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Seagrass restoration in Gulf of Mannar, Tamil Nadu, Southeast India: a viable management tool

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Abstract Seagrass beds are important marine ecosystems that provide significant ecological services. The global decline of seagrass beds is becoming severe due to the increasing pressure of human-induced factors and changing climatic conditions. Restoration of seagrasses is an evolving science that started in 1939. In this study, we report a remarkably successful restoration activity carried out in the Gulf of Mannar (GoM), Southeast India. This is the first wide-scale effort in Indian waters. After the initial experimentation, manual transplantation of seagrass sprigs was carried out near Vaan and Koswari islands in GoM. Transplantation was performed with PVC quadrats and jute twines in areas of 800 m^2 in both the islands during February to May 2014. An increase from 16.4 ± 0.3 to 32.3 ± 0.6% in Vaan and from 15.1 ± 0.2 to 35.1 ± 0.9% in Koswari was observed in seagrass percentage cover during the period from June 2014 to May 2016. Area cover, shoot density, macrofaunal density and fish density increased at the restoration sites after the transplantation. Bottom trawling was found to be the most serious threat to the seagrass beds in these islands. This method of transplantation can be replicated in other areas of degraded seagrass in India to carry out wide-scale restoration of seagrasses.

Keywords Seagrasses · Restoration · Gulf of Mannar · Transplantation · Sprigs

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Introduction

Seagrass beds are immensely productive marine ecosystems that supply food and often shelter a large population of marine fauna. Seagrasses are marine flowering plants well adapted to live submerged in salt water, endowed as they are with strong anchoring mechanisms and unique reproductive systems (Ehringer 2000). The services that seagrasses render include the provision of nutrition and acting as habitat and as nursery grounds to several ecologically and economically important marine organisms (Beck et al. 2001; Heck and Valentine 2006; Barbier et al. 2011; Liu et al. 2013). Despite their inherent worth, seagrass beds have been victims of negligence and targets of harmful practices globally. Various natural and human-induced factors like coastal development, pollution, destructive fishing practices and changing climate conditions have worked together and brought about the present plight (Duarte et al. 2004). It has been reported that about 29% of global seagrass cover has already been lost since 1879 (Waycott et al. 2009). Increasing instances of seagrass degradation have prompted researchers to seriously think about the restoration of seagrass beds.

Measures for the restoration of seagrasses were first taken in Europe in 1939 (Reigersman et al. 1939). Since then, various techniques have been attempted for the restoration and recovery of seagrasses, with varying success rates (Fonseca 1992; Gordon 1996; Seddon 2004; Treat and Lewis 2006). As noted above, restoration of seagrasses is an evolving science, and so, there have been many failed projects and there have been many successful ones too (Paling et al. 2009). However, research work on seagrass restoration in India is still in a primitive stage as very few experimental activities have been undertaken (Bensam and Udhayashankar 1990; Thangaradjou 2000; Edward et al. 2008; Thangaradjou and Kannan 2008). Before the execution of any project for seagrass restoration, the following points are to be taken into consideration: planning, formulating a policy, defining management guidelines, prescribing apt planting methods, selecting suitable sites, developing methodology appropriate to site conditions, improving seagrass spreading and coverage rates, accounting for self-facilitative properties, minimising donor bed damage, means of reducing costs, saving labour and time and preventing bioturbation (Gordon 1996; Fonseca et al. 1998; Paling et al. 2009).

Over the years, there has been a significant improvement in the techniques related to seagrass restoration (Paling et al. 2009). Some such improvements are enhanced knowledge in the selection of suitable sites, application of exposure indices, development of erosion sensors, employment of sprig planters, boat-based hydraulic extraction of large sods, use of submerged machinery, etc. They have yielded increased success rates in seagrass restoration projects (Fonseca et al. 1998; Kelly et al. 2001; Nakase and Shimaya 2001; Suzuki 2002; Fonseca et al. 2002; Paling et al. 2003; Lewis et al. 2006; Orth et al. 2006; Paling et al. 2009). Site-specific manual methods have also been proved to be comparatively efficient (Davis et al. 2006; Montin and Dennis 2006; Orth et al. 2006; Paling et al. 2009).

In India, seagrass beds occur in the Gulf of Mannar (GoM) (Fig. 1) and Palk Bay along the coasts of Tamil Nadu, Lakshadweep Island in the Arabian Sea and Andaman and Nicobar islands in the Bay of Bengal (Jagtap et al. 2003). Significantly large beds of seagrasses have been reported in GoM (Jagtap 1991; Parthasarthy et al. 1991; Thangaradjou and Kannan 2010; Mathews et al. 2010). In GoM, dense seagrass beds occur between the mainland and the islands, while patches of seagrasses occur beyond the islands. Altogether, 13 species of seagrasses have been recorded in GoM. The species *Thalassia hemprichii*, *Syringodium isoetifolium* and *Cymodocea serrulata* have been found to dominate the seagrass beds in GoM (Mathews et al. 2010). Seagrass beds in GoM support a wide array of marine fauna including fishes, sea turtles, sea horses, sea cucumbers, sea urchins, star fishes, gastropods, bivalves, ascidians, sponges and crustaceans as well as the endangered marine mammal *Dugong dugon* (Mathews et al. 2010).

Seagrasses in GoM are threatened by various natural and human-induced factors, especially by certain detrimental fishing practices like bottom trawling (Mathews et al. 2010; Raj et al. 2017). In GoM, the three types of bottom trawling that happen are mechanised trawling by the big trawlers, push net operation in which bottom trawling is done using wind sails and shore seine operation in which bottom trawling is done manually along the shore (Asha et al. 2015; Raj et al. 2017). Whether mechanised or manual, bottom trawling is very dangerous to the bottom-dwelling seagrass beds as these bottom trawls bury the benthic organisms and cause physical and biological damages which are not only extensive but are also irreversible (Kaiser and De Groot 2000). Apart from destructive fishing practices, there are other factors such as pollution, coastal development, elevated sea surface temperature, sea level rise and sedimentation that also threaten the very survival of seagrasses. Therefore, effective management actions and proactive restoration measures that benefit seagrasses in GoM assume critical importance to sustain the communities that relay on the ecological functions and resources provided by seagrasses. The present study consolidates and summarises the seagrass restoration initiatives undertaken in GoM by the Suganthi Devadason Marine Research Institute (SDMRI).

