

Quantifying Eelgrass Habitat Loss in Relation to Housing Development and Nitrogen Loading in Waquoit Bay, Massachusetts

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ABSTRACT: Change analysis of eelgrass distribution in Waquoit Bay demonstrated a rapid decline of eelgrass habitat between 1987 and 1992. Aerial photography and ground-truth assessments of eelgrass distribution in the Waquoit Bay National Estuarine Research Reserve documented progressive loss in eelgrass acreage and fragmentation of eelgrass beds that we relate to the degree of housing development and associated nitrogen loading, largely via groundwater, within various sub-basins of the estuary. The sub-basins with greater housing density and higher nitrogen loading rates showed more rapid rates of eelgrass decline. In eelgrass mesocosm studies at the Jackson Estuarine Laboratory, excessive nitrogen loading stimulated proliferation of algal competitors (epiphytes, macroalgae, and phytoplankton) that shade and thereby stress eelgrass. We saw domination by each of these three algal competitors in our field observations of eelgrass decline in Waquoit Bay. Our study is the first to relate housing development and nitrogen loading rates to eelgrass habitat loss. These results for the Waquoit Bay watershed provide supporting evidence for management to limit development that results in groundwater nitrogen loading and to initiate remedial action in order to reverse trends in eelgrass habitat loss.

Introduction

Seagrasses around the world are declining as a result of various types of disturbance in coastal and estuarine environments (Short and Wyllie-Echeverria 1996). Losses of eelgrass (*Zostera marina* L.) habitat in Chesapeake Bay, United States, have resulted primarily from declines in water quality linked to upland development, agriculture, and shoreline construction (Orth and Moore 1983; Dennison et al. 1993). Extensive seagrass declines in Australia have been ascribed to a variety of human-induced and natural changes (Larkum et al. 1989; Walker and McComb 1992). The best documented example of pollution-induced seagrass decline in Australia occurred in Cockburn Sound, where both industrial and domestic pollution were implicated as causes of extensive die-off (Cambridge and McComb 1984). Numerous studies in the Mediterranean, along the Atlantic coast of Europe, and elsewhere have also linked losses in seagrass habitat with increased levels of pollution (Nienhuis 1983; Hanekom and Baird 1988; Giesen et al. 1990; Short et al. 1991; De Jonge and De

Jonge 1992; den Hartog 1994). The effects of nutrient pollution on seagrass distribution have been clearly demonstrated in a case study of Hillsborough Bay, Florida, United States (Johansson and Lewis 1992). The study showed extensive loss of seagrass under polluted conditions during the 1970s, followed by a rapid recovery of seagrass during the 1980s after fertilizer and waste entering Hillsborough Bay were reduced. In this situation, the recovery response of seagrass was related to the decline in nutrient loading. A similar resurgence has been observed in freshwater rooted aquatic plants in the upper reaches of Chesapeake Bay following a dry spring with low runoff and low nitrogen loading rates (Stevenson et al. 1993).

Besides pollution, declines in seagrass populations have been caused by outbreaks of disease (Short et al. 1986), particularly by the eelgrass wasting disease during the 1930s on the Atlantic coast of both Europe and the United States. The wasting disease has recurred during the 1980s and 1990s along the east coast of the United States (Short et al. 1986, 1988; Muehlstein et al. 1991). Severity of wasting disease outbreaks in eelgrass populations can now be assessed quantitatively (Burdick et al. 1993) using an index that deter-

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mines if disease is a factor in eelgrass decline (Short et al. 1993).

Eutrophication from nutrient loading in coastal and estuarine environments is a growing problem due to increased human population and development (Hinga et al. 1991). In southern New England, nitrate-enriched groundwater was found to be the source of 71% to 97% of nitrogen loading to several shallow coastal estuaries (Lee and Olsen 1985; Valiela et al. 1990). Increased housing in coastal areas with well-drained, sandy soils leads to increases in nitrogen input to the groundwater because even properly operating traditional septic systems in these permeable soils do not adequately remove nitrate before the wastewater enters the groundwater (Valiela et al. 1992). The impact of eutrophication on seagrass, and specifically on eelgrass, has been associated with the growth of competitive algal forms stimulated by excess nutrients. Descriptive field studies have found that the algae appeared to inhibit or eliminate eelgrass (Kemp et al. 1983; Short et al. 1991; Costa et al. 1992; Denison et al. 1993). Additionally, experimental mesocosm studies have shown that nutrient loading can reduce eelgrass productivity and health by stimulating algal competition (Short et al. 1991, 1995; Neckles et al. 1993; Williams and Ruckelshaus 1993) and by direct nitrate toxicity (Burkholder et al. 1992). In our mesocosm experiments, competition for available light resources under eutrophied conditions always included substantial eelgrass losses coupled with algal gains and resulted in either macroalgal, epiphytic algal, or phytoplankton dominance (Short et al. 1989, 1995).

