DREDGING EFFECTS ON SEAGRASSES: CASE STUDIES FROM NEW ENGLAND AND FLORIDA

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ABSTRACT

While speculation on effects of dredging on seagrass beds is plentiful, actual empirical data documenting these effects are not. We present two case studies, one from coastal New England and the other from the Florida panhandle, in which seagrass beds in the immediate vicinity of coastal dredging sites were monitored in detail before and after dredging operations. Acoustic-based seagrass mapping techniques were used to generate detailed maps of seagrass distributions. Eelgrass (*Zostera marina*) within Scituate Harbor, MA, was monitored during mid-summer in 2001, 2003, and 2004; navigation maintenance dredging of the harbor was performed during fall 2002. In addition to eelgrass removal from the navigation channel, a substantial reduction in coverage occurred in adjoining undredged areas suggesting possible indirect impacts. A modest recovery was evident in the un-dredged areas between the first and second post-dredging years. No direct measurements of dredge-induced turbidity or photosynthetically available radiation (PAR) were made during dredging at this site. Monitoring of another un-dredged site within the region showed natural year-to-year variations in eelgrass coverage to be almost as large as those occurring at the dredged site.

In the second case study, mixed beds of turtlegrass (*Thalassia testudinum*) and shoalgrass (*Halodule wrightii*) adjoining and remote from a dredging site in St. Andrews Bay, FL (Panama City) were monitored in late summer 2002 and 2003, with dredging conducted during the intervening winter and spring. At this site extensive measurements were made of dredge-induced turbidity plumes associated with open-water discharge of hydraulically pumped dredged material. Measurements of photosynthetically active radiation (PAR) were also made within the seagrass beds. In post-dredging surveys, seagrass coverage and maximum depth of growth declined in both the adjoining and the remote beds indicative of a system-wide response. Direct measurements of light and suspended solids indicated that dredge-induced turbidity did not extend to seagrass beds within the project area. Results emphasize the need for long-term data sets to discern any potential effects of dredging on seagrass dynamics as opposed to a host of other factors contributing to high variability in measured parameters.

Keywords: Suspended solids, acoustic vegetation mapping, light attenuation, sediment plume

INTRODUCTION

Seagrasses play an important ecological role in nearshore coastal ecosystems (Thayer et al. 1984; Zieman and Zieman 1989). Seagrass is known to provide food and shelter for a diverse array of fishes and invertebrates (Thayer et al. 1984; Hughes et al. 2002). Many of these species reach their maximum abundance and biomass in areas of high seagrass complexity (Hughes et al. 2002). Seagrass seeds, roots and rhizomes can be an important source of food for over-wintering waterfowl (Ganter 2000). Seagrasses also baffle wave and current energy, increase sediment deposition, and stabilize bottom sediments, thereby improving water quality (Fonseca et al. 1982). Seagrass resources world-wide have been declining, due to a host of factors, both natural and anthropogenic, which could lead to changes in nearshore ecosystem structure and function (Short and Wyllie-Echeverria 1996).

The amount of light, or photosynthetically active radiation (PAR), is a primary limiting factor in the photosynthesis, growth, and depth distribution of seagrasses (Bulthuis 1983; Dennison 1987; Abal et al. 1994; Kenworthy and Fonseca 1996). During dredging and dredged material disposal operations, a certain amount of sediment is re-suspended in the water column. Turbidity changes induced by dredging, either on a short-term (during dredging) or long-term basis (due to altered bathymetry or circulation), can conceptually be linked to increased light attenuation in the water column. Consequently, concerns have been raised regarding the potential impacts of dredging activities

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on seagrass resources (Onuf 1994; Short and Wyllie-Echeverria 1996). Detecting the specific impacts of dredging against a background of natural spatial and temporal variability is challenging, however, and in many cases the impacts of dredging on seagrass resources have not been clearly established (e.g., Quammen and Onuf 1993; Long et al. 1996).

The magnitude and extent of the seagrass impacts are affected by a host of factors, including proximity of seagrasses to dredging and disposal sites, sediment characteristics, hydrodynamic regime, seagrass species present, operational characteristics of the dredging and disposal, and techniques used to detect and quantify the impacts. In this paper we examine impacts of dredging operations on well-established seagrasses beds in a small boat harbor in coastal New England and in St. Andrews Bay, FL. In these two case studies, the environments and dredging operations were very dissimilar, resulting in differences in nearly all of the contributing factors listed above. We describe conditions and operations for each dredging site, and compare and contrast results in order to identify those factors that were most likely to affect seagrass resources.

NEW ENGLAND CASE STUDY

Site Description

Interannual patterns of eelgrass (*Zostera marina*) distribution were examined at two small boat harbors in New England; Scituate Harbor, Massachusetts, and Wood Island Harbor, Maine. Scituate Harbor was dredged during fall 2002. Wood Island, Maine, has not been dredged since 1992. Both sites are extensively colonized by eelgrass. Seagrass surveys at both harbors were conducted in 2001, 2003, and 2004 during the month of July, when eelgrass is near peak annual biomass. Timing of the three surveys corresponds to one pre-dredging survey and two post-dredging surveys (5-months and 17-months after dredging) at Scituate Harbor. The undredged Wood Island Harbor site, located 145 km north of Scituate Harbor, was used to obtain a measure of natural interannual variability of eelgrass within the region.

