INTRODUCTION

The estuaries in the southeastern United States contain about 17.2 million acres of marsh and other estuarine habitat and 5.1 million acres of intertidal area. These wetlands comprise about 83 percent of the coastal wetlands in the conterminous United States (Mitsch and Gosselink, 1993). Both tidal marshes and barrier islands play a major role in coastal states by providing storm protection, flood prevention, and erosion control (Mitsch and Gosselink, 1993). Marsh ecosystems also serve as primary nurseries for many fish species and also provide important recreational and economic resources from which many coastal states derive both economic and social benefits.

Through the years, many tidal marsh systems have deteriorated because they have lost sediment naturally through erosion, then sediment supplied to them preventing accretion, or they have had problems retaining sediment or have subsided. A loss of marsh sediment can mean loss of habitat for benthic flora and fauna along with a loss of nutrients that are found in the sediments. Tidal wetlands must accumulate sufficient materials to maintain their position in the intertidal zone and to counteract the loss or movement of sediment associated with tidal action, sea level rise, and organic soil decomposition. Wetlands do so with allochthonous inputs of sediments (riverine or marine origin) or autochthonous additions (plant and animal debris) (Hackney and Cleary, 1987). Aggradation or vertical accretion rates depend on the quality as well as the quantity of material available and are influenced by the distance from the source and local hydrology (DeLaune et al., 1990). Marsh sedimentation is affected by factors such as freshwater input, wave action, tidal amplitude, storms such as hurricanes and nor'easters, and change in sea level.

As population growth and development have occurred in the coastal zone, multi-use coastal management practices have negatively impacted the natural processes of marsh accretion (Leonard, 1996). An inadequate supply of sediment to the interior of a salt marsh can result in vegetation dieback or marsh deterioration. The mechanisms of marsh deterioration are both biological and physical. Restricted sediment input can reduce vertical accretion while submergence or changes in marsh surface water-level relationships can accelerate organic decomposition, and stress or kill organic matter available for accretion processes (DeLaune et al., 1990).

Overwash deposits and abandoned flood tidal deltas are the main source of inorganic sediment for back barrier marshes in North Carolina. In many locations, however, sediment inputs to barrier marshes by these two mechanisms are affected by coastal engineering practices (Hackney and Cleary, 1987). Construction on barrier islands limits the amount of sediment that would be supplied by the natural overwash process. Overwash is a process that occurs in conjunction with large storm events such as hurricanes and nor'easters, which usually have storm surges capable of carrying sediment to the backside of a barrier island. During storm events, as much as 17 cm of sediment can be added to the marsh during a short time (Hackney and Cleary, 1987). Previous research (Hackney and Cleary, 1987) has indicated that marine sands are in fact much more important than autochthonous organic production in maintaining the marsh elevation in southeastern North Carolina back barrier marshes relative to sea level rise. If marine sands are denied to these ecosystems, rising sea level will eventually drown the marshes and they will disappear (Hackney and Cleary, 1987).

Inlet migration is a natural process that is often hindered by human involvement. In southeastern North Carolina, inlet stabilization (i.e. maintenance dredging and the building of jetties) can reduce or stop inlet migration, which can limit the spatial distribution of flood tide delta deposits. These flood tide delta deposits are the platform upon which marshes develop. The low mesotidal regime along some parts of the coast of North Carolina produces moderate size flood tide deltas. These sandy deltas typically provide substrate for salt marsh establishment (Cleary and Hosier 1979).

Sea level rise is a mechanism of coastal marsh loss. Studies have documented the apparent inability of some marshes to maintain structure and position with respect to present and predicted near-future sea level rise (DeLaune et al. 1983). Other studies have indicated that marsh loss is exacerbated by anthropogenic effects (Daiber 1986, Ward et al. 1998) and warn that coastal wetland habitats may become more vulnerable to sea level rise and its effects if current management practices continue (Hensel et al. 1998). Marsh loss occurs when sediment accretion does not keep up with the rate of the rising sea level. Along the eastern seaboard of the United States it is estimated that the rise in sea level is approximately 1.9 mm/yr while the rate of sediment accretion is only 1.2 mm/yr (Hackney and Cleary 1987).

Recently, attempts have been made in areas such as the coast of Louisiana, to offset sediment loss in marshes by introducing new sediment into the system. The replacement sediment is often dredged material from nearby waters. The addition of dredged material is beneficial in two ways. First, the stability of the marsh is enhanced and second, the dredged material is used beneficially. This is important because spoil banks have also been implicated as one of the contributing factors to the high wetland

loss rate in southern Louisiana (Swenson and Turner 1987). The connection between spoil banks and wetland loss is presumably through their influence on marsh hydrologic conditions, which in turn influence biotic and physical factors important in sustaining land building processes.