The documentation of seagrass habitat loss has become a major focus for a number of federal programs in the United States (e.g., Klemas et al. 1993) as well as in Australia (Shepherd et al. 1989). Aerial photography has been recommended as the preferred methodology for measuring seagrass habitat loss (Ferguson and Wood 1990; Klemas et al. 1993). Change analysis of seagrass distribution is based on maps developed from aerial photography acquired over a period of time. To date, only qualitative studies have been used to demonstrate changes in seagrass distribution (Orth and Moore 1983; Shepherd et al. 1989; Short et al. 1989; Kirkman and Kuo 1990; Ferguson and Wood 1990; Costa et al. 1992). Here we quantitatively measure changes in eelgrass habitat in Waquoit Bay using computer analysis of spatial images in a series of four maps based on aerial photography spanning six years. We then relate these changes in eelgrass habitat to housing and nitrogen loading rates in different parts of the Waquoit Bay estuarine system.

Methods

Spatial distribution of eelgrass in Waquoit Bay, Massachusetts, a National Estuarine Research Reserve, was assessed on August 11, 1987, August 31, 1988, October 6, 1989, and July 31, 1992. Near-vertical aerial photography taken with a hand-held 35 mm camera from a light plane at 1,000 m elevation was used from 1987 through 1989, while in 1992 large format vertical photography was obtained. In the past, change analyses of eelgrass habitats have been done using large format (23 × 23 cm) vertical imagery in color, color infrared, and black and white (Orth and Moore 1983; Ferguson and Wood 1990). Additionally, black and white historical photography has been used (Costa et al. 1992). Although each of these methods has its advantages and limitations, the National Oceanic and Atmospheric Administration's Coastal Ocean Program (Coastal Change Analysis Program: C-CAP) has recommended the use of large format vertical color imagery (Klemas et al. 1993). To this end, our aerial surveys of eelgrass in 1992 followed C-CAP protocol (Klemas et al. 1993); the vertical imagery was taken at 400 m altitude. Photography for all years was obtained as true color transparencies. Although the 1987 through 1989 photography was not obtained using the preferred large format, our experience indicates that 35 mm near-vertical photography is a reliable aerial assessment technique for seagrass (Short et al. 1986, 1989, 1993), as has been shown for emergent wetlands (Clegg and Sherz 1975).

Ground-truth assessment was conducted each year by surveying the various sub-basins of the estuary from a small boat during mid-to-low tides (Fig. 1). From ground truthing, it was evident that Waquoit Bay was shallow enough that the deep edge of all eelgrass beds could be distinguished in aerial photography. The presence and absence of eelgrass and the occurrence of various algae within the eelgrass meadows were noted on a standard base map of Waquoit Bay with outlines of submerged vegetation distribution created from a particular year's aerial photography. The standard base map was created from 1986 aerial imagery since no map existed that adequately depicted the Waquoit Bay shoreline. Samples of eelgrass were collected in each of the sub-basins to characterize the morphology of the eelgrass populations and to assess wasting disease infection; these data have been reported elsewhere (Short et al. 1993).

The ground-truthed eelgrass distribution was mapped from the aerial photography onto the base map. Distribution maps were created for each year and electronically scanned into a Macintosh II computer for reproduction and image analysis.

