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SeagrassNet monitoring across the Americas: case studies of seagrass decline

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Abstract

Seagrasses are an important coastal habitat worldwide and are indicative of environmental health at the critical land–sea interface. In many parts of the world, seagrasses are not well known, although they provide crucial functions and values to the world's oceans and to human populations dwelling along the coast. Established in 2001, SeagrassNet, a monitoring program for seagrasses worldwide, uses a standardized protocol for detecting change in seagrass habitat to capture both seagrass parameters and environmental variables. SeagrassNet is designed to statistically detect change over a relatively short time frame (1–2 years) through quarterly monitoring of permanent plots. Currently, SeagrassNet operates in 18 countries at 48 sites; at each site, a permanent transect is established and a team of people from the area collects data which is sent to the SeagrassNet database for analysis. We present five case studies based on SeagrassNet data from across the Americas (two sites in the USA, one in Belize, and two in Brazil) which have a common theme of seagrass decline; the study represents a first latitudinal comparison across a hemisphere using a common methodology. In two cases, rapid loss of seagrass was related to eutrophication, in two cases losses related to climate change, and in one case, the loss is attributed to a complex trophic interaction resulting from the presence of a marine protected area. SeagrassNet results provide documentation of seagrass change over time and allow us to make scientifically supported statements about the status of seagrass habitat and the extent of need for management action.

Problem

Seagrasses are important marine and estuarine plants worldwide. They create underwater meadows, forming an ecologically critical habitat throughout much of the world's coastal oceans (Larkum *et al.* 2006). Found in the nearshore waters of every continent except Antarctica, seagrass habitat is linked to mangroves and coral reefs in tropical oceans by both geographic proximity and trophic

interaction; in temperate oceans, seagrasses are similarly linked to salt marshes, kelp forests, and shellfish reefs (Hemminga & Duarte 2000). Seagrass beds are nursery areas for many commercial and recreational fisheries species, including finfish, shrimp, lobsters, and other shellfish (Heck *et al.* 2003). Seagrasses act as a filter of coastal waters, taking up nutrients and contaminants from the water and causing suspended sediment to settle (Short & Short 1984). Additionally, seagrasses are a part of the

food chain: as old plants die and break down, they become a part of the detritus that is eaten by small crustaceans, which in turn are preyed upon by fish. The important ecological roles of seagrass habitats have been identified and found to represent a high value of ecosystem services to the planet (Costanza *et al.* 1997).

Seagrasses are considered indicators of marine habitat degradation (Orth & Moore 1983; Dennison *et al.* 1993); increasingly, scientific studies document this concept (Johansson & Lewis 1992; Short & Burdick 1996). Seagrasses are positioned at the land–sea interface, encountering runoff and terrestrially derived pollution that decreases seawater clarity and quality. Under normal conditions, seagrasses thrive at this interface and are able to absorb some level of excess nutrients and other pollutants, but when pollution levels and coastal development increase, seagrasses can be stressed and decline or disappear (Dennison *et al.* 1993; Kirkman 1997; Boudouresque *et al.* 2000; Green & Short 2003). Shoreline development, industrial and residential pollution, and damaging fishery and aquaculture practices, all have contributed to losses of seagrasses around the world. Globally, the issue of seagrass habitat loss requires attention (Short & Wyllie-Echeverria 1996; Spalding *et al.* 2003): all too often, because of their submerged nature, seagrasses are overlooked when environmental consequences are examined. By monitoring these habitats, early detection of coastal environmental degradation can be made before coastal ecosystems experience irreparable loss so that corrective measures can be taken.

Despite progress in coastal management and information gathering, there is insufficient assessment of marine and estuarine habitat resources, particularly long-term trend evaluation based on scientific monitoring of critical habitats. Worldwide, consistent information on the seagrass habitat has not been previously collected; the monitoring that has been done has used various protocols, yielding few long-term comparable records (Short & Coles 2001). Through analysis of consistently gathered scientific data from a fixed transect within an indicator habitat, early detection of trends is possible (Burdick & Kendrick 2001).

The aim of SeagrassNet is to establish such seagrass monitoring worldwide, tracking the status of seagrasses as a measure of the trends in environmental health, using a standard protocol (Short *et al.* 2004, 2005). The program is an international effort that empowers participants in host countries with both scientific knowledge and environmental awareness. SeagrassNet teams are now monitoring the seagrass habitat four times a year, using the same protocol, in 18 countries at 48 sites worldwide.

Here, we present five case studies across the Americas (Fig. 1, Table 1), ranging from New Hampshire, USA to



Fig. 1. Location of the SeagrassNet sites across the Americas included in the study.

Abrolhos, Brazil, all based on SeagrassNet monitoring over the past 2–4 years. The sites include two north temperate and three tropical locations. Each of the five case studies presented here is a change analysis of a specific seagrass habitat with its particular seagrass species and other environmental components. As a group, the case studies encompass many of the impacts to seagrasses throughout the hemisphere and indeed worldwide. In each case, it is the quarterly, repeated analysis of a series of specific locations across a seagrass meadow that provides evidence of change in the seagrass environment.

Methods

The goal of the SeagrassNet program is to scientifically measure change in seagrass, including distribution, species composition, and abundance. The protocol for seagrass monitoring is the same throughout the world, with some specific methods adapted to the suite of seagrass species in a region (Short *et al.* 2004, 2005, <http://www.SeagrassNet.org>) and is based on a statistically valid and peer-reviewed sampling scheme (Burdick & Kendrick 2001). Uniformity of methods assures that results are comparable worldwide and over time at a given site, and the repeated measures, quarterly monitoring allows trend detection over relatively short time periods (1–2 years). SeagrassNet does not survey or map the seagrasses of a region, nor does it characterize region-wide trends. However, it captures small increments of change within a specific meadow that is representative of the area, and widespread use of the protocol allows comparisons across countries, regions, and the world.

