and translucent. Rhizomes are typically 1 mm in diameter. The entire plant community can easily be dislodged from the sediment although because of it's high fecundity and rapid rhizome extension rate (Josselyn et al. 1986) it is a ruderal or opportunistic species. Plants cannot tolerate burial and will disintegrate beyond recognition in 24 h if buried, making the carbon and other nutrients fixed by *Halophila* the most labile of any seagrass. Although diminutive, this seagrass genus can significantly enhance sediment stabilization (Fonseca 1989b). In general, this genus can withstand very low light conditions (~1% of insolation as opposed to ~20% for other genera) and occurs at great depths (>40 m; Josselyn et al. 1986, Kenworthy et al. 1989), or in shallow turbid water, under docks or as an understory to other seagrass species.

Halophila engelmanni: Similar in an ecological context to *H. decipiens* but rather than having leaf pairs at each rhizome node, there is one stem per rhizome node with upwards of 7 leaves on each stem. Blades are more elongated and pointy than *H. decipiens.*

Halophila johnsonni (threatened species): Also similar in an ecological context to *H. decipiens* and also has a leaf pair at each rhizome node. Blades are more elongated and pointy than *H. decipiens* and veins sweep upwards from the center line of each leaf at nearly 45 degrees, but blade margins are not serrated.

Halophila hawaiiana: Again, similar in an ecological context to *H. decipiens* and also has a leaf pair at each rhizome node, Blades are more elongated and pointy than *H. decipiens* although unlike *H. johnsonii*, leaf tips are more rounded.

Halophila ovalis:Very similar in appearance to *H. decipiens* except *H. ovalis* has 10-25 leaf veins whereas *H. decipiens* only 6-9 leaf veins.

Halophila minor: Smaller than *H. decipiens* with a maximum length of 14 mm and has only 3-8 pairs of leaf veins.

***Halodule wrightii* (contrast with *Ruppia maritima*):** This species was once classified under the genus *Diplanthera.* References from earlier than 1975 often refer to this species by that name. It has a lower depth limit equal to turtle grass and manatee grass. It also can occur in very shallow water and it is noted for its relative tolerance to desiccation once rooted. It often forms large pancake-like patches reaching 30 m diameter) or extensive meadows on shallow shoals and flats, experiencing regular exposure at low tides (the basis for its common name). The fine (1-3 mm width) blades occur in groups of two to four on a shoot and vary in length according to depth as does turtle grass. Blade lengths range from as small as 5 cm to over 40 cm. This species forms very dense beds, with upwards of 5000 shoots per m^2 (although 11000 can occur; pers. obs). Flowers are difficult to locate as they occur on the base of the shoots near the sediment surface. Rhizomes are fairly shallow, rarely being deeper than 5 cm, although roots may extend for 25 cm or more. Rhizomes may extend into the water column with attached short shoots which appears to be a form of vegetative propagation. These rhizomes may be easily harvested and are efficiently transplanted with the staple method.

This species can easily be confused with widgeon grass. Four visual clues separate them: (1) widgeon grass produces extensive flowering stalks often reaching a meter in length, with numerous seed clusters resembling miniature rattlesnake rattles while flowers in shoalgrass are rarely seen; (2) the blade tip of shoalgrass forms a miniature three-point crown, with the two leaf margins and central vein of the leaf forming the points. Widgeon grass blades taper to a single sharp point; (3) shoalgrass rhizomes are usually very straight and white often somewhat zig-zagged when viewed from above and may be green or white; and 4) shoalgrass has two roots per node on the rhizome whereas widgeon grass has one root per node.

Phyllospadix scouleri: Found on rocky substrate and appears much like *Zostera*; blades can be 2 meter long but compared with *Zostera*, are much narrower, only up to 4 mm wide. Rhizomes can be covered with many fibers. Abundant, but not limited to, north of Monterey, California.

Phyllospadix serralatus: Found on rocky substrate primarily north of southern Oregon. Leaf margins show fine serrations.

Phyllospadix torreyi: Also found on rocky substrate but deeper than the other *Phyllospadix* spp.; abundant but not limited to south of Monterey, California. Blades are different from the others of this genus in that they emerge from a sheath that is wider than the blade itself with only one blade per sheath.

