Seagrass Restoration: A Story of Success or Not?

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Summary

Throughout the world, ecosystems provide countless of services for humanity. Seagrass ecosystems are one of such systems. They are one of the most productive and valuable primary producers in the world, as they are of great importance for marine biodiversity, ecological and economical reasons (Burke *et al*., 2001; Costanza *et al*., 1997; Duarte and Chiscano, 1999). Seagrass meadows, varying from small patches to vast fields, inhabit shallow estuarine environments of all the world's continents except Antarctica (Green and Short, 2003; van der Heide *et al*., 2009). Seagrasses have come to dominate many marine coastal ecosystems and have proven to be a robust species, but during the last decades, over 90000 ha of seagrass loss has been documented worldwide (Short and Wyllie-Echeverria, 1996; Burke *et al*., 2001). Multiple stressors, such as eutrophication, disease, sedimentation and toxicity events (van der Heide *et al*., 2009) can act simultaneously at different temporal and spatial scales, resulting in large scale seagrass-loss (Orth *et al*., 2006). Currently there are approximately 19 monitoring programs that involve 30 seagrass species in 44 countries (Orth *et al*., 2009). Regardless of all our efforts, restoration and transplantation projects have a low success rate; only 30% of conservation projects are successful (Fonseca *et al*., 1998). In this thesis, I wish to examine various case studies to gain insight into which abiotic and biotic factors influence the outcome of a conservation project. In this thesis, four different case studies have been discussed; Orbetello lagoon, Tampa Bay, Chesapeake Bay and the Wadden Sea (Lenzi *et al*., 2003, Greening *et al*., 2006, Orth *et al*., 2009 and van Katwijk *et al*., 2009, respectively). During the thesis a distinction is made between two different means of conservation projects; 1) Habitat restoration, where abiotic an factors of a system are brought back into favorable conditions, either directly or indirectly, for seagrasses to thrive in and 2) Seagrass transplantation, where seagrasses shoots are transplanted through various techniques, hereby (re)introducing seagrass in the hope that it can populate the area. Habitat restoration, in the case of Orbetello lagoon, showed 50 times more seagrass recovered in ha, and in Tampa Bay, seagrass recovered was at most 10 times more, when compared to the transplantation efforts in the Chesapeake Bay and the Wadden Sea are It can be concluded that the main factor influencing the success of restoration projects is to what extend a seagrass meadow is exposed to currents and wave action. Even if moderate levels of eutrophication persist, restoration can still be a success. When comparing the nutrients concentration between the Orbetello lagoon and the Wadden Sea, it is shown that in the Orbetello lagoon restoration was a success, whereas in the Wadden Sea it was not. This confirms the conclusion stated above. The two different means of restoring seagrass habitats do not exclude one another.

Contents

Introduction

Throughout the world, ecosystems provide countless of services for humanity. Our entire civilization is based on the resources attained by reaping the fruits of nature (Costanza, 1997). Therefore it is of utmost importance that ecosystems are kept in a healthy and stable condition. Sadly though, for too long have we consumed without realizing the consequences of our actions. Ecosystems worldwide are being damaged either directly through waste and disease or indirectly by processes as eutrophication and global warming (Burke *et al*, 2001). Even more, in terms of evolutionary history, the rate at which the climate is changing is unparalleled. Therefore, in order for us to be able to keep on reaping the fruits that mother nature provides for us, we need a better understanding of how ecosystems function and respond to human as well as natural induced stresses so that a more conscience and sustainable approach can be applied while managing our ecosystems (Burke *et al*, 2001).

Seagrass ecosystems are one of such systems where a better understanding is needed about how they react to stressors. They are one of the most productive and valuable primary producers in the world, as they are of great importance for marine biodiversity, ecological and economical reasons (Burke *et al*., 2001; Costanza *et al*., 1997; Duarte and Chiscano, 1999). Seagrass meadows, varying from small patches to vast fields, inhabit shallow estuarine environments of all the world's continents except Antarctica (Green and Short, 2003; van der Heide *et al*., 2009). Seagrasses have come to dominate many marine coastal ecosystems and have proven to be a robust species, but during the last decades over 90000 ha of seagrass loss has been documented worldwide (Short and Wyllie-Echeverria, 1996; Burke *et al*., 2001). Multiple stressors, such as eutrophication, disease, sedimentation and toxicity events (van der Heide *et al*., 2009) can act simultaneously at different temporal and spatial scales, resulting in large scale seagrass-loss (Orth *et al*., 2006). These stressors have caused many seagrass systems to collapse, and undergo a transition into a different stable state where either macroalgal or phytoplankton communities dominate (Viaroli, 2008). A shift in stable states can result in hysteresis and making it all the more difficult for the seagrass to recover when a community shift has occurred even if external stressors are removed. As a result a pristine seagrass ecosystem can become a barren underwater landscape (e.g. The Wadden Sea. (van Katwijk *et al*., 2009; Viaroli, 2008)).

In response to these stressors and worldwide seagrass habitat loss, seagrass restoration programs have been initiated on a global scale(van Katwijk *et al*., 2009) and over the last decade marine protected areas that include seagrass have increased(Orth *et al*., 2009). Currently there are approximately 19 monitoring programs that involve 30 seagrass species in 44 countries (Orth *et al*., 2009). Regardless of all our efforts, restoration and transplantation projects have a low success rate; only 30% of conservation projects are successful (Fonseca *et al*., 1998).

Therefore in this thesis, I wish to examine various case studies in order to gain insight into which abiotic and biotic factors influence the outcome of a conservation project. I aim to understand how two different methods of conservation influence to recovery of a depauperate seagrass ecosystems. I have chosen this approach to understanding seagrass ecosystems and the stressors that act upon it, as it has a more practical value. During the thesis a distinction is made between two different means of conservation projects; 1) Habitat restoration, where abiotic an factors of a system are brought back into favorable conditions, either directly or indirectly, for seagrasses to thrive in and 2) Seagrass transplantation, where seagrasses shoots are transplanted through various techniques, hereby (re)introducing seagrass in the hope that it can populate the area. While comparing these two methods, while first gaining a global understanding of how seagrass ecosystems function, I ask what the difference is between them and what the possible causes of success or failure are.

Seagrass

Biology and Ecology

Seagrasses are meadow forming marine rhizomatous angiosperms that look and although their name states otherwise, are not related to terrestrial grasses (Burke *et al*., 2001; van der Heide *et al*., 2009). Seagrasses compromise out of leaves, stems, rhizomes and roots (FIG X). Seagrass meadows, varying from small patches to vast fields, are present in shallow estuarine environments in all the world's continents, except Antarctica. (van der Heide *et al*., 2009; Green and Short, 2003).

