Maintaining Tropical Beaches with Seagrass and Algae: A Promising Alternative to Engineering Solutions

REBECCA K. JAMES, RODOLFO SILVA, BRIGITTA I. VAN TUSSENBROEK, MIREILLE ESCUDERO-CASTILLO, ISMAEL MARIÑO-TAPIA, HENK A. DIJKSTRA, RENÉ M. VAN WESTEN, JULIE D. PIETRZAK, ADAM S. CANDY, CAROLINE A. KATSMAN, CARINE G. VAN DER BOOG, RICCARDO E. M. RIVA, CORNELIS SLOBBE, ROLAND KLEES, JOHAN STAPEL, TJISSE VAN DER HEIDE, MARIEKE M. VAN KATWIJK, PETER M. J. HERMAN, AND TJEERD J. BOUMA

Tropical beaches provide coastal flood protection, income from tourism, and habitat for flagship species. They urgently need protection from erosion, which is being exacerbated by changing climate and coastal development. Traditional coastal engineering solutions are expensive, provide unstable temporary solutions, and often disrupt natural sediment transport. Instead, natural foreshore stabilization and nourishment may provide a sustainable and resilient long-term solution. Field flume and ecosystem process measurements, along with data from the literature, show that sediment stabilization by seagrass in combination with sediment-producing calcifying algae in the foreshore form an effective mechanism for maintaining tropical beaches worldwide. The long-term efficacy of this type of nature-based beach management is shown at a large scale by comparing vegetated and unvegetated coastal profiles. We argue that preserving and restoring vegetated beach foreshore ecosystems offers a viable, self-sustaining alternative to traditional engineering solutions, increasing the resilience of coastal areas to climate change.

Keywords: coastal erosion, nature-based engineering, ecosystem services, coastal protection, coastal management

Beaches are key ecosystems in coastal zones, making up 31% of the world's shoreline in ice-free regions of the world (Luijendijk et al. 2018). They have a vital role in flood defense, provide a source of income as a tourist attraction, and are essential habitats for various tropical flagship species, such as sea turtles and sea birds (Defeo et al. 2009). Beach erosion, however, has become a major global problem, with a recent analysis showing that 24% of the world's sandy beaches experience chronic erosion (Luijendijk et al. 2018). The development of human infrastructure along the coast and waterways (figure 1a–c) has led to the rapid loss of natural systems that accumulate and stabilize sediment—such as coastal dunes, seagrass meadows, and mangroves—disrupting the regular pathways of sediment transport (Feagin et al. 2015, Luijendijk et al. 2018). Moreover, the combination of sea-level rise with increasing storm occurrence and intensity will exacerbate beach erosion in the future (Defeo et al. 2009, Nicholls and Cazenave 2010). This is of great concern for many tropical areas, which typically have a high dependency on beaches for flood safety and also economically, for local tourism (see the red shading in figure 1d). For example, the Caribbean islands together received over 23 million tourist visitors in 2015, creating a revenue of $26.5 billion (UNWTO 2016). On average, 23% of the gross domestic product (GDP) of countries within the Caribbean is obtained from tourism (figure 1d), with most tourists being attracted by the sandy beaches. Cost-effective solutions to prevent or mitigate beach erosion are therefore urgently needed for the long-term economic sustainability in these countries (UN 2016, Morris et al. 2018).

Many tropical countries lack the infrastructure and finances to undertake engineering solutions for beach protection. Therefore, beaches continue to disappear into the sea, increasing the vulnerability of coastal areas to flooding, and threatening coastal structures and beach tourism (figure 1b). Where there are sufficient resources, two schemes of coastal engineering strategies are used to counter beach erosion: hard and soft (Finkl and Walker 2005, Castelle et al. 2009, Stive et al. 2013, Silva et al. 2016), both of which incur a high capital cost. Hard coastal defense schemes are employed to mitigate wave attack and reduce local erosion (figure 1a; Ranasinghe and Turner 2006, Ruiz-Martínez et al. 2015, Walker, Dong, and Anastasiou 1991). Such physical barriers typically inhibit the natural sand transport pathways, thereby depleting sand from neighboring areas (Ranasinghe and Turner 2006, Ruiz-Martínez et al. 2015, Luijendijk et al. 2018). Soft defense schemes, such as beach or foreshore nourishments, have recently become more popular