Even though sediment inputs have been postulated to offset marsh degradation (Mendelssohn and McKee 1988, Nyman et al. 1990) by increasing elevation, enhancing nutrient exchange, and stimulating *Spartina alterniflora* productivity (DeLaune and Pezeshki 1988), most coastal vegetation can only tolerate sediment input up to a limit that, once surpassed, may be lethal (Zhang and Maun 1989). A recent study conducted in southeastern North Carolina marshes following Hurricane Fran found that *Spartina alterniflora* has a low rate of survival when sediment inputs exceed 10 cm (Moundalexis 1996). Not only is it important not to overload sediment because of the possibility of killing existing vascular plants and benthic microalgae (Hilterman 1998), but the depth of sediment can affect the duration and frequency of tidal inundation (Swenson and Turner 1987) thereby interrupting sediment and nutrient supply to the marsh. The sedimentological characteristics of back barrier dredged material are often similar to those of marsh overwash deposits, making them ideal for this purpose which is why the dredged material is usually placed on marshes and barrier islands (USACE 1987).

Benthic microalgae (BMA) have been identified as ecologically significant primary producers in a variety of coastal habitats including rivers (Amspoker and McIntire 1978), salt marshes (Freeman, 1989), tidal flats (Cadee and Hegeman 1977), and the continental shelf (Cahoon 1999) contributing a substantial amount of primary production to these systems (Peterson 1981). BMA are numerous and productive but do

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not accumulate the highly visible biomass created by marsh and sea grasses (Freeman 1989). It has been shown that benthic microalgal biomass can be many times greater than water column biomass, accounting for >50% of the total system primary production in many coastal environments (Cahoon and Cooke 1992, Krom 1991, Sundbäck and Snoeijis 1991). Other studies have determined that benthic microalgal production may range from one-third to as much as 1.4 times that of marsh angiosperm production (Freeman 1989). The availability of benthic microalgal biomass (BMB) to consumers may be enhanced by its spatial distribution (Freeman 1989). Availability of BMB may be affected by resuspension, and benthic microalgae are also concentrated at the sediment-water interface, which makes them easily accessible and consumed by deposit feeders (McIntyre et al. 1996). BMA are, in general, much more nutritious to the fauna that consume them than vascular plant material. They are easily digested and their rapid growth and higher nitrogen content make them a potentially important base for estuarine food webs (McIntyre et al. 1996, Freeman 1989).

Addition of sediment to the surface of the marsh affects several factors. Sediment addition increases the surface elevation of the marsh, which helps to keep that area of the marsh from being inundated by water for the entire flood cycle. Increasing the surface elevation will also increase the light flux reaching the marsh surface, which could promote an increase in benthic microalgae biomass. Seasonality can also affect the biomass of benthic microalgae. Certain seasons of a year may promote "blooms" in which the productivity will increase. Seasonal stability of BMB and production further enhances the importance of BMA as a food source in estuaries (Baillie and Welsh 1980, Freeman 1989). Previous studies have determined that BMA are the only significant

photoautotrophic component of saltmarshes in Delaware, and northward and that are functional over the entire year. In fact, a large portion of the period of BMA productivity may occur while nearby marsh grasses are dormant (Freeman 1989). Also, BMA biomass is more seasonally stable than that of phytoplankton (Freeman 1989). All of these factors need to be taken into consideration so as not to confound them when interpreting the results of this experiment.

Changes in benthic microalgae biomass and diversity are likely to exert secondary impacts on higher trophic levels, although untested with respect to engineered sediment additions (Posey et al. 1999). Benthic microalgae in the marsh systems have been shown to display taxonomic reorganization in response to sediment inputs associated with hurricane induced overwash (Hilterman 1998). Changes in diatom composition, therefore, will influence organisms such as benthic vertebrates and invertebrates that graze on the diatoms as a food source. Ultimately, those higher organisms that rely on benthic invertebrates as a food source may affect the stability of the marsh system.

The ultimate goal of marsh re-nourishment is to slow or reverse the effects of wetland loss due to rising sea level and anthropogenic effects. Also, the addition of the dredged material may provide nutrients for vascular plants and may help with the precipitation of toxic sulfides (Evans 1989). In turn, healthy vegetation in the marsh or barrier island may promote a higher quality of food for higher organisms such as fish and other wildlife. If executed properly, sediment additions may indirectly enhance higher level consumers by increasing primary production, specifically benthic microalgae and vascular plants. However, while it is generally accepted that sediment additions can be beneficial to salt marsh and barrier island function (e.g. increase diversity, promote

accretion, enhance nutrient exchanges, and raise elevation), too few data exist to establish tolerance limits.

This goal of this study was to determine if: the addition of dredged material increased biomass of benthic microalgae or did not decrease benthic microalgal biomass. Accordingly, the following null hypotheses were examined: (1) the addition of sediment to the marsh surface had no effect on the biomass of the benthic microalgae (BMA); (2) the amount of sediment (depth) placed on the marsh surface had no effect on the amount of benthic microalgae biomass (BMB); and (3) there was no significant difference in the biomass of BMA between the amended and non-amended areas.