Table 1. SeagrassNet sites across the Americas: locations, climate, species, environmental parameters, and monitoring results.

locations (climate)	latitude (longitude)	seagrass species (substrate)	temp. [°C]	salinity [p.s.u.]	depth m.s.l. [±tide] [m]	monitoring reveals	cause of loss (impact)
New Hampshire, USA (temperate)	43°05' N (70°42' W)	<i>Zostera marina</i> (fine sand)	-3–35	22–32	1.4–1.6 (±1.5)	declining cover, biomass, density and area	climate change (increased grazing)
Maryland, USA (temperate)	38°04' N (75°20' W)	<i>Zostera marina</i> , <i>Ruppia maritima</i> (silty sand)	0–35	25–40	1.2 (±0.2)	declining cover	eutrophication (algal overgrowth)
Placencia, Belize (tropical)	16°38' N (88°22' W)	<i>Thalassia testudinum</i> , <i>Syringodium filiforme</i> , <i>Halodule wrightii</i> (muddy sand)	26–31	35–38	0.6–4.2 (±0.3)	declining cover and density	eutrophication (tourist development/increased nutrients)
Tamandaré, Brazil (tropical)	08°47' S (35°07' W)	<i>Halodule wrightii</i> , <i>Halophila decipiens</i> (carbonate sand)	24–30	34–39	1.2–2 (±1)	declining biomass and area	climate change (storm frequency/shifting sediment)
Abrolhos, Brazil (tropical)	17°58' S (38°43' W)	<i>Halodule wrightii</i> (carbonate sand)	25–29	38–42	5–7 (±1)	declining biomass and density, position shift	MPA (altered trophic cascade)

MPA = marine protected area.

A site consists of three fixed parallel 50 m cross transects, their midpoints on a transect laid out seaward, perpendicular from shore (Fig. 2). Sites are chosen based on a set of parameters designed to locate a typical or representative seagrass bed for the area under consideration (Short *et al.* 2004, 2005). Quarterly sampling is done at twelve 0.25 m² quadrats placed at predetermined random locations along the cross transects. Distribution of seagrasses is captured by measuring the position of the meadow relative to the permanent cross transects over time. Species composition is measured along the cross transects, and seagrass abundance is determined via measurements of cover, canopy height, density, and biomass.

Voucher specimens are collected and prepared as herbarium sheets of each seagrass species (with flowering parts if present) and then archived at the International Seagrass Herbarium at the Smithsonian, Washington, DC, USA. Each quadrat is photographed to create a permanent record of the seagrass along each cross transect. Seagrass percent cover by species is visually estimated per quadrat by using a photo guide of various cover conditions. Canopy height is measured for the dominant seagrass species. Any evidence of grazing is identified (Supanwanid *et al.* 2001; Zupo *et al.* 2001). Biomass is collected with a 0.0035 m² core outside each quadrat at least 0.5 m distance from the quadrat but in an area of the same seagrass species and cover. The sediments are washed from the core sample and the plant material is separated into leaves, stems, and root-plus-rhizome. Epiphytes are scraped from the leaves and the leaves rinsed in a weak acid (vinegar) if calcareous epiphytes are present. All plant parts are rinsed in fresh water, dried to constant weight (60 °C for 24 h) and weighed. Shoot density by species is determined from the core sample by counting all shoots having intact meristems. Environmental data are obtained, including continuously measured water temperature (by using an iButton[®] temperature data logger) and salinity.

Statistical analysis of case study results was performed by using correlation analysis and least squares regression to identify trends over time. ANOVA was used for testing significant change at individual cross transects. Significance was determined at $P < 0.05$.

Results and discussion: case studies

Seagrass loss from goose grazing in New Hampshire, USA

The Fishing Island SeagrassNet site is a 10-ha intertidal eelgrass meadow located at the mouth of the Piscataqua River, in Portsmouth Harbor, on the border of New Hampshire and Maine, USA (Table 1). Although intertidal, much of the site is effectively a shallow tide pool at low tide; eelgrass (*Zostera marina* L.) throughout the

Fishing Island meadow typically has short, narrow leaves compared to eelgrass in the region. The site is regularly flushed with Gulf of Maine water and shows no evidence of anthropogenic impacts.

During the first year of SeagrassNet monitoring (October 2001 through July 2002), data for the eelgrass parameters of percent cover, canopy height, and aboveground biomass showed the typical seasonal changes of eelgrass abundance at Fishing Island (Rivers & Short 2006). Eelgrass canopy height and aboveground biomass showed similar seasonal fluctuations at all three cross transects, with low values occurring in January and peak values occurring in July; percent cover and shoot density data followed a similar trend (Fig. 3). Eelgrass at the site during the first year of monitoring showed no evidence of having been grazed by waterfowl (Fig. 4). Based on two decades of annual aerial photography (F.T. Short, unpublished data), eelgrass at Fishing Island had no waterfowl grazing impacts prior to January 2003.

During the second year of SeagrassNet monitoring (October 2002 through July 2003), grazing impacts of Canada geese (*Branta canadensis* L.) on eelgrass were first seen in the January 2003 quarterly sampling (Figs 3 and 4). Eelgrass shoots at the site were very short-leaved and lacked leaf tips; shoot density was greatly reduced. All eelgrass parameters declined from January 2003 to July 2003 with no evidence of the typical spring growth period; biomass remained low. All three cross transects had significantly lower values for eelgrass percent cover, canopy height, and aboveground biomass in April and July 2003 compared to the same sampling periods in 2002 (Fig. 3). Over the course of the 3-year monitoring, and despite seasonal fluctuations, both percent cover and density showed significant declines ($r^2 = 0.652$ and 0.720 , respectively; $P < 0.05$; Fig. 3). The change in eelgrass structural characteristics at the site is evident in the voucher records. The largest overall declines in eelgrass parameters were seen in percent cover (from 98% to 1%) and aboveground biomass (from 685.4 to 2.3 g·m⁻²).

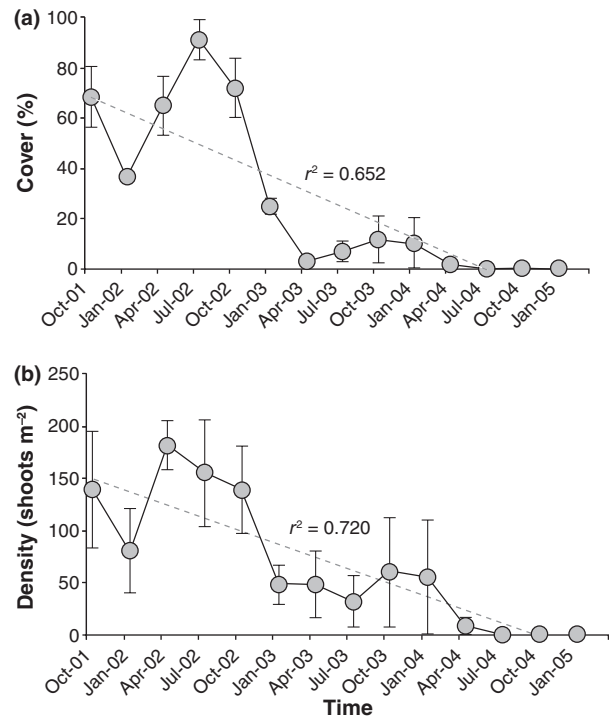


Fig. 3. Eelgrass (a) percent cover and (b) shoot density (mean ± SE) at the Fishing Island, New Hampshire SeagrassNet site from October 2001 to January 2005. Coefficient of determination (r^2) of each parameter versus time indicates significant declines (dotted line) of eelgrass cover and density ($P < 0.05$).