***Ruppia maritima* (contrast with *Halodule wrightii*):** This species is a favorite food of migratory waterfowl, a fact on which its common name is based. This species does not usually form a rhizome mat as dense as that of shoalgrass, but does much to stabilize the bottom. This species is set apart from all other seagrasses in that it can grow in both fresh water and hypersaline conditions (> 70 ppt). See shoalgrass for further description and contrast.

Syringodium filiforme: This species is easily distinguished from all the other seagrass species. It is nearly cylindrical. Its long erect blades are ~ 2-3 mm in diameter and there are usually only two leaves per shoot. These beds often accumulate a large under story of unattached macroalgae. The rhizome system varies in depth, between 1-10 cm. Flowering produces extensive branching which extends up into the water column, similar to widgeon grass grass but not as extravagant. Rhizomes may extend into the water column with attached shoots as described for shoalgrass, again presumably as a means of producing vegetative (as opposed to seed) propagules. As with shoalgrass, these propagules make excellent transplanting stock with no app

Thalassia testudinum: This species is one of the most well-known seagrasses in the subtropical regions of the U.S. It is a favorite food of the endangered green sea turtle, hence its common name. Its broad (often > 1 cm wide) deep green, strap-like blades (usually three to a plant but may often have upwards of five) cannot easily be mistaken for any other marine submerged aquatic macrophyte. Leaf length of the plants depends on water depth (as is the case with most seagrasses) and varies from ~10 to 75 cm. The leaves emerge from the sediment at the top of a vertical rhizome which rarely protrudes above the sediment surface. The thick, fibrous rhizomes from which the individual shoots originate are often located in excess of 20 cm into the sediment. This species develops flowers which emerge from the sediment next to the short shoot. Once fertilized, a round seed the size of a small acorn will be produced.

Seeds have been successfully used in planting projects. This species is noted for its longevity (sometimes > 10 yr. for an individual shoot) and the dense, extensive stands.

Zostera asiatica: Similar in appearance to *Z. marina* except that there are many more roots per rhizome node and sometimes emerge from the side of a short shoot. Multiple sheathes can occur on a single shoot which does not occur in *Z. marina*.

Zostera marina: The most common temperate seagrass species. Tremendous variation in size is reported from 30 cm to over 200 cm long. This species occurs on both coasts of the U.S. The rhizome is brown with roots located only at the rhizome nodes. A single shoot occurs at the end of each rhizome. A sheath encompasses 3-5 strap-shaped leaves. The leaf tip is rounded, sometimes with a very small point at the apex.

Zostera japonica: Appearing to be a smaller version of *Zostera marina*, this recently introduced species occurs as a *Z. marina* under story or in intertidal areas not colonized by *Z. marina*. Leaves are narrow, rarely wider than 15 mm and only ~30 cm long. Also unlike *Z. marina*, the blade tips may be asymmetrical or have longer margins than the center (like *Halodule*).

APPENDIX C

Partial List of Equipment

This Appendix was modified from Fonseca 1994 and Merkel 1992; see Merkel for a more detailed list and integrated discussion of how to implement that list using his field methods.

Depending on the location of the sites, access by boat may be required. Local knowledge of wind, tide and navigational hazards should be obtained prior to operations. A prominent listing of emergency numbers and emergency procedures should be worked out in advance, with particular attention to the needs of SCUBA divers. Reliable, seaworthy vessels which can work in a range of sea conditions and water depths should be procure. More than one vessel type might be required. If you are not fully knowledgeable in these areas and do not posses basic training in navigation and seamanship, retain trained personnel as boat operators, divers, etc. Precautions must be taken against injury during heavy lifting which is typical for seagrass planting.

STAPLE METHOD

Paper coated twist ties (e.g., tomato plant tie-up material)

Dive knife (or similar tool for loosening the bottom to insert the staple)

Mesh float buckets (for holding plants washed free of sediment)

Site markers (stakes, buoys, etc. 3/4" diameter thin wall electrical conduit is relatively inexpensive, comes in 10' lengths and can be easily driven into the sediment although it must be cleaned out between uses). Conduit is difficult to see and should be marked with reflectors. When possible, white PVC pipe should be used.