Methodology

Study site

The Gulf of Mannar in Southeast India encompasses 21 uninhabited islands situated off the coast of Tamil Nadu stretching from Rameswaram to Tuticorin. All the 21 islands and the shallow waters around them come under the jurisdiction of Gulf of Mannar Marine National Park

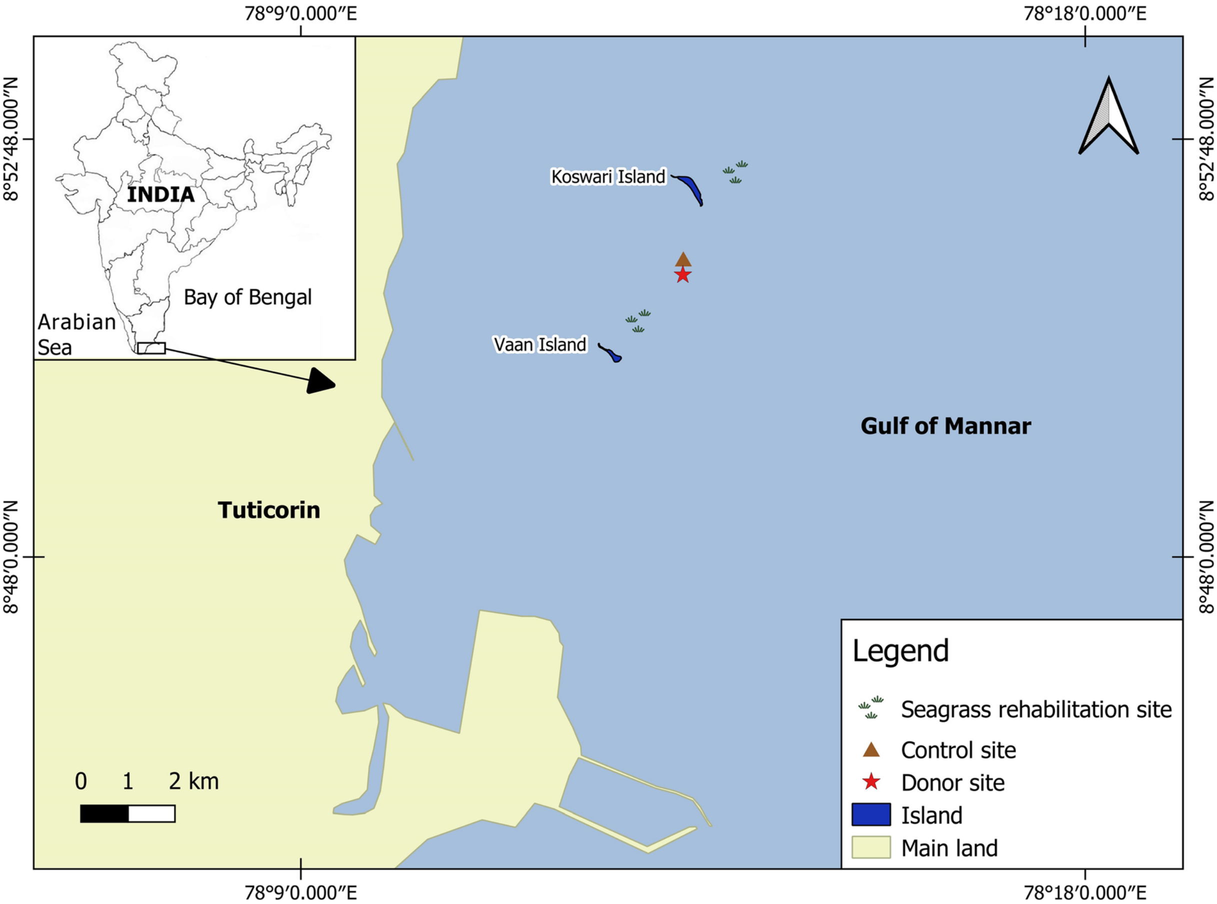

Fig. 1 Map showing the restoration study area

(GoMMNP). Starting from Tuticorin in the south, Vaan and Koswari are the first and the second islands of the chain that continue in a northeasterly direction towards Rameswaram (Fig. 1). The shortest distance between the mainland and Vaan Island is 5 km, while it is 6 km in the case of Koswari Island. The distance between the islands is 3.7 km. There is a reasonably extensive seagrass cover around these islands; in terms of percentage cover, Vaan Island has 31% and Koswari 34.9% (Mathews et al. 2010). Several human-induced and natural factors are threatening the survival of the seagrasses in GoM (Mathews et al. 2010). For example, Thangaradjou et al. (2008) reported a significant reduction of seagrass cover in Pamban area where a loss of 56.08% happened in 4 years. This unprecedented loss of seagrass has prompted the need of restoration as a management tool. Seagrass restoration activities were carried out in the bare sandy areas near these islands during the period between February and May in 2014.