METHODS AND MATERIALS

This study was conducted on Masonboro Island, North Carolina (Fig. 1). Masonboro Island is a 13.3 km long barrier island ($34\,^{o}\,11'$ N to $34\,^{o}\,05'$ N) with a narrow beach and dune ridge system (60 m to 1.6 km across). An extensive back barrier *Spartina alterniflora* marsh (1,771 ha., 87% of the total island area) occurs on the sound side of the island (Freeman 1989). Tides at Masonboro Island are semi-diurnal with a mean amplitude of 1.15 m making this a microtidal system.

This study utilized four marked plots in the back barrier marshes of Masonboro Island (Fig. 2). There were a total of four plots and two of the plots were non-deteriorated (or healthy) and two of the plots were deteriorated (or unhealthy).

Figure 1. Map of Masonboro Island. NAD 83 North Carolina State Plane Units (ft)
(courtesy of CJ Jackson, 2003)
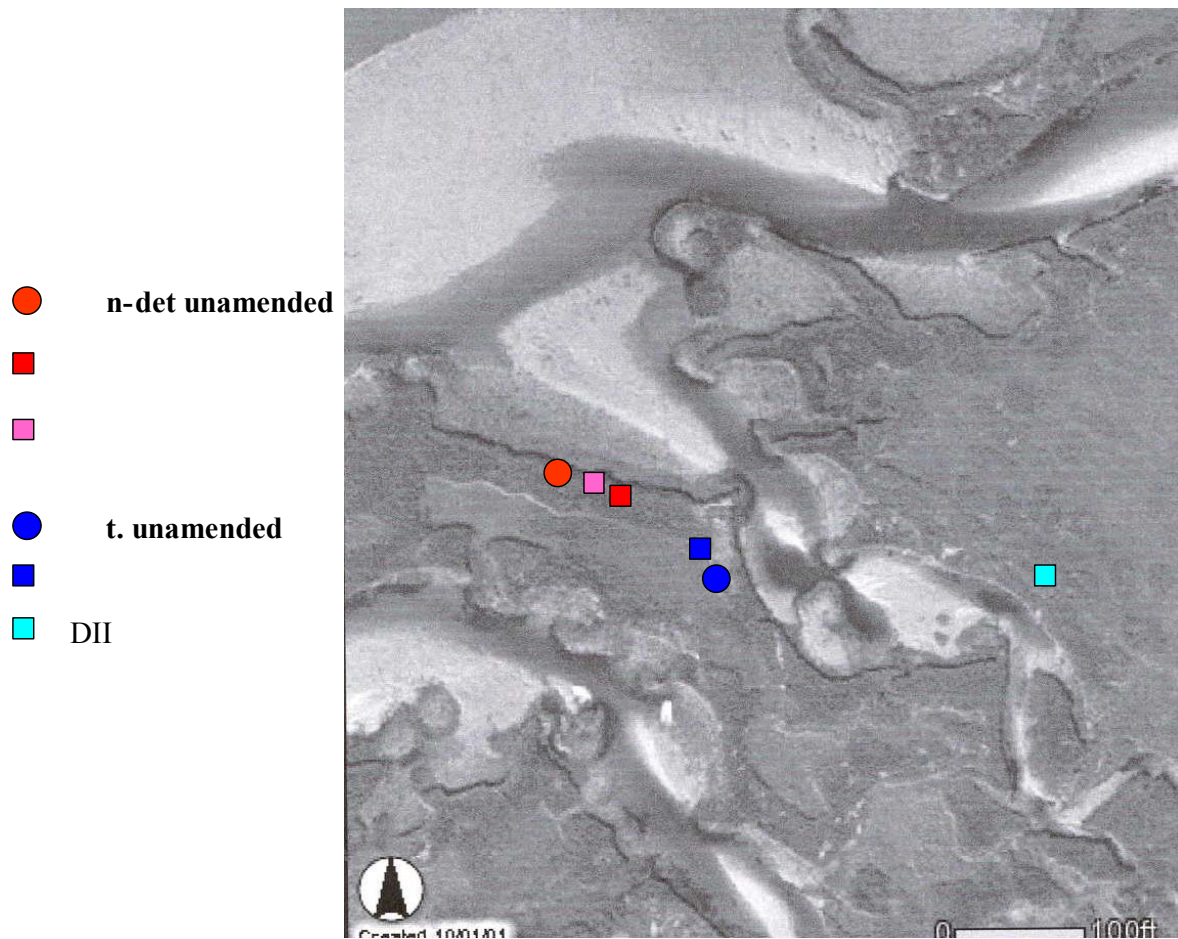
n-det unamended

t. unamended

DII

Figure 2. Aerial photograph of location of sample sites along channel in back barrier marsh of Masonboro Island. Non-det unamended= non-deteriorated site with no sediment addition while Det. Unamended= deteriorated site with no sediment addition. I and II represent the first and second non-deteriorated and deteriorated sites, which are determined by color. Red blocks represent non-deteriorated sites while blue blocks represent deteriorated.