Waterproof tape measures (100 m variety)

Lead core lines with ribbons on plating intervals if precise spacing is desired or visibility is so poor that a means of orientation is required at depth (line is manufactured for gill nets, survey ribbon must be added)
or

Polypropylene line with ribbons may be floated on the surface as a planting guideline for surface-oriented (non-diving) operations. Buoys are also helpful for orientation as are 3/4" diameter thin wall electrical conduit poles, over which PVC pipe may be slid to mark boundaries. We recommend the conduit because it cuts into the bottom, is more stable in the bottom than re-bar and rusts less (make sure the conduit is cleaned out between uses; fill the conduit with water and shake vertically or tap on a hard substrate - the pressure of the water will eventually force out the sediment in the conduit).

Snorkeling or SCUBA equipment (certified divers only).

If SCUBA diving is required, develop and rehearse a Dive Accident Management Plan. Follow emergency procedures as recommended by recognized safety groups such as the Divers Alert Network (D.A.N.). An ample supply of SCUBA tanks, extra weights (divers work better negatively buoyant here). Dive Flags should be plentiful and obvious.

Tide tables and updated weather forecast.

First aid kit including sun screen and insect repellent and, for divers, an emergency O_2 kit.

Redundant communications equipment.

Appropriate clothing for exposure cannot be overstated. Equipment such as wet suits, wool clothing and foul weather gear which can be worn in the water as well as a wind breaker. Waders may be preferred by some people but as seagrass planting requires much bending over, it is not unusual to overtop waders.

Warm or cold fluids (depending on season), fresh water, and high energy foods.

Polarized sun glasses (enhances visual penetration of the surface).

Peat Pot Method

All of the same operational equipment except for the first four items. These should be replaced by the following:

Peat pots (3" square)

Plugger (same size as peat pot)

Tree planting bar

PVC (or equivalent) float collars to support ~ 30 peat pots in a tray.

Durable plastic trays to contain ~ 30 peat pots.

Heavy (~ 10 ga.) wire mesh to fit over peat pots in tray to prevent them from floating out as any air pockets are displaced by water.

Core Tube Method

All of the equipment for staples except for the first four items. Replace with sufficient core tubes so as to fully utilize the capacity of the transport vehicle.

Surveys

Transit
Meter sticks
Waterproof tape measures (100 m variety)
Random number table
Writing tablets (photocopiable underwater paper on clipboards with prepared data collection sheets are useful). Pencils or grease pencils tethered to the tablet with a generous length of surgical tubing is inexpensive.

(OPTIONAL - Depending on survey technique)

Quadrat: 1 x 1 m 1" PVC sand-filled frame with parachute cord (thin braided nylon line) on 25 cm intersections.

APPENDIX D

Suggested Minimum Components of Proposals and Reports

(modified from Fonseca 1994).

A. Mitigation Proposal

1. Identification of goals
   a. compensatory mitigation or restoration
   b. specify replacement ratio and final acreage

2. Description of impact site survey methodology

3. Site selection criteria and list of sites

4. Location and availability of donor material/demonstration of appropriate collection permits

5. Planting methodology

6. Spacing and spatial arrangement on site

7. Monitoring specifications
   a. identification of variables and methods of collection
   b. monitoring and reporting frequency and duration (suggest minimums of 4 times in year 1, 2 times in year 2, and annually thereafter; this frequency allows implementation of Item 8.
   c. monitoring interpretation criteria

8. Specify criteria for remedial planting

9. Specify criteria for success (e.g., acreage of seagrass cover to be generated [1b], species, duration of unassisted persistence).

10. Specify duration of responsibility and consequences of non-compliance with Items 1–9.

B. Time Zero Report

1. Results of impact site survey and statistical relevance of the survey methodology.

2. Documentation of implementation as compared to the descriptions of Items 1–6 above.

C. Progress Reports

1. Results of monitoring as described in Section A, Item 7, above.

2. Identify and document any remedial action taken

3. Provide best professional estimate of likelihood of meeting Item 9, Section A.

D. Final Project Report

1. To improve subsequent projects, review operational errors/shortcomings in the context of the original work statement.

2. Identify and document compliance with all stated requirements, with particular attention to Section A, Items 1, 7b, 8, 9, and 10.

APPENDIX E

Example Propeller and Mooring Scar Restoration Plan

I. BACKGROUND

The appropriate scale for the restoration should be determined using Habitat Equivalency Analysis (HEA) methodology (Attachment I).