Canada goose grazing activity was prevalent at Fishing Island throughout the winter months, with a flock of approximately 100 geese observed frequently at the site. The abnormally heavy snows of 2003 might have forced the geese out toward the coast as their inland food resources were buried. Historically, Canada geese migrated south, but warmer winters in the past decade have produced overwintering populations in New England. During the particularly snowy winter of 2003, the geese turned to eelgrass beds as the only green areas

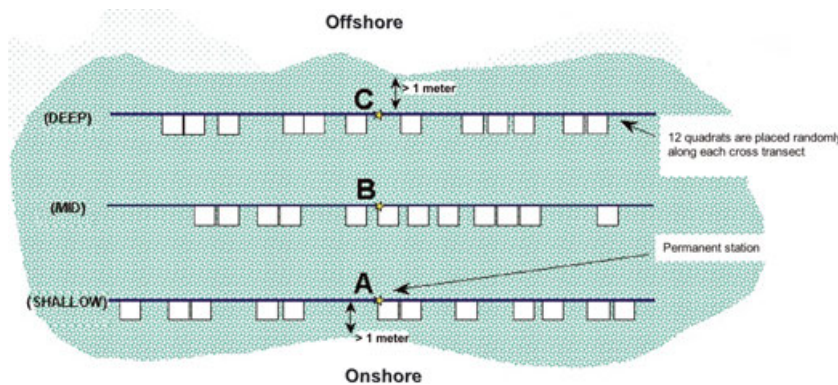
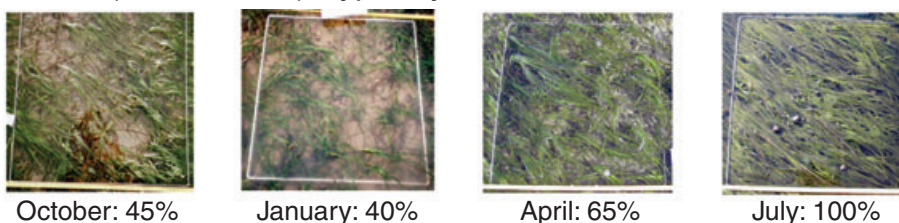


Fig. 2. Schematic diagram of the SeagrassNet sampling design: three fixed cross transects with predetermined random quadrats on the shoreward side (Short *et al.* 2005). The shaded area is the seagrass bed.

Year 1 (2000 – 2002): Typical year



Year 2 (2002 – 2003): Goose grazing event

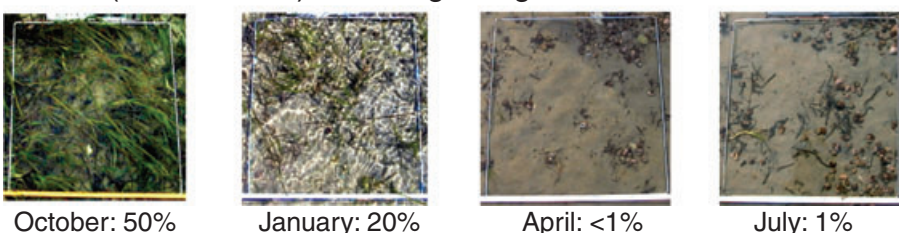


Fig. 4. Photographs of quadrat no. 1 on the mid-depth cross transect at Fishing Island, New Hampshire for eight quarterly sampling times from October 2001 to July 2003 with percent seagrass cover.

available for foraging. These geese continue to feed at the intertidal eelgrass bed near Fishing Island in winter, despite the very low eelgrass biomass, even with open corn fields and golf courses available. As a result of information based on SeagrassNet sampling of this site, the New Hampshire Department of Fish and Game is now discussing a reduction in Canada goose hunting restrictions by lengthening the hunting season and increasing the allowable take of birds. To date, however, monitoring data show no reduction in goose grazing impacts at Fishing Island.

In conclusion, monitoring showed that 96% of the seagrass (eelgrass, *Zostera marina* L.) at the SeagrassNet monitoring site in New Hampshire (USA) was lost in the first quarter of 2003, a consequence of intensive grazing by Canada geese, *Branta canadensis* L. (Rivers & Short 2006), birds which historically migrated south during the winter. With a warmer climate, these birds now reside in New Hampshire throughout the year, increasing the grazing pressure on seagrasses. Intensive grazing by the geese of the eelgrass bed at the New Hampshire SeagrassNet site has continued to the extent that the bed has not recovered by 2005.

Loss of seagrasses in Maryland, USA: eutrophication of a coastal lagoon

The SeagrassNet site in Maryland, USA, near the southern end of the species range for *Z. marina* on the east coast of North America, is located in a lagoon subject to anthropogenic inputs of nutrients from groundwater

sources. The lagoon also has widgeon grass, *Ruppia maritima* L., growing at the shallow edge of the lagoon. Monitoring has shown a general seagrass decline between 2002 and 2004, particularly near the shore.

Tizzard Island is located on the west side of Chincoteague Bay, a coastal lagoon in the states of Maryland and Virginia, USA. Chincoteague Bay is a fairly shallow lagoon with a low tidal range and an inlet at its southern end and a connection to Sinepuxent Bay at the northern end. A residence time of 63 days (Pritchard 1960) leads to relatively high salinities (Table 1). To the east, Chincoteague Bay is bordered by the Assateague Island National Shoreline, a pristine barrier island, and to the west by a mostly undeveloped coastline dominated by marshes and small fishing villages. Slightly larger developments are located at the northern and southern ends of Chincoteague Bay, and farms dominate the inland area. Although the surrounding shorelines look pristine, Chincoteague Bay has shown clear signs of eutrophication in recent years. Brown tides (*Aureococcus anophagefferens*) and macroalgal (*Chaetomorpha linum*) blooms were first observed in 1998. The source of nutrients is mainly groundwater (from inland farms and septic tanks) and the atmosphere, as the watershed is relatively small. Eroding marsh shorelines are also a possible source of nutrients.