Wood Island Harbor is located on the southern coast of Maine near the mouth of the Saco River. The project consists of a 3.0-m deep [All depths are referenced to mean low low water (MLLW)] outer channel leading to a 1.8 m- deep anchorage basin within Biddeford Pool (Figure 1). For this study, a 5.25-ha area in the outer channel, containing dense established eelgrass beds, was surveyed. The mean tidal range is approximately 2.75 m.

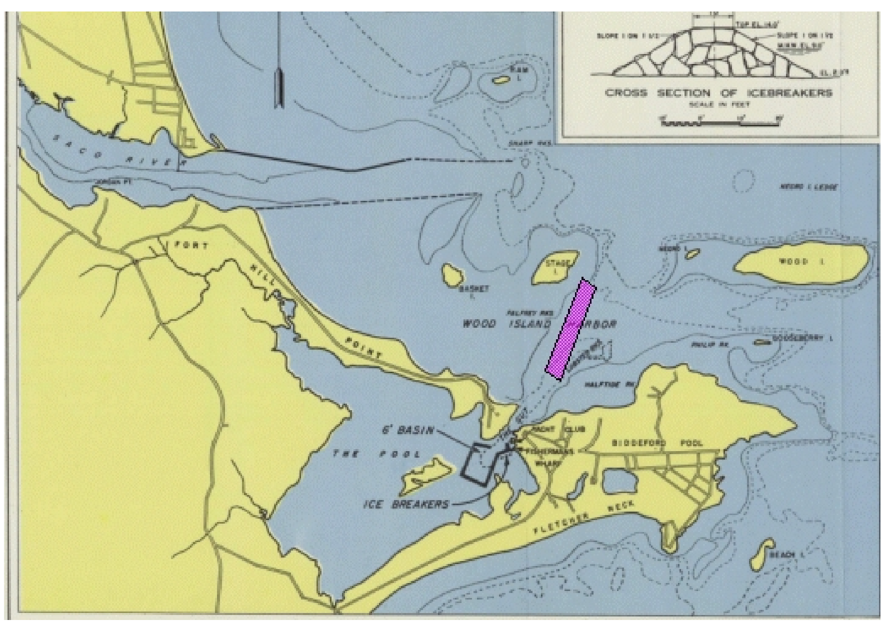

Figure 1. Detailed map of Wood Island Harbor with surveyed area highlighted in purple.

Scituate Harbor lies on the southern shore of Massachusetts Bay about 29 km north of Plymouth Harbor and 37 km south-east of Boston. The harbor is bounded on the east and north by the Atlantic Ocean and has a tidal shoreline of about 10 km. The mean tidal range is approximately 2.75 m. The project consists of a 3.7-m deep entrance channel, a 3.0-m deep and a 2.4-m deep outer harbor anchorage basin, a 3.0-m channel in the inner harbor area leading to a 3.0-m deep inner harbor anchorage basin, and two rubblemound breakwaters (Figure 2). Prior to dredging, appreciable siltation had occurred in the outer harbor anchorage basin. Materials requiring dredging included fine sediments (silt) in the anchorage and a limited amount of very coarse sand and cobble in the entrance channel. Dredging was performed from September 2, 2002 to February 10, 2003, using a bucket dredge. Approximately 199,000 m^3 of sediment were removed, placed on a barge, and transported to the Massachusetts Bay Disposal site, 24 km northeast of the harbor, where it was placed in an open water disposal area.

For the purposes of this study, the surveyed area of the Scituate Harbor was divided into two sections, entrance channel and anchorage, based on differences in sediment type, bottom slope, and current regime. The area surveyed within the entrance channel area was 3.7 ha in size and was characterized by coarse-grained sediments, steep bottom slopes, and relatively high current velocities. The area surveyed in the anchorage area was 10.9 ha in size and was characterized by a relatively flat bottom slope, fine-grained sediments, and relatively slow current velocities.

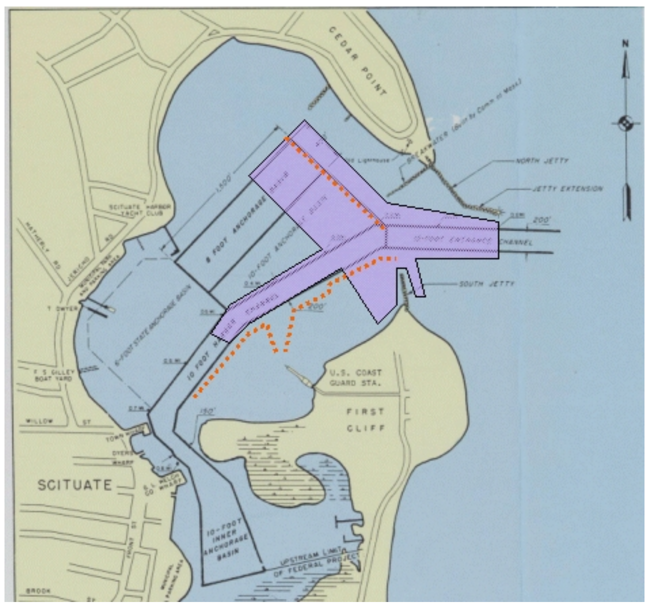

Figure 2. Detailed map of Scituate Harbor denoting project boundary and surveyed area (purple).