II. RESTORATION APPROACH

The restoration of seagrass scars created by vessel impacts represents is a viable approach to off-site restoration. Restoration efforts will should on seagrass transplanting of scars in heavily injured *Thalassia testudinum* (turtlegrass) seagrass meadows such as described by Sargent et al. (1995) within the Florida Keys. Seagrass beds can be scarred by many activities, but scars are most commonly made when a vessel is moored and the ground tackle gouges the bed or the vessel operates in areas vegetated by seagrasses that are too shallow for the vessel to avoid contact with the seafloor. The vessel's hull and/or propeller tears and cuts up the leaves, stems and roots of the seagrasses, typically leaving long, narrow, trench-like furrows devoid of seagrass. A typical prop scar created by a small vessel (less than 6.5 m in length) is approximately 0.25-0.5 m wide and 0.1-0.5 m deep. Larger vessels with twin propellers or inboard engines (greater than 6.5 m in length) can produce deeper (0.25-0.75 m) and wider trenches (0.5-1.5 m). While smaller scars may naturally recolonize over several years, some scars, especially in *Thalassia* seagrass beds experiencing moderate tidal currents or wave action, persist for decades and can enlarge from erosion (Zieman, 1976). The slow growth rate of *Thalassia* contributes to its comparatively slow recolonization of the bare sediments in scars.

One technique for restoring slow-growing seagrass species such as *Thalassia* focuses on planting another seagrass species such as shoalgrass, *Halodule wrightii*

(*Halodule*) to achieve a "compressed succession" (Durako et al., 1992; sensu Fonseca, 1994). The compressed succession is a planting technique intended to achieve a more rapid rate of seagrass recovery by temporarily substituting the faster growing *Halodule* for the slower growing *Thalassia*. This sequence promotes more suitable conditions for *Thalassia* to recolonize the scar while stabilizing the sediment and establishing functional seagrass habitat.

III. RESTORATION SITE SELECTION

There are, however, a number of management decisions that can be made within the permit process to ameliorate a loss in habitat and better approaches the goal of no net habitat loss. Mitigation in its broader definition typically also includes impact avoidance and minimization (the latter term unfortunately implying an acceptable net loss of acreage). In practice, avoidance and minimization are sometimes difficult to achieve. The existence of techniques to transplant seagrass has often been used to justify the destruction of existing, productive habitat but this approach has consistently produced a net loss of habitat. This net loss of habitat occurs for a number of reasons: one reason is that because the permit-associated activities that destroy seagrass beds in the first place typically are long lasting (i.e., creation of channels, bridges, bulkheads) and do not allow enough area for on-site planting to offset the loss of habitat.

If planting is considered at a location not on the original impact site (off-site compensatory mitigation), that site would preferably not be an area that itself has lost seagrass to some other impact (i.e., if one permits a loss of seagrass for some form of coastal development [e.g. -1 acre] and plants an equivalent area [+1 acre] onto a site which had previously lost seagrass [a previous loss of -1 acre] but was not associated with the project at hand, then the net change in habitat is: [-1 + -1)] +1 = -1 acre; because only the repair of the original problem was addressed, the new, most recently impacted site then constitutes a net loss of local habitat).

Moreover, what if a site chosen for planting does not currently support seagrass? Selecting an appropriate planting site is the single most important step in the entire process. If an off-site planting area must be selected, whether it be for restoration or mitigation, it must pass a simple, but exacting, test: "If seagrass does not currently exist at the (chosen) site, what makes you believe it can be successfully established?" (Fredette et al. 1985). In the absence of site history information, one must then assume absence of seagrass indicates some inherent difficulty in colonization or persistence of seagrass. The events influencing the colonization process are sometimes difficult to document because they are often aperiodic, yet acute events (e.g., extreme low tides, storms, migrating rays excavating the bottom). Naturally unvegetated sea floor should not be substituted for vegetated bottom as this typically creates only a transient seagrass bed and alters, not necessarily improves, exiting habitat functions. There are few off-site compensatory mitigation sites that do not involve habitat substitution or can satisfy the no-net-loss goal.

Planting sites must meet (at least) the following criteria:

1) they are at similar depths as nearby natural seagrass beds;
2) they were anthropogenically disturbed;
3) they exist in areas that were not subject to chronic storm disruption;
4) they are not undergoing rapid and extensive natural recolonization by seagrasses;
5) seagrass restoration had been successful at similar sites;
6) there is sufficient acreage to conduct the project; and,
7) similar quality habitat would be restored as was lost.