Seagrass transplantation

Different techniques have been employed for seagrass restoration in the different parts of the world with different success rates (see Fonseca et al. 1998). In the plug method, seagrasses with attached sediment are harvested using core tubes and transported with the tube to the restoration site. At the planting site, a hole is made to accommodate the planting plug. In the staple method, seagrasses are dug up using shovels and the attached sediment is removed from the roots and rhizomes. Seagrasses are then attached to staples by inserting the root-rhizome portion, and the staples are, in turn, inserted into the sediment so that the roots and rhizomes are buried into the sediment (Christensen et al. 2004). Manual transplantation of sprigs was found to be the best choice for seagrass restoration in GoM. Initially, experimentation was performed to ascertain the most feasible technique in GoM, including plug, staple and manual transplantation of sprigs (Edward et al. 2008). Of them, the sprig method was found to be the best one, with an initial survival rate of 87% and 89% for *Cymodocea serrulata* and *Syringodium isoetifolium*, respectively. During the study period, the percentage cover increased from 30 to 73.5% and from 30 to 68.3% for *Cymodocea serrulata* and *Syringodium isoetifolium*, respectively (Edward et al. 2008). In the sprig method (Perrow and Anthony 2002), mature seagrass sprigs are collected manually in a mesh bag by scuba divers. The sprigs were thoroughly washed in seawater to free them from sediment, and they were then transferred to large containers filled with seawater. One apical shoot with intact roots was attached at a regular interval to a biodegradable jute twine, and the twine was tied to a 1-m^2 PVC quadrat (Fig. 2a, b). Six rows of such jute twines were tied to each quadrat such that each row had 20 shoots and 120 shoots in total per quadrat. Holes were predrilled into the quadrats to allow the seawater to enter and to make them negatively buoyant (Fig. 2c). Then, these quadrats with sprigs were immediately taken underwater and fixed at the restoration site (Fig. 2d). Hook-shaped iron rods, 30 cm in length, were used to anchor the frames to the sediment (Fig. 2e). The quadrats and the jute strings keep the seagrass intact and in contact with and firmly established in the bottom, thus minimising disturbance from waves, tides and currents. Seagrass restoration quadrats were left in place until the shoots were firmly rooted to the sediments. It was ensured that all the shoots tied to jute strings maintained contact with the seafloor.

With a view to reduce the stress to the donor site and to allow recovery there, sprigs not more than 3–5% were collected from the nearby dense seagrass meadows between Vaan and Koswari islands. Three abundant seagrass species, *C. serrulata*, *T. hemprichii* and *S. isoetifolium* (Mathews et al. 2010), were used for the restoration. In total, 400 quadrats with seagrass shoots were deployed at Vaan Island at a depth of 3 m, whereas a similar number of quadrats were deployed at Koswari Island at a depth of 6 m which covered a total area of 800 m^2 at both islands together. The donor site was dense and healthy enough to allow the collection of as many sprigs as necessary for fixing onto the 800 quadrats. The recovery of the donor site was achieved within 6 months of collection. Quadrats were removed from the restoration sites 3 months after the restoration when shoots got established themselves. Ten random quadrats were left at each restoration area to tag the site for future reference. A total of 25 quadrats were found to have been dragged to the shore by the shore seine operators during the restoration activities. Using a standard protocol, sediment texture was characterised as sand, silt and clay (Ingram 1970).

Monitoring

Data collection on survival and shoot density was started after the initial loss of plants by June 2014. Both the restoration sites in Vaan and Koswari islands and the control site were monitored for 2 years from June 2014 to May 2016. The control site is a natural seagrass bed with a seagrass cover of 45.7% as measured per methods detailed in English et al. (1997). Altogether, 8 permanent transects (100 m) were laid on the restoration sites perpendicular to the island shore. Along each transect, 10 quadrats (50 cm × 50 cm) were laid at a distance of 10 m for regular monitoring. The percentage cover of seagrasses and the percentage cover of each species were assessed every month (Saito and Atobe 1970). Individual shoots were counted underwater within the quadrats, and shoot density was calculated as the number of shoots per square meter. Extension of restored seagrass site was delineated during May 2016 to identify the increase in seagrass area cover. Densities of the macrofaunal categories such as echinoderms, molluscs, ascidians, sponges and sea anemones were also estimated and calculated. Five 1 m × 1 m quadrats were laid at each transect lines with a distance of 20 m, and the value for each transect was taken as the number per 5 m^2. Density and diversity at the restoration sites were also assessed by applying the belt transect method (English et al. 1997). Fish assessments were done before 9 a.m. to get better visibility. The length of each belt transect was 20 m and the width 1 m (0.5 m each side); the transects were separated by an interval of minimum 20 m. The length and width of transects reduced because of comparatively poor visibility. Three transects were laid during each assessment, and the values were calculated as the number per 60 m^2. The actual number of each fish species seen within the transect strip was counted and recorded onto the data sheets. A monthly analysis using standard protocols was carried out for the environmental parameters such as water temperature, salinity, pH, turbidity, total suspended solids (TSS), sedimentation and dissolved oxygen content. Trend analysis and two-way ANOVA were performed to identify the deviation of parameter values between sites and over time using

Fig. 2 Seagrass transplantation. **a** PVC quadrats. **b** Shoot with intact roots attached to biodegradable jute twine. **c** Six rows of such jute twines tied to each quadrat. **d** Quadrats with shoots fixed at the restoration site using iron clamps. **e** Hook-shaped iron clumps of 30 cm length

StatistiXL software. Paired *t* test was performed to analyse the initial and final observations on seagrass cover in the study sites. Correlation analysis was done to understand the relationship between seagrass cover and fish abundance in the restoration sites. The Tukey post hoc tests were performed to understand the deviation between sites in terms of shoot density, fish density and macrofaunal density. The Shannon-Weiner diversity index was applied on fish data.

Results

Vaan restoration site

In the restoration site at Vaan Island, a significant increase in the percentage cover and shoot density was observed after the transplantation. Seagrass percentage cover (Fig. 3) in this site gradually increased from June 2014 (16.4 ± 0.3%) to May 2016 (32.3 ± 0.59%). A sharp decline from 28.8 to 17.7% in the seagrass percentage cover was observed between May and August 2015. From a total extent of 400 m^2 during June 2014, the area cover of seagrasses in the restored site increased to 575 m^2 during May 2016. The overall shoot density increased from 124.9 ± 2.1 to 245.3 ± 4.7 m^{-2} during the study period of 2 years (Fig. 4). Shoot densities of all the three transplanted species increased: shoot density of *C. serrulata* increased from 56.3 ± 1.3 to 113.4 ± 3.3 m^{-2}, that of *T. hemprichii* increased from 45.2 ± 1.1 to 84.2 ± 2.4 m^{-2} and that of *S. isoetifolium* increased from 23.4 ± 1.1 to 47.7 ± 1.4 m^{-2} (Fig. 5). Statistically, the trend increased and the shoot density varied significantly between months ($F = 65.687$; $p = 0.00$; $p < 0.01$). By the end of the 24-month study period, the values of the shoot density of the restoration sites were identical to those of the shoot density of the control site.