Materials and Methods

Hydroacoustic Surveys

The Submersed Aquatic Vegetation Early Warning System (SAVEWS), developed at the US Army Engineer Research and Development Center (ERDC), was used for all surveys. SAVEWS hardware consists of a digital echosounder, a global positioning system (GPS), and a laptop computer. The hydroacoustic component is a *Biosonics* DT-series digital echosounder (Biosonics, Inc., Seattle WA) with a 420-kHz, 6-degree single-beam transducer that generates monotone pulses at a rate of 10 Hz, and a 0.1-ms duration. Return echoes are digitized at high frequency and dynamic range (22 bits) to generate a return envelope that is sampled at 41.67 kHz, corresponding to a depth increment of approximately 1.8 cm. Data are stored on the hard drive of the computer that operates the system. Interspersed with these digitized echo signals are NMEA-format position reports (latitude and longitude, NAD83) recorded at 1 Hz from the real-time differentially corrected GPS (DGPS), using U.S. Coast Guard broadcast corrections. A Trimble NT 300D GPS system, which has a horizontal root mean squared error of approximately 1.5 m, was used for all surveys. The SAVEWS transducer and co-located GPS antenna were mounted on a Corps of Engineers survey vessel that navigated a pre-planned set of transects through each study site. Further details on the SAVEWS hardware can be found in Sabol et al. (2002).

Data were collected along pre-planned survey transects running parallel to the longitudinal axis of the channel or anchorage at a separation interval of 7.6 m. The survey vessel navigated transects using its resident DGPS at an average speed of 2.5 m sec^{-1}. The actual path surveyed was typically within a meter of the intended transect line. Tide measurements were recorded for every 0.03 m change in depth on the local tide gage. Surveys were conducted in 2001, 2003, and 2004 during the month of July, corresponding to one pre-dredging survey and two post dredging surveys.

The SAVEWS processor examines the signal strength and spatial distribution of echo signals to determine the bottom depth and detect bottom-attached vegetation. SAVEWS outputs include bottom depth, SAV coverage (percentage of pings within a localized region in which SAV was detected) and mean SAV canopy height (average height of detected plants within the localized region). Under typical operating conditions SAVEWS can detect vegetation exceeding 0.09 m in height and 60 g m^{-2} (wet weight) biomass (Sabol et al 2002). In this study, emphasis was placed on identifying locations containing eelgrass. During the 2001 survey, rake sampling revealed that locations in which the apparent SAV height (based on echosounder screen display) exceeded 0.3 m contained at least some eelgrass. Locations with an apparent SAV height less than 0.3 m typically contained only *Fucus*, a brown marine macroalgae. The SAVEWS processor only measures canopy geometry and currently does not have the ability to discriminate between species. Accordingly, the general height difference between eelgrass and *Fucus* was used as a discriminating feature. The plant detection threshold was set to 0.3 m, so that only pings with a detected vegetation height of 0.3 m or more were declared to contain eelgrass. While this discrimination rule appeared to work in July, during peak eelgrass biomass and height, it may not be appropriate at other times of the year. These SAVEWS processing parameters were held constant for all New England site surveys. Following initial SAVEWS processing, position data were converted from latitude/longitude in NAD83 to the local state plane coordinates (ft) and SAVEWS-detected bottom depths were corrected to MLLW by adjusting for tidal amplitude and transducer depth.

Study Design and Data Analysis

Corrected depth and vegetation coverage outputs for the closely spaced transects were spatially interpolated (3 m grid spacing) to achieve exactly matching grid coverages between the three surveys for the two sites (Sabol et al. 2005). Scituate Harbor was further partitioned into treatment areas (dredged vs. undredged) and locations (anchorage area with low energy and fine sediments, and channel with high energy and coarse sediments). This facilitated discerning direct (physical removal) versus indirect (impacts due to increased light attenuation and sedimentation) impacts on the eelgrass. Vegetation maps were generated by site and survey to facilitate visual comparison. Statistical and spatial analyses of vegetation coverage were performed to quantify: 1) total vegetated area by site and year, 2) changing vertical distribution of eelgrass, and, 3) horizontal changes in eelgrass coverage. Quantified changes in dredged and adjoining undredged areas at Scituate Harbor were simultaneously compared with changes at the un-dredged Wood Island Harbor site, which served as an indicator of natural interannual variability. Vegetation height was not examined since high tidal flows greatly affect canopy height (Sabol et al. 1997).

Results

Coverage maps (Figure 3 and 4) depict the spatial distribution of SAV within each harbor. Prior to dredging (2001) both harbors exhibited well-established eelgrass beds. At the un-dredged site (Wood Island Harbor, Figure 3), dense established beds at the southern and northern end of the site diminished in area and coverage for each of the two subsequent surveys. During the last survey (2004), sparse vegetation appeared in the relatively shallow middle section of the site. Eelgrass decline in the dredged site (Scituate Harbor, Figure 4) appeared to be even more severe. Largest declines were observed in the inner harbor (anchorage), both within the dredged area and in adjoining un-dredged areas outside the project bounds. During the final survey, some recovery of vegetation was evident within the un-dredged portion of the anchorage however, none was apparent within the dredged areas.