In the case of scarred habitat, scarring can be an ongoing impact on seagrass meadows and restoration efforts should be conducted at locations that provide protective management, such as restrictions on power vessel operation where restoration is less likely to be disturbed by further scarring. Additional sites may be considered outside of such management areas where other site characteristics or circumstances exist which will minimize the threat of future injury from vessel groundings.

Preliminary site selection of scars should encompass the inspection of existing high resolution vertical aerial photography or detailed on-site mapping Low level vertical photos are required to quantitatively delineate areas. Photographs should be inspected, and scars identified and measured to calculate total area. If existing aerial photographs are not adequate, new aerial photography should be collected. Following preliminary selection, the sites should be verified for the presence of seagrass adjacent to the scar and for plantable unconsolidated sediments within the scar. Verification should be conducted by snorkel or SCUBA divers, depending upon water depth at the site. Plantable unconsolidated sediments in a scar should be medium to fine grain and at least 10 cm thick. Sediment thickness should be determined by inserting a probe into the sediment approximately every 5 m along the length of the scar. Scars should be targeted in areas that at the time of the survey appear to be susceptible to additional erosion and scar expansion, particularly as the result of disturbance caused by water motion (e.g., waves, tidal currents).

IV. METHODS

A. Planting Area, Site Marking and Site Preparation

We present here guidance for propellor scar restoration; planting broader areas (e.g. mooring gouged areas) would follow similar procedures. Using a conservative estimate for scar width of 0.5 meters, the planned acreage should require delineating X linear meters of scar (Area in m^2 / 0.5); this estimate is conservative given that narrower scars translates into a greater linear distance needed to restore. The location of each scar selected for planting should be established using a differentially corrected Global Positioning System (GPS). Each end of the selected scar should be identified with a permanent marker for positioning and the distances calculated in a Geographical Information System (GIS). Maps delineating the sites and the location of scars should be produced with GIS.

B. Planting Species and Technique

The selected scars should be planted with planting units of shoalgrass, *Halodule wrightii* to achieve a "compressed succession" (attributed to M. Moffler, Durako et al., 1992; *sensu* Fonseca, 1994). The compressed succession is a planting technique intended to achieve a more rapid rate of seagrass recovery by temporarily substituting the faster growing *Halodule* for the slower growing *Thalassia*. This sequence promotes more suitable conditions for *Thalassia* to recolonize the scar while stabilizing the sediment and establishing functional seagrass habitat.

C. Planting Methods

Planting should occur during April and May, months which present optimal environmental conditions for planting. The planting method to be used should use commercially available "peat pots" (Fonseca et al., 1994). Peat pots (one peat pot = one planting unit) are made of an organic, compressed peat material with a surface area of 7.6 cm^2 and approximately 7 cm deep. A sod plugger of the same dimensions as the peat pot is used to extract plugs from the donor seagrass bed which is then extruded into the peat pots (see Fonseca, 1994 and Fonseca et al, 1994 for detailed description of method). The donor beds should be located on shallow, sandy shoals where *Halodule* grows at densities of at least 3,000 shoots per m^2 yielding planting unit shoot densities of at least 17 shoots per planting unit. Donor plugs should be extracted at no less than 25 cm between plugs to minimize any effects on the donor beds.

Prior to extruding a plug of *Halodule*, approximately 10 grams of constant release (70 day) phosphorus fertilizer (0-39-0, nitrogen-phosphorus-potassium) or an equivalent form should be added to each peat pot. Phosphorus has been shown to be a highly limiting nutrient in carbonate sediments such as are found in the Florida Keys (Powell et al., 1989; Fourqurean et al., 1995). Plugs should then be planted at 1.0 m intervals in the scars selected for restoration.

V. SEAGRASS TRANSPLANT MONITORING

Monitoring of the restoration project is necessary to provide data required to evaluate the viability of the restoration project based on the performance standards identified in Section VI and to permit timely identification of problems or conditions that may require corrective action to ensure the success of the restoration project. Restoration monitoring herein should be in accordance with the following terms and specifications.