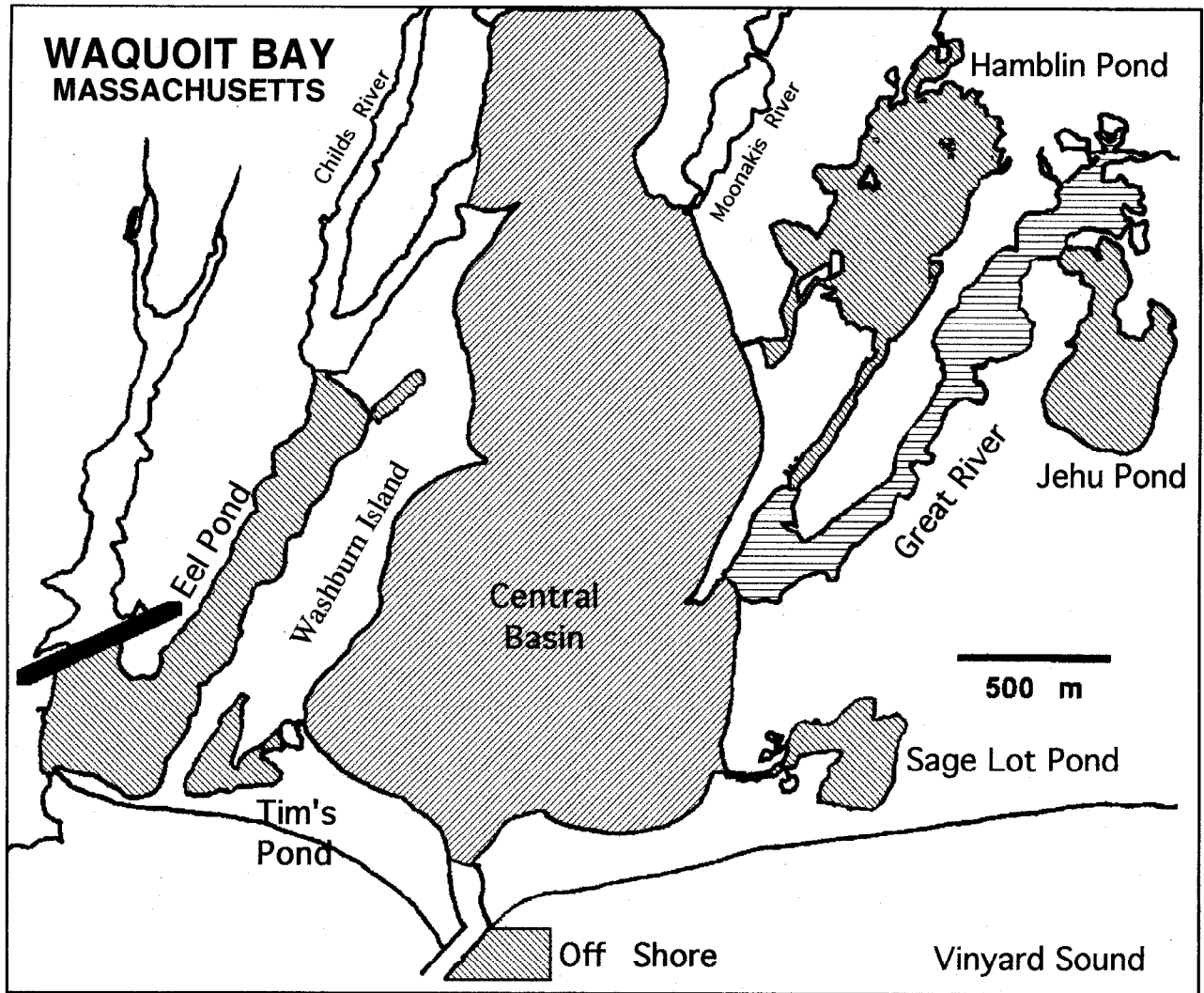


Fig. 1. Map of Waquoit Bay, Massachusetts, showing the sub-basins of the estuary.

Images were analyzed on the computer to count the number of pixels in each sub-basin designated as eelgrass using IMAGE software (Rasband 1993). Spatial analysis of each distribution map included the area of eelgrass habitat in each of the estuarine sub-basins as well as in the entire estuarine system. The minimum detectable size of a shallow eelgrass patch for the near-vertical aerial photography at low tide was 0.25 m^2 , as determined in Great Bay, New Hampshire, by intermittently photographing transplanted eelgrass shoots on 0.5 meter centers at 1,000 m elevation. Change analysis of eelgrass distribution was performed for each of the sub-basins of Waquoit Bay and for the entire estuary by subtracting the area of eelgrass in an annual assessment from that determined for the subsequent year.

Information on nitrogen loading and the mag-

nitude of housing development within the Waquoit Bay watershed was obtained from National Science Foundation-funded Waquoit Bay Land Margin Ecosystem Research (LMER) scientists (Valiela et al. 1992, In press). These researchers determined nitrogen loading by calculating the measured area of upland according to land use type within each sub-basin watershed and multiplying by nitrogen inputs for each land use type within the watershed. The number of houses within each watershed was determined by direct counts from aerial photography by LMER scientists. To compare these two measures of development, nitrogen loading rates were regressed against the number of houses in the watershed of each sub-basin using simple linear regression. Housing density was calculated for each sub-basin based on associated watershed area (Valiela et al. 1992).

Linear regression analysis was used to examine and compare trends in the eelgrass coverage in the sub-basins of Waquoit Bay over time (1987–1992). Sub-basins were blocks and year was the independent variable in the SuperANOVA procedure (Abacus Concepts 1989); eelgrass area was log transformed to reduce error heterogeneity. Loss in eelgrass area from 1987 to 1989 was examined by regressing the log of eelgrass area for all sub-basins (total) on year (Abacus Concepts 1992), and a prediction for eelgrass area was made for 1992 for comparison with the results of the 1992 large format photography.

Eelgrass area (% of the total area within each sub-basin) for the years 1987–1989 was analyzed using two regression models to assess the effects of housing development on eelgrass losses. The first model included year, number of houses for each watershed, and their interaction; the second model included year, estimated nitrogen loading for each watershed, and their interaction. To improve fit and reduce error heterogeneity, eelgrass area was transformed using the common logarithm. Although loading is estimated, it is used as a regressor in the analysis of eelgrass cover because the range of loadings examined is relatively large (150 to 25,000 kg km⁻² yr⁻¹), and the variability is small relative to the number of houses in each sub-basin.

Results

Aerial surveys of Waquoit Bay made on August 11, 1987, together with ground-truth observations, provide our baseline of eelgrass distribution (Fig. 2a). We found a large area of eelgrass within the south central part of Waquoit Bay that was split by dredged boat channels, extensive eelgrass beds covering both sides of the channel in Eel Pond, nearly continuous beds along the shore of Great River, wide distribution of eelgrass in Jehu, Hamblin, and Tim's ponds, and small beds of eelgrass outside the channel mouth in Vinyard Sound (Fig. 2a). Childs River, Moonakis River, and the northern end of the central bay (Fig. 1) were found to have no eelgrass in 1987–1992. We found extensive eelgrass in Sage Lot Pond during our ground-truthing survey, but this sub-basin was not included in the 1987 aerial photography.