Vegetated area and mean coverage within vegetated areas (Table 1), computed from gridded data, quantify what is shown in the maps. Wood Island Harbor exhibited a 13% relative decline in total vegetated area between 2001 and 2003 and was basically unchanged between 2003 and 2004 (+1.4% increase). However, coverage within the vegetated area progressively declined over the entire time period. Relative fluctuations in vegetated area for the entire surveyed area of Scituate Harbor were larger than Wood Island Harbor. When Scituate Harbor changes were examined by treatment and location factors, the effects were even more pronounced. Both dredged and un-dredged anchorage areas lost two thirds or more of their vegetated areas between pre- and immediate post-dredging surveys; percent coverage within vegetated areas likewise declined. Between the two post-dredging surveys, the un-dredged anchorage area exhibited a substantial recovery of terms of total vegetated area and coverage, however no recovery was evident within the dredged anchorage area. The dredged channel area showed a response very similar to the dredged anchorage – decline following dredging with no apparent recovery. The un-dredged channel exhibited very limited fluctuations in vegetated area and coverage within vegetated area compared to the other areas.

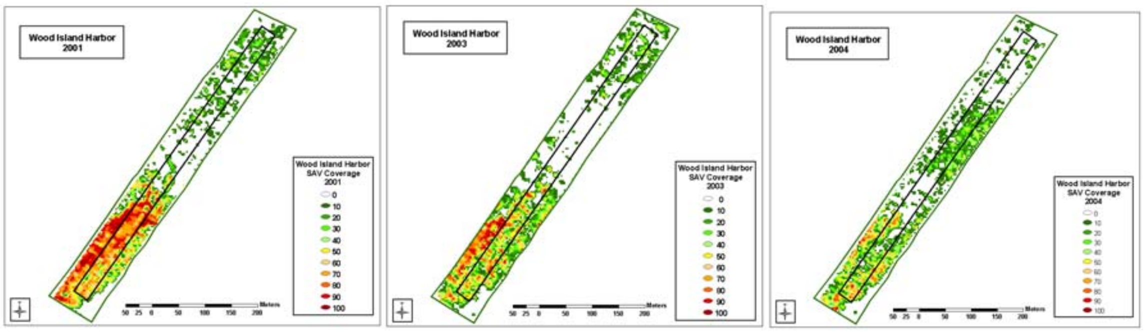

Figure 3. Vegetation coverage maps for Wood Island Harbor; (a) 2001, (b) 2003, and (c) 2004.

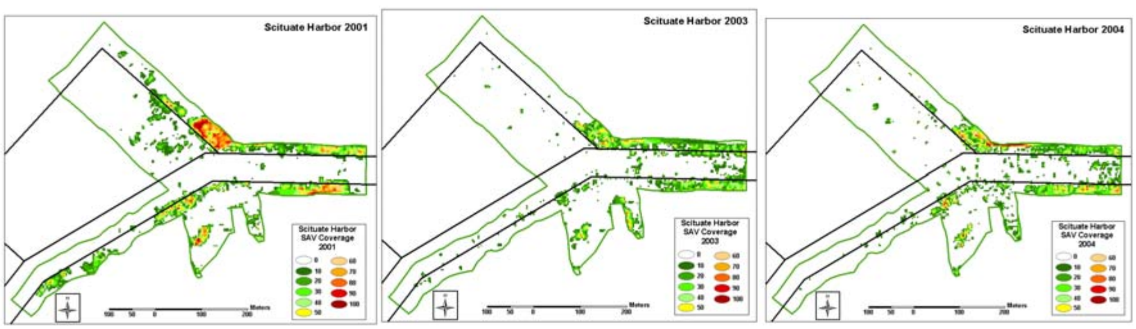

Figure 4. Vegetation coverage maps for Scituate Harbor; (a) 2001, (b) 2003, and (c) 2004.

Table 1. Vegetated areas, and mean coverage within vegetated area, by site, treatment area, and survey.

Site	Treatment area	2001 (pre-dredging)		2003 (5 months post-dredging)		2004 (17 months post-dredging)	
		Area (ha)	Mean cover (%)	Area (ha)	Mean cover (%)	Area (ha)	Mean cover (%)
Wood Island	Total	3.24	47	2.83	36	2.87	28
Scituate	Total	3.30	30	2.18	21	2.35	25
	Dredged anchorage	0.61	18	0.14	13	0.15	13
	Un-dredged anchorage	1.42	35	0.49	21	0.71	27
	Dredged channel	0.17	28	0.12	18	0.13	15
	Un-dredged channel	1.10	30	1.43	22	1.36	25