A. Monitoring schedule and activities

Field collection of data for performance monitoring should occur for four years after planting. Original plantings should be monitored for three years and potential remedial plantings in Year 2 should be monitored for three years for a total monitoring period of four years. Under this schedule the monitoring would be conducted as follows:

Year 1 - day 60, 180, 365
Year 2 - day 180, 365
Year 3 - day 180, 365
Year 4 - day 180, 365

The precise dates are weather dependent. At day 60 of Year 1, each surviving planting unit should receive an additional spike of constant release phosphorous fertilizer (0-39-0, nitrogen-phosphorus-potassium) to be delivered to each planting unit. Semi-annual refertilization of surviving *Halodule* planting units should be required at each planting unit. Alternatively, bird roosting stakes could be installed ~ every 5-10 m along scars; the roosting birds defacate into the water and fertilize the plantings with their phosphorous-rich defecant. This has been shown to be an extremely effective means of accelerating *Halodule* growth in shallow water and in prop scar restoration (Powell et al. 1989, Progress Report II, Prop Scar Restoration Pilot Study, December 1995). The drawback is that rows of stakes may be mistaken for navigational aids, drawing additional boaters to run alongside, further scarring the adjacent beds. Bird roosting stakes can only be used with appropriate navigational markings to discourage such action.

B. Data Collection

Monitoring should focus on documenting the numbers of apicals at planting time, planting unit survival, shoot density and areal coverage under the following schedule and definitions. This monitoring protocol applies to original plantings for three years (Year 1-3) and to remedial plantings under Section VII for three years (Year 2-4).

1. Apical counts

Prior to planting, one planting unit (i.e. peat pot core) out of every one hundred (100) collected should be examined for the number of rhizome apicals.

2. Survival

Each scar should be examined for survival of all planting units during each survey in Year 1 (60, 180 and 365 days) or until coalescence. Survival of each species should be expressed as a percentage of the original number, but the actual whole number should also be reported.

3. Shoot density

A separate (from survival) random selection of three (3) planting units of *Halodule* per scar or per 100 planted PU (whichever yield the higher sample size) should be assessed for number of shoots per planting unit at each survey time until coalescence begins. After some planting units begin to coalesce, 3 randomly selected locations per scar or per 100 m (100 PU) should be surveyed for shoot density over a 1 meter linear distance along each planted scar at 0.0625 m^2 (25 cm x 25 cm) resolution. Shoot density should be monitored for three (3) years.

4. Areal coverage

The randomly selected planting units (may be same as shoot density selection) should be surveyed for coverage at each survey time starting at day 180 of Year 1. Measurements should be taken at a 0.0025 m^2 (5 cm x 5 cm) resolution prior to coalescence and over a 1 meter linear distance along a scar at $0.0625m^2$ (25 cm x 25 cm) resolution after coalescence for each seagrass species present at each survey time. Areal coverage should be monitored for three (3) years.

5. Video Tape Transects

Five 100 meter transects along randomly selected portions of the planted scars should be video-tape recorded to establish permanent visual documentation of the progression of areal coverage of seagrass through time. A metric tape measure should be laid along the central (long) axis of the scar and should be included in the video tape to allow physical reference of locations within the scar. Video recordings should be taken at each survey time during the monitoring period of three (3) years. Observation-based assessment of success may be substituted for 3 and 4 above, if quadrats are used in accordance with a Braun-Blanquet survey method (see Attachment II) if the data are obtained from the video tape (making the observational data base available for cross-checking). The same number of sample points must be obtained with the same spatial extent (i.e., survey each scar). Similarly, Braun-Blanquet observations of cover at every meter along each scar may also be obtained from the video tape to obtain estimates of planting performance.

C. Reports

Monitoring reports (up to a total of 9) should include copies of raw data gathered in each survey, an analysis of the data, and a discussion of the analysis. Originals of all video tapes recorded since the previous report should be provided with each new report. Originals of all video tapes and other photography should be turned over to the permitting agency following project completion by the party conducting the monitoring.

VI. PERFORMANCE STANDARDS

Although it is the overall objective to restore *Thalassia* at the selected scar sites, performance criteria should be based on the success of the *Halodule* planting as the planting methodology used is designed to expedite the recovery of *Thalassia*.

A. Apicals

A minimum average of one horizontal rhizome apical per unit should be maintained in all original planting and remedial planting.