On August 31, 1988, eelgrass beds were again photographed in each of the estuarine sub-basins. Some reductions in bed size in the Central Basin, Eel Pond, and the Great River were apparent, as was some expansion of eelgrass beds in Hamblin Pond (Fig. 2b). In 1989, aerial eelgrass photographic surveys were made on October 6, and the losses of the previous year were still evident (Fig. 2c). Even greater losses of eelgrass area were seen in Eel Pond, the Central Basin, and the Great Riv-

er, as well as substantial losses in Hamblin Pond (Fig. 2c). The losses in Hamblin Pond were sizeable and more than negated the gains of 1988 (Fig. 2b). Also in 1989, Sage Lot Pond was mapped by air for the first time and was found to have 80% of its total area covered by eelgrass (Fig. 2c). After a 2-yr hiatus, the eelgrass beds in Waquoit Bay were mapped with large format vertical photography taken July 31, 1992 (Fig. 2d). Further declines in eelgrass distribution were evident within the Central Basin, Eel Pond, Great River, Hamblin Pond, and Jehu Pond. No substantial changes within Sage Lot Pond and Tim's Pond were evident from the 1992 aerial photography.

The area of eelgrass in each of the Waquoit Bay sub-basins (Fig. 1) was calculated and plotted versus time to show temporal trends in eelgrass distribution (Fig. 3). The rate of loss in eelgrass area within each sub-basin was calculated as the slope of the linear regression of change in eelgrass area over time. After blocking on sub-basins, which had significant differences in eelgrass bed area, we found significant declines in most sub-basins and significantly different rates of decline between sub-basins. Our model describes 98.5% of the variance, with $F = 67.6$ and $p < 0.0001$. The Central Basin and Great River sub-basins had significantly greater rates of loss than Tim's Pond and the Off Shore site, with the rates of loss in the other sub-basins intermediate (Fig. 3). The Central Basin of Waquoit Bay as well as the Great River showed the highest and most variable absolute rates of eelgrass loss over the 5-yr period of our study (ranging from 0.019 km² yr⁻¹ to 0.028 km² yr⁻¹). Hamblin, Eel, and Jehu ponds showed steady declines in eelgrass area, with similar rates of eelgrass areal loss for each (loss rate ranging from 0.014 km² yr⁻¹ to 0.017 km² yr⁻¹). Eelgrass in Tim's Pond on undeveloped Washburn Island and at the Off Shore site showed little-to-no evidence of area change (gain of 0.0003 km² yr⁻¹ and loss of 0.0003 km² yr⁻¹, respectively).

We used information from the Waquoit Bay LMER study to develop a regression of percent eelgrass area (relative to total area of the sub-basin) as a function of year and the number of houses associated with each sub-basin (Fig. 4). This model was highly significant ($F = 52.3$, $p < 0.0001$, $r^2 = 0.923$). The percent area of eelgrass was significantly lower in sub-basins with more houses, and decreased over time, but neither the effect of time nor the time-by-house interaction was significant. Therefore, only the house effect averaged over time is presented in Fig. 4. Besides direct counts of housing, LMER scientists calculated the rate of nitrogen loading from various sub-watersheds of the Waquoit Bay Estuary (Valiela et al. in press). We related these nitrogen loading rates to the ex-