In the restoration site of Vaan Island, the total density of macrofauna was 0.93 (5 m^{-2}) in June 2014 which increased during the study period and was 14.13 (5 m^{-2}) by May 2016. Molluscs exhibited the highest mean density with 13.69 ± 1.4 (5 m^{-2}) followed by echinoderms with 11.31 ± 1.3 (5 m^{-2}). A total of 1329 fishes were counted, and a total of 20 species belonging to 15 families were recorded during the study period. Fish abundance varied from 0 to 140.67 (60 m^{-2}) between June 2014 and May 2016. Among the fish species, *Sardinella* sp. was the dominant one, with a mean density of 9.53 ± 1.5 (60 m^{-2}) followed by *Terapon puta* with 7.59 ± 1.1 (60 m^{-2}).

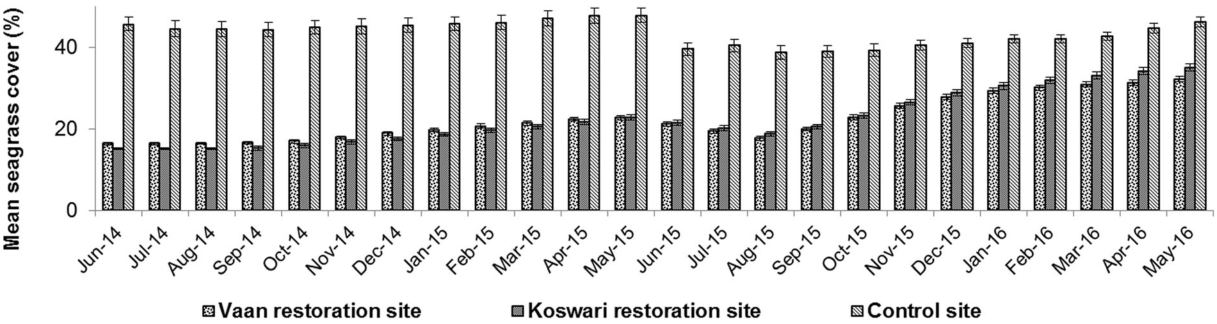

Fig. 3 Temporal variations in seagrass percentage cover in the restoration sites and control site (% ± SE)

Koswari restoration site

In the restoration site at Koswari Island, seagrass percentage cover increased gradually from 15.1 ± 0.2 to 35.1 ± 0.9% during the course of the study (Fig. 3). At Vaan Island, here too, a significant decrease from 22.8 to 18.8% in seagrass percentage cover was noticed during May to August 2015. The total area of seagrasses in the restoration site at Koswari Island increased to 650 m^2 during May 2016 from the initial area of 400 m^2. The overall shoot density increased from 120.1 ± 2.3 to 277.9 ± 6.7 m^{-2} from June 2014 to May 2016 (Fig. 4). *C. serrulata* increased from 56.7 ± 1.7 to 133.4 ± 3.3 m^{-2}, *T. hemprichii* increased from 39.9 ± 1 to 95.4 ± 3.5 m^{-2} and *S. isoetifolium* increased from 23.5 ± 1.4 to 49.1 ± 2.6 m^{-2} from June 2014 to May 2016 (Fig. 5). In the Vaan restoration site, the trend increased and the shoot density varied significantly between months in Koswari Island ($F = 47.257$; $p = 0.00$; $p < 0.01$). By the end of the 24-month study period, the figures of the shoot density of Koswari restoration site were almost similar to the estimates of the control site.

At Koswari Island, the overall density of macrofauna increased from 0.96 to 16.47 (5 m^{-2}) during the study period. Molluscs had the highest mean density of 14.57 ± 1.61 (5 m^{-2}) followed by echinoderms with 12.97 ± 1.36 (5 m^{-2}). A total of 1417 fishes were counted during the study period, from a total of 19 species belonging to 16 families. Fish abundance varied from 0 to 105.83 (60 m^{-2}) in 2 years. Among the observed fish species, *Sardinella* sp. was the dominant one with 9.14 ± 1.57 (60 m^{-2}) followed by *T. puta* with 8.73^2 ± 1 (60 m^{-2}).

Control site

At the control site, seagrass percentage cover was 45.7 ± 1.63% in June 2014 and it was 46.2 ± 1.14% in May 2016 with fluctuations in between (Fig. 3). The overall shoot density also increased slightly from 319.1 ± 25.3 m^{-2} in June 2014 to 340 ± 32.6 m^{-2} in May 2016 (Fig. 4). Shoot density of *C. serrulata* increased from 150.3 ± 6.9 to 167.2 ± 4.5 m^{-2}, *T. hemprichii* increased from 106 ± 4 to 118.28 ± 5.6 m^{-2} and *S. isoetifolium* increased from 62.8 ± 4.7 to 54.6 ± 2.4 m^{-2} from June 2014 to May 2016 (Fig. 5). At the control site, the overall density of macrofauna increased from 8.8 to 18.13 (5 m^{-2}) between June 2014 and May 2016. Here also, molluscs had the maximum density of 22.73 ± 0.83 (5 m^{-2})

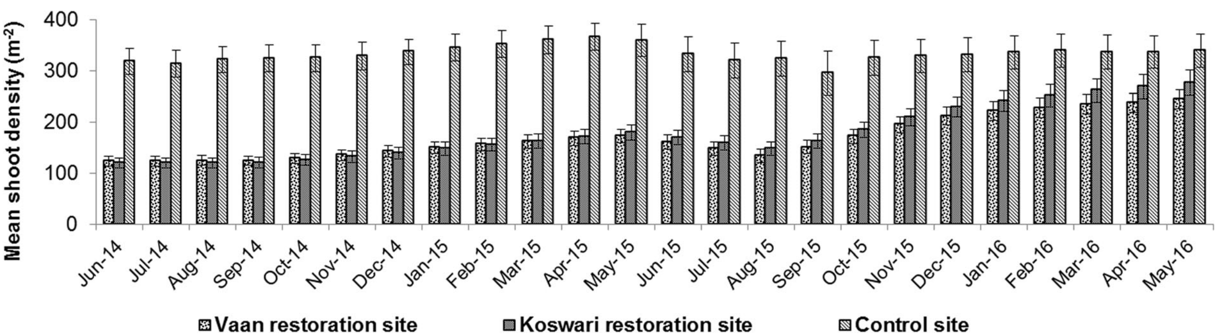

Fig. 4 Temporal variations in shoot density in the restoration sites and control site (m^{-2} ± SE)