Seagrasses are fully equipped for a complete submerged existence in saline waters. They have developed unique ecological, physiological and morphological adaptations (Orth *et al*., 2006). Some of these adaptations are: internal gas transport, epidermal chloroplasts, submarine pollination and marine dispersal (Les *et al*., 1997; Orth *et al*., 2006).

Although seagrasses have a worldwide distribution, they have a low taxonomic diversity, approximately 60 species (Orth *et al*., 2006; Green and Short, 2003). Seagrasses have evolved to three different lines from a single lineage of monocotyledonous flowering plant about 70-100 million years ago (Orth *et al*., 2006). The three different seagrasses are Hydrocharitaceae, Cymodoceaceae complex and Zosteraceae. In the most general sense, seagrass can be placed into two different groups, temperate and tropical.

Vegetative growth occurs through the spreading of rhizomes, resulting in seagrass meadows that are formed through the expansion of genetic individuals (genets), i.e. clones. Even though seagrass is a flowering plant, sexual reproduction occurs infrequent and can be hampered by hydrodynamic conditions, as seed dispersal is at maximum range of only a few meters (Duarte et al, 2006. Chapter 11). How seagrass meadows disperse depend on factors as rhizome growth rate and frequency of sexual reproduction and the abiotic factors influencing both mechanism. Because seagrass has two different means of dispersal, the genetic diversity of a meadow is the result of a balance between clonal growth and sexual reproduction (Dorken and Eckert, 2001). In this sense, seagrass habitats are highly dynamic, as they are maintained through the recruitment of new clones and sexual reproduction (Duarte et al, 2006. Chapter 11). Recruitment can take place on the outskirts of a seagrass meadow or in empty patches created by disturbances (anthropogenic or natural origin).

Seagrasses are able to take up nutrients through the water column as well through the interstitial water of the sediment. Seagrasses are able to take in nutrients through their leaves as well as through their roots and rhizomes, hereby coping with the fact that seagrasses usually occur in open systems, which can results in nutrient run off (Stapel, 1996). Although not all nutrients are assimilated, as excess detritus is transported to deeper offshore parts of the ocean (Suchanek et al., 1985).

Generally seagrass grow on soft substrate such as sand or mud. In most seagrass ecosystems the top few millimeter or centimeter below the sediment service are anaerobic, mainly caused by the slow diffusion rate into the sediment and the high oxygen demands of microbial processes and of seagrass itself (Terrados *et al*., 1999). Therefore, they need to transport oxygen to their rhizomes and roots. In order to support the transport of O^2, seagrasses require some of the highest light levels of any plant group worldwide (Orth *et al*., 2006).

Seagrasses are known to be habitat engineers, meaning that they alter the physical and chemical conditions in the water column and sediments (Marba *et al*., 2006. Chapter 6). Carbon and nutrient dynamics can be altered by the metabolic activities of seagrass, significantly lowering

nutrient levels in the water column (Moore, 2004). The structure of the canopy also influences hydrodynamic processes such as water current velocity and waves, thereby decreasing the amount of suspended particles (Koch *et al*., 2006. Chapter 8) and preventing the resuspension of sediments (Garcia and Duarte, 2001; van der Heide *et al*., 2007). The reduction of nutrients and water velocity both influence turbidity levels of the water column, as both these processes reduce phytoplankton, epiphytes and suspended sediment (van der Heide *et al*., 2007). As turbidity levels decrease, higher concentrations of light are able to penetrate deeper into the water column, favoring conditions for seagrass ecosystems.

Not only do the seagrasses create conditions favorable for themselves but they also serve as wildlife habitat and provide stability for coastal systems (Campanella, 2009). Wildlife diversity in seagrass meadows is vast in numbers as it functions as heaven for many species. Also the meadows serve as nursery grounds, where organisms in their juvenile stage can benefit from the protection granted by the canopy in order to avoid predation (Orth *et al*. 2006). Top predators that are associated with seagrass ecosystems include: sea turtles, dugongs and manatees.

Stressors

Because coastal zones have become highly urbanized and correspondingly human population densities have increased immense, anthropogenic-derived nutrient inputs to coastal waters has increased dramatically (Deegan *et al*., 2002). Through pathways as rivers and estuaries, sewage discharge has been reaching coastal waters, increasing nutrient loads directly. This process known as eutrophication has degraded many marine coastal systems, and is one of the main drivers behind the drastic loss of seagrass (Green and Short, 2003). Although the increase in nitrogen and phosphorus does not harm seagrass directly (Despite the fact that vast amounts of nitrate can cause nitrate inhibition, impending seagrass growth directly (Hauxwell and Valiela, 2004), it is the growth of phytoplankton, epiphytes and macroalgae in nutrient rich waters which imposes stress on seagrass (Burkholder *et al*., 2007). As can be seen in Fig. 1, as Macroalgae, Epiphytes and Phytoplankton are no longer nutrient limited, it is through the competition for light that seagrass meets its demise. Because epiphytes appropriate light and Phytoplankton increase turbidity levels seagrass no longer receives the necessary amount of energy required