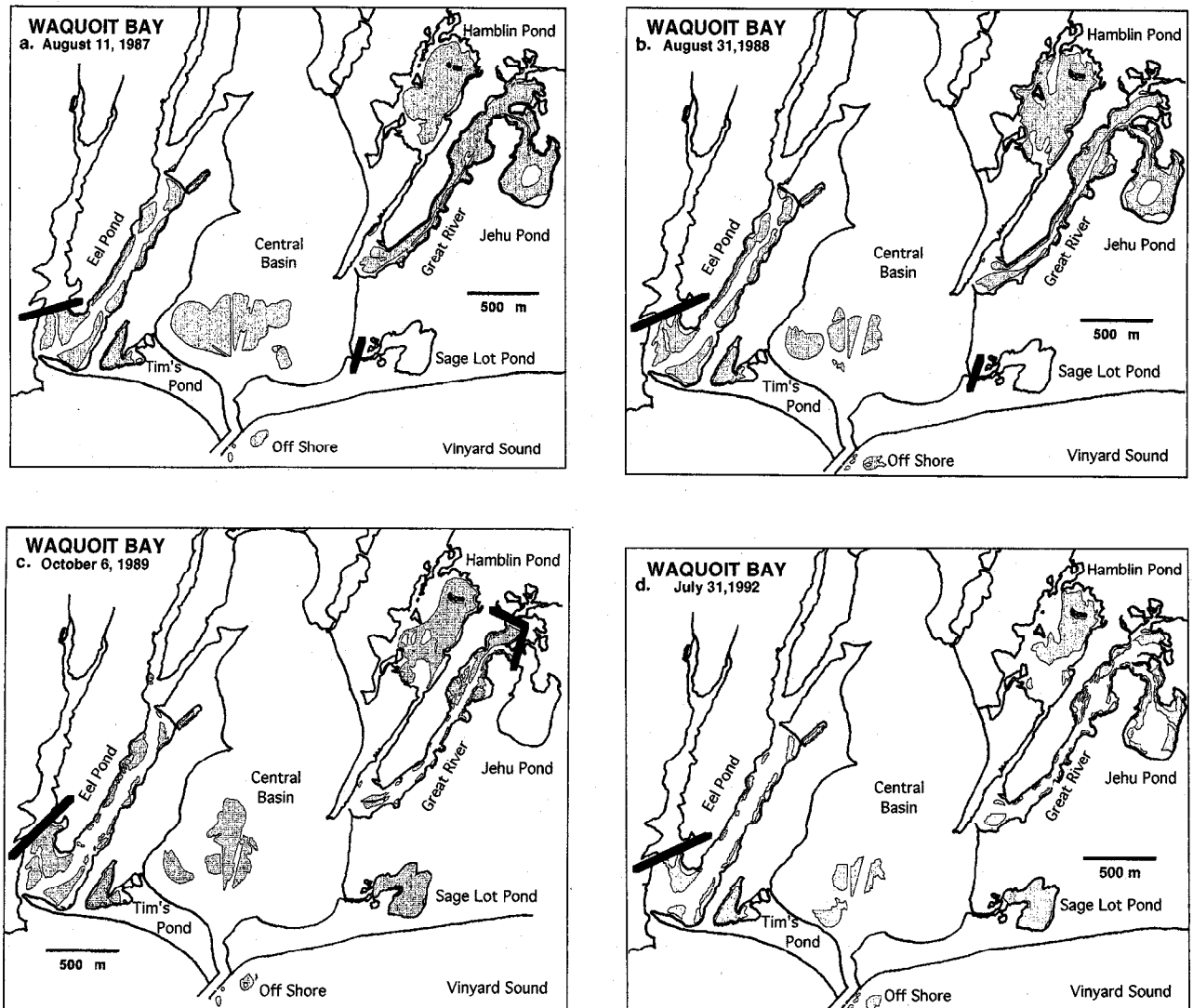


Fig. 2. Maps of eelgrass, *Zostera marina*, distribution (shaded area) in Waquoit Bay on (a) August 11, 1987, (b) August 31, 1988, (c) October 6, 1989 and (d) July 31, 1992. Black bars denote the extent of estuary surveyed.

tent of housing development (Fig. 5) for each of the sub-basins where we assessed eelgrass area. There was a significant linear regression between nitrogen loading rates and number of houses in the watershed for each sub-basin ($F = 655.0$, $p < 0.0001$, $r^2 = 0.962$).

Information on nitrogen loading rates and eelgrass area in the estuary's sub-basins over the first 3 yr of study was used to develop a regression model relating eelgrass area to estimated nitrogen loading rates for Waquoit Bay. As in the housing model, year was used as a covariate, and the model was highly significant ($F = 43.6$, $p < 0.0001$, $r^2 = 0.910$), but again the year and interaction effects were not significant. A statistically significant loga-

rithmic relationship ($F = 127.3$, $p < 0.001$, $r^2 = 0.888$) expresses the change in percent eelgrass area within the sub-basins as a function of nitrogen loading rate (Fig. 6).

Total combined eelgrass area for the Waquoit Bay Estuary was determined for each of the first 3 yr of distribution assessments, and regressed on years to produce a model of eelgrass area change over time ($F = 80.5$, $p < 0.0707$; Fig. 7) prior to the 1992 photographic analysis. Using this model, a predicted eelgrass area for 1992 (0.321 km^2) was calculated for comparison to the area measured from aerial photographs (0.358 km^2). The measured area was 12% higher than that predicted by the regression.