Determination of whether the site was deteriorated was based on stem densities of *Spartina alterniflora*. Areas with stem density >300 stems m^{-2} were considered non-deteriorated, while areas with stem density <200 stems m^{-2} were considered deteriorated.

Each site that was amended with sediment was 6.4 m x 6.4 m and marked at each corner with PVC pipe. A control area for non-deteriorated marsh and a control area for deteriorated marsh were similarly marked. The control sites were adjacent to non-deteriorated site I, and the deteriorated control was adjacent to deteriorated site I. Sediment samples for BMA analysis were taken from all sites (control, deteriorated, and non-deteriorated) for both pre- and post-sediment addition. Samples were taken using a 2.5 cm diameter acrylic coring tube, to approximately 15-20 mm depth of sediment.

Pre-sediment addition samples were taken once per month from January 2000 until the final addition (addition began in May 2000) of dredged sediment material in June 2000. A total of eight samples were taken at each of the four sites in which sediment was to be added prior to the addition.

The sample plots were amended in May 2000 by spreading sediment collected from a nearby dredged island onto the marsh surface in a wedge shape from 0 mm deep at one edge of the site to 102 mm deep at the opposite end of the site. All four sample areas were placed along a channel edge for consistency, and the sediment wedge was oriented in the same direction, relative to the channel, within each area. Each area had the thin edge (0 mm) of the wedge oriented facing the same direction relative to the channel and increased in thickness (102 mm) with distance from the channel. The wedge was divided in three increments with each section having a width of 0.914 m. The first

area was thin (0mm-25.4mm), then intermediate with 25.4 mm-76.2mm of sediment amended, while the third was thick (76.2mm-102 mm) (Fig. 3).

Benthic microalgal biomass sediment cores were taken following completion of sediment addition to all sites in June 2000. For the next thirteen months (June 2000- July 2001) eight samples were taken within each of the individual sediment addition depth segments, within each of the four amended locations, on a monthly basis. Eight samples were also taken at each of the two control sites during all sample collections. A total of 112 total samples were taken every sampling period post-sediment addition. Samples were analyzed in the lab for benthic chlorophyll *a* mg m^{-2} using the double extraction, spectrophotometric method of Whitney and Darley (1979).

The effects of sampling time and marsh health on pre-sediment addition concentrations of benthic microalgal biomass were analyzed by two-way analysis of variance (ANOVA) that examined both direct and interactive effects. The effects of varying sediment additions at different experimental sites over a 14-month period post-sediment addition on benthic microalgal biomass were analyzed by a three-way analysis of variance (ANOVA) that examined different effects. Differences among sediment addition treatments were analyzed using Tukey-Kramer *a posteriori* tests using the SAS statistical software package.