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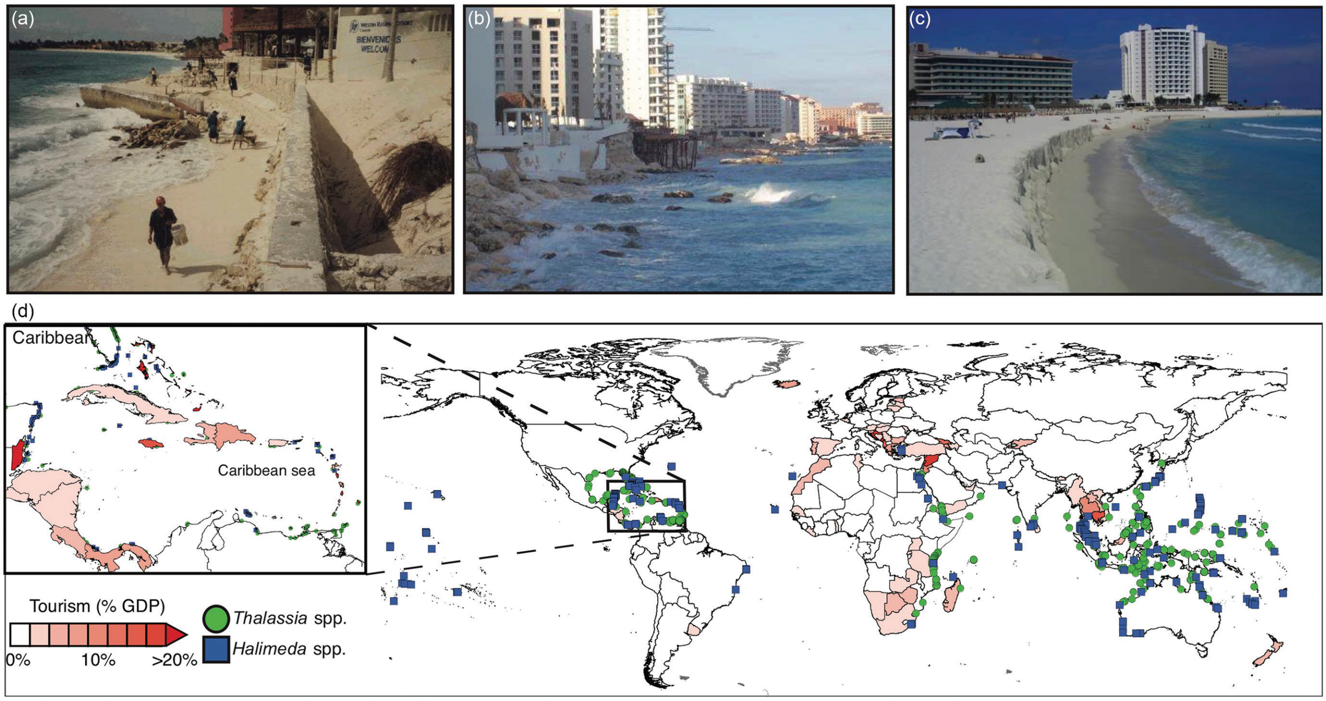


Figure 1. The building of hard structures to prevent coastal erosion, such as seawalls (a), the overdevelopment of coastlines (b), and beach nourishments (c) only serve to exacerbate coastal erosion. The global map (d) shows the proportion of the gross domestic product (GDP) obtained from tourism in 2015 (data sourced from World Bank and World Tourism Organization), with the darker red shading indicating that a higher proportion of the GDP is obtained from tourism for that country. The effective sediment-stabilizing seagrass Thalassia spp. is globally distributed (the green circles, sourced from UNEP-WCMC 2005) and can be found alongside the sediment-producing calcifying macroalgae Halimeda spp. (the blue squares, sightings reported in peer reviewed literature).

(figure 1c; Bishop et al. 2006, Castelle et al. 2009, Ruiz-Martínez et al. 2015, Stive et al. 2013). Although it is effective, soft engineering requires continuous maintenance, resulting in repeated smothering and disturbance of the natural beach communities (Bishop et al. 2006, Defeo et al. 2009) and their neighboring ecosystems (e.g., coral reefs). In the long term, nourishments can alter beach grain characteristics (Hanson et al. 2002), which can potentially cause permanent changes to the benthic community (Bishop et al. 2006).

By combining experimental field measurements with data from the literature, we demonstrate that the combination of foreshore stabilization by seagrass and natural foreshore nourishment by calcifying macroalgae can provide long-term maintenance of tropical beaches. In general, foreshore nourishment (both natural or engineered) is effective in beach protection, because a shallow foreshore reduces wave attack on the beach (Hanson et al. 2002, Christianen et al. 2013). Because a natural foreshore stabilization and nourishment regime requires no maintenance and operates gradually over long timescales with locally produced sediment, it offers a cost-effective and sustainable alternative to human-engineered solutions. Comparing unique long-term beach profiles of vegetated, transitioning, and unvegetated coasts illustrates the effectiveness of this approach.

Natural foreshore nourishment by vegetation: Sediment stabilization and production

Shallow inter- and subtidal foreshores of natural tropical sandy beaches are predominately composed of locally produced calcium carbonate ($CaCO_3$) sediments. These carbonate sediments are biogenically produced and need to be continually captured and retained within the foreshore for a beach to resist erosion and remain stable, something that seagrass is extremely effective at achieving.

With a newly developed portable flume designed to be used in the field, the ability of different vegetation types—bare, vegetated with only calcifying macroalgae, sparse seagrass (50% cover of *Thalassia testudinum*) and dense seagrass (100% cover of *T. testudinum*)—to stabilize sediment was measured directly within Galion Bay, Saint Martin (in the Caribbean). Regulating the speed of two motor-driven propellers allowed the flow velocity within the flume tunnel to be modified (see the photo in figure 2a and further methods in supplement 1). The point at which the surface sediment began to move was recorded as the threshold shear velocity. We found that, in bare areas and areas with only calcifying macroalgae, the coarse carbonate sediments (median grain size = 337 micrometers [μm], standard error [SE] = 33) that are present in these areas start eroding already at flow speeds caused by moderate breezes (i.e., a wind of 10 meters [m]