In the 1930s, most eelgrass in the lagoon was lost due to wasting disease. Its natural recovery was remarkable, steadily increasing in area (an average of 305 ha·year⁻¹) until eutrophication slowed the recolonization process in 1999. Recovery occurred mainly on the eastern shore of

Chincoteague Bay; more recently, the seagrass beds also expanded to the western shore where Tizzard Island is located.

At the Tizzard Island SeagrassNet site, since 2002, when monitoring first began, there has been a significant overall

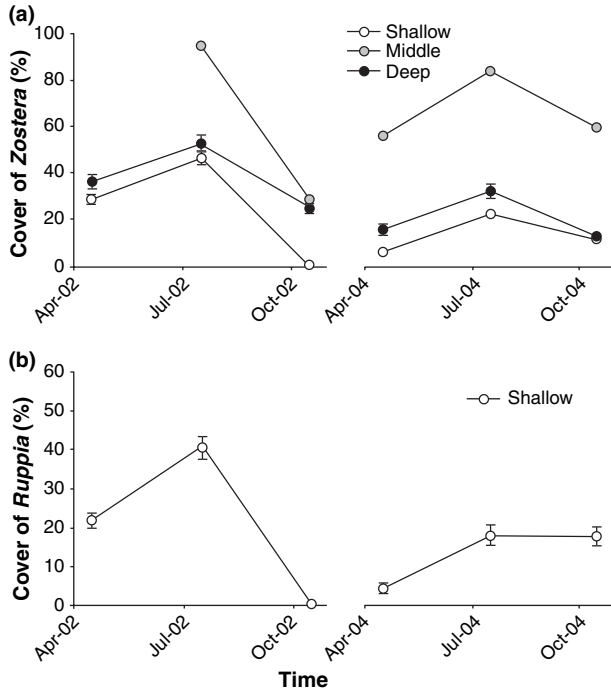


Fig. 5. Percent cover of (a) *Zostera marina* at three depths (the three SeagrassNet cross transects) and (b) *Ruppia maritima* at the shallow depth at Tizzard Island, Maryland in 2002 and 2004. Vertical bars represent one standard error. Percent cover of *Z. marina* in the shallow area in October 2002 was 0.08% (average of 12 quadrats). Note that *R. maritima* does not occur at the mid and maximum depths.

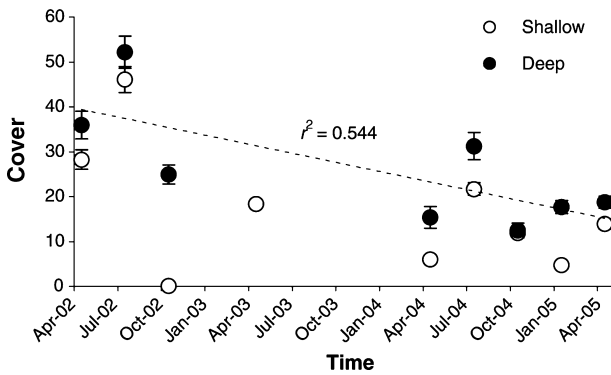


Fig. 6. Shallow and deep *Zostera marina* at Tizzard Island, Maryland between April 2002 and April 2005. Coefficient of determination (r^2) of percent cover versus time indicates a significant decline (dotted line) of eelgrass for the deep cross transect ($P < 0.05$).

decline in seagrass percent cover (Figs 5 and 6). When comparing eelgrass cover for the month of July, when percent cover is high, a decline is observed between 2002 and 2004 ($P < 0.05$). Additionally, for the 3 years of monitoring, percent cover at the deep cross transect showed a significant linear decline ($r^2 = 0.544$, $P < 0.05$).

Percent cover consistently reached values over 80% in mid summer at the mid-depth cross transect, while percent cover only reached values of 46% (2002) and 22% (2004) in the shallow cross transect and 52% (2002) and 31% (2004) in the deep cross transect (Fig. 5a). Percent cover of *Z. marina* and *R. maritima* (found in the shallow area only) suggested a seasonal cycle in 2002, with highest values in July (Fig. 5).

Seagrass abundance at Tizzard Island seems to follow the classical pattern for the area, highest at mid depth and mid summer (Orth & Moore 1988). The deep edge of seagrass beds colonizing turbid waters such as those of Chincoteague Bay tends to be relatively sparse because of limitation of light (Dennison *et al.* 1993). In contrast, the shallow area is exposed to stresses such as occasional desiccation, high light/UV levels and elevated temperatures.

As eutrophication has become a problem in Chincoteague Bay, blooms of micro- and macro-algae have reduced the availability of light to the seagrasses. The nuisance macroalga *C. linum* has become a problem especially for shallow-water seagrasses where the alga tends to accumulate and shade them. In 2002, no *C. linum* blooms were observed at Tizzard Island, but in July 2004 the alga was a major problem along the shallow cross transect.

The general trend of seagrass loss in the sampling area suggests a broad-reaching problem, probably degraded water quality, a result of eutrophication. Indeed, total nitrogen and phosphorus concentrations have been steadily increasing in Chincoteague Bay (Wazniak *et al.* 2005). As a result, extensive brown tides have contributed to degraded water quality and are the most likely cause of the significant decline seen in percent seagrass cover at the deep cross transect (Fig. 6).

Seagrass loss from development impacts in Belize

Placencia, Belize is a rapidly developing seaside town at the end of a narrow sand spit peninsula between Placencia Lagoon and the Western Caribbean. The SeagrassNet site is located off Placencia Village adjacent to a residential area and just up the shore from a marina and docking facility for live-aboard sailboat rentals. The site is far from pristine, with rapid increase of tourist facilities nearby, but was established to capture the effects of expanding development within the region. The seagrass species present are mixed stands of *Syringodium filiforme* and *Thalassia testudinum* at the shallow cross transect, to

pure stands of *T. testudinum* at the deep edge. Some *Halodule wrightii* is found scattered within the meadow (Table 1). Manatee forage extensively in the seagrass beds of Placencia Lagoon and are often seen off the end of the peninsula.