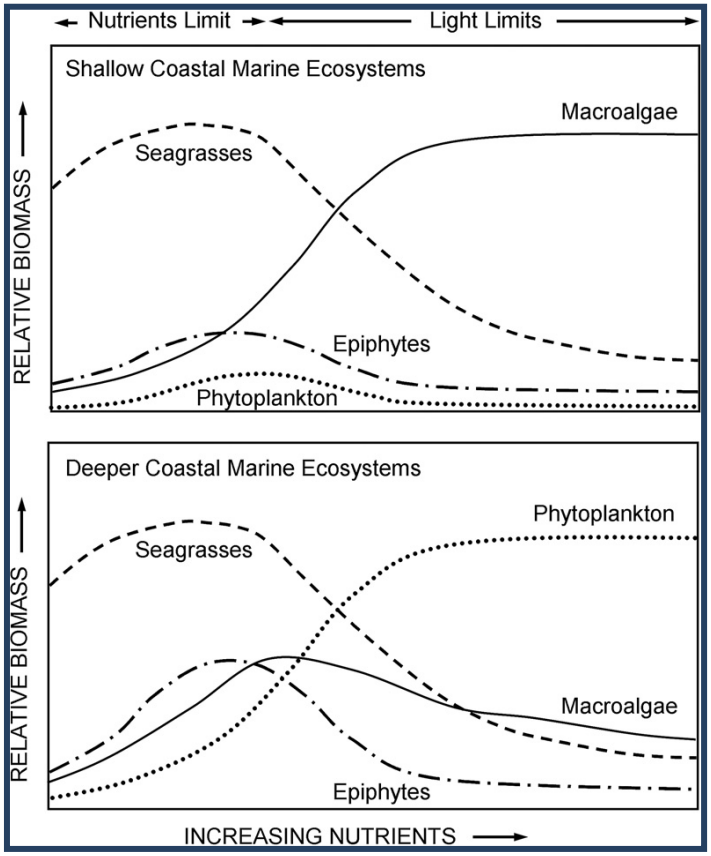


Fig. 1.
Generalized scheme which represents the shift in the biomass of major groups of primary producers with increasing nutrient enrichment to shallow and deeper coastal marine waters (upper and lower panels, respectively). In this generalized format, it can be seen that nutrients limit production. As eutrophication progresses, light becomes the primary limiting factor. Macroalgae (in shallow waters) or phytoplankton (in deeper-waters) dramatically increases, while sea-grasses decline. As seen in Burholder *et al*., 2007.

to compete with the other floral species.

It is generally agreed that eutrophication is one of the most widely reported causes of seagrass decline (Ralph *et al.*, Chapter 24. 2006). But there are still other stressors that play an important role in the decline of seagrass.

Fish farms also contribute to the decline of seagrass. Unconsumed food and fish excretion adds pollution and eutrophication (e.g. nutrients, organic matter) to the water column. Also, the cages create unfavourable conditions by altering light regimes and turbidity on a local scale (Pergent-Martini et al. 2006a).

Fishing activities can cause direct damage to seagrass meadows. Trawling for example can create vast gaps, hypothesized to form metapopulations. Other activities as dredging and land reclamation also directly impact seagrass. Furthermore, they can also alter the physical conditions of the water column by increasing turbidity caused by (re)suspended particles (Erftemeijer 2006). All large scale physical altering activities which take place in marine habitat contribute in the same manner (e.g. the burying of underwater cables, construction of harbors etc.).

Case studies

Habitat restoration

Like so many other lagoons (Morand and Briand, 1996), the Orbetello lagoon in Italy has developed a macroalgae community through the extend discharge of nutrient rich waters which have anthropogenic origins. Not only has the shift of community from seagrass dominated (*Zostera Noltii*) to seaweed *(Chaetomorpha linum*, *Cladophora vagabunda*, *Gracilaria verrucosa* and *Ulva rigida*) dominated further altered the chemical composition of the water column and sediment, which led to mortalities of aquatic fauna, but also has caused problems for local tourism by discoloring water to the adjacent beaches (Lenzi *et al*., 2003).

As a response to these problems, the Orbetello Lagoon Environmental Reclamation Authority (OLERA) was formed. OLERA had formulated a mission in the hope of creating conditions less favorable for seaweeds and thereby giving seagrass communities the space to thrive. The mission encompassed three main goals: 1) Removal of macroalgae from the lagoon, 2) Increase the clean sea water entering the lagoon and 3) decreasing nutrient discharge of anthropogenic origin, i.e. all waste water from anthropogenic sources was first pumped into treatment plants and phytotreatment areas, hereby decreasing the nutrients entering the lagoon. Lenzi and others (2003) researched what the relationship was between nutrient loading and macroalgae and seagrass communities

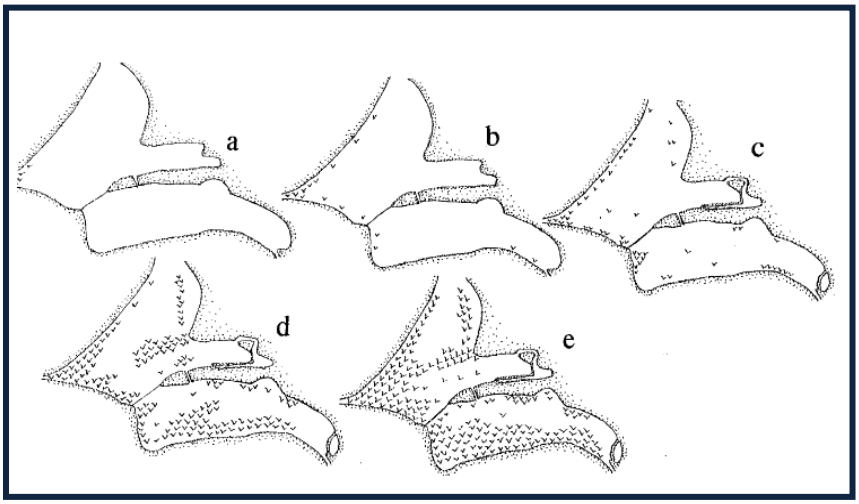


Fig. 2.
Phanerogam distribution in the Orbetello lagoon, between 1993 and 2000: (a) 1993 (Bombelli and Lenzi, 1996; modified); (b, c) 1995, 1997, respectively (Lenzi, unpub. data); (d) 1999; (e) 2000. As seen in Lenzi *et al*., 2003.

Through August 1999 and July 2000 Lenzi and others carried out a monitoring program, during which: Seagrass and seaweed distribution were measured, annual nitrogen (N) and phosphorous (P) budgets were measured and dissolved organic nitrogen (DIN. The sum of NNH_4^+, NNO_2^-, NNO_3^-) and soluble reactive phosphorous (SRP) were measured in the lagoon and at waste water treatment areas. Also water quality parameters as pH, temperature and dissolved oxygen (DO) in percents of saturation were measured.

Results showed an average pH of 8.27±0.58 and DO content ranged between 40 and 230, with an average of 92.72±44.00 in the lagoon. High fluctuations were caused by contrasting photosynthetic activities in spring and summer, increasing pH and DO in the spring and decreasing in the summer. DIN as well as SRP showed wide temporal ranges. DIN varied between 12.0 and 85.1 µM, highest values being between November and January (1999-2000). SRP had values ranging from 0.1 to 0.9µM. DIN and SRP values showed significant reduction, 85% and 79% respectively, in treated plants. Despite the system's efficiency, nutrient values were still extremely high compared to lagoon values. DIN and SRP values ranging approximately

between 600 µM to 1800µM and 42 µM to 120 µM, respectively. Annual N and P budget of all anthropogenic sources were 272.83 and 17.61 tons, respectively. Water treatment areas were able to reduce 35.02% of N and 40.05% of P. Total annual budget for N and P values for the top 3 cm of sediment, were 1,444 and 198 tons respectively.