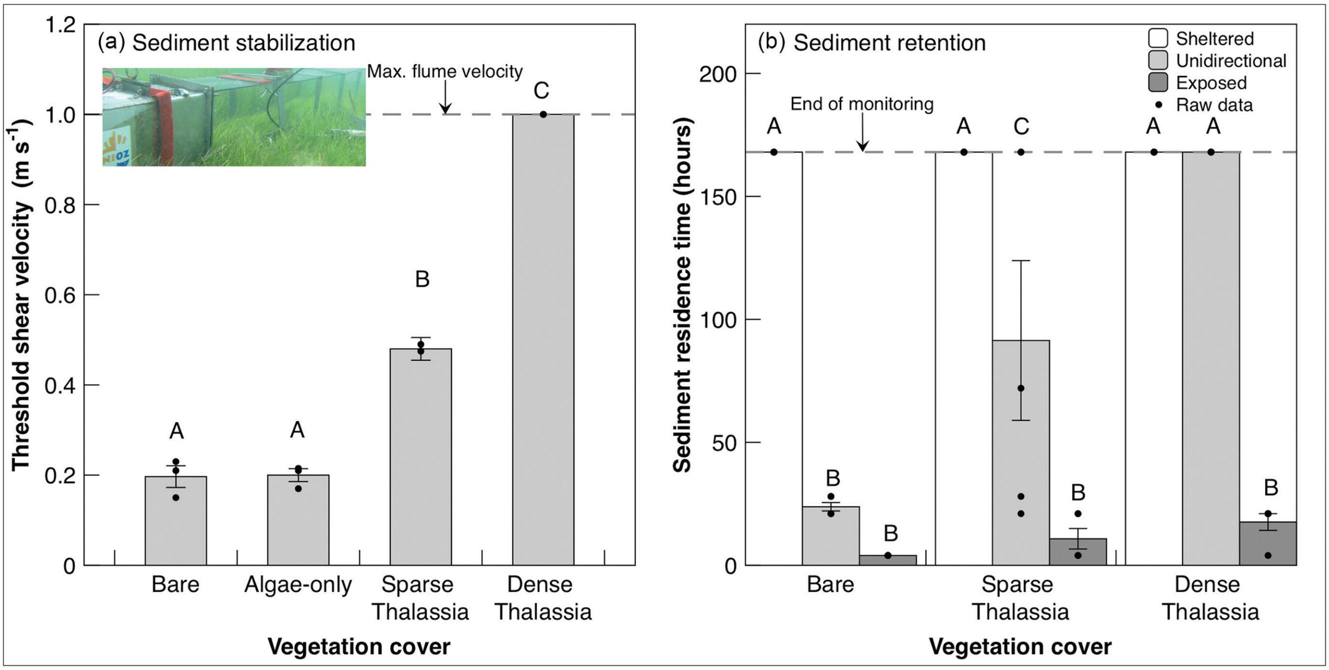


Figure 2. Carbonate sediment is stabilized by seagrass, as was indicated by measuring the critical threshold for bed-load transport with a field flume in contrasting vegetation types: bare, calcifying algae only, sparse Thalassia (50% cover of T. testudinum), dense Thalassia (100% cover of T. testudinum) (a). This was corroborated by measuring the retention time of stained sediments for contrasting vegetation types in the different physical environments (b): wave sheltered (mean wave height = 0.15 m, SE = 0.004, n = 370), unidirectional (mean flow rate = 0.15 m per second, SE = 0.025, n = 18), and wave exposed (mean wave height = 0.22 m, SE = 0.005, n = 429). The error bars represent the standard error ($n_{sed.stab} = 3$, $n_{sed.ret} = 5$), and the black points indicate individual data points. The different letters above the bars denote significant difference (p < .05), tested with Tukey HSD pairwise comparisons.

per second can cause flow speeds of 0.2 m per second within shallow areas; Hughes 1956). However, where a sparse cover of seagrass is present, the sediment is finer (median grain size = 297 μm, SE = 17), because the protected seagrass canopy promotes fine grains to settle (De Boer 2007), but the flow required to erode the carbonate sediment doubles. And when *T. testudinum* seagrass cover is dense, the sediment is finer again (median grain size = 129 μm, SE = 7) but remains stable at flows stronger than 1.0 m per second (figure 2a), the maximum flow velocity of the flume. These flume results were confirmed by retention time of stained sediment that was placed in dense seagrass beds, which was seven times longer than those of bare areas in a high unidirectional flow environment within Galion Bay, and the retention time in a wave-exposed area was four times higher (figure 2b).

Although the sediment stabilizing effect of seagrass has been directly measured in relatively few studies (Scoffin 1970, Widdows et al. 2008), the available literature widely supports our findings. For example, Christianen and colleagues (2013) found that even low-density, heavily grazed seagrass meadows significantly reduce sediment erosion in Indonesia. A global review by Potouroglou and colleagues (2017) shows an average accretion rate of 5.33 millimeters (mm) per year occurring within seagrass meadows compared with adjacent unvegetated areas, which experience an average erosion rate of 21.3 mm per year. Seagrasses reduce erosion and cause sediment accretion by stabilizing the sediment with their root–rhizome mat (Potouroglou et al. 2017) and by attenuating water flow and waves. Hansen and Reidenbach (2012) reported that dense seagrass canopies of *Zostera marina* can attenuate flow velocity by 70%–90%, whereas Fonseca and Cahalan (1992) showed a wave energy reduction of 34%–44% for four varying species of seagrass, including *T. testudinum*. Flow and wave attenuation cause sediment particles to settle and reduces resuspension; in addition, seagrass leaves can bend over the sediment surface, further stabilizing the sediments. For a beach to remain stable over the long term, however, a continuous supply of sediment is required to offset any erosion that occurs during storm events or from seaward currents that may transport unprotected sediment out of the beach system.