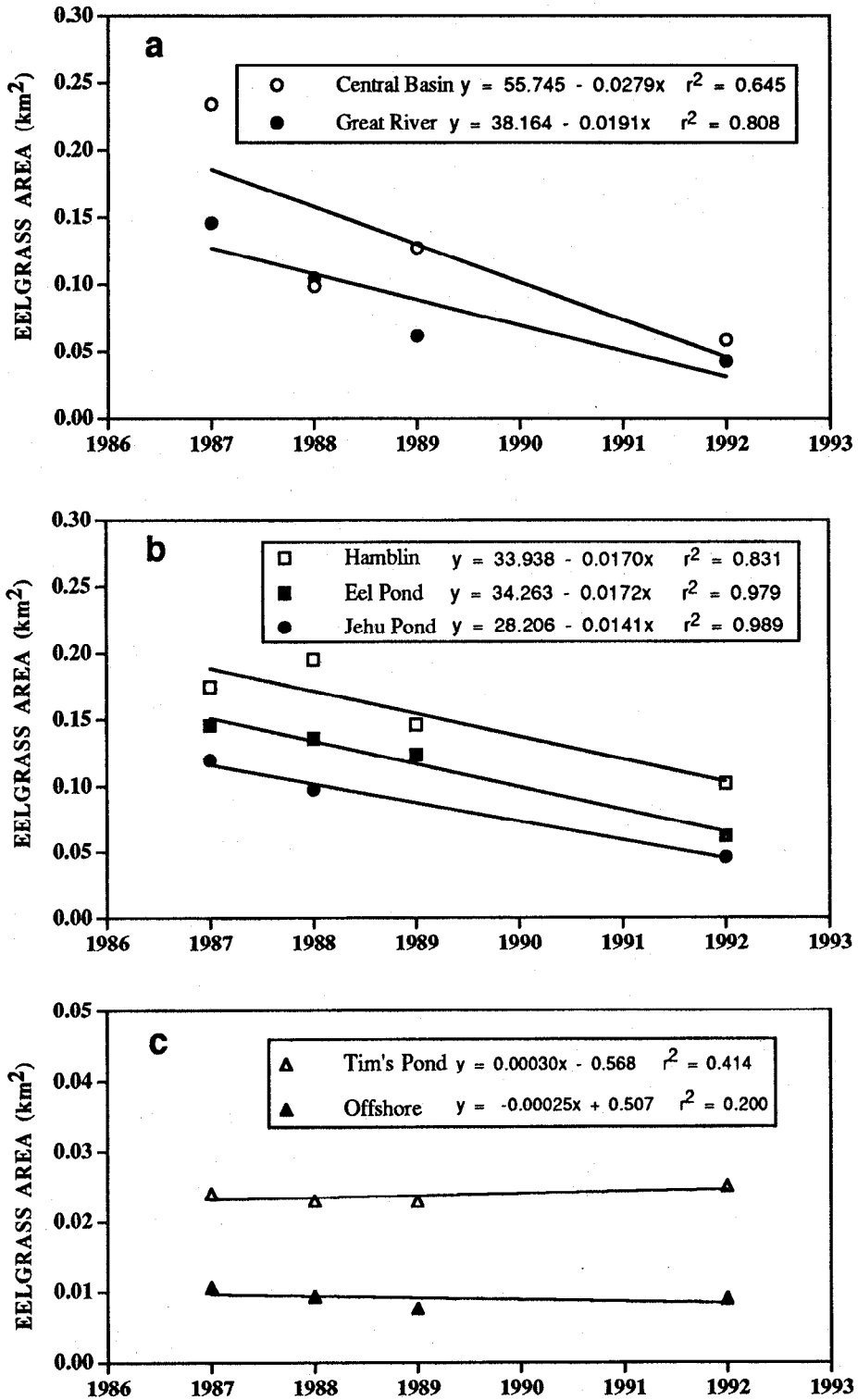


Fig. 3. Area of eelgrass beds in each of the sub-basins of Waquoit Bay for the years 1987–1989 and 1992. Eelgrass areas derived from image analysis of eelgrass distribution maps (Figs. 2a–d). Linear regressions for each sub-basin show the rate of habitat loss based on change in eelgrass area over the four mapping dates.

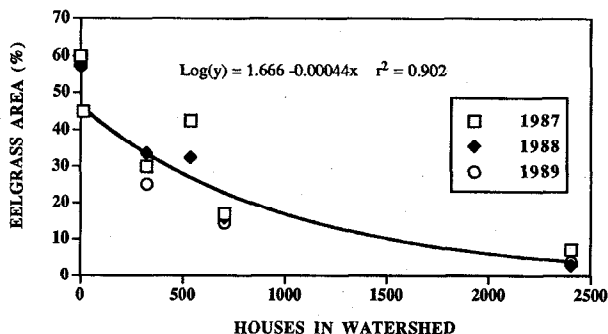


Fig. 4. Number of houses in the watersheds of Waquoit Bay compared to percent eelgrass area within each associated sub-basin. The log of eelgrass area for the four years of study was regressed against number of houses.

Discussion

Nutrient loading from human activities has long been identified as a source of eutrophication in coastal and estuarine waters (Ryther and Dunstan 1971; Nixon and Pilson 1983). The impacts of nitrogen loading on plankton-dominated estuarine systems have been clearly shown (see reviews: Elmgren 1989; Hinga et al. 1991; Kautsky 1991). Also, the degrading eutrophic effects of nitrogen loading on seagrass-dominated ecosystems have been identified (Short et al. 1991, 1995; Costa et al. 1992; Valiela et al. 1992; Dennison et al. 1993). However, these studies did not directly relate nitrogen loading rates or upland development to seagrass habitat loss. A Florida study provides the best and most compelling evidence to date of the effects of nutrient loading on seagrass ecosystems (Johansson and Lewis 1992). Their study illustrated the link between seagrass distribution and anthropogenic inputs by demonstrating the rapid recovery of seagrass habitat after reduction of nutri-

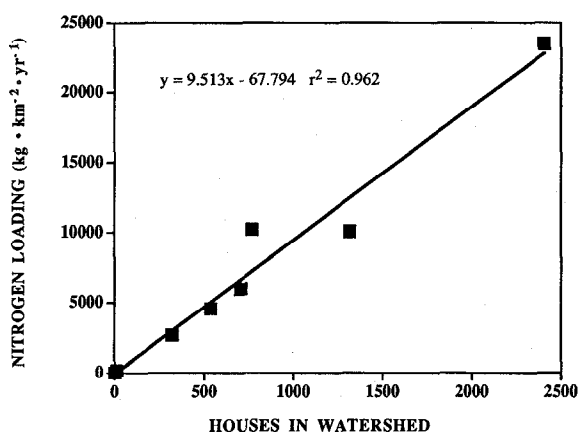


Fig. 5. The nitrogen loading rates for each sub-basin compared to the extent of housing development in the watershed associated with each sub-basin using linear regression.