The results showed that the macroalgal growth was increased by the input of nutrient rich waters into the lagoon. This was the main factor decreasing the growth of seagrass, as they were unable to compete with the macroalgae in eutrophic conditions. Because nutrients were no longer the limiting factor, macroalgae could out compete seagrass and sunlight became the limiting factor for seagrass. But, as nutrient input was decreased macroalgae was only present near areas were anthropogenic nutrient rich discharge persisted. Also, the removal of approximately 200 tons of macroalgae from the period 1993-2000 contributed to the reestablishment of seagrasses. Seagrass slowly yet steadily started to recover as its presence started to increase in the period from 1993-2000, as it finally covered 50% of the lagoon (Fig. 2), equivalent to 10,000 ha. Areas where seagrass did not dominate were inhabited by macroalgae and correspondingly the areas where nutrient discharge still persisted.

There are also other factors which strongly influence the restoration of seagrass. Increased seawater volume pumped in the lagoon ensured faster water flow, anthropogenic sources were confided to specific areas and only discharged nutrient rich waters locally and the removal of seaweeds disturbed the sediment in less than one meter deep water; all these factors contributed to the oxygenation of organic matter in the sediment, favoring orthophosphate adsorption by the carbonate detritus and clays. Since the macroalgae present in the lagoon are P limited, the onset of these conditions favor seagrass.

The main focus of the previous restoration project lay on finding out how nutrients influenced seaweed and seagrass communities. Another, similar, restoration project in Tampa Bay, Florida USA, had a different approach to the same Dilemma. Greening and others (2006) did research on how water quality, in terms of turbidity, and seagrass communities responded to nitrogen loading.

Human populations surrounding the Tampa Bay area have been increasing drastically over the last decades, as numbers have quadrupled since the 1950's; over two million people live in 5617 km^2 watershed. The increase of human population in the Tampa Bay area resulted in increased nutrient discharge. As seen in Lenzi and others (2003), the effects of increased nutrient loading were visible. The increase nutrient loading resulted in reduction of dissolved oxygen in the water column when excess organic matter decomposes, more frequent occurrences of noxious algal blooms, hypoxia/anoxia, which in the end led to the drastic decline of seagrass and a substantial shift in ecosystem processes. The most visible symptom of eutrophication was the decrease in light penetration through the water column. As a result massive loss of seagrass habitat toke place, in the 1950's over 16,000 ha covered the bay area, but by the early 1980's, more than half was lost.

Most of the nitrogen input in the water originated from single identifiable localized sources (point sources). Although contributing the most to the nitrogen loading of the bay, point sources, unlike non point sources of pollution, can be addressed directly and regulated easier as the origin of pollution is known. Other sources contributing which were specified are; fertilizer run-off, ground water and springs, and atmospheric deposition. By the 1990's the restoration goals had reduced the contribution of point sources from 54% to 12%, and fertilizer input from 5% to 1%. Afterwards, nonpoint sources were the main source contributing to nitrogen, as it increased from 16% to 62%.

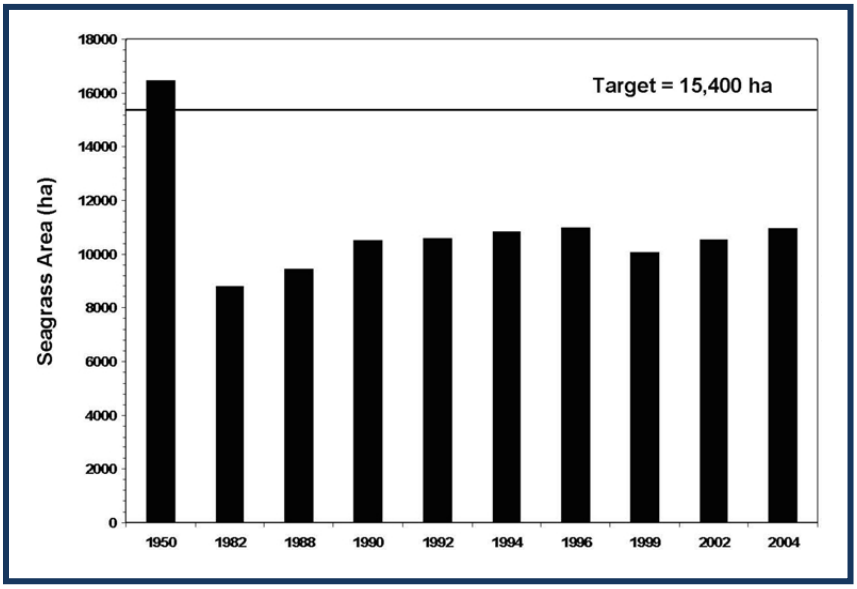
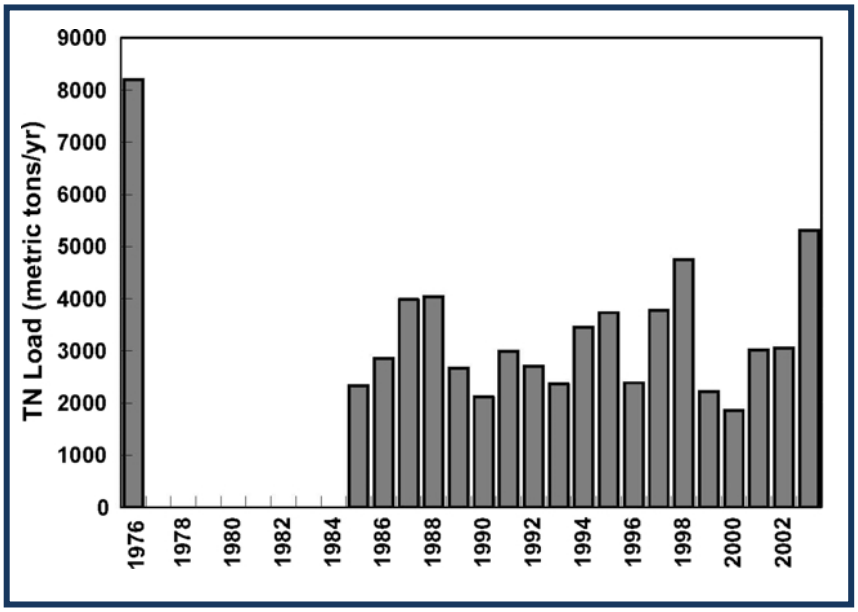


Fig. 3.
Changes in the area coverage (ha) of seagrasses in Tampa Bay since 1950. The long term seagrass recovery target of 15,400 ha is shown. As seen in Greening *et al.*, 2006.