The breakdown and erosion of nearby coral reefs can provide a large contribution of sediment when the reefs are present (Chave et al. 1972, Hallock 1981). Another sediment contributor is calcifying macroalgae from the Halimedaceae family, which are composed of 70%–90% $CaCO_3$ (van Tussenbroek and Van Dijk 2007). Because they grow directly within and adjacent to seagrass meadows on tropical beach foreshores, the sediment they produce is deposited where it is most valuable for providing a natural foreshore nourishment.

This sediment production does vary significantly, depending on the season, species, and their abundance; however, the fast growth and rapid turnover rates mean that the average sediment production reported for *Halimeda* spp. growing within seagrass meadows in the Pacific region is 337 grams dry weight (g_{dwt}) $CaCO_3$ per square meter (m^2) per year (SE = 70, $n = 10$; supplement 2; Garrigue 1991, Merten 1971, Payri 1988), and in the Caribbean region, 166 g_{dwt} $CaCO_3$ per m^2 per year (SE = 93, $n = 8$; supplement 2; Neumann and Land 1975, Wefer 1980, Armstrong and Miller 1988, Multer 1988, Freile 2004, van Tussenbroek and Van Dijk 2007). Although this average rate contributes less than 0.28 (Pacific) and 0.15 mm (Caribbean) of sediment to the bed level per year (assuming a dry bulk density of 1.08 grams per cubic centimeter), the deposition of this $CaCO_3$ occurs directly within the foreshore, where seagrass is present. The algae-produced sediment is therefore immediately captured and retained within the beach foreshore ecosystem by the seagrass, thereby supplying a continuous and natural nourishment.

Engineering and natural nourishment as contrasting management regimes

We postulate engineering solutions and natural foreshore nourishment as contrasting management regimes, each having its own positive feedback (figure 3a). The engineered regime involves an unvegetated disturbed foreshore ecosystem with little or no biogenic sand production and highly mobile sediments. Such a regime results in a beach vulnerable to erosion and, therefore, requires regular engineering nourishments of the beach foreshore system to maintain its form. The alternative regime, a natural self-sustaining foreshore ecosystem with seagrass and calcifying macroalgae fronting a stable beach, forms a self-stabilizing and self-nourishing system.

The combined sediment stabilization by seagrass and sediment production by calcifying algae yields a biologically driven landscape with self-maintaining feedback loops. Specifically, by attenuating waves, preventing excessive erosion, and replenishing lost sediments, seagrass meadows, and calcifying algae together create a self-reinforcing loop (Maxwell et al. 2017). Stable sediment has been shown to be a main requirement for the long-term persistence of seagrass meadows (Reise and Kohlus 2008, Christianen et al. 2014, Suykerbuyk et al. 2016) and, in areas with fine sediment, can lead to a higher water transparency needed to sustain growth (van der Heide et al. 2007, Adams et al. 2018). This means that disruption of these self-reinforcing feedback loops may result in rapid losses of the seagrass–algae community (Maxwell et al. 2017). That is, in beach foreshore systems without seagrasses and algae, the sediment surface is freely agitated by currents and waves, yielding highly mobile sediments (Widdows et al. 2008, Marbà et al. 2015). Such unstable sediment conditions make it very difficult for seagrasses and algae to establish or reestablish (Williams 1990, Infantes et al. 2011, Balke et al. 2014, Suykerbuyk et al. 2016) and can increase turbidity levels if smaller sediment particles become suspended in the water column (van der Heide et al. 2007, Adams et al. 2018).

Human engineering through frequent beach nourishments can increase the sand supply to such disturbed beach foreshore systems (Finkl and Walker 2005, Castelle et al. 2009, Stive et al. 2013). However, these repeated nourishments smother establishing seagrasses and algae and create an unstable sediment surface that is more likely to erode (figure 3a). Therefore, although engineered nourishments may save the beach in the short term, they may, paradoxically, generate the necessity for recurrent beach nourishments in the long run (Trembanis and Pilkey 1998), creating an expensive and unsustainable management cycle in developing tropical regions (Silva et al. 2014).

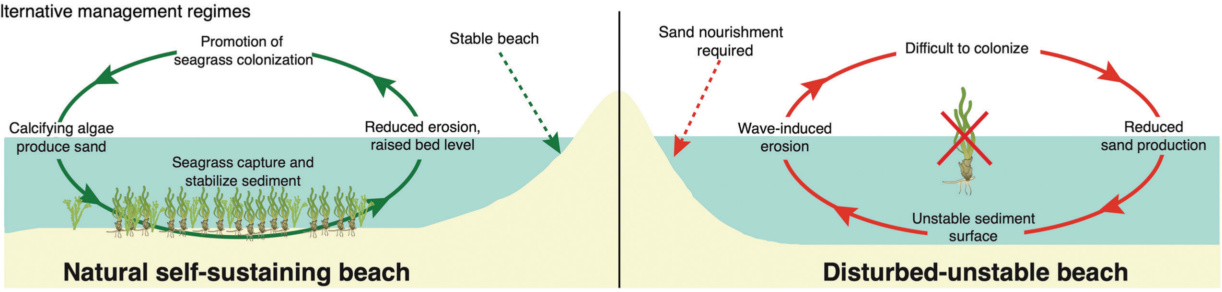
Examples of the two alternative management regimes and one in transition are found along the coast of Mexico (see map in supplement 1). In coastal areas in which seagrass and calcifying macroalgae dominate the system, beach shore profiles conducted from 2008 to 2012 (methods detailed in supplement 1) are stable (figure 3b). In contrast, areas devoid of these species are typified by continuous erosion, which persists after engineered nourishments (figure 3d). A transition between these contrasting management regimes is observed in a third area. In the present article, extensive seagrass meadows of *T. testudinum* disappeared from the first 60 m of the foreshore in 2015 because of a large brown tide of drifting *Sargassum* spp. (van Tussenbroek et al. 2017). As a result of these losses, beach profiles taken in 2007 and 2017 show the beach foreshore experienced strong vertical erosion, up to 0.4 m in some areas (figure 3c). However, a small area of the beach foreshore on which seagrass was not lost, experienced only minor erosion and remained relatively stable (figure 3c). Overall, these examples impressively illustrate the effectiveness of vegetated foreshore ecosystems for maintaining stable beaches and shorelines.