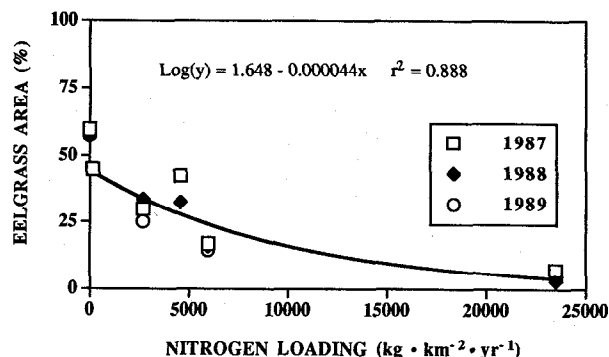


Fig. 6. Nitrogen loading rates and eelgrass area in the Waquoit Bay estuary's sub-basins over the first three years of study are compared. The log of eelgrass area is regressed against loading.

ent stress. In Waquoit Bay, the source of nutrient stress has been identified as groundwater-borne nutrients (largely domestic wastewater) from housing within the watershed, which have resulted in nutrient enrichment of the estuary (Valiela et al. 1990, 1992). Our study takes the additional step of quantitatively relating the area of seagrass habitat in sub-basins of Waquoit Bay to the magnitude of anthropogenic nutrient loading in the sub-watersheds of this estuarine system.

In Waquoit Bay the distribution of eelgrass showed a dramatic decline from 1987 through 1992 (Figs. 2a-d), which may be a continuation of earlier eelgrass declines reported by Costa et al. (1992). We found that the character and rate of decline were not uniform throughout the estuary, with substantial variation between sub-basins (Fig. 3). The areas most isolated from groundwater nitrogen inputs showed little or no net change in eelgrass extent over the period of our study (Fig.

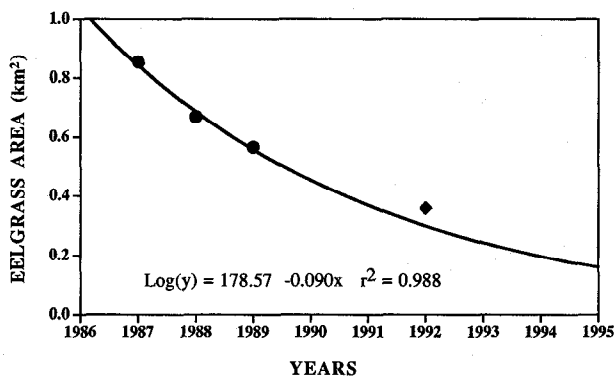


Fig. 7. Total combined eelgrass area (log) for the Waquoit Bay estuary determined for each of the first three years of the distribution assessments (circles) regressed on years to produce a regression model (line) of eelgrass area change over time. The regression model prediction is then compared to the observed eelgrass distribution for 1992 (diamond).

3c). The narrow parts of the estuarine system, including the sub-basins of Eel Pond and Great River, saw a decline and fragmentation of the once-continuous eelgrass beds that formed bands along the main channels of these waterways. Within the Central Basin of Waquoit Bay and in Hamblin Pond, loss of eelgrass was somewhat more variable from year to year, with some losses apparently offset by bed expansions, perhaps due to seed dispersal (Figs. 2a-d).

In assessing the cause of dramatic eelgrass decline in Waquoit Bay, we reviewed the most likely causes of eelgrass loss, including the possibility of eelgrass die-off from wasting disease, storms, boating, fishing, and the effects of suspended sediments, in addition to groundwater nitrogen loading. Assessment of wasting disease between 1987 and 1989 showed Waquoit Bay to have a low incidence of wasting disease, the lowest of any of four National Estuarine Research Reserves examined on the East Coast (Short et al. 1993). Thus, disease was not a major factor leading to the observed declines in eelgrass distribution in Waquoit Bay. Some of the eelgrass losses in the Central Basin of Waquoit Bay and in Eel Pond may have been caused by Hurricane Bob and a severe October storm, both in 1991, but we have no distribution data near the time of these events. In any case, these storms would have affected only the 1992 eelgrass distribution. The losses of eelgrass in Great River and Eel Pond may have been accelerated by increases in boating activity, although neither boats nor docks are a primary cause of decline (Burdick and Short 1995) and there have been no new boat channels dredged in Waquoit Bay in the past several decades. Recreational and commercial shell-fishing activity have the potential to impact eelgrass distribution (Short et al. 1989), but the major declines in shellfish occurred prior to our study (Valiela et al. 1992), resulting in reduced fishing effort in the remaining eelgrass beds. Suspended sediments can impact eelgrass distribution, but in Waquoit Bay there is very little surface runoff, with virtually all the freshwater flow entering the bay via groundwater plus spring-fed streams (Valiela et al. 1992). Thus, there is little anthropogenic sediment input, although there is some wind-driven sediment resuspension.