The first seagrass monitoring in Placencia was conducted in November 2003. Over the relatively short duration of monitoring, a significant decrease in seagrass percent cover ($r^2 = 0.740$, $P < 0.05$) and shoot density ($r^2 = 0.715$, $P < 0.05$) has been observed (Fig. 7). The increased development on the shoreline adjacent to the SeagrassNet site and in the village are the likely cause. It is early to draw conclusions, but the large reductions seen in percent cover (46%) and shoot density (66%) during the second year of SeagrassNet monitoring are of concern. Anecdotal evidence from the Placencia peninsula supports this concern regarding the direct developmental effects on the marine coastal environment. SeagrassNet is poised to document any further changes in the seagrass habitat and will provide valuable evidence for management action.

The coast in Placencia Village and the sand spit leading to the village are subject to high development pressures, as a result of the rapidly growing tourist industry in the

area. Nutrient enrichment from groundwater intrusion, local shrimp farms and live-aboard boats contribute to signs of eutrophication both along the shoreline and within Placencia Lagoon, including phytoplankton blooms, growth of nuisance macroalgae, and anecdotal reports of seagrass decline (Short *et al.* 2006). Additionally, mangroves are being clear cut, and shoreline hardening is occurring as developers and home owners attempt to protect their investments. Furthermore, recent government relaxation of restrictions on development and reduction in government support for the coastal zone management program do not bode well for the near future of natural resources in Placencia.

Storm-related seagrass decline in Tamandaré, Brazil

Tamandaré is characterized by the presence of three reef formations parallel to the beach, as well as mangroves, seagrass ecosystems, and tropical forests along the shore. Due to its high ecosystem and biodiversity values, the area is preserved by two national Environmental Protection Areas (EPAs): the Reef Coast EPA, which protects the marine area, and the Guadalupe EPA which protects both marine and continental systems. The site is tropical and humid with annual rainfall over 2000 mm (Table 1). The rainiest months are March to August and the dry season is from September to February.

Halodule wrightii A. is the dominant seagrass in Tamandaré, but *Halophila decipiens* O. is found regularly in ephemeral, small populations. The SeagrassNet site is located in Carneiro Bay, one of the three bays of Tamandaré, where the seagrass bed is limited by reefs at the deep edge. All cross transects are subtidal, approximately 2 m below MSL, and there is a tidal range of about 2 m. The sediment is predominantly calcareous sand and gravel, derived from *Halimeda* sp. segments and reef debris.

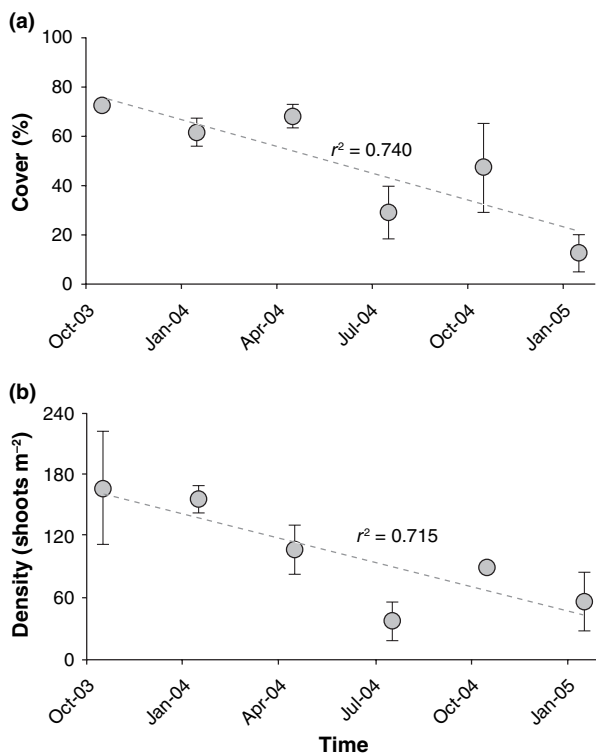


Fig. 7. Seagrass (a) percent cover and (b) shoot density (mean \pm SE) at the Placencia SeagrassNet site. Coefficient of determination (r^2) of each parameter versus time indicates significant declines (dotted line) of seagrass cover and density ($P < 0.05$).

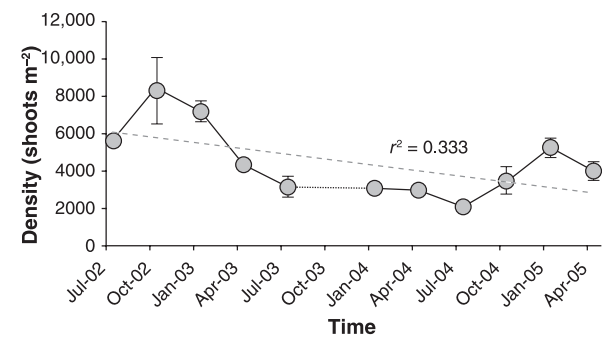


Fig. 8. Mean shoot density (\pm SE) for all transects of *Halodule wrightii* in Tamandaré, Brazil. October 2003 was not sampled. Coefficient of determination (r^2) of density versus time indicates a significant decline (dotted line) of *H. wrightii* for the three cross transects ($P < 0.05$).

Statistically significant differences in seagrass shoot density ($P < 0.001$) and biomass ($P < 0.001$) were observed between the dry and rainy seasons. In general, dry months showed the highest values for all plant parameters. Shoot density (mean \pm SE) followed a seasonal pattern in 2002–2003 (Fig. 8), with the deepest cross transect showing the highest density values. In 2003, especially between July and October, an increase in storm frequency and intensity led to the loss of sensors and transect markers and the movement of sand waves over the seagrass meadows led to their partial loss (Fig. 8). Over the course of the 3 years of monitoring, shoot density showed a significant linear decrease for the site overall (Fig. 8; $r^2 = 0.333$, $P < 0.05$).

Percent cover showed seasonal variation during the period of sampling, with the lowest mean percentages in July 2004 ($8 \pm 9\%$) for the deep cross transect and the highest in October 2002 ($59 \pm 33\%$) for the mid-depth cross transect. Algae percent cover was also estimated but was not significantly related to variations in seagrass cover, although in other meadows in Tamandaré, algae such as *Halimeda* sp. and *Caulerpa* sp. are replacing *H. wrightii*. Algae percent cover was highest during the rainy months with a maximum of $26 \pm 14\%$ in July 2003.