Implications and challenges for future management of tropical beaches

To create stable long-term management solutions for tropical beaches, beach management would benefit from shifting away from frequent engineered nourishments and hard structures, toward maintenance by natural ecosystems. With current insights, anthropogenic use of beaches could be designed to halt and reverse current decline of natural foreshore ecosystems. Tropical seagrass and *Halimeda* spp. usually co-occur and can be found in tropical sandy regions all around the world (figure 1d; Green and Short 2003, UNEP-WCMC 2005), so there is widespread potential to restore these systems (Orth et al. 2006) to create a natural, self-sustaining beach management regime.

Conservation of areas in which natural foreshore vegetation still persists will improve the condition of foreshore ecosystems, maximizing their ability to protect beaches against erosion. Where foreshore vegetation has become degraded, an effort to protect what remains and to restore the ecosystem to a healthy self-reinforcing state may be necessary to implement effective natural beach management regimes. Preserving and restoring foreshore vegetation that still exists

(a) Alternative management regimes



(b) Natural beach, vegetated foreshore (Puerto Morelos) (c) Partially disturbed beach (Mirador Nizuc) (d) Disturbed beach, bare foreshore (Cancun)

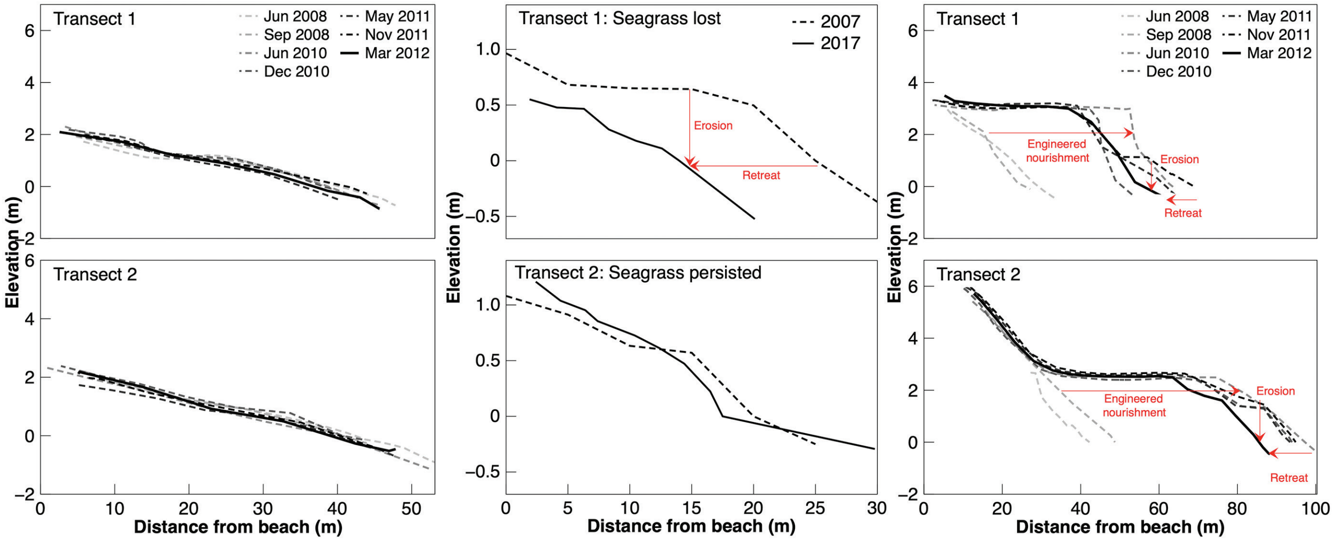


Figure 3. Self-reinforcing feedback loops drive the contrasting beach management regimes as schematized in panel (a). The natural beach is driven by seagrass stabilizing the sediment, which encourages further ecosystem development, whereas the system devoid of vegetation has increasingly mobile sediment, discouraging the growth of vegetation and leading to an unstable beach system, requiring engineering that further contributes to sediment mobility and erosion. These types of beach regimes can be seen in examples from the coastline of Mexico (map in supplement 1). Regular beach profiles taken from two transects at the natural beach of Puerto Morelos from June 2008 (the dashed lines) to May 2012 (the solid line) show that this relatively undisturbed beach with extensive seagrass-calcifying algae meadows has remained stable over many years (b). Although beach profiles at Mirador Nizuc in 2007 (the dashed line) and June 2017 (the solid line) show that the beach had significant erosion after a Sargassum brown tide that persisted from July 2015 to May 2016 resulted in the loss of seagrass (c, upper graph), in an area of the same beach on which seagrass persisted, very little erosion occurred (c, lower graph). Although Cancun has no natural reef or seagrass meadows and although development along the sand dunes has led to constant beach erosion, a sand nourishment in 2010 helped to restore the beach, but it continues to erode (d). Elevations are relative to mean sea level. Seagrass illustration: IAN image library, Saxby.