All of the eelgrass losses we observed within the Waquoit Bay estuarine system occurred where a variety of algal forms normally found in these environments exhibited proliferation to nuisance levels (Valiela et al. 1992). In different sub-basins of the estuary the character of eelgrass decline, and the algal competitor we found associated with that decline, varied. In Eel Pond and the mouth of Great River, competition between eelgrass and its epi-

phytic algae appeared to have led to the demise of eelgrass, the epiphytes growing so extensively that eelgrass blades were completely covered and the plants disappeared. In parts of Hamblin Pond and the Central Basin, the main algal competitor causing a decline in eelgrass habitats was unattached macroalgae (*Gracilaria* sp., *Cladophora* sp.), which smothered and crowded out eelgrass plants. Phytoplankton was the third type of algal competitor that was stimulated by nitrogen loading and out-competed eelgrass for available light. We found phytoplankton blooms in Jehu Pond that appeared to reduce the amount of light reaching eelgrass beds. These blooms obscured the aerial photography of eelgrass in 1989 to the extent that they could not be mapped in Jehu Pond, and may be responsible for the eelgrass decline in that portion of the estuary.

The effects of nutrient loading on eelgrass habitats have been investigated in experimental mesocosm studies (Short et al. 1991, 1995; Neckles et al. 1993; Williams and Ruckelshaus 1993). These studies show a decline in eelgrass biomass as a result of increased loading and a concomitant increase in algal competitors. Excessive nutrient loading to mesocosms at rates similar to those found in Waquoit Bay can result in the elimination of eelgrass (Short et al. 1995). Our mesocosm experiments quantitatively demonstrated that nitrogen loading stimulates the growth of algal groups, which outcompete eelgrass for available light. The same algal forms (epiphytes, macroalgae, and phytoplankton) responsible for eliminating eelgrass in the mesocosm experiments (Short et al. 1995) were found dominating the eelgrass habitats in Waquoit Bay's sub-basins that experienced eelgrass loss. The mesocosm experiments indicate that when nutrient loading to the water column greatly exceeds eelgrass requirements, one of the three types of algal competitors will gain dominance over eelgrass.

The measure of total eelgrass area remaining in the various sub-basins of Waquoit Bay in 1992 was related to housing development and nitrogen loading rates for each of the respective sub-basins. The percent of eelgrass remaining in the sub-basins was highest for watersheds with few houses and low loading rates and was exponentially lower in watersheds with progressively more houses and higher nitrogen loading rates (Figs. 4 and 6). Impacts to eelgrass from increasing housing development with septic systems located on sandy glacial outwash plains also have been observed along the southern coast of Rhode Island (Short et al. 1996). The statistically significant decline in eelgrass area as a function of house number (Fig. 4) and loading

rate (Fig. 6) clearly showed the impact of elevated nitrogen inputs on eelgrass habitat distribution.

Change analysis of eelgrass between 1987 and 1989 for the Waquoit Bay estuarine system showed a steady decline in eelgrass area that was fit with an exponential loss rate model prior to mapping the 1992 eelgrass distribution (Fig. 7). Projecting the loss of eelgrass area from this equation, we predicted eelgrass area in Waquoit Bay for the year 1992. Our prediction was 0.32 km², 12% lower than the measured 1992 eelgrass area of 0.36 km², based on assessment of large format vertical imagery. The relatively small difference between the predicted extent of eelgrass and the observed indicates that the descriptive regression model is useful in predicting eelgrass area. There is always some year-to-year variation in eelgrass distribution, particularly with respect to sampling date. Our observations of Waquoit Bay in 1993 suggest 1992 was a relatively good year for eelgrass in Waquoit Bay. The estimate of 0.32 km² for 1992 was lower than the measured area, but may better reflect the long-term trend for eelgrass distribution in the estuary, since we found that beds had shrunk substantially in the Central Basin, the lower Great River, and Eel Pond in 1993. This analysis demonstrates the utility of short term (3 yr) observations of nutrient-loading-related eelgrass declines in predicting longer term trends.

Our model predicts further decreases in eelgrass distribution in Waquoit Bay unless there is a reduction in nitrogen loading or management action that increases the flushing of the estuary. Without management, further eelgrass declines seem inevitable since the slow rate of groundwater movement insures that high nitrogen loading to the estuary will continue for some time, even if nutrient discharge from residential wastes ceased immediately. Our evidence indicates that there has been a dramatic loss of eelgrass in Waquoit Bay estuary, that this loss is predictable, and that it is closely related to the number of houses in the watershed and to nitrogen loading estimates developed from land use data.

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