Biomass values also showed seasonal variation with dry season peaks (Fig. 9). The values measured for leaf biomass varied from 8.92 ± 8.86 g·dw·m⁻² in July 2004 for the deep transect to 94.84 ± 76.19 g·dw·m⁻² for October 2002 for the mid-depth cross transect. Rhizome biomass varied between 24.31 ± 24.63 g·dw·m⁻² in July 2004 for the deep cross transect and 165.14 ± 113.33 g·dw·m⁻² in January 2003 for the shallow cross transect.

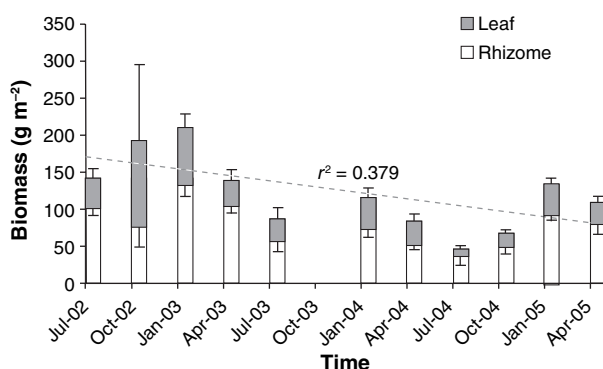


Fig. 9. Biomass (mean \pm SE) of seagrass *Halodule wrightii* in Tamandaré, Brazil. The data show the mean leaf and rhizome biomass estimated in 12 permanent quadrats for three transects at each quarterly sampling period. October 2003 was not sampled. Coefficient of determination (r^2) of total biomass versus time indicates a significant decline (dotted line) of *H. wrightii* for the three cross transects ($P < 0.05$).

In general, biomass in Tamandaré followed the same seasonal pattern as shoot density (Figs 8 and 9). *Halodule wrightii* leaf biomass decreased significantly over the 3 years of monitoring (Fig. 9; $r^2 = 0.335$, $P < 0.05$), although there was no significant trend in rhizome biomass (Fig. 9; $r^2 = 0.221$). Total biomass trends indicate a significant linear decline in *H. wrightii* over the course of our monitoring (Fig. 9; $r^2 = 0.379$).

In Tamandaré, *H. wrightii* shows a seasonal pattern in shoot density, percent cover, and biomass. The dramatic seasonal change in both above and below ground biomass demonstrates clear interannual trends even at 8° south latitude (Fig. 9). Seasonality has not previously been documented in this tropical area (Magalhães 2004), although it has been noted for *H. wrightii* at more southerly latitudes in Brazil (Oliveira *et al.* 1997).

Although seasonal fluctuation was observed, in 2004 *H. wrightii* showed the lowest biomass values in SeagrassNet sampling to date (Fig. 9). It may be early to assume that it is not a natural fluctuation of the population, but the possible explanation for the decrease of biomass and shoot density in the period is the increased sediment dynamics in the area caused by an increased storm frequency. During 2004, sand waves were observed to be moving through the area, causing the complete disappearance of seagrasses on the right side of the shallow cross transect. As of May 2005, this area was still unvegetated.

For the last 2 years, the mean air temperature in Pernambuco has been 3 °C higher than the 30 year mean (F. Lacerda, personal communication). The increase has caused changes in storm events and rain patterns and influenced the coastal hydrodynamics, all of which may have affected the seagrass meadows via excessive sediment transport. In Itamaracá Island, just 100 km away, this problem was already verified and identified as one of the causes of seagrass and fish stock decline (Magalhães 2004).

In summary, at the Tamandaré SeagrassNet site, sediment movement, a result of stronger and more frequent storms, is the main limiting factor to *H. wrightii*. In this area, SeagrassNet proved to be an excellent mechanism to bring attention to the importance of seagrasses. Since the beginning of this project in Tamandaré, seagrasses are getting more attention from scientists, managers and decision makers who are now beginning to reconsider state legislation, in order to include seagrasses.

Shifting seagrass boundaries at Abrolhos, Brazil

Further south in Brazil, the Abrolhos SeagrassNet site presents an interesting picture from an offshore location with a complex combination of trophic interactions which may have been modified by the establishment and eventual enforcement of a marine protected area (MPA).

The Abrolhos archipelago, located 65 km off the southern coast of Bahia State, Brazil, within the tropics, is part of the Abrolhos Marine National Park (Leão & Kikuchi 2001). The archipelago consists of five small islands with small fringing reefs and lies on the Abrolhos Bank, at a widening in the continental shelf. The region is mainly influenced by the Brazilian Current, responsible for the high temperatures of the surface seawater. Monitoring has revealed stable temperatures at the site with oceanic salinities (Table 1).

The shallow soft-bottomed marine communities near the coral reefs of the Abrolhos archipelago are macrophyte-dominated, being composed of algae and the seagrasses *H. wrightii* and *H. decipiens* (Creed & Amado Filho 1999; De Paula *et al.* 2003). The SeagrassNet site is situated by the island of Santa Barbara, in the subtidal at a depth of 5–7 m. *Halodule wrightii* is the only seagrass species at the site, where the seagrass forms a narrow mono-specific subtidal strip parallel to the shore. Typically, the spatial pattern of habitats from the shore toward the sea is fringing reef, unvegetated sand, *H. wrightii* and rhizophytic macroalgae (Fig. 10).

SeagrassNet monitoring was started at the site in April 2002; macroalgae percent cover was also estimated in the quadrats. Since 2002, the seagrass bed has expanded in the direction of the reef (shoreward) at a rate of 1.68 m·year⁻¹ (Fig. 11; $R^2 = 0.7126$, $P < 0.05$). Similarly, the deep edge of the seagrass bed has also moved shoreward, at a rate of 0.86 m·year⁻¹ (Fig. 11; $R^2 = 0.7225$, $P < 0.05$). Seagrass shoot density, biomass, and percent cover have also shown distinct and consistent changes. In the shallow cross transect, an initial increase in shoot density was detected, followed by a continuous decline (Fig. 12a; ANOVA $F = 64.7$, $P = 0.015$).