ST. ANDREW BAY CASE STUDY

Site Description

The Saint Andrew Bay system, located in northwest Florida, consists of Saint Andrew, North, West, and East bays with a combined surface area of approximately 28,000 hectares. Because of the lack of significant riverine input, the St. Andrew Bay system has been described as a relatively clear water, high salinity estuary (Keppner and Keppner 2001). Tides in the Saint Andrew Bay system are diurnal with a tidal range of approximately 0.37 meters (Brim 1998). St. Andrew Bay has the largest seagrass stocks in the Florida panhandle (Wolfe et al. 1988), estimated at more than 6,200 acres (Brim 1998). The dominant species is turtlegrass (*Thalassia testudinum*), but there are also extensive beds of manatee grass (*Syringodium filiforme*) and shoal grass (*Halodule wrightii*) (Brim 1998). The study area adjacent to the port of Panama City, Florida, is depicted in Figure 5. Primary areas of interest for this study included the dredging location, immediately south and abreast of Dyers Point, and the dredged material placement area, located approximately 500 m west of Dyers Point.

This study was designed to evaluate potential impacts on seagrass resources resulting from Federal navigation dredging activities at the Dyers Point Turning Basin in Panama City. The project involved excavation of approximately 99,000 cubic meters of dredged material using a 0.66-m hydraulic cutterhead pipeline dredge in order to deepen the turning basin to a depth of 11.6 m. Dredged material was deposited within a deep portion of St. Andrew Bay in proximity to the turning basin using an open water pipeline discharge with a down-turned, submerged diffuser. Dredging operations were initiated on February 4, 2003, and completed by March 1, 2003. This study was performed in three parts. First, the plume of suspended solids generated during disposal operations was tracked using an acoustic-based device. Second, light availability in adjoining seagrass beds was monitored before, during, and after dredging operations. Finally, an extensive SAVEWS survey was performed before and after dredging in seagrass beds adjoining and remote from the dredging site.

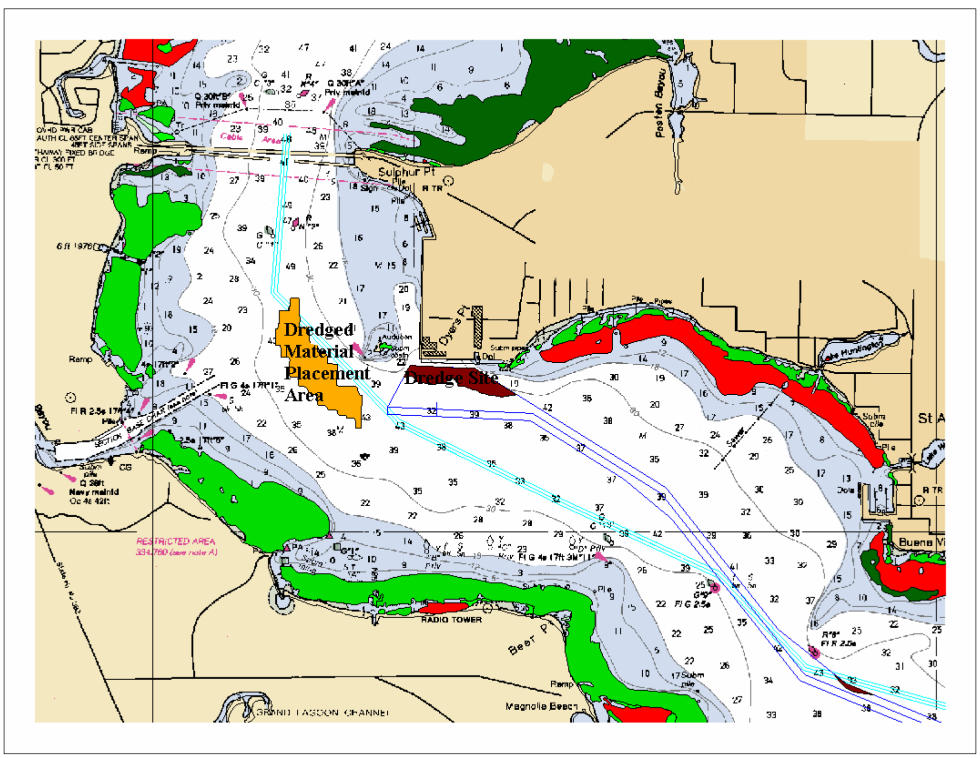

Figure 5. Map of the study area showing the dredging and dredged material placement areas in relation to nearby seagrasses, depicted by density (red = sparse, light green = medium, dark green = heavy).

Plume Tracking

Methods and Materials

The plume generated by dredging and disposal operations was monitored due to concerns regarding the effects of the elevated suspended sediment concentrations and their subsequent deposition. Because plumes can change dramatically over large spatial scales and short time scales, particularly when driven by tidal forces, characterizing plumes has presented severe challenges to many previous monitoring efforts. Data collected at points in time at fixed locations are not sufficiently rigorously to assess the potential effects of dredging. However, acoustic technologies offer advantages in capturing data at appropriate spatial and temporal scales to allow accurate interpretation of plume dynamics. In this study an Acoustic Doppler Current Profiler (ADCP) was employed to characterize both ambient conditions and dredging-induced plumes resulting from disposal operations. In brief, the ADCP determines current velocities and direction vectors based on acoustic backscatter from particles moving through the water column. Because ADCP surveys can cover large areas within a single tidal phase, plume signatures can be mapped and their proximity to seagrass resources easily determined. ADCP backscatter data were then used to derive estimates of suspended sediment concentrations according to the methods described by Land and Bray (2000). Physical characterizations also included temperature, salinity, suspended sediment concentration measurement (gravimetric, optical and acoustic), and current structure surveys.