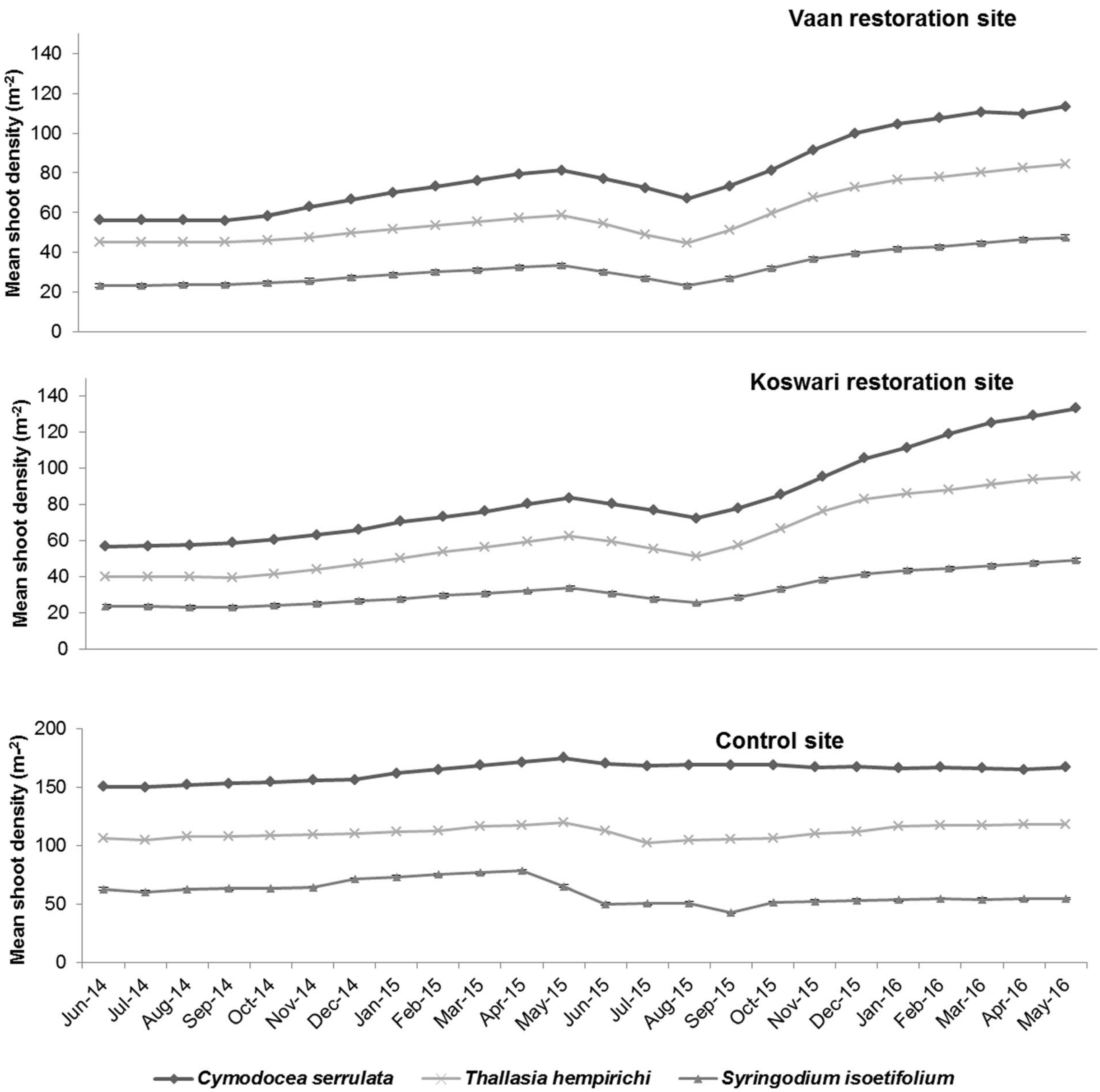

Fig. 5 Species-specific temporal variations of shoot density in the restoration sites and control site

followed by echinoderms with 13.66 ± 0.41 (5 m^{-2}) (Figs. 6 and 7). At the control site, a total of 3630 fishes were counted and a total of 21 species belonging to 18 families were recorded during the study period. Fish abundance varied from 120 to 219 (60 m^{-2}) during the study period. Among all the fishes observed, *T. puta* was the dominant species with 18.20 ± 1.2 (60 m^{-2}) followed by *Leiognathus splendens* with 13.03 ± 1.5 (60 m^{-2}) (Table 1, Figs. 8 and 9).

Water quality parameters

There was no substantial variation between the restoration sites and the control site in terms of environmental parameters. Water temperature ranged between 28.2 and 32.3 °C during the study period, and it peaked during the summer months. Salinity was recorded between 33 and 36 psu; pH ranged between 7.1 and 7.9; turbidity ranged between 4.1 and 5.6 NTU; total suspended solids ranged between 61 and 89 mg/l; sedimentation rate ranged between 45.13 and 60.23 mg cm^{-2} day^{-1}; dissolved oxygen ranged between 4.9 and 5.2 mg/l. Sediment texture at the control site was 18.3%, 49.5% and 32.2%, respectively, for sand, silt and clay; at the Vaan Island restoration site, it was 40.9, 40.3 and 18.8, respectively, whereas at the Koswari Island restoration site, it was 18.3%, 49.5% and 32.2%, respectively.

