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Eight years of community structure monitoring through recreational citizen science at the "SS Thistlegorm" wreck (Red Sea)

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Abstract

Large artificial coral reef communities, such as those thriving on sunken shipwrecks, tend to mirror those of nearby natural coral reefs and their long-term dynamics may help future reef resilience to environmental change. We examined the community structure of the world-renown "SS Thistlegorm" wreck in the northern Red Sea from 2007 through 2014, analyzing data collected during the recreational citizen science Red Sea monitoring project "Scuba Tourism for the Environment". Volunteer divers collected data on 6 different diving parameters which included the date of the dive, maximum depth, average depth, temperature, dive time, hour of dive, and gave an abundance estimation of sighted taxa from a list of 72 target taxa. Although yearly variations in community structure were significant, there was no clear temporal trend, and 71 of all 72 target taxa were sighted throughout the 8 years. The 5 main taxa driving variations among year clusters in taxa presence/absence (Soft Tree Coral—*Dendronephthya spp*., Giant Moray—*Gymnothorax javanicus*, Squirrel Fish—*Sargocentron spp*., Humpback Batfish—*Platax spp*., and Caranxes—Carangidae) and taxa abundance (Soft Tree Coral, Giant Moray, Red Sea Clownfish—*Amphiprion bicinctus*, Napoleon Wrasse—*Cheilinus undulatus*, and Caranxes) data were determined. The "SS Thistlegorm" provides a compelling example of how artificial coral reefs can sustain a well-established community structure similar to those of their natural counterparts.

Introduction

The biological communities of tropical and sub-tropical coral reefs support some of the highest biodiversity in the world [1] and provide a wide array of socio-economic services including coastal protection, water quality and chemical cycling, fisheries and materials markets (e.g. sponges, ornamental use, medicine, etc), and experiential benefits (e.g. education, recreation) [2]. Despite their assortment of values, coral species homogenization [3], species extinctions

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[4], and overall changes of coral communities are expected in future [5]. In fact, the Intergovernmental Panel on Climate Change projections [6] and current research indicate a multitude of threats to coral reefs including but not limited to ocean warming and acidification, overfishing, and rise in coastal human populations and impacts [7].

One of the largest coral reef systems of the world, with notably high endemism, lies within the Red Sea [8]. Due to its peculiar geo-evolutionary past resulting in large latitudinal gradients of sea surface temperature and salinity [8] the Red Sea provides an attractive location for studying the response of coral reefs to threats of their projected immediate future. The northern Red Sea is considered a coral refuge because, although it has experienced some of the highest sea surface temperature anomalies, there have been very few coral bleaching events compared to its southern portion [7, 9]. Notwithstanding the relative resistance of reefs in the northern Red Sea, there is still evidence of general coral colony size decline and species homogenization throughout [10], having the potential to induce changes in the biological communities they support. The importance of monitoring community variation lies not only in its fundamental value but also the economic income from fisheries [11] and especially the tourist industry, which provide an array of socio-economic opportunities for many coastal countries [12]. Unfortunately, there is general lack of information regarding Red Sea coral communities, which are relatively understudied when compared to the Great Barrier Reef or reef systems of the Caribbean Sea [13]. This can pose conservation challenges when attempting to monitor and analyze crucial changes in the biological communities of the Red Sea.

An innovative and upcoming approach to gathering large amounts of data in a time efficient way is the practice of citizen science. This approach is not so novel in terms of human history, but current technological advances of citizen science in the spreading and sharing of information and data [14, 15] have benefits and uses that are now an integral part of modern science [16]. Citizen science can be a critical asset in helping researchers, policy makers, and stakeholders overcome resource limitations (e.g., economical, temporal, geographical). Community-based management, and community-based monitoring are two of the