Eleven ADCP surveys were conducted during flood tidal cycles, and four during ebbing tides. The disparity in survey allocations was predicated by the fact that ebbing tides in the during-dredging period occurred primarily at night. Safety considerations limited survey vessel operations during nighttime hours. Three additional surveys were completed during slack tide conditions and another two surveys were completed during transitions from flood to slack tide. These surveys illustrate the typical conditions and spatial scales of the plumes encountered.

Results

The ADCP surveys effectively characterized suspended sediment plume structure for open-water pipeline discharges as conducted by hydraulic cutterhead dredging operations in the Dyers Point Turning Basin in St. Andrew Bay. Data from ADCP surveys, OBS sensors and TSS water samples produced a detailed characterization of the spatial extents and concentration gradients of disposal-induced plumes (Figures 6 and 7). These data consistently indicated that plumes tended to follow bathymetry contours. The majority of plume-borne sediments remained in the lower portion of the water column, as the plumes were entrained in tidal flows. The plumes dissipated without indications of transport up side slopes and into shoals peripheral to the navigation channel and natural deep basin.

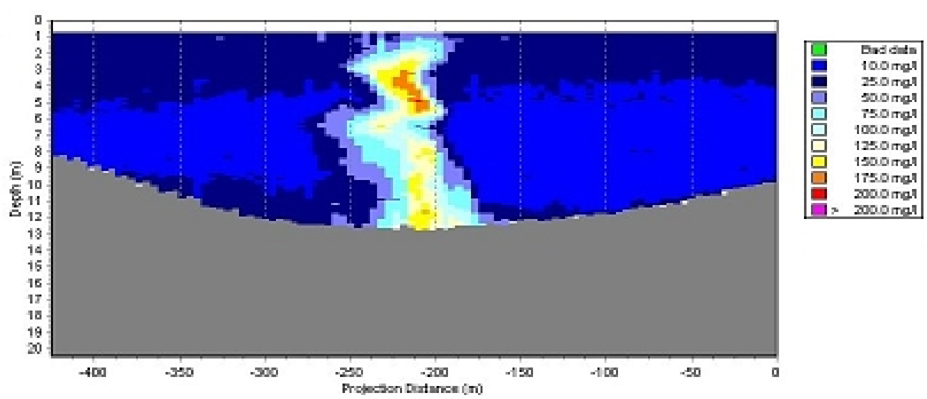

Figure 6. Vertical cross-section profile of acoustic estimate of total suspended solids on transect B located 30 m down current for discharge during a flood tide survey.

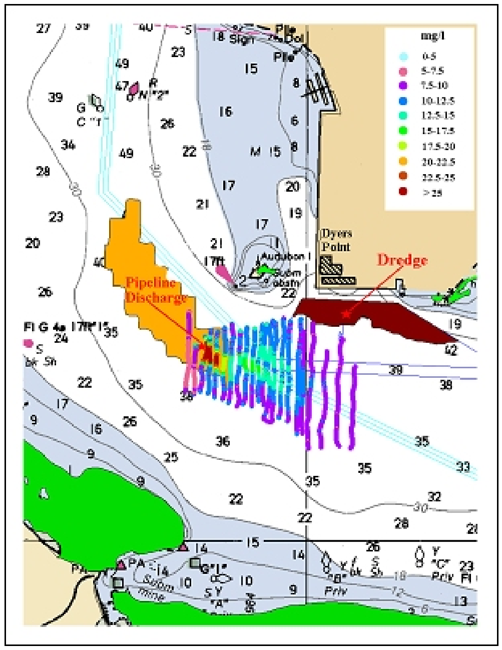

Figure 7. Depth-averaged suspended sediment concentration for a plume tracked during an ebbing tide, 9 February 2003.

Light Availability

Methods and Materials

Spherical quantum light sensors (LI 193SA, LICOR, Inc.) were used to record simultaneous underwater light data at three sites in St. Andrew Bay. These sensors are accurate to within ±5%, stability is ±2% within any given 1-year period, and the data are recorded with a precision of ±0.01 μm m^{-2} s^{-1} (LICOR, Inc.). Two sensor arrays (Sites P1 and P2) were located in the vicinity of the project site; a third was used as a reference. Light data were collected during the period from January 19 to March 11, 2003. Two sensors, separated by 32 cm in depth, were deployed at each location to allow calculation of diffuse attenuation coefficients. To minimize the error associated with sediment deposition and fouling of the sensor, the sensors were cleaned by hand every 2 days. Water depth was measured at the time of sensor installation, and corrected for tidal amplitude using National Ocean Service water level gage located in Panama City, Florida (www.co-ops.nos.noaa.gov).