Statistical analyses

Two-way ANOVA showed significant deviations in shoot density between sites ($F = 849.679$; $p = 0.00$; $p < 0.01$) and between months ($F = 177.233$; $p = 0.00$; $p < 0.01$). Significant deviation in the fish abundance was found between restoration and control sites ($F = 79.529$; $p = 0.00$; $p < 0.01$), but the deviation between the two restoration sites was not statistically significant ($F = 0.140$; $p = 0.710$; $p < 0.05$). There was a significant deviation ($p < 0.01$) in fish abundance between months in all the three sites. Density of macrofauna deviated significantly between sites and between months in all the three sites ($F = 79.529$; $p = 0.00$; $p > 0.01$). Seagrass cover and fish abundance showed a positive correlation during the study period in Vaan ($r^2 = 0.823$; $p < 0.001$) and Koswari ($r^2 = 0.617$; $p < 0.001$) islands as an increase of seagrass cover increased the fish abundance. Results of paired t test revealed a significant deviation between initial and final seagrass covers during the study period in Vaan ($t = 24.19$; $p = 0.00$; $p < 0.001$) and Koswari ($t = -28$; $p = 0.00$; $p < 0.001$) restoration sites while it was non-significant for the control site ($t = -0344$; $p = 0.739$; $p > 0.05$). The Turkey post hoc HSD analysis indicated that deviations in seagrass shoot density, macrofaunal density and fish density between the control site and both the restoration sites are significant. The Shannon-Weiner diversity index (H') ranged from 0 to 2.628 at the Vaan restoration site and from 0 to 2.745 at the Koswari restoration site while it ranged between 2.324 and 2.841 at the control site. Evenness (J) ranged from 0 to 0.940 at the Vaan restoration site and from 0 to 0.956 at the Koswari restoration site while it ranged between 0.870 and 0.965 at the control site.

Discussion

A global decline in the extent of pristine seagrass meadows and in the dependent biodiversity has accelerated in the recent times. Threats to seagrass in GoM include bottom trawling, bottom laid gill nets, boat anchoring, invasion of exotic seaweed, pollution and developmental activities along the coasts (Mathews et al. 2010). Restoration of seagrasses using different techniques has been undertaken in many countries around the world (Paling et al. 2009). The present wide-scale restoration work in GoM is the first of its kind in Indian waters though experimental works have been done before (Thangaradjou and Kannan 2008). Thangaradjou and Kannan (2008) experimented with different restoration techniques in GoM with survival rates between 4.3 and 79.3% and concluded that plug and turf methods are comparatively better. Manual methods have been reported to be more successful (Davis et al. 2006; Montin and Dennis 2006; Orth et al. 2006; Paling et al. 2007). Seagrass restoration is generally expensive (Paling et al. 2009) as it is highly laborious and involves scuba diving. But when we consider the invaluable ecosystem services provided by the restored seagrass beds, the cost appears justified. Fonseca et al. (2001) estimated that an amount of 245,000 USD per acre is required for seagrass restoration. Bayraktarov et al. (2016) estimated that 399,532 USD per hectare is required for seagrass restoration in developed countries while there are no estimates for the restoration of seagrasses from developing countries. The method applied in this study did not require much money and was appropriate for wide-scale restoration projects in GoM. An 800-m^2 area restored in the present study encompasses about 20 ha which costs about 62,000 USD.

A low success rate of seagrass restoration may be attributed to poor site selection, high sedimentation, reduced light, strong waves and currents, animal foraging, etc. (Treat and Lewis 2006). Seagrass cover in absolute values, as well as in terms of percentage, and shoot density have all increased during the course of this study after the transplantation. Before the transplantation, the restoration sites were barren sandy areas and have now become verdant seagrass beds capable of providing ecosystem services similar to the nearby natural seagrass beds. Though statistically not reflected, the results after the end of the 24-month period show that the restoration sites have become more or less similar to the control sites. The population of the associated organisms such as fish and other macrofauna increased with the increase of seagrass cover. Diversity and density of fishes were similar between restoration and control sites after the study period. Koswari Island had a

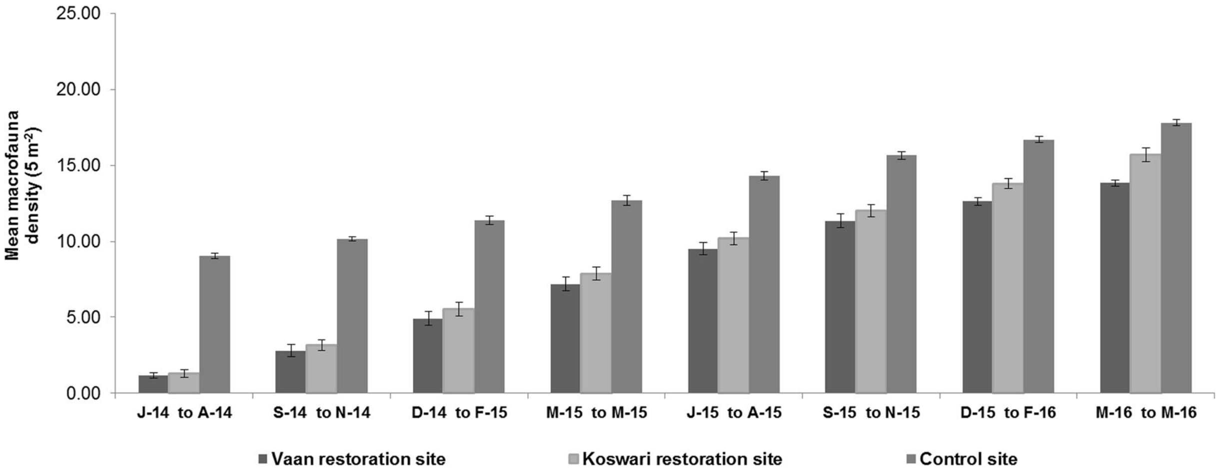

Fig. 6 Macrofaunal density in the restoration sites and control site (5 $m^{-2} \pm$ SE)

slightly higher area cover, percentage cover and shoot density than Vaan Island by the end of the monitoring period, and this may be attributed to the former's comparatively greater depth. Depth controls the water clarity, light availability, sedimentation and temperature, and hence, it is an important factor in seagrass restoration (Bologna and Sinnema 2006). Because of the comparatively greater depth of the restoration site at Koswari Island, it is better to protect it from increased temperature and turbidity. Sediment texture is another factor which affects the survival and growth of seagrasses (Thangaradjou and Kannan 2005; Bradley and Stolt 2006). It has been reported that silt/clay domination is favourable for the seagrass abundance in the Gulf of Mannar (Thangaradjou and Kannan 2005) and the combination of silt and clay was higher in Koswari Island than in Vaan Island. Sedimentation rate is also an important factor which affects the survival of the transplants (Turner 1995). Sedimentation rate in the restoration sites of Vaan and Koswari islands did not differ from the donor site, and hence, it was not an influencing factor as the range of sedimentation was between 45.13 and 60.23 mg cm^{-2} day^{-1} in all the three sites.