Fig. 4.
Annual nitrogen loads (metric tons) to Tampa Bay for 1976 and 1985–2003. As seen in Greening *et al.*, 2006.



The increase in light attenuation was accompanied by an increase in algal biomass, caused by an increase in nitrogen loads. So, through a "simple" model, it was presumed that by decreasing nutrient load, so would light attenuating decrease and give seagrass communities the space to thrive once again. Unlike in Lenzi and others (2003) Greening and others (2006) used chlorophyll-*a* concentrations in Tampa Bay to determine the water quality, as a direct result from decreased nitrogen load entering the bay. A linear regression showed a reasonably strong correlation between chlorophyll-*a* (µg/L) and the depth of 20.5% light penetrating the water (r^2=0.67). This amount of light is the minimum needed for seagrass to sustain itself. Lenzi and others (2003) were focused on the amounts and different sources of N and P, whereas Greening and others (2006) were interested in water clarity in terms of chlorophyll-*a* concentration. Using Secchi discs, local government monitored light penetration depth of different parts of the bay from 1975 to 2005. All locations which were measured, showed a decline in light penetration during the late 1970's, but starting from 1980 and onwards, the bay showed lower depths at which light could penetrate. In the mid 1970's chlorophyll-*a* concentrations varied between 25 and 35 µg/L. Finally, around 2000, chlorophyll-*a* concentration goals were achieved, current mean annual values are typically less than 15 µg/L. Nitrogen loads entering the bay

were nearly cut in half, were in 1976 annual nitrogen loads were above 8000(metric tons/yr), in 2002 this was reduced to just above 5000 (metric tons/yr)(Fig.4). Accordingly, seagrass coverage increased from 1982 to 2004, although the goal to achieve 95% of coverage as in 1950 was not achieved. In 2004 seagrass covered over 10000 ha of the bay area, whereas the goal was set at 15,400 ha (Fig. 3). None the less, for most part all chlorophyll-*a* concentration and light attenuation targets were met. Of the four segments of the Tampa Bay area, only one remained in a degraded state.

Seagrass Transplantation

In the Chesapeake Bay, the mid Atlantic region of the USA, seagrass populations of *Zostera marina* showed a sudden and widespread change in the 1930's. Seagrass decline was thought to be caused by a fungal parasite and a 1933 hurricane which devastated the entire Chesapeake Bay area. This area consists out of three different bays, the Virginia Coastal Bay, the Chincoteague coastal Bay and the inlet Chesapeake Bay. Although seagrass populations started to recover after the 1930's, current distributional limits are dramatically different than those in the late 1950's and 1960's. This second decline in seagrass populations is caused by increased nutrient loading, causing reduced water clarity and an overgrowth of Macroalgae (Fig. 5). After this decline, the Chesapeake Bay started a slow recovery, in the 1990's total seagrass cover was 12,105 ha, but a third decline took place between 2003 and 2007. By 2007, only 5,829 ha of seagrass covered Chesapeake Bay, 2,100 ha in the Chincoteague coastal Bay and 591 ha in the

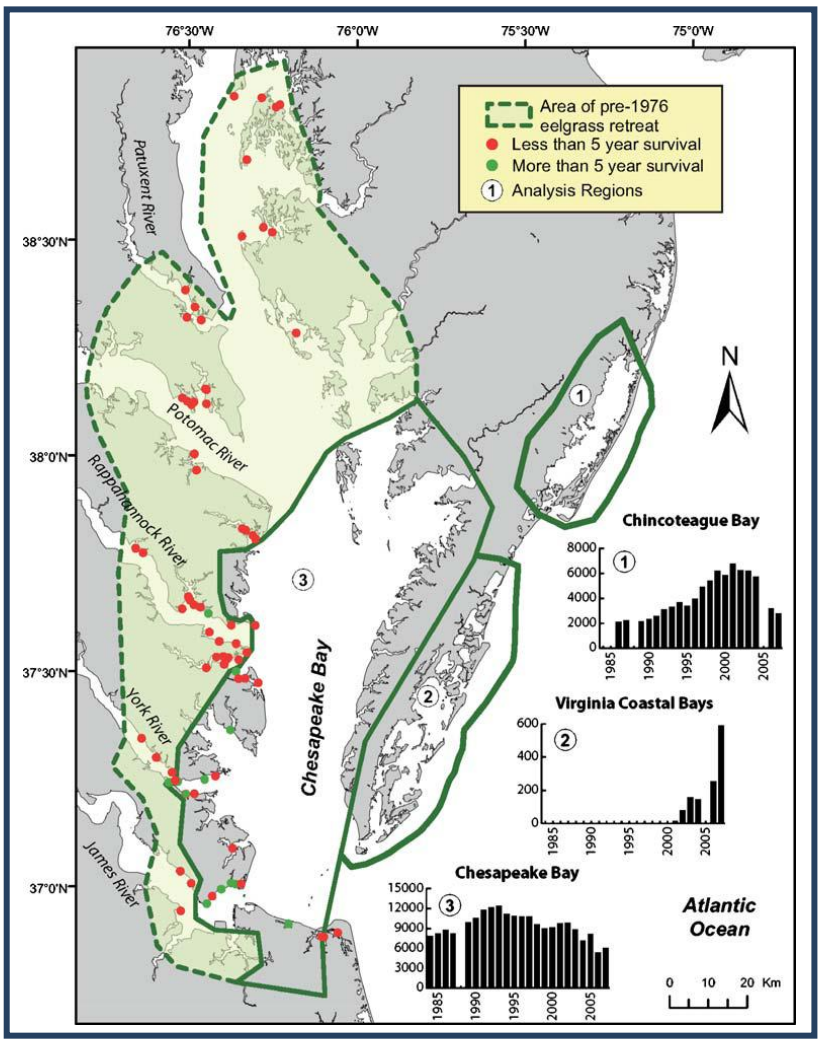


Fig. 3.
Map of Chesapeake Bay region showing three analysis regions and eelgrass abundance in each from 1984 to 2007. The shaded polygon in the Chesapeake Bay shows the upper extent of eelgrass distribution in the 1960s while the lower, clear polygon represents the current distribution. Red and green dots show eelgrass transplant projects between 1978 and 2006. The three graphs show the total coverage of seagrass (ha) according to year. Each graph represents a different location. As seen in Orth *et al.*, 2009.