Figure 3. Diagram of sediment addition segments within study sites

6.4 m

6.4 m

Thin

Intermediate

Thick
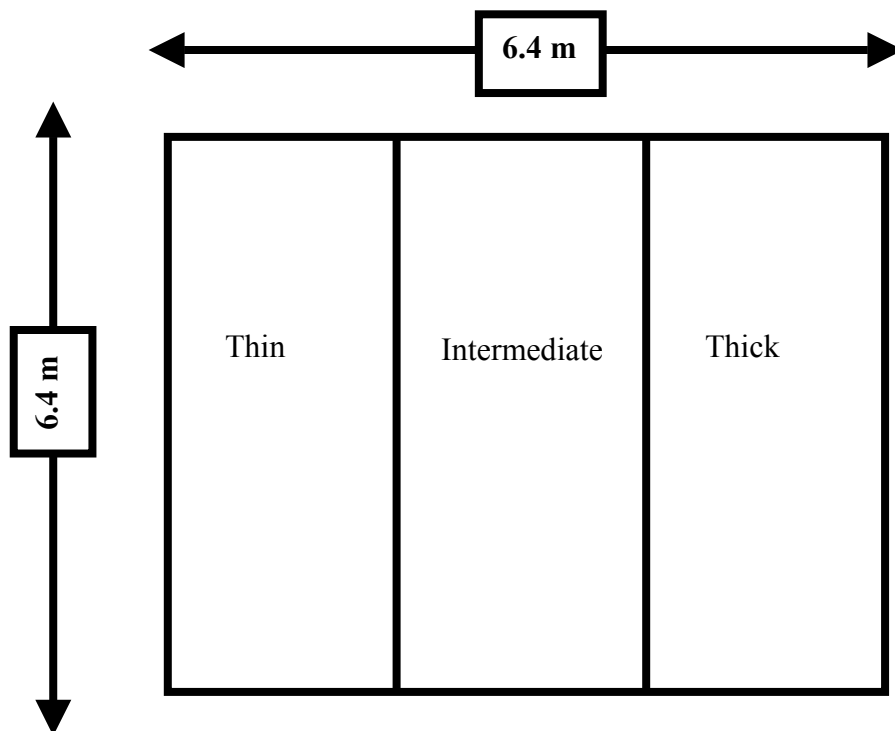
RESULTS

Pre-sediment addition

Individual values of benthic microalgal biomass taken at all sites prior to sediment placement ranged from 0.70mg chlorophyll a m^{-2} to 134.8 mg chlorophyll a m^{-2}. Two-way analysis of variance (ANOVA) of the effects of sampling time (January 2000-May 2000) and marsh health (non-deteriorated vs. deteriorated) on benthic microalgal biomass showed that both had significant ($p<0.05$) direct and interactive effects (Table 1). Benthic microalgal biomass was significantly higher in non-deteriorated sites prior to sediment addition sites (Fig. 4). Mean monthly chlorophyll a among all non-deteriorated sites before sediment addition ranged from 24.9 mg chlorophyll a m^{-2} to 117mg chlorophyll a m^{-2}. Non-deteriorated sites had a significantly higher mean of 66.2mg chlorophyll a m^{-2} while deteriorated sites had a mean of 13.3 mg chlorophyll a m^{-2}.

Prior to sediment addition, benthic microalgal biomass in the non-deteriorated sites varied over time (Fig. 4). Seasonal patterns were seen in the non-deteriorated site values with benthic microalgal peaks in January 2000 (monthly mean of 102.7mg chlorophyll a m^{-2}) and in May 2000 (monthly mean of 116.9mg chlorophyll a m^{-2}). Both of these values were well above the 66.7mg chlorophyll a m^{-2} average. Deteriorated sites deteriorated only in late May 2000 when the monthly mean of all pre-sediment addition sites increased to 68.8 mg chlorophyll a m^{-2}, which was well above the 21.7 mg chlorophyll a m^{-2} average.

Table 1. Comparison of pre-sediment addition benthic microalgal biomass *vs.* sampling time (Jan. 2000-May 2000) and marsh health (non-deteriorated *vs.* deteriorated) using two-way analysis of variance (ANOVA)

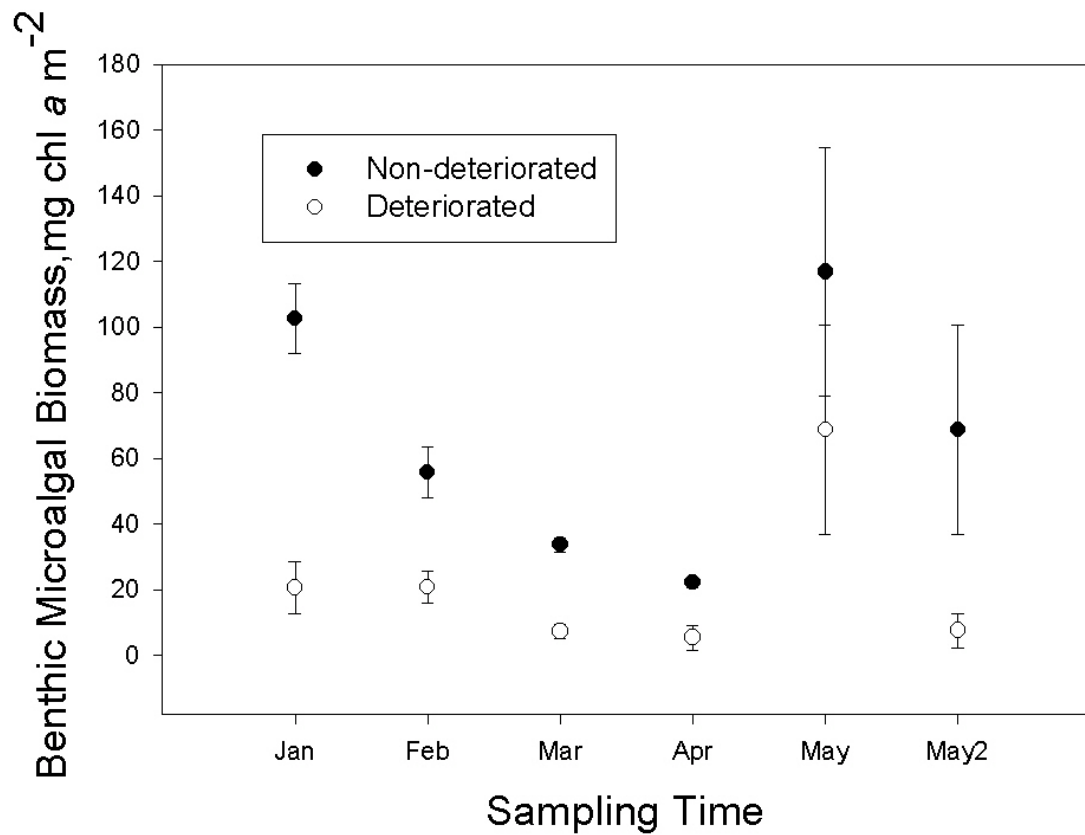| Effect	F	DF	*p*-level
Non-deteriorated/deteriorated	264.7	1, 95	<0.001
Time	27.74	5, 95	<0.001
Interaction	22.01	5, 95	<0.001