B. Survival

The survival rate shall be considered successful if a minimum of 75% of the planting units have established themselves by the end of Year 1. If it is determined that less than 75% survival has occurred by the end of Year 1, then remedial planting should occur during the next available planting period to bring the percentage survival rate to the minimum standard by the next monitoring survey.

C. Growth

The third success criteria should be the measured growth rate of bottom coverage. The growth rate should be considered successful if, starting after one year, the planted seagrass in the scars (restoration sites) is projected to achieve 1.55 acres of bottom coverage, with 95% statistical confidence, within the three year monitoring period for original plantings. If this criteria is not met, then remedial planting should occur during the next available planting period.

VII. REMEDIAL PLANTINGS AND/OR PROJECT MODIFICATIONS

If data from a monitoring report establishes that the performance standards described in Section V are not being met or are projected not to be met, remedial plantings of those affected seagrass species should occur. If there is a recurring problem with survival of plantings or replantings in a particular area, remedial planting should occur in another scar within the Sanctuary subject to the approval of permitting agencies.

Based on past experience in seagrass restoration efforts, it is assumed that 30% of the planted area should require remedial planting in Year 2. All original plantings should be monitored for three (3) years in accordance with Sections V and VI. Remedial plantings should be monitored for three (3) years subsequent to the date of a remedial planting.

VII. PROJECT PERMITTING

The seagrass restoration and monitoring outlined in this plan should be implemented consistent with any applicable state or federal permitting requirements. The format of the restoration and monitoring plan outlined in this document may be amended in order to comply with applicable permitting requirements.

IX. CONTRACTOR(S)

The permitting agecny should utilize the services of one or more qualified contractors to implement this restoration and monitoring plan.

X. PERMITTING AGENCY OVERSIGHT OF SEAGRASS RESTORATION PROJECT

The agency should oversee the implementation and monitoring of the seagrass restoration project in order to ensure its implementation in accordance with the terms of this plan. Costs which the agency should incur to provide effective oversight are part of the costs of implementing this seagrass restoration and monitoring plan. Activities which the agency may undertake in order to provide for effective implementation of this plan include, but are not limited to, the following:

a. actions associated with the identification, selection and hiring of any contractor(s) who should implement any part of this plan, including monitoring or remedial actions,

b. oversight of any field work at the project site, including remedial actions,

c. inspection of any completed field work, including remedial actions, to determine whether implementation is in accordance with this plan, including any applicable contract or permitting requirements,

d. review and evaluation of monitoring reports,

e. identification and direction of any actions needed to bring field work, including remedial actions, into compliance with standards for project performance identified in this plan,

f. actions to address NEPA and permitting processes, and

g. actions associated with final site selection.

XI. REFERENCES

Durako, M.J., M.O. Hall, F. Sargent, S. Peck. 1992. Propeller scars in seagrass beds: an assessment and experimental study of recolonization in Weedon Island State

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Powell, G.V.N., W.J. Kenworthy and J.W. Fourqurean. 1989. Experimental evidence for nutrient limitation of seagrass growth in a tropical estuary with restricted circulation. Bull. Mar. Sci. 44: 324-340.

Progress Report II, Prop Scar Restoration Pilot Study, December 1995.

Sargent, F.J., T.J. Leary, D.W. Crewz, and C.R. Kruer. 1995. Scarring of Florida's seagrasses: assessment and management options. Florida Marine Research Institute, Tech. Rep. TR-1.

Zieman, J.C. 1976. The ecological effects of physical damage from motorboats on turtle grass beds in Southern Florida. Aquat. Bot. 2: 127-139.

ATTACHMENT I. Assessment of Interim Losses

Another goal of many seagrass plantings is an attempt to recoup interim loss of ecosystem functions. This was mentioned earlier as an attribute of functional equivalency. Because the concept of success and functional equivalency are so closely tied, planning for successful restoration and/or mitigation requires early incorporation of interim loss considerations. The manner in which interim loss has been addressed historically has been through adjusting replacement ratios (how much acreage to plant per unit acreage lost). However, the manner in which interim ecosystem losses have been computed has not been consistent. Replacement ratios of less than 1:1 to as high as 5:1 have been proposed (pers. obs.), based on a number of criteria, but that ratio is usually inversely proportional to the degree which a project was in the public interest.