Though the islands of GoM are situated in the Marine National Park area where entry is prohibited, they are still prone to illegal shore seine activities (Raj et al. 2017). Because of this, some of the restoration quadrats were dragged ashore. During the monitoring period, a significant decrease in the seagrass cover was observed during the months between May and August 2016 which may be attributed to the wind-driven turbidity caused by southwest monsoon. Seagrasses of the restoration sites recovered from the damage and started to increase in percentage cover and shoot density. The control site is outside the dragging area of shore seine operation and hence was not affected by shore seines but is affected by push net operation.

Fig. 7 Groupwise macrofaunal density in the restoration sites and control site (5 $m^{-2} \pm$ SE)

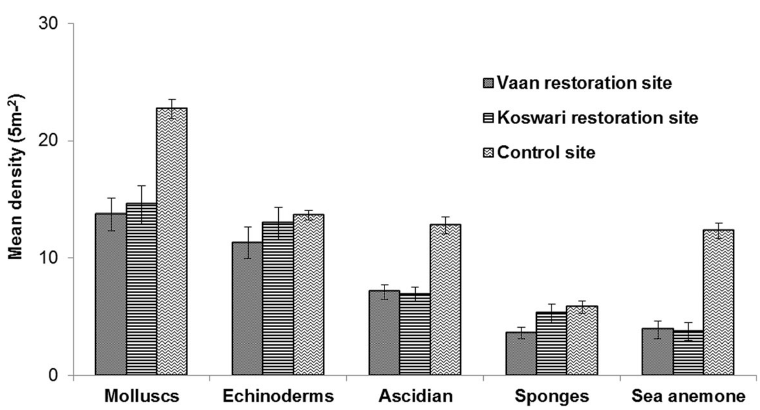

Table 1 Fish diversity and mean abundance in the restoration sites and control site

Species	Family	Vaan restoration site (60 m^{-2})	Koswari restoration site (60 m^{-2})	Control site (60 m^{-2})
Strongylura strongylura	Belonidae	2.19 ± 0.31	2.11 ± 0.34	6.66 ± 0.96
Alepes djedaba	Carangidae	7.01 ± 1.19	3.22 ± 0.74	4.64 ± 0.58
Scarus ghobban	Scaridae	0 ± 0	0 ± 0	2.88 ± 0.54
Epinephelus formosa	Serranidae	0 ± 0	0 ± 0	3.25 ± 0.38
Odonus niger	Balistidae	0 ± 0	0 ± 0	2.74 ± 0.41
Sardinella sp.	Clupeidae	9.53 ± 1.5	9.14 ± 1.57	5.38 ± 0.8
Parupeneus indicus	Mullidae	1.57 ± 0.17	2.51 ± 0.32	4.88 ± 0.59
Upeneus sulphureus	Mullidae	1 ± 0.2	1.78 ± 0.25	8.15 ± 1.07
Sphyraena barracuda	Sphyraenidae	1.49 ± 0.3	2.14 ± 0.34	9.94 ± 1.4
Lactoria cornuta	Ostraciidae	0.3 ± 0.05	1.14 ± 0.21	4.08 ± 0.61
Lutjanus sp.	Lutjanidae	5.07 ± 0.71	5.11 ± 0.75	12.08 ± 1.18
Mugil cephalus	Mugilidae	1.64 ± 0.23	2.37 ± 0.31	10.5 ± 1.37
Leiognathus splendens	Leiognathidae	1.74 ± 0.2	1.9 ± 0.33	13.03 ± 1.52
Terapon puta	Terapontidae	7.59 ± 1.15	8.73 ± 0.91	18.2 ± 1.22
Amphiprion sp.	Pomacentridae	0.8 ± 0.13	1.45 ± 0.25	2.58 ± 0.24
Syngnathoides biaculeatus	Syngnathidae	0.19 ± 0.07	1.29 ± 0.2	8.26 ± 1.16
Siganus canaliculatus	Siganidae	1.33 ± 0.34	2.4 ± 0.5	12.74 ± 1.12
Siganus javus	Siganidae	2.21 ± 0.4	2.85 ± 0.49	9.86 ± 0.85
Plotosus lineatus	Plotosidae	3.5 ± 0.61	2.78 ± 0.55	7.18 ± 1.82
Hippocampus sp.	Syngnathidae	0.1 ± 0.04	0.76 ± 0.23	1.83 ± 0.56
Stolephorus commersonnii	Engraulidae	6.48 ± 1.54	5.66 ± 1.23	13.92 ± 1.73
Sphyraena obtusata	Sphyraenidae	0.83 ± 0.19	0 ± 0	0 ± 0
Caranx para	Carangidae	0.85 ± 0.31	1.72 ± 0.33	0 ± 0
Diversity indices				
No. of species (S)		20	19	21
Diversity (H')		0–2.628	0–2.745	2.324–2.841
Evenness (J)		0–0.940	0–0.956	0.870–0.965

The complexity of habitat and the availability of food are the key factors that contribute to higher biodiversity in seagrass ecosystems (Crowder and Cooper 1982; Gilinsky 1984; Bostrom and Mattila 1999; Hughes et al. 2002). Seagrass ecosystems provide shelter to a wide range of macrobenthic communities (Bostrom and

Fig. 8 Mean fish abundance in the restoration sites and control site (60 m^{-2} ± SE)