Virginia coastal bays. For the greater part of the Chesapeake Bay area, seagrass is either absent or rare. Where seagrass is present, it grows on either sand or muddy-sand substrate, at depth range just below mean low water to approximately 2 meters in depth.

Orth *et al.* (2009) examined the Chesapeake Bay area and conducted research in order to understand which environmental issues influence the change of seagrass distribution *in situ*. Orth showed that the recovery of seagrass in the Chesapeake Bay was closely related to salinity, water depth, temperature and turbidity. Recovery rates differed according to salinity: upriver no natural recovery took place and all restoration efforts have been fruitless, midriver where eelgrass had disappeared completely but thanks to natural recovery and restoration efforts seagrass has repopulated the area and downriver where seagrass never disappeared completely and was able to recovered naturally. Light attenuation was highest at upriver sites, having the highest value at 3.1 K_d m^{-1} in August compared to the 1.9 K_d m^{-1} and 1 K_d m^{-1} at the midriver and river mouth, respectively. Furthermore, Secchi depths showed a continued decline since the mid 1980's till present. Light penetrated on average 1.5m into the water in 1985 and decreased to 1.1m by 2005. Throughout a year chlorophyll-*a* concentrations peaked in March and were highest at upriver and midriver location valuing at 10 and 11 CHL *a* µg l^{-1} respectively. At the river mouth chlorophyll valued at 6 CHL *a* µg l^{-1}. Dissolved inorganic nitrogen (DIN) did not show a distinct difference between river locations, peaking in September and October, with DIN values at 23 µM, 17 µM, and 9 µM, although the river mouth showed the lowest values throughout the year. Dissolved inorganic phosphorus (DIP) showed highest values at upriver location during the entire year, then mid river and lowest values at the river mouth. The river mouth varied between 0.4 µM and 0.5 µM, midriver varied between 0.5 µM and 1.0 µM, and upriver varied between 0.7 µM and 1.7 µM.

Since 1978 transplantation activities have failed to significantly increase seagrass abundance in

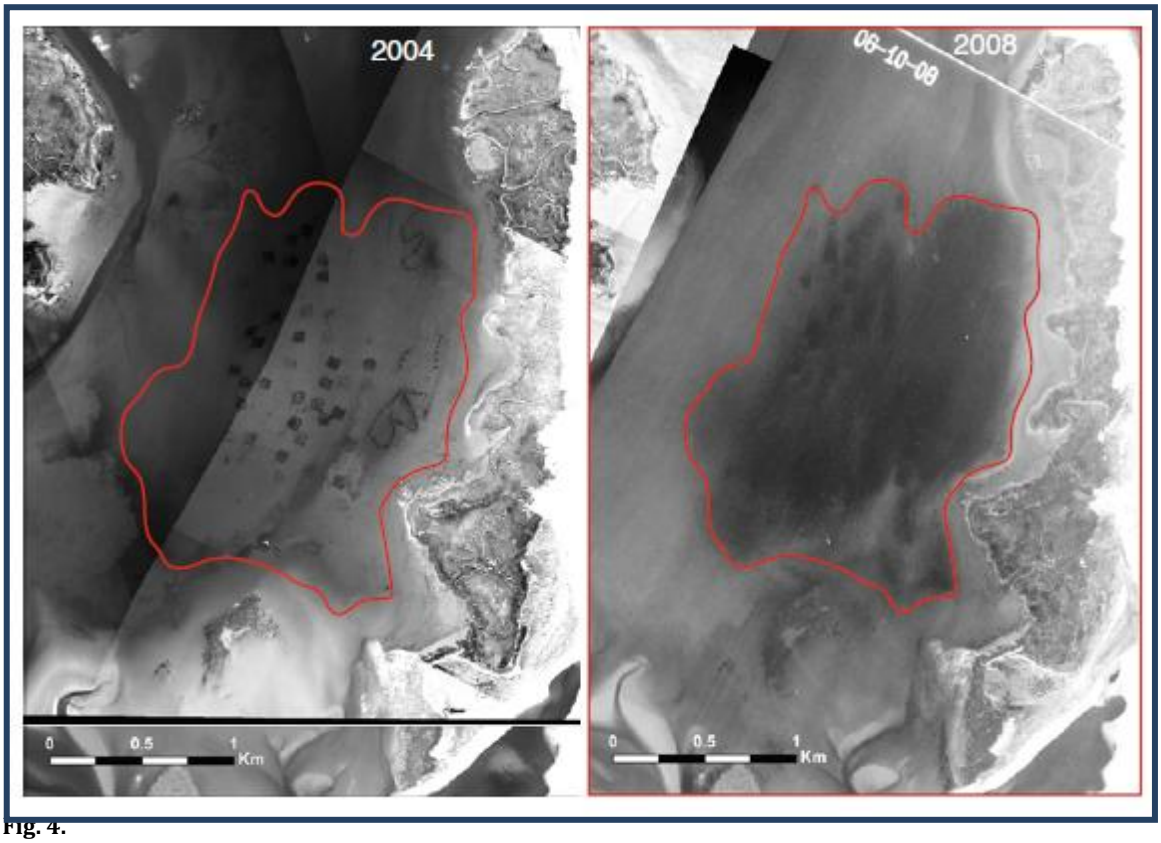


Fig. 4.
Aerial photographs of a seagrass restoration site in one of the four Virginia Coastal Bays (in 2004 (left) and 2008 (right)). The total area of cover by 2008 is approximately 300 ha, as shown by the red polygon. Dark squares in 2004 are 0.4-ha plots of eelgrass that were seeded in 2001 and 2002; by 2008, the area within the polygon has become almost completely vegetated with eelgrass. As seen in Orth *et al.*, 2009.

the Chesapeake Bay area. Although different techniques were used, e.g. transplanting adult plants or seeds, most restoration efforts failed, as the seagrass transplants died within 1-2 years of transplantation and barely any survived longer than 5 years. Seagrasses were transplanted between 1996 and 1998 and have continued to spread ever since; resulting in a total restored seagrass area of 35.5 ha. In the Virginia Coastal bays restoration efforts were more successful. By spreading 23 million seeds in 2007, 77 ha of seagrass were restored. Furthermore, seeds produced by the plants propagated the expansion of the seagrass meadow. Currently, the Virginia Coastal Bays is estimated to encompass 590 ha of seagrass (Fig. 6). Although no active restoration efforts toke place in the Chincoteague Bay, seagrass recovered naturally between 1986 and 2001, indicating that measures, which were taken by local governments, taken to improve water quality plays an important role in reestablishing seagrass habitat. The declines from 2002 through 2007 are related to the decrease in water quality. Orth et al. (2006b) showed that for chlorophyll-*a* concentrations and DIN most habitat requirements were met. Yet, for DIP and Total Suspended Solids (TSS) the habitat requirements were not met. Threshold values are: Chl a (15 ug/l), DIN (0.15 mg/l), DIP (0.02 mg/l) and TSS (15 mg/l).