The percent surface irradiance for each underwater record was determined by comparison with a surface mounted sensor located on the runway approach tower of the Bay County Airport, Panama City, Florida. Mean daily percent surface irradiance values were calculated by averaging these values over a 4-hour period from 1000 to 1400 CST. Diffuse attenuation coefficients (K_d) were calculated (Carruthers et al. 2001) according to the equation:

$$K_d = (\log_{10} I_t - \log_{10} I_b)/0.32 \qquad (1)$$

where, I_t and I_b = irradiance recorded at the top and bottom sensors, respectively, separated by 32 cm in depth. In order to investigate the potential linkages between weather events and light availability, meteorological data from the Bay County Airport in Panama City were obtained from the National Climatic Data Center database.

Results

The seagrass community in St. Andrew Bay is dominated by *Thalassia testudinum*, with extensive areas of *Halodule wrightii* and *Syringodium filiforme*. Minimum light requirements of these species, expressed as a percentage of surface irradiance, range from 10-22% surface irradiance (SI), but there is strong evidence to suggest that long-term averages at the upper end of this range are needed to maintain stable biomass and density of *Thalassia testudinum* (Czerny and Dunton 1995, Shafer 1999, Dixon 2000).

During the period prior to the initiation of dredging activities, the average daily percent of SI available at the level of the seagrass canopy at Site P1 and the reference area were 27% and 30%, respectively. During dredging operations, light levels at Sites P1 and P2 averaged 29% and 31% of SI, while average light levels at the reference site remained unchanged compared to those observed during the pre-dredging period. Lower average light levels, ranging from 6-10 % SI, were observed in both the reference area and areas P1 and P2 during the post-dredging period (Table 2).

Table 2. Mean daily percent surface irradiance at three locations (± standard deviation).

Site	Pre-Dredging	During Dredging	Post-Dredging
P1	27.03 ± 6.95	29.31 ± 7.12	6.68 ± 2.20
P2		31.85 ± 8.40	8.27 ± 1.82
R	30.11 ± 4.08	30.63 ± 6.32	10.44 ± 2.12

Based on the average minimum seagrass light requirements, light levels in the seagrass beds at all three sites were well above threshold levels both prior to and during dredging and disposal activities (Table 2). Light levels during the post-dredging period of March 1-11, 2003 were consistently below these minimum thresholds at all sites, including the reference area. Since light reductions were not observed in the seagrass beds near the project area during periods of active dredging and disposal, this change was more likely to represent a short-term system-wide response to climatic events rather than a consequence of dredging operations.

Hydroacoustic SAV Surveys

Methods and Materials

The same equipment and generally the same procedures were used in St. Andrews Bay as described for the New England case study. Survey lines were established perpendicular to the shoreline at a separation interval of 50 m. Surveys were conducted in September 2002 (pre-dredging) and October 2003 (post-dredging). The deep end of each transect was consistently beyond the depth limit of vegetation colonization. The survey vessel navigated these transects using its resident DGPS at an average speed of 2.5 m sec^{-1}. Depth measurements were corrected to MLLW by using a National Ocean Service water level gage located in Panama City, Florida. SAVEWS processing was performed at the highest level of sensitivity, which has previously shown detection capabilities of 9 cm SAV height and 60 g m^{-2} (wet weight) SAV biomass (Sabol et al. 2002).

The bay was divided into two treatment regions. The outer bay, which was in close proximity to the dredging and disposal operations, was designated as the project (treatment P) region, most likely to be subject to any indirect impacts from dredging and disposal operations. The inner bay, north of the Hwy 98 Hathaway Bridge, designated as the reference (treatment R) region, was least likely to be influenced by dredging and disposal operations. Five localized areas (numbered 1-5, and referred to as "reaches") were selected within the P treatment region, and four (1-4) were selected within the R treatment region (Figure 8). The number of transect lines per reach ranged between 20 and 73.

Potential indirect impacts in this study were related to changes in water column turbidity and light availability to seagrasses. Accordingly, we examined the maximum depth limit of SAV colonization. The deepest extent of SAV presence along each transect was determined (defined as maximum depth of contiguous growth of ≥20% coverage). The maximum depths of each transect within reach were differenced between years. A nested analysis of variance (ANOVA) was performed to examine the potential differences between treatments and reaches.

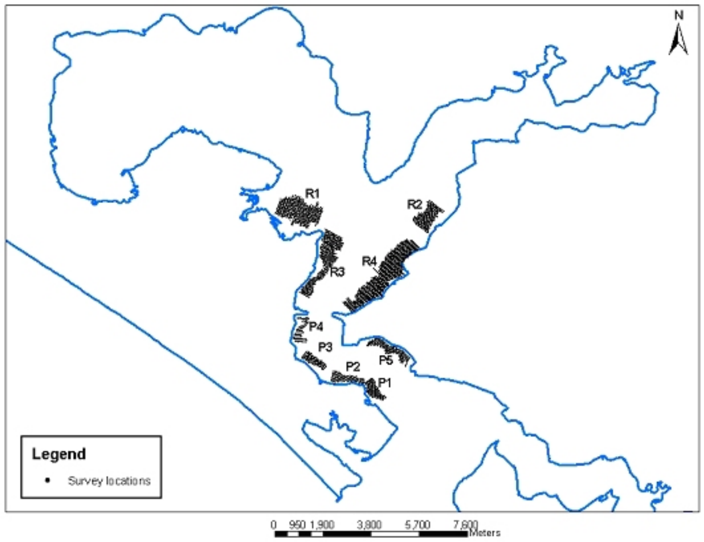

Figure 8. Seagrass survey reaches (R = Reference, P = Project) in the study area.