Figure 4. Comparison of mean mg chlorophyll *a* m^{-2} of non-deteriorated sites and deteriorated sites by month (January 2000-May 2000) prior to sediment addition. May2 indicates a second set of samples taken in May 2000 and error bars indicate standard deviation

Post-sediment addition

Individual values of benthic microalgal biomass taken at all post-sediment sites ranged from 2.81mg chlorophyll *a* m^{-2} to 189.7mg chlorophyll *a* m^{-2}. A three-way analysis of variance (ANOVA) of the effects of sampling time (June 2000-July 2001), sampling site (non-deteriorated vs. deteriorated), and sediment added (thin, intermediate, thick) on benthic microalgal biomass showed that sediment addition had significant ($p<0.05$) effects (Table 2). Mean benthic microalgae biomass was significantly ($p<0.05$) different between control sites and amended sites post-sediment addition. The unamended deteriorated control site continued to exhibit low monthly benthic microalgal biomass means during the fourteen months of sampling post-sediment addition. Monthly means were consistently low with no apparent seasonal patterns. Average benthic microalgal biomass ranged from 5.01 mg chlorophyll *a* m^{-2} to 49.0 mg chlorophyll *a* m^{-2}. Monthly means were always lower than the unamended non-deteriorated and all amended sites (Fig. 5).

In the fourteen months following sediment addition, the unamended non-deteriorated control site exhibited higher mean benthic microalgal biomass than the unamended deteriorated site, but lower monthly means than all (non-deteriorated and deteriorated) amended sites (Fig. 5). Monthly means ranged from 26.9mg chlorophyll *a* m^{-2} to 160.2mg chlorophyll *a* m^{-2} in the non-deteriorated control. The unamended non-deteriorated site also exhibited a seasonal pattern with a spring peak in benthic microalgal biomass in late May 2001.

Table 2. Comparison of post-sediment addition benthic microalgal biomass *vs.* sampling time (June. 2000-July 2001), sediment addition (thin, intermediate, thick) and sampling site (non-deteriorated *vs.* deteriorated) using three-way analysis of variance (ANOVA). *A posteriori* test (Tukey-Kramer) showed that all sediment additions (thin, intermediate, thick) were significantly ($p < 0.05$) different from no sediment addition but not different among them. An * indicates no significant difference

Variable		DF	p-level
Site		7	<0.001
Time		7	<0.001
Sediment addition:	Control *vs.* thin	7	<0.001
	Control *vs.* intermediate	7	<0.001
	Control *vs.* thick	7	<0.001
	*Thin *vs.* intermediate	7	<0.131
	*Thin *vs.* thick	7	<0.224
	*Intermediate *vs.* thick	7	<0.719

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Figure 5. Comparison of all unamended sites and post-sediment addition average
monthly chlorophyll *a* of amended sites where ND=non-deteriorated sites,
D=deteriorated sites, and U=unamended sites.

The increases in BMA biomass in both amended non-deteriorated and amended deteriorated sites were significantly higher than the respective control sites (Fig. 5). Post-sediment addition exhibited a rapid rise in benthic microalgal biomass. The mean BMA biomass increased from an average of 66.7 mg chlorophyll *a* m^{-2} in non-deteriorated sites to 90.1mg chlorophyll *a* m^{-2}, while deteriorated sites increased from an average of 21.8 mg chlorophyll *a* m^{-2} to 92.7 mg chlorophyll *a* m^{-2}. The levels obtained and subsequently sustained post-sediment addition were similar to, if not even higher than, benthic chlorophyll *a* levels observed in the unamended non-deteriorated site during the spring peak.

Furthermore, *a posteriori* Tukey-Kramer means comparison showed that there were significantly greater amounts of benthic microalgae biomass present in all amended sites compared to the control sites. However, a least squares means comparison of the effects of sediment addition (thin, intermediate, thick) on marsh health (non-deteriorated vs. deteriorated) of benthic microalgal biomass showed that while the addition of sediment was significant, the amount of sediment added showed no direct effect (Table 2). Significant changes in mean benthic microalgae biomass seen only between zero sediment addition (or control sites) and all sediment additions (thin, intermediate, thick), indicate that the addition of sediment, not the depth of the sediment added, was the important effect.