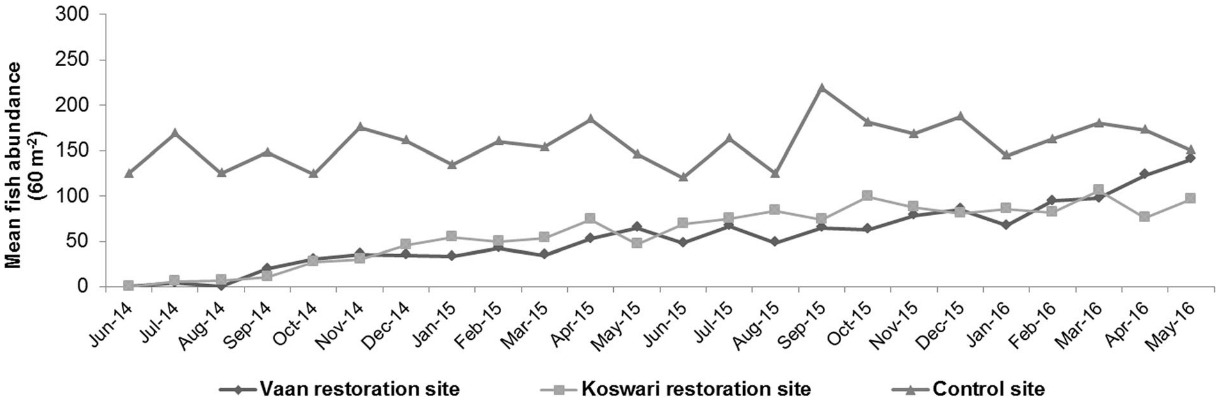

Fig. 9 Temporal variations in fish abundance in the restoration sites and control site (60 $m^{-2} \pm$ SE)

Mattila 1999; Moncreiff and Sullivan 2001). It is well known that seagrass beds offer a wide range of resources to fishes in both the temperate and the tropical systems (Bostrom et al. 2006). There are reports that an increase in seagrass cover triggers an increase in fish abundance (Hemminga and Duarte 2000). The present study shows that densities of fish and other benthic macrofauna started to increase along with the increase of seagrasses at both the restoration sites. There is a positive correlation between seagrass cover and fish abundance in the restoration sites. By the end of May 2016, the macrofaunal density and fish abundance of the restoration sites were almost similar to those of the control site, where natural seagrass bed occurs. Ecological functions of the restoration sites are expected to be similar to those of the natural sites (Evans and Short 2005). In 24 months, the restored sites have become functionally equivalent to the natural seagrass beds in terms of faunal usage. Fish assemblage at the restoration site of Vaan Island appeared to be the same as control site by the end of study period during May 2016. The trend is in the same direction at the restoration site of Koswari Island (Fig. 10).

Extreme anomalies in the parameters such as sea surface temperature, salinity and seawater pH have impacted marine ecosystems globally. Like mangroves and coral reefs, seagrasses too are sensitive indicators of water quality and ecological integrity of the ecosystem (Spurgeon 1992; English et al. 1997). Marine habitats associated with seagrasses, mangroves and coral reefs provide important ecosystem services with respect to local climate management (Camp et al. 2016). Poor water quality impedes the growth and affects the survival of transplanted seagrasses (Orth and Moore 1982; Moore et al. 1996, 1997; Treat and Lewis 2006). No anomalies were found in the analysed environmental parameters in the restoration sites, and they did not have any substantial impact on seagrass cover and shoot density. There was no significant deviation between the sites as they lie close to each other. The

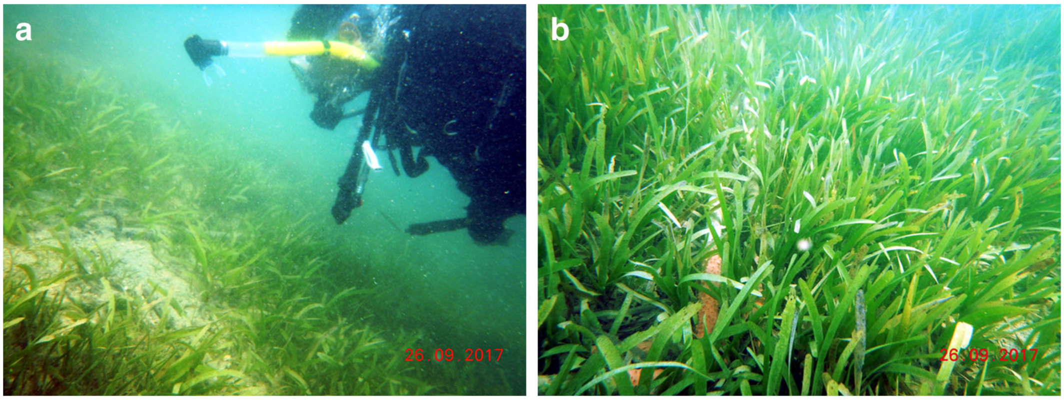

Fig. 10 **a**, **b** Restored seagrass beds after the monitoring period

environmental conditions of the restoration sites were conducive to the well-being of the transplanted seagrasses at both Vaan and Koswari islands, and the conditions were equivalent to those prevailing at the control site. However, sea surface temperature was very high during the summer of 2016, when it rose above 32 °C. Corals in GoM suffered a huge mortality due to these elevated temperature levels (Edward et al. 2018), but no obvious impact was observed on seagrass beds.

The thousands of fishermen living along the coast of GoM depend exclusively on seagrasses for their fishery resources. Persistent human-induced threats along with deteriorating climatic conditions have made survival difficult for seagrasses. Wide-scale recovery of seagrasses is needed to restore the lost seagrass areas in GoM. The manual transplantation of seagrass sprigs using quadrats and jute twines carried out in GoM can be termed successful. The present low-tech seagrass restoration method can be used to do extensive seagrass restoration in GoM and in the nearby Palk Bay. Environmental conditions in GoM are supportive for the transplantation. However, the harmful practices like bottom trawling which are fatal to seagrasses should be checked to conserve the existing seagrass beds and to allow their recovery.

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