At this transect, total biomass followed a similar pattern for the middle and deep cross transects (Fig. 12b; ANOVA Middle $F = 23.5$, $P < 0.001$ and Deep $F = 16.0$, $P < 0.001$), although biomass declined half a year later, in April 2003. In the two deeper cross transects, declines have

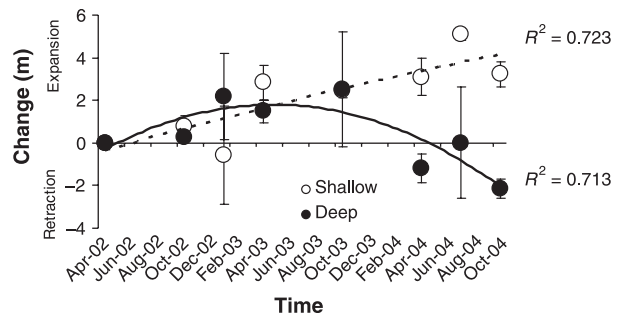


Fig. 11. Change in shallow and deep boundaries of the seagrass bed at Abrolhos, Brazil. Values are mean \pm SE relative to position (zero) at the start of monitoring. Curves were fit by second-order polynomial regression (dotted line = shallow transect; continuous line = deep transect).

been nearly constant between 2002 and 2004 (36–39% year⁻¹ decline in density and 34–37% year⁻¹ decline in total biomass). In the deep cross transect, seagrass cover decreased in a similar fashion (39% year⁻¹, Fig. 13) and the shallow cross transect showed a pattern of increase and subsequent decrease in seagrass cover similar to that found in shoot density and biomass. In the shallow cross transect, algae had lower cover than seagrass, but in the deep cross transect algae were 2.3–6 times more abundant than seagrasses. In these cross transects, a highly significant negative correlation was found between mean seagrass and algae cover (Pearson's $r = -0.803$, $P < 0.001$).

The seagrass bed at Abrolhos has shown a systematic reefward shift over the 3-year period, that is, a shift toward the island. At Abrolhos, as in the Caribbean, the fringing reefs are bordered by an area of reduced seagrass cover, the 'halo' (Sweetman & Robertson 1994), which is caused by the grazing activities of fishes foraging out from the reefs (mainly parrot and surgeon fishes) (Hay 1984; Sweetman & Robertson 1994) or by hydrodynamic factors (Ogden & Zieman 1977) (Fig. 10). The advance of the seagrass reefward would therefore suggest: (i) the

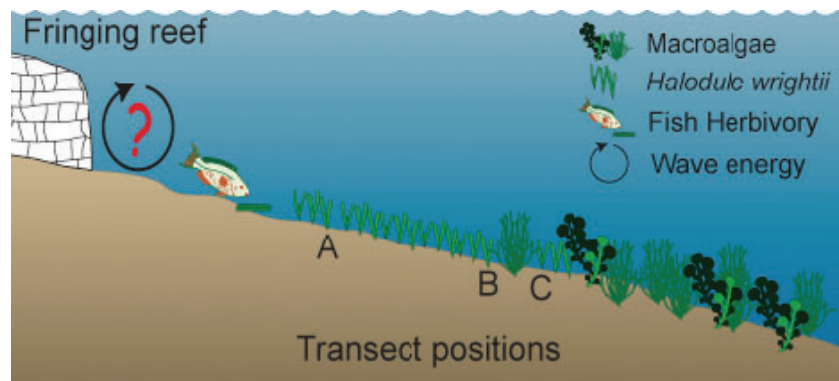


Fig. 10. Position of principal habitats and conceptual model of probable factors influencing the spatial distribution and change of the seagrass bed monitored by SeagrassNet at Abrolhos, Brazil. The position of the three cross transects are shallow (A), mid-depth (B), and deep (C).

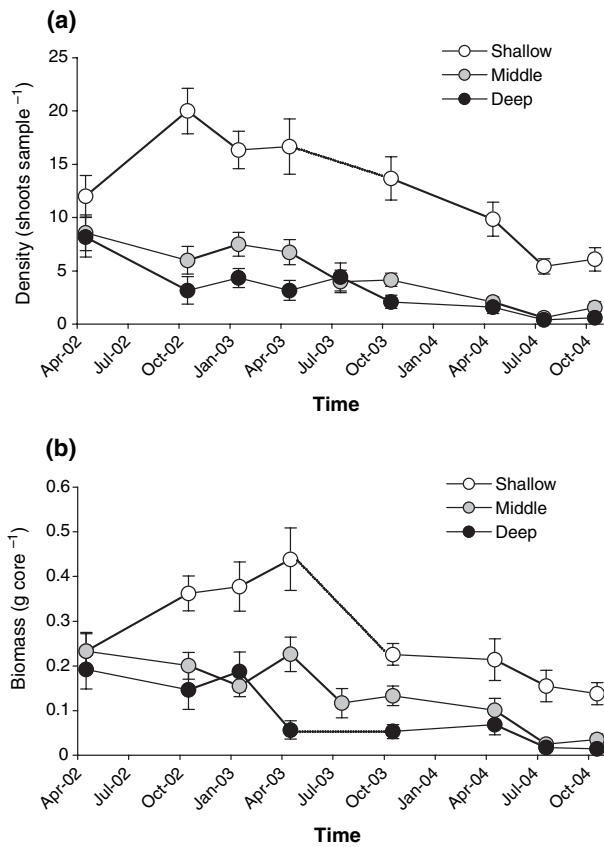


Fig. 12. *Halodule wrightii* (a) shoot density and (b) total biomass (mean ± SE) per core in the three cross transects at Abrolhos, Brazil.

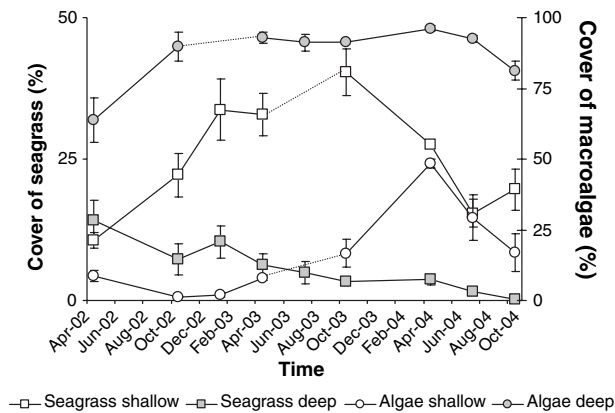


Fig. 13. Seagrass and macroalgae cover (mean ± SE) in shallow and deep transects at the SeagrassNet site at Abrolhos, Brazil.

growth of seagrass into shallower areas that had been previously impacted by a strong hydrodynamic regime at some time before the monitoring program started; (ii) a reduction in grazing pressure from reef-associated fish, the result of a trophic cascade in which predatory fishes have become more abundant and herbivores less so.