To compute losses though, requires some assessment of not only acreage lost, but how long a time the functions of that acreage was lost to the ecosystem at large before it was returned to pre- or unimpacted levels. Depending on how long one wishes to amortize a loss will influence how much replanting must be done. In theory, if one hectare of seagrass were destroyed today and three hectares were replanted tomorrow and, for argument sake, reached standards of equivalency in three years, then after those three years, the planting would have largely compensated for the total loss of production; the net loss of production over this three year period would be very low. However, it rarely works this way. First, it is very difficult to consistently locate and successfully create new seagrass habitat that meet our site selection criteria (which precludes simply substituting naturally unvegetated bottom for vegetated bottom). Finding large acreage for planting in close proximity to the impacted area is rare, meaning that planting is often done at a site physically removed from the impact area and any functions affected by spatial elements of ecosystem linkages (i.e., geographic setting) are lost. Second, the production that was lost was removed from a specific point in time; ecosystem functions were disrupted and those specific resources are not replaced, such as that year's spawn of herring (e.g. as in the Pacific northwest). Further, if there was a greater hiatus between the time of impact and recovery or the spatial separation were greater, then one could argue that plantings conducted longer after an impact or further away from an impact have less value than ones conducted sooner or nearer.

The injury assessment strategy to calculate interim loss is based on four steps of analysis: 1) documentation and quantification of the injury, 2) identification and evaluation of restoration options, 3) scaling of the restoration project to compensate for the injury over time, and 4) determine the appropriate means of compensation (e.g., monetary or planting). The scaling aspect is the portion of the process that helps standardize the way in which interim losses are computed, irrespective of the habitat type involved. Interim lost services can be considered to be the integral of service lost from some baseline level over time. To compare services lost with those recovered by some remedial action (such as planting seagrass), the product:

square m of habitat lost x time = square m-years,

is set against square m-years of services provided by the planting project, but discounted as a function of time since the initial injury. Discounting is a widely-known economic principle and is a way of computing value of a commodity (such as a unit area of planted seagrass) based on how long it has been since the impact occurred. Plantings that occur longer after an impact are worth less than plantings conducted shortly after an impact and therefore more planting must be done as more time elapses.

ATTACHMENT II. Synopsis of Braun-Blanquet Technique (Braun-Blanquet, J. 1965. Plant Sociology: The study of plant communities (translated, revised and edited by C.D. Fuller and H.S. Conrad). Hafner, London.

The B-B coverage abundance scale is computed beginning at the zero point and at X m intervals along a transect (frequently enough to get some detailed variation, say a absolute minimum of 30 quadrats per transect), place a quadrat (for seagrass I recommend 0.25 m^2 , i.e., 50 cm on a side) on the seafloor. Visually inspect the content of the quadrat and assign a cover-abundance scale value to the seagrass coverage. The scale values are:

0.1 = solitary shoots with small cover
0.5 = few shoots with small cover
1.0 = numerous shoots but less than 5% cover
2.0 = any number of shoots but with 5-25% cover
3.0 = any number of shoots but with 25-50% cover
4.0 = any number of shoots but with 50-75% cover
5.0 = any number of shoots but with > 75% cover.

From the survey of quadrats along a transect, frequency of occurrence, abundance and density of seagrass (or macroalgae, by species in either case) can be computed as follows:

frequency of occurance = number of occupied quadrats / total number of quadrats,

abundance = sum of B-B scale values / number of occupied quadrats,

density = sum of B-B scale values / total number of quadrats.

One might wish to average the frequency, abundance and density values of the various transects or consider stratifying the data in some way that defines the site in some meaninful way. These values can then be used as a comparative basis among sites, pre- and immediately post-impact or post-planting as a means of assessing recovery.

APPENDIX F

Recommendations for Further Reading

The following community profiles on seagrass published by USFWS:

Phillips, R. C. 1984. The ecology of eelgrass meadows in the Pacific northwest: a community profile. U.S. Fish Wildl. Serv. FWS/OBS-84/24. 85 p.

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Thayer 1992 (Entire document).

Thom 1990.

Wyllie-Echeverria et al. 1994a. (Entire document).

Muehlstein, L.K. (1989): review of wasting disease phenomena.

Phillips, R.C. and C.P. McRoy (1990): handbook of seagrass research method.