Results

Maximum depth of SAV growth (Table 3) exhibited appreciable change across time and treatment area. The Project treatment area (outer bay) exhibited greater mean maximum depth than the Reference area (inner bay) for both years. A decline in maximum depth between 2002 and 2003 occurred for both treatment areas, but was largest for the Project area. The last columns of Table 3 summarize the difference in maximum depth of individual transects between years, grouped by treatment and by reaches within treatment areas. Analysis of variance of paired depth differences, by treatment area and reach, were highly significant between reaches (p=0.0010) and treatments (p=0.0287). The decline in the Project area was significantly larger than that for the Reference area**.**

Table 3. Summary of maximum SAV depth and differences between surveys.

Treatment	Reach	Transects	Maximum SAV depth (m)				Paired transect difference in maximum depth (m) between years	
			2002 (pre-dredging)		2003 (post-dredging)			
			Reach mean	Treatment mean	Reach mean	Treatment mean	Reach mean	Treatment mean
Project	1	20	1.96	1.92	1.54	1.63	-0.42	-0.29
	2	26	2.21		1.82		-0.39	
	3	20	1.91		1.64		-0.27	
	4	20	1.94		1.71		-0.24	
	5	35	1.66		1.48		-0.18	
Reference	1	34	1.42	1.67	1.30	1.53	-0.12	-0.14
	2	22	1.80		1.60		-0.20	
	3	52	1.60		1.50		-0.10	
	4	73	1.82		1.64		-0.18	

DISCUSSION

Although light availability is the predominant factor influencing seagrass abundance and distribution, seasonal changes in water temperature, salinity, sediment characteristics, freshwater discharge, nutrient availability, nutrient enrichment, exposure to waves and currents, and a host of other natural and anthropogenic factors, also affect seagrasses (Koch 2001). Detection of dredging impacts against this background of natural variability presents a difficult technical challenge. Single point-in-time data, even taken at periodic intervals, are only marginally useful in discerning differences or trends in conditions. Sampling on appropriate spatial and temporal scales is critical to interpretation of the data obtained in any monitoring study of dredging effects. Plume tracking studies can provide an indication of the magnitude and behavior of the sediment plume at relatively small spatial and temporal scales (hours to days). Continuous measurement of light availability over the course of the dredging activity (intermediate time scale) allows changes in water column turbidity to be interpreted in relationship to the minimum light requirements of the seagrass species. Hydroacoustic mapping of seagrass distribution conducted at large spatial (reaches) and temporal scales (annual) can be used to examine the potential for long-term changes in seagrass distribution in the vicinity of the project area. Careful interpretation of the combined datasets provides insights regarding the spatial and temporal patterns of dredging effects that would not be possible if each dataset were examined individually.

These two case studies illustrate the potential for both direct and indirect impacts to seagrasses associated with dredging activities. In the New England case study, both direct (physical removal of eelgrass along with the sediments) and indirect (changes in eelgrass distribution in adjacent un-dredged areas due to elevated turbidity and/or siltation) impacts were evaluated. In the Florida case study, only potential indirect impacts were involved. Direct impacts are easily quantified using the hydroacoustic mapping techniques. Indirect impacts are considerably more difficult to assess. In both case studies, the potential for indirect impacts appears to be minimized if the dredged sediments are coarse-grained. Coarse-grained sediments settle rapidly and contribute little to water column turbidity and re-suspension. However, the probability of indirect impacts may be increased if the dredged sediments are fine-grained, since these materials remain suspended in the water column for longer periods, which could lead to light limitation impacts to seagrasses.

These two case studies also point to the need for a multi-disciplinary approach in the assessment of potential dredging impacts to seagrass resources. In St. Andrew Bay, Florida, results of the annual hydroacoustic vegetation mapping surveys suggested a potential indirect impact, as evidenced by the greater decline in the maximum depth of seagrass colonization in the Project reaches as opposed to the Reference reaches. However, continuous measurement of PAR within the seagrass beds, prior to and during dredging and disposal operations revealed that turbidity levels within the seagrass beds during dredging and disposal operations were consistently well above the minimum light requirements for these seagrass species. Furthermore, the real-time plume tracking studies indicated that the plume resulting from disposal operations settled rapidly to the bottom and did not disperse to within 500 m of the closest seagrass beds. In the absence of the plume-tracking and continuous PAR measurements, the decline in the maximum depth may have been interpreted as an impact of dredging. However, continuous measurement of PAR showed that the elevated turbidity levels that followed after dredging and disposal operations were occurring on a bay-wide scale and appeared to be associated with the passage of several consecutive winter cold fronts with strong winds and precipitation. Therefore, the changes in seagrass distribution observed in the annual hydroacoustic surveys were unlikely to be related to the dredging operations.

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