It is not clear which factor is responsible for the changes observed, but experiments have shown that in Abrolhos grazing pressure in the seagrass bed is 20% of that quantified in the halo (J.C. Creed, unpublished data, using *H. wrightii* as a bioassay). Furthermore, during the study period, sand ripple marks (caused by hydrodynamic impacts) have not been observed in the halo, so it seems probable that a reduction in grazing fishes is responsible, at least in part, for the observed shift. Such a reduction may demonstrate herbivore overfishing or predator recovery. The national park was instituted in 1983 and it is possible the MPA has effectively increased predatory fish stocks, as has happened in other locations (reviewed by Mous *et al.* 2005).

At its deeper edge, the seagrass was impacted by taller, multi-species rhizophytic macroalgae, principally composed of *Caulerpa* spp. The advance of these macroalgae into the seagrass bed suggested that the deep edge of the seagrass is controlled by macroalgal competition, a condition which may be mediated by grazing pressure on macroalgae.

Manipulative experiments are needed to tease apart the factors which have resulted in the shifting seagrass bed at the Abrolhos site. However, we can conclude that:

- 1 The seagrass bed monitored by using the SeagrassNet protocol has shown a considerable change. Further monitoring will reveal more about frequency and severity of impact to the seagrasses.
- 2 SeagrassNet methods are sensitive enough to detect change over the timescales of interest.
- 3 Seagrass abundance is impacted by biological interactions such as competition with macroalgae, herbivory and/or predation. Seagrass beds are therefore habitats which reflect a long-term systematic global change in the coastal zone related to factors which act directly or indirectly upon them.

Conclusions

Seagrasses are showing declines throughout the Americas: this is the conclusion that our combined case studies suggest. The declines discussed within the case studies have a variety of causes, all of which relate directly or indirectly to human activity, from eutrophication to global climate change.

In Portsmouth Harbor, on the border of New Hampshire and Maine (USA), generally warmer winters have caused Canada geese to overwinter in the state instead of migrating south. The occasional cold, snowy winter then forces the overwintering geese to use *Z. marina* as a winter-long food resource, severely impacting the intertidal coastal eelgrass beds (Rivers & Short 2006). In response to the SeagrassNet data, state environmental managers

have invoked increased goose population controls. In the southern hemisphere, the SeagrassNet monitoring in Tamandaré, Brazil, also revealed a climatic change-related response of seagrass, with increased sediment transport due to increased storm frequency negatively impacting distribution of *H. wrightii*. SeagrassNet has drawn increased attention to seagrasses in Tamandaré and legislation is now being considered by the state to protect seagrasses from more direct human impacts.

Eutrophication is a major cause of seagrass decline worldwide (Short & Wyllie-Echeverria 1996), and two of the case studies illustrate the adverse impacts of coastal development and increased nutrient loading. In Tizzard Island, Maryland (USA) eutrophication has resulted in increased macroalgal blooms, which smother *Z. marina* and *R. maritima*, particularly at the shallow edge of the seagrass bed, as well as extensive brown tides which have impacted water quality throughout the area and thus seagrass at all depths. Similarly, in Placencia, Belize, eutrophication is the main threat to seagrasses, a mix of *T. testudinum*, *S. filiforme*, and *H. wrightii*. During a relatively short period, percent cover and density of seagrasses have decreased dramatically, while coastal housing and tourism development have proceeded rapidly. Cutting mangroves in Placencia appears to increase the impacts to seagrasses, as normally mangroves are a first buffer, protecting the shoreline and reducing runoff.

In Abrolhos, Brazil, an established offshore marine national park, SeagrassNet has documented both a seagrass decline and a shift in seagrass bed distribution of *H. wrightii*. The seagrass decline appears related to an increase in macroalgal abundance, likely caused by a decrease in fish herbivory on macroalgae, which in turn resulted from increased predatory fish populations brought on, ironically, by successful MPA management. The concomitant shift in seagrass distribution, resulting in more seagrass at the shallow edge of the bed, may have also resulted from a reduction in direct seagrass herbivory, again possibly related to MPA management. SeagrassNet monitoring has prompted the development of testable hypotheses regarding these changes in seagrass habitat.

The relative importance of the causes of seagrass loss is not measured by SeagrassNet, except to identify specific sites around the world where impacts like eutrophication are occurring (Table 1). Two of our sites in this study [Maryland (USA) and Belize] are experiencing relatively rapid declines because of eutrophication, an impact that is a significant issue worldwide (Short & Wyllie-Echeverria 1996). At two of the sites [Great Bay (USA) and Tamandaré (Brazil)], the cause of the decline is related to global climate change, another impact captured by SeagrassNet sampling elsewhere in the world.

SeagrassNet monitoring differs from other seagrass monitoring efforts in being a fixed-transect, repeated-measures-based analysis with a high degree of frequency and replication (<http://www.SeagrassNet.org>). Most other monitoring programs collect random samples from a meadow, bay, or coastal area (e.g., CARICOMP 2001; Fourqurean *et al.* 2002) or monitor over multiple-year time increments (e.g., Klemas *et al.* 1993; Boudouresque *et al.* 2000); such a sampling, even with a lot of replication, makes it difficult to detect statistically significant change over moderate time frames (<5 years), due to the high degree of heterogeneity within seagrass meadows. The SeagrassNet protocol provides a statistically valid replicated assessment, allowing comparison within sites over time and between sites, and is stratified by depth, yielding insight into impacts which influence both deep and shallow edges. The thrust of the SeagrassNet protocol is change detection, and it is designed to rapidly capture change (1–2 years) in seagrass abundance at various depths and to assess any change in the spatial dimension of the meadow by monitoring bed movement in reference to fixed cross transects. The protocol was designed by a group of international scientists specifically to allow change detection anywhere in the world for any seagrass species assemblage or meadow type; other monitoring programs are targeted toward specific locations or species.

In each case study, it is the continuous scientific monitoring of the seagrass habitat using the SeagrassNet quarterly protocol that has produced valid data regarding changes in the seagrass environment. While the monitoring is still in the early stages, we have seen statistically significant declines related to various specific impacts at SeagrassNet sites across the Americas, with each requiring continued monitoring and hypothesis testing. At the same time, we provide scientific evidence that management action is needed to arrest seagrass loss.

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