The epidemic that took place in the 1930's not only damaged seagrass habit in the Eastern coast of the USA, but occurred on a transatlantic scale. *Zostera marina* was also endemic to the Dutch Wadden Sea, a group of populated islands in a tidal flat system situated of the northern coast of the Netherlands(Fig. 7). De Jonge *et al.*, (1997) states that the drastic change of seagrass cover of *Zostera marina* is attributed to the epidemic along with the closure of the Zuiderzee, an inlet, which changed physical and chemical conditions of the mudflats. These two major events attributed to the rapid and widespread decline of the 65-150 km² of seagrass that covered the Dutch Wadden Sea in the 1930's. Currently, less than 1 km² of seagrass is still present. The closure of the Zuiderzee caused the tidal range to increase by 15-25 cm in the tidal inlets. Furthermore, current velocities also changed, in some locations in decreased by 30%, while on a general scale velocities increased by 1.2 to 2.9.. These values indicate that a maximum potential growth depth for seagrass in the Wadden Sea varies. Other stressors such as land reclamation projects and fisheries can harm seagrass directly or indirectly by increasing turbidity levels. These factors could account for the large variation in annual turbidity levels. Furthermore, as seen in the previous case studies, nutrient loading also affected seagrass directly and indirectly. High nutrient loads can give rise to phytoplankton and macroalgae species (i.e. *Ulva* spp. and *Enteromorpha* spp.). As phytoplankton concentrations increase so do turbidity levels. The macroalgae species can hinder *Z.marina* by suffocating it. De Jonge *et al.*, (1996) shows that chlorophyll-*a* concentrations have been increasing annually since the mid 1970's. On average the

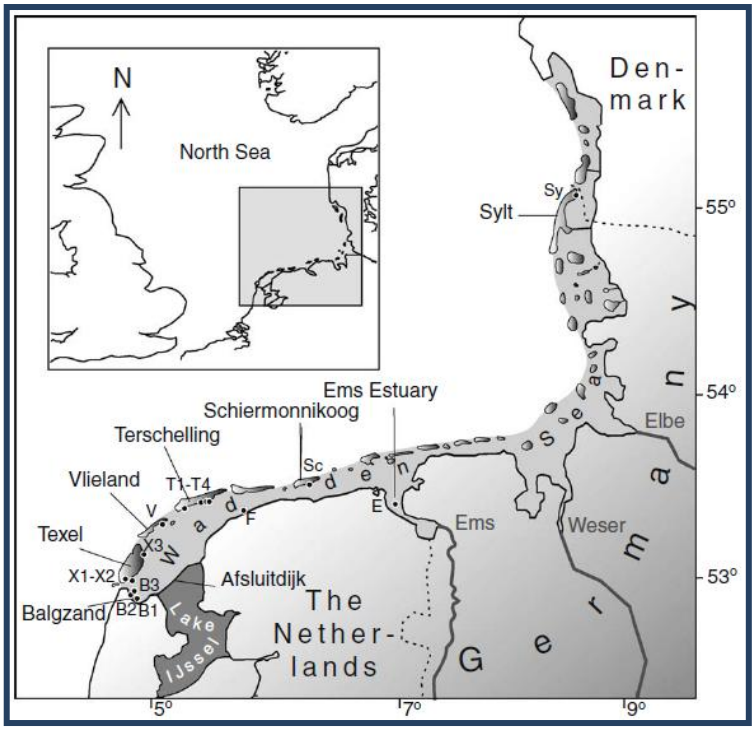


Fig. 5.
Map of the Wadden Sea, NW Europe with transplantation locations. As seen in van Katwijk *et al.*, 2009.

Wadden Sea showed 4 mg m^{-3} of chlorophyll-*a* in 1975 and increased to 11 mg m^{-3} in 1995. Till the late 1970's DIN and DIP increased. The main sources that contributes to eutrophication in the Wadden Sea has an anthropogenic origin. In 1975 DIN had reached its highest measured value at approximately 60 mmol m^{-3}, but by 1995 it decreased to 38 mmol m^{-3}. DIP showed a relatively steeper decline, in 1983 the highest value of DIP was measured valuing at 2.5 mmol m^{-3} and decreased to 0.7 mmol m^{-3} in 1995. The declines in DIP, DIN and chlorophyll-*a* concentrations were the result of new regulations imposed by different Dutch governmental bodies (Katwijk *et al*., 2009). Although these regulations led to the better water quality of the Wadden Sea, they weren't put into action for the purpose of seagrass restoration.

Katwijk et al. (2009) reviewed most, if not all, of the major seagrass transplantations that took place in the Wadden Sea till 2004. In total, 10,000 *Zostera noltii* shoots and 23,000 *Zostera marina* plants were transplanted between 1991 and 2004 (Fig. 7). The transplants were conducted in order to see the viability of *Z.noltii* and *Z.marina* and their survival in the Wadden Sea. In total 39 different seagrass transplantations were reviewed. Of the 39 transplants, 33 either failed or only survived for one season. Of the remaining 6 transplants, survival was short term, varying between 2-3 years. Two transplants survived longer i.e. one 8 years and the other 13 years and were still present in 2009.

Conclusion and Discussion

In this thesis, four different case studies have been discussed; Orbetello lagoon, Tampa Bay, Chesapeake Bay and the Wadden Sea (Lenzi *et al*., 2003, Greening *et al*., 2006, Orth *et al*., 2009 and van Katwijk *et al*., 2009, respectively). The first two case studies show how habitat restoration can facilitate and improve seagrass meadows in areas which were once dominated by seagrass. This can be seen as a "passive" form of seagrass restoration, by improving conditions favorable for seagrass and thereby can insuring its return. A more active approach is by physically transplanting seagrasses via different methods (e.g. seed transplants or shoot transplants), in order to restore seagrass habitat. This is showcased in the later two case studies.

The four different locations were the restoration activities took place, showed seagrass declines for similar reasons. The most important factor influencing seagrass decline is eutrophication, as it had detrimental effects on total seagrass cover. In all case studies, the increase of nutrients caused a specific change in species composition. Seagrass dominated communities shifted toward macroalgae and/or phytoplankton dominated communities as high nutrient levels in the water column and sediment persisted.