is especially important as climate-driven disturbance events, such as extreme wave action, cyclones (Saunders and Lea 2008), and the occurrence of brown tides from *Sargassum* spp. drifts (van Tussenbroek et al. 2017), become more frequent with rising global temperatures. As climate-driven factors are hard to manage at a local scale, management should primarily aim at reducing local human-induced impacts (Scheffer et al. 2001). Local impacts, such as greater turbidity (Orth et al. 2006), nutrient enrichment and pollution (Kemp et al. 2005), physical damage to seagrass meadows from trampling and boat anchoring (Eckrich and Holmquist 2000), and modification of natural sediment transport and increased wave reflection caused by the construction of hard structures (Defeo et al. 2009, Ruiz-Martínez et al. 2015, Luijendijk et al. 2018), are all intensifying as coastlines develop further. The installation of sewage treatment plants and limiting construction of hard structures along the coast are the most obvious steps to help protect and restore natural foreshore vegetation. Another is to limit accessibility of people to vulnerable areas and provide boat anchoring facilities outside regions of vegetation. Ensuring coral reefs remain in abundance and their sediment input to tropical beaches persist would also improve the prospects of tropical beaches to keep up with sea-level rise.

Given that the engineering management regime of a disturbed beach is self-reinforced by a feedback that maintains sediment instability (figure 3a), it will be difficult to induce a transition to the natural beach systems in areas in which engineering management regimes already take place or in which vegetation has been completely lost. Developing ways to stimulate natural vegetation development may be necessary, such as using temporary structures that protect establishing seagrass and calcifying macroalgae, until they grow to a point that they can self-stabilize the sediment (Suykerbuyk et al. 2016, van Katwijk et al. 2016). Engineered nourishments will need to either cease or be modified to ensure that any added sediment encourages the growth of the natural ecosystem rather than smothers it (Cheong et al. 2013). This may be achieved by using methods that give a gradual sediment flux, such as the sand engine in The Netherlands (Stive et al. 2013), or by using smaller doses of sediment.

It is imperative that we recognize the benefits of a vegetated foreshore ecosystem in preventing beach erosion and, therefore, increase the resistance of coastal areas to storm surges and flooding. Switching disturbed beach systems to natural self-sustaining ecosystems for coastal defense will require financial investments (e.g., from the World Bank, in the context of climate adaptation; UN 2016, World Bank 2017), the development of effective restoration methods, and altered governance. Only a collaborative approach of many stakeholders will ensure both economic and ecological benefits. This will require interdisciplinary collaboration among economists focusing on tourism, ecologists focusing on ecosystem functioning, and natural values, engineers focusing on physical processes and design measures, and sociologists focusing on governance processes and public support. With this article, we aim to provide an alternative beach management regime to traditional engineering solutions, by highlighting the viable and self-sustaining capacity of vegetated beach foreshore ecosystem in preventing erosion. Using an effective natural solution to coastal erosion will help to increase the resilience of tropical coastal areas to climate change in a sustainable way.

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Data availability

Data associated with this study are available from 4TU. Centre for Research Data at *https://doi.org/10.4121/uuid:a5f07774-9a90-4aa2-ae03-690da7d36a77*.

References cited

Adams MP, Ghisalberti M, Lowe RJ, Callaghan DP, Baird ME, Infantes E, O'Brien KR. 2018. Water residence time controls the feedback between seagrass, sediment and light: Implications for restoration. Advances in Water Resources 117: 14–26.

Armstrong ME, Miller AI. 1988. Modern carbonate sediment production and its relation to bottom variability Grahams Harbor, San Salvador, Bahamas. Pages 23–32 in Mylroie JE, Gerace DT, eds. Proceedings of the Fourth Symposium on the Geology of the Bahamas. Bahamian Field Station.

Balke T, Herman PMJ, Bouma TJ. 2014. Critical transitions in disturbance-driven ecosystems: Identifying Windows of Opportunity for recovery. Journal of Ecology 102: 700–708.

Bishop MJ, Peterson CH, Summerson HC, Lenihan HS, Grabowski JH. 2006. Deposition and long-shore transport of dredge spoils to nourish beaches: Impacts on benthic infauna of an ebb–tidal delta. Journal of Coastal Research 223: 530–546.

De Boer WF. 2007. Seagrass–sediment interactions, positive feedbacks and critical thresholds for occurrence: A review. Hydrobiologia 591: 5–24.

Castelle B, Turner IL, Bertin X, Tomlinson R. 2009. Beach nourishments at Coolangatta Bay over the period 1987–2005: Impacts and lessons. Coastal Engineering 56: 940–950.

Chave KE, Smith SV, Roy KJ. 1972. Carbonate production by coral reefs. Marine Geology 12: 123–140.

Cheong SM, Silliman B, Wong PP, Van Wesenbeeck B, Kim CK, Guannel G. 2013. Coastal adaptation with ecological engineering. Nature Climate Change 3: 787–791.

Christianen MJA, van Belzen J, Herman PMJ, van Katwijk MM, Lamers LPM, van Leent PJM, Bouma TJ. 2013. Low-canopy seagrass beds still provide important coastal protection services. PLOS ONE 8 (art. e62413).

Christianen MJA, et al. 2014. Habitat collapse due to overgrazing threatens turtle conservation in marine protected areas. Proceedings of the Royal Society B 281: 20132890–20132890.

Defeo O, McLachlan A, Schoeman DS, Schlacher TA, Dugan J, Jones A, Lastra M, Scapini F. 2009. Threats to sandy beach ecosystems: A review. Estuarine, Coastal and Shelf Science 81: 1–12.