DISCUSSION

Results of this study suggest that a healthy or non-deteriorated pre-existing marsh habitat, which was determined by percent cover of healthy *Spartina alterniflora* growth, supports higher benthic microalgae biomass. The overall increase in benthic microalgal biomass observed in this study was paralleled by an increase in *Spartina alterniflora* stem densities as measured in a complimentary part of this project by Croft (2003). *Spartina alterniflora* stem densities significantly increased following sediment additions in both non-deteriorated and deteriorated sites. The increase in *Spartina* stem densities, which would create shading or a canopy over the marsh surface, might have had a negative effect on benthic microalgal biomass, but did not. One possible reason is that there was not enough light attenuation by the *Spartina* to prevent microalgal growth. A second possibility could be that there was, in fact, another covariate related to *Spartina* growth that had a direct positive effect on increasing benthic microalgal biomass.

Previous studies conducted in other local marshes, including Masonboro Island found benthic microalgal biomass values similar to those measured in the non-amended sites during this study (Table 3). Sigmon (1995) and Freeman (1989) looked specifically at Masonboro Sound and documented average monthly chlorophyll *a* concentration of 20-150 mg chlorophyll *a* m^{-2} and >100 mg chlorophyll *a* m^{-2}, respectively comparable to monthly chlorophyll *a* means found in this study. Slightly lower sediment chlorophyll *a* means were reported by Becker (1996) in both Pages Creek (45-50 mg chlorophyll *a* m^{-2}) and Bradley Creek (25-35 mg chlorophyll *a* m^{-2}), North Carolina. Benthic microalgae biomass in amended sites from this study ranged from 3.61 up to 189.7 mg chlorophyll *a* m^{-2}, which is significantly higher than all averages from previous studies of nearby

Table 3. Mean benthic microalgal biomass in mg chlorophyll a m^{-2} from previous studies in nearby North Carolina estuaries

Study	Study site	Mean chl a mg m^{-2}
Becker (1996)	Pages Creek	25-35 mg chl a m^{-2}
Cahoon and Cooke (1992)	Onslow Bay	36.4 mg chl a m^{-2}
Freeman (1989)	Masonboro Sound	>100 mg chl a m^{-2}
Sigmon (1995)	Masonboro Sound	20-150 mg chl a m^{-2}
Panasik (2003)	Masonboro Island	pre-sed: 0.70 - 134.8 mg chl a m^{-2} post-sed: 3.61 -189.7 mg chl a m^{-2}

marsh systems.

Addition of dredged sediment material supported mean monthly BMB that was almost always greater than at the control sites with the exception of samples taken at amended sites immediately following dredge sediment addition. This drop in chlorophyll *a* levels could be attributed to the change in sediment when the dredged material was added. The amended material was abiotic, therefore new colonization by benthic microalgae would be necessary. Samples were taken within one week of marsh amendment, possibly not giving enough time for new colonies to completely form and began production of chlorophyll *a*.

Changes in substrate grain size also may have affected benthic microalgal biomass. Mean sediment grain size has often been indicated as a possible determinant of BMA biomass (Freeman 1989, Kennett and Hargraves 1985, Amspoker and McIntire 1978, Davis and Lee 1983, Cahoon et al. 1999). These studies suggest that an increase in the grain size of the sediment added is consistent with an increase in benthic microalgal biomass. In this study, the mean sediment grain size of fill sediment was approximately 0.56 mm at the time of sediment addition (Croft 2003). These grain sizes are usually associated with epipsammic diatom assemblages (Freeman 1989). Prior to sediment addition, the mean grain size was approximately 0.10 mm in non-deteriorated sites and 0.11 mm in deteriorated sites (Croft 2003). These are usually associated with epipelic diatom assemblages (Freeman 1989). The marsh sediment composition prior to sediment addition was estimated at approximately 50 % fine silts and 50 % fine grained sands. Mean grain size post-sediment addition was 0.244mm and 0.321mm for non-deteriorated and deteriorated sites, respectively. Chlorophyll *a* increased in all sites post-sediment

addition and there seemed to be a significant positive correlation between benthic

microalgal biomass and grain size measured in 2000 and 2001 (Fig. 6). As the percent of

sediment particles that are <125μm in diameter decreased, the monthly mean chlorophyll

a mg m-2 generally increased (Fig. 6). These results are similar to those of previous

studies, such as Cahoon et al. (1999), which showed similar relationships between grain

size and benthic microalgal biomass in other North Carolina estuarine systems.