In the Orbetello lagoon, situated of the west coast of Italy in Tyrrhenian Sea, water treatment plants had reduced DIN and SRP values by 85% and 79%, respectively. Although values in the water treatment plants were very high compared to values in the lagoon, DIN values had decreased and ranged between 12.0 and 85.1 μM (1999-2000). As for SRP values, they ranged between 0.1 and 0.9 μM, and averaged at 0.38±0.24. In Tampa Bay, Florida USA, regulations against nitrogen loading had resulted in the decline of total amount of nitrogen entering the Bay. By 2003 annual nitrogen entering the Bay area was approximately 5000 metric tons/yr. As a result, chlorophyll-*a* concentrations decreased in all sections of the Bay area. Accordingly, Secchi disc depth increased, reaching 2.5 meters in certain parts of the Bay. This means that more light can penetrate further into the water column, creating conditions favoring seagrass communities. In the Chesapeake Bay, eastern coast of the USA, conditions are also less favorable. Chlorophyll-*a* concentrations ranged between 6 and 11 μM, DIN between 9 an 23 μM and DIP between 0.4 and 1,7 μM. In the Wadden Sea, by the late 1990's, DIN valued 38 μM, DIP 0.7 μM and chlorophyll-*a* 11 mg m^{-3}.

As a result of habitat restoration activities, seagrass covered 50% of the Orbetello lagoon by the end of 2000. In a timeframe of 7 years (1993-2000), 10,000 ha had seagrass had been recovered. The Tampa Bay area had recovered approximately 2,000 ha of seagrass, 9,000 ha cover in 1982 increased to 11,000 ha of cover by 2004. Active restoration efforts by transplanting seagrass in Chesapeake Bay and the Wadden Sea were far less successful. In the former, a total of 625.5 ha were restored; of which 105.5 ha was the result of surviving seagrass transplants and the remaining 520 ha was through the natural expansion of seagrass transplants in a period of 10 years. In the Wadden Sea almost all transplants failed, only one site still survives to date, and encompasses approximately 200 ha.

By comparing the total seagrass coverage restored it is easy to see that habitat restoration is preferable to seagrass transplantation. Habitat restoration, in the case of Orbetello lagoon, showed 50 times more seagrass recovered in ha, and in Tampa Bay, seagrass recovered was at most 10 times more. But when comparing nutrient concentrations at the Chesapeake Bay to that of the Tampa Bay area, nutrient concentrations do not differ all that much. Then why was the total seagrass cover restored in the Tampa Bay so much more than in the Chesapeake Bay Area?

The biggest difference between these two locations is that in the Tampa Bay area, seagrass never completely disappeared. The presence of seagrass plays an important role in seagrass

restoration and is in my opinion underrated. Van der Heide *et al*. (2007) explains in one of his articles the self facilitating properties of seagrass habitats. Seagrass are notorious habitat engineers, as a meadow grows large enough they can; (1) maintain oxygen balance, (2) buffer perturbations, (2) decrease wave action and current velocity and hereby decrease suspended particles. As a result turbidity decreases in the water column, and further enhances conditions favorable for seagrass, as more light is able to penetrate. Van der Heide et al. (2007) illustrated by using a deferential equation that a stable equilibrium can be achieved when a seagrass meadow has at least 3,400 shoots per m^2. The higher the density the more effective a seagrass meadow becomes at decreasing current velocity (van der Heide et al., 2007). As can be seen in Fig. 6, once seagrass shoots were transplanted, "natural" recovery took over, probably caused by the natural facilitating properties of seagrass. If these results hold true on a global scale, then it is not strange why most of the small scale transplantations were a failure.

Furthermore, if the full extent of positive feedbacks in seagrass communities is to be believed, then it becomes all the more difficult to restore a collapsed seagrass ecosystem. Viaroli et al, (2008) hypothesize that this collapse can be seen as a transition between two alternative stable states. On one end, an oligotrophic pristine seagrass ecosystem, and on the other end a degraded eutrophic state, dominated by opportunistic seaweeds and phytoplankton (e.g. cyanobacteria). Once an ecosystem has transitioned into a seaweed dominated habitat, restoration efforts focused on creating conditions favorable for seagrass might not be enough, as hysteresis is likely to occur. Even if conditions that initially modified seagrass habitats are reversed, it might not be sufficient to result in a reciprocal shift. Hence, abusing the positive feedbacks in restoration attempts of seagrass communities becomes a given.

The only location were seagrass transplantations did not fail in the Wadden Sea, were areas which were protected from strong prevailing water currents and wave action. This leads me to presume that the current conditions in the exposed areas in the Wadden Sea, even if light penetration is sufficient and nutrient levels have been reduced, are still not favorable for the proliferation of seagrass meadows. The success of the case studies which encompassed habitat restoration can also be attributed to low velocities of wave currents, although no specific data is given. Three of the four case studies took place in either a lagoon or bay, whereas Wadden Sea is semi exposed, as the barrier islands partially decrease water currents. This leads to further speculations that all factors influencing turbidity in the enclosed lagoons and bay areas are mainly caused by the eutrophic conditions. The Wadden Sea borders with the North Sea were extensive trawling still takes place and sand is still mined for land reclamation purposes. Therefore, it becomes an even more encumbering labor to decrease turbidity levels in the Wadden Sea. It can be concluded that the main factor influencing the success of restoration projects is to what extend a seagrass meadow is exposed to currents and wave action. Even if moderate levels of eutrophication persist, restoration can still be a success. When comparing the nutrients concentration between the Orbetello lagoon and the Wadden Sea, it is shown that in the Orbetello lagoon restoration was a success, whereas in the Wadden Sea it was not. This confirms the conclusion stated above. Although it must be noted that in the Orbetello lagoon macroalgae was actively removed, hereby insuring that seagrass could grow extensive meadows.

The two different means of restoring seagrass habitat do not exclude one another. Nothing is further from the truth, creating abiotic and biotic conditions favorable for seagrass, while at the same time actively transplanting seagrass seems an excellent way of restoring lost habitat. Nonetheless, surprisingly many seagrass transplantations have taken place in eutrophic conditions. This seems to me like "mopping the floor with the tab running". In order for seagrass transplantations to truly help restore seagrass habitat, conditions must be altered so that they can thrive. Even if this is not enough, the transplants can facilitate the positive feedbacks needed, in order to ensure the shift from eutrophic and turbid conditions to an oligotrophic state with low turbidity levels. We can use the habitat engineering qualities of seagrass communities in order to achieve restoration goals.

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