Eckrich CE, Holmquist JG. 2000. Trampling in a seagrass assemblage: Direct effects, response of associated fauna, and the role of substrate characteristics. Marine Ecology Progress Series 201: 199–209.

Feagin RA, Figurelus J, Zinnert JC, Sigren J, Martínez ML, Silva R, Smith WK, Cox D, Young DR, Carter G. 2015. Going with the flow or against the grain? The promise of vegetation for protecting beaches, dunes, and barrier islands from erosion. Frontiers in Ecology and the Environment 13: 203–210.

Finkl CW, Walker HJ. 2005. Beach nourishment. Pages 147–161 in Schwartz ML, ed. Encyclopedia of Coastal Sciences. Springer.

Fonseca MS, Cahalan JA. 1992. A preliminary evaluation of wave attenuation by four species of seagrass. Estuarine, Coastal, and Shelf Science 35: 565–576.

Freile D. 2004. Carbonate productivity rates of Halimeda in two different locations, San Salvador Island, Bahamas. Pages 95–106 in Lewis RD, Panuska BC, eds. Proceedings of the 11th Symposium on the Geology of the Bahamas and Other Carbonate Regions. Gerace Research Centre, Auburn University.

Garrigue C. 1991. Biomass and production of two Halimeda species in the southwest new Caledonian lagoon. Oceanologica Acta 14: 581–588.

Green EP, Short FT. 2003. World Atlas of Seagrasses. Published in association with UNEP-WCMC by the University of California Press.

Hallock P. 1981. Production of carbonate sediments by selected large benthic foraminifera on two pacific coral reefs. Journal of Sedimentary Research 51: 467–474.

Hansen JCR, Reidenbach MA. 2012. Wave and tidally driven flows in eelgrass beds and their effect on sediment suspension. Marine Ecology Progress Series 448: 271–287.

Hanson H, Brampton A, Capobianco M, Dette HH, Hamm L, Laustrup C, Lechuga A, Spanhoff R. 2002. Beach nourishment projects, practices, and objectives: A European overview. Coastal Engineering 47: 81–111.

van der Heide T, Van Nes EH, Geerling GW, Smolders AJP, Bouma TJ, Van Katwijk MM. 2007. Positive feedbacks in seagrass ecosystems: Implications for success in conservation and restoration. Ecosystems 10: 1311–1322.

Hughes P. 1956. A determination of the relation between wind and sea-surface drift. Quarterly Journal of the Royal Meteorological Society 82: 494–502.

Infantes E, Orfila A, Bouma TJ, Simarro G, Terrados J. 2011. Posidonia oceanica and Cymodocea nodosa seedling tolerance to wave exposure. Limnology and Oceanography 56: 2223–2232.

van Katwijk MM, et al. 2016. Global analysis of seagrass restoration: The importance of large-scale planting. Journal of Applied Ecology 53: 567–578.

Kemp WM, et al. 2005. Eutrophication of Chesapeake Bay: Historical trends and ecological interactions. Marine Ecology Progress Series 303: 1–29.

Luijendijk A, Hagenaars G, Ranasinghe R, Baart F, Donchyts G, Aarninkhof S. 2018. The state of the World's beaches. Scientific Reports 1–11.

Marbà N, Arias-Ortiz A, Masqué P, Kendrick GA, Mazarrasa I, Bastyan GR, Garcia-Orellana J, Duarte CM. 2015. Impact of seagrass loss and subsequent revegetation on carbon sequestration and stocks. Journal of Ecology 103: 296–302.

Maxwell PS, et al. 2017. The fundamental role of ecological feedback mechanisms for the adaptive management of seagrass ecosystems: A review. Biological Reviews 92: 1521–1538.

Merten M. 1971. Ecological observations of Halimeda macroloba Decaisne (Chlorophyta) on Guam. Micronesica 7: 27–44.

Morris RL, Konlechner TM, Ghisalberti M, Swearer SE. 2018. From grey to green: Efficacy of eco-engineering solutions for nature-based coastal defense. Global Change Biology 24: 1827–1842.

Multer HG. 1988. Growth rate, ultrastructure and sediment contribution of Halimeda incrassata and Halimeda monile, Nonsuch and Falmouth Bays, Antigua, W.I. Coral Reefs 6: 179–186.

Neumann ACC, Land LS. 1975. Lime mud deposition and calcareous algae in the Bight of Abaco, Bahamas: A budget. Journal of Sedimentary Research 45: 763–786.

Nicholls RJ, Cazenave A. 2010. Sea level rise and its impact on Coastal Zones. Science 328: 1517–1520.

Orth RJ, et al. 2006. A global crisis for seagrass ecosystems. BioScience 56: 987–996.

Payri CE. 1988. Halimeda contribution to organic and inorganic production in a Tahitian reef system. Coral Reefs 6: 251–262.

Potouroglou M, Bull JC, Krauss KW, Kennedy HA, Fusi M, Daffonchio D, Mangora MM, Githaiga MN, Diele K, Huxham M. 2017. Measuring the role of seagrasses in regulating sediment surface elevation. Scientific Reports 1–11.

Ranasinghe R, Turner IL. 2006. Shoreline response to submerged structures: A review. Coastal Engineering 53: 65–79.

Reise K, Kohlus J. 2008. Seagrass recovery in the Northern Wadden Sea? Helgoland Marine Research 62: 77–84.

Ruiz-Martínez G, Mariño-Tapia I, Mendoza Baldwin EG, Silva Casarín R, Enríquez Ortiz CE. 2015. Identifying coastal defense schemes through morphodynamic numerical simulations along the Northern Coast of Yucatan, Mexico. Journal of Coastal Research 651–670.