Placing larger grain sized sediment on the marsh surface could change several

parameters in the system, which might also contribute to an increase in benthic

microalgal biomass. A larger grain size plus more inorganic quartz may allow for greater

light penetration into the sediment, possibly increasing biomass. There will also be more

pore space between sediment particles, which will facilitate greater pore water flow and

diffusion. This pore water could carry nutrients essential to microalgal productivity,

including nitrogen and phosphorus. Although sufficient data were not collected during

this study to examine the effect of grain size, sediment texture can not be ruled out as a

contributing factor in increased benthic microalgal biomass.

Increase in marsh elevation due to the addition of sediment to the marsh surface

most likely did not have an effect leading to the observed increase of BMB in all

amended sites. The original deteriorated control site was 22.5 cm lower in elevation than

the non-deteriorated control site. After sediment addition, deteriorated sites were
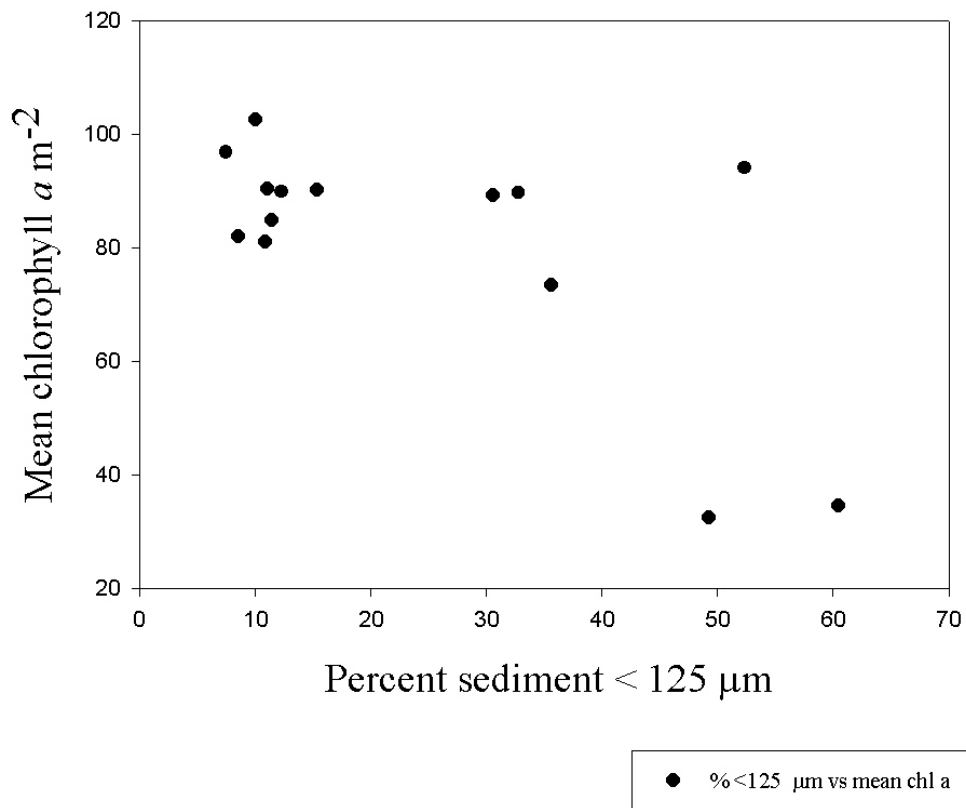
Figure 6. Scatterplot of mean chlorophyll *a* mg m^{-2} when plotted against mean sediment grain size that is <125 μm. Higher chlorophyll *a* numbers are expressed when the percent of sediment <125 μm is low. Grain size data from Croft (2003).

13.4 cm and 7.3 cm lower than the non-deteriorated control site yet an increase in BMA occurred at all amended sites. These data suggest that the addition of sediment may have affected the increase in BMA biomass, but that effect of the amount of sediment added was not significantly different between sediment additions within the amended areas (Table 2). Because elevation was not manipulated in this study and because all sites had different beginning elevations, there is little evidence to evaluate the role that a change in elevation may have had. However, elevation, within certain bounds, can not be ruled out as a factor that could help to increase benthic microalgal biomass.

Further experimentation is necessary to assess the full effects of sediment addition to marsh surfaces because of the important role that marsh ecosystems play. Healthy marsh systems can potentially supply large amounts of benthic microalgae, which are important in nutrient cycling (Sigmon and Cahoon 1997), support deposit and suspension feeders (Cahoon et al., 1999), and are significant primary producers (Mallin et al. 1992, MacIntyre et al. 1996, Cahoon et al. 1999). The amount of sediment added appears to be a non-factor, at least within the ranges used here, but the addition of too much sediment to the marsh surface could affect other factors such as benthic infauna, which are important consumers of BMA (Hilterman 1998). It is clear, however, that the addition of a fine to medium grained sediment to the marsh surface, which allowed for an increase in marsh elevation, was beneficial to the production of BMA in sites that were not only previously deteriorated, but also to those that were previously classified as non-deteriorated. Overall, the effects of sediment addition showed positive effects for both types of marsh.