Saunders MA, Lea AS. 2008. Large contribution of sea surface warming to recent increase in Atlantic hurricane activity. Nature 451: 557–560.

Saxby T. Saxby. Integration and Application Network, University of Maryland Center for Environmental Science. ian.umces.edu/imagelibrary/.

Scheffer M, Carpenter S, Foley JA, Folke C, Walker B. 2001. Catastrophic shifts in ecosystems. Nature 413: 591–596.

Scoffin TP. 1970. The trapping and binding of subtidal carbonate sediments by marine vegetation in Bimini Lagoon, Bahamas. Journal of Sedimentary Petrology 40: 249–273.

Silva R, et al. 2014. Present and future challenges of coastal erosion in Latin America. Journal of Coastal Research 71: 1–16.

Silva R, Mendoza E, Mariño-Tapia I, Martínez ML, Escalante E. 2016. An artificial reef improves coastal protection and provides a base for coral recovery. Journal of Coastal Research 75: 467–471.

Stive MJF, et al. 2013. A new alternative to saving our beaches from sea-level rise: The sand engine. Journal of Coastal Research 29: 1001–1008.

Suykerbuyk W, Bouma TJ, Govers LL, Giesen K, de Jong DJ, Herman P, Hendriks J, van Katwijk MM. 2016. Surviving in changing seascapes: Sediment dynamics as bottleneck for long-term seagrass presence. Ecosystems 19: 296–310.

Trembanis AC, Pilkey OH. 1998. Summary of Beach Nourishment along the U.S. Gulf of Mexico Shoreline. Journal of Coastal Research 14: 407–417.

[UN] Secretary general of the United Nations. 2016. United Nations Economic and Social Council Progress towards the Sustainable Devlopment Goals. United Nations.

van Tussenbroek BI, Van Dijk JK. 2007. Spatial and temporal variability in biomass and production of psammophytic Halimeda incrassata (Bryopsidales, Chlorophyta) in a Caribbean reef lagoon. Journal of Phycology 43: 69–77.

[UNEP-WCMC] United Nations Environment Programme World Conservation Monitoring. 2005. Global distribution of seagrasses (version 3.0). Third update to the data layer used in Green and Short (2003). http://data.unepwcmc.%0Aorg/data sets/7.

UNWTO. 2016. UNWTO Tourism Highlights 2016 Edition. World Tourism Organization (UNWTO).

van Tussenbroek BI, Hernández Arana HA, Rodríguez-Martínez RE, Espinoza-Avalos J, Canizales-Flores HM, González-Godoy CE, Barba-Santos MG, Vega-Zepeda A, Collado-Vides L. 2017. Severe impacts of brown tides caused by Sargassum spp. on near-shore Caribbean seagrass communities. Marine Pollution Bulletin 122: 272–281.

Walker DJ, Dong P, Anastasiou K. 1991. Sediment transport near groynes in the Nearshore Zone. Journal of Coastal Research 7: 1003–1011.

Wefer G. 1980. Carbonate production by algae Halimeda, Pencillus and Padina. Nature 285: 323–324.

Widdows J, Pope ND, Brinsley MD, Asmus H, Asmus RM. 2008. Effects of seagrass beds (Zostera noltii and Z. marina) on near-bed hydrodynamics and sediment resuspension. Marine Ecology Progress Series 358: 125–136.

Williams SL. 1990. Experimental studies of Caribbean seagrass bed development. Ecological Monographs 60: 449–469.

World Bank. 2017. Atlas of Sustainable Development Goals 2017: From World Development Indicators. World Bank.

Rebecca K. James (rebecca.james@nioz.nl) and Tjeerd J. Bouma are affiliated with the Department of Estuarine and Delta Systems at the Royal Netherlands Institute for Sea Research and Utrecht University, in Yerseke, The Netherlands. Rodolfo Silva and Mireille Escudero-Castillo are affiliated with the Instituto de Ingeniería at the Universidad Nacional Autónoma de México, in Mexico City. Brigitta I. van Tussenbroek is affiliated with the Unidad Académica de Sistemas Arrecifales—Puerto Morelos, the Instituto de Ciencias del Mar y Limnología, and the Universidad Nacional Autónoma de México, in Puerto Morelos. Ismael Mariño-Tapia is affiliated with the Departamento de Recursos del Mar, in Mérida, Mexico. Henk A. Dijkstra and René M. van Westen are affiliated with the Institute for Marine and Atmospheric research, Utrecht, in the Department of Physics at Utrecht University, in Utrecht, The Netherlands. Julie D. Pietrzak, Adam S. Candy, Caroline A. Katsman, Carine G. van der Boog, and Peter M. J. Herman are affiliated with the Department of Hydraulic Engineering, at the University of Technology, in Delft, The Netherlands. Riccardo E. M. Riva, Cornelis Slobbe, and Roland Klees are affiliated with the Department of Geoscience and Remote Sensing at the University of Technology, in Delft, The Netherlands. Johan Stapel is affiliated with the Caribbean Netherlands Science Institute, in Sint Eustatius, in the Caribbean Netherlands. Tjisse van der Heide is affiliated with the Department of Coastal Systems at the Royal Netherlands Institute for Sea Research and with Utrecht University, in Texel, The Netherlands. Marieke M. van Katwijk is affiliated with the Department of Environmental Science at the Institute for Water and Wetland Research, in the Faculty of Science of Radboud University, in Nijmegen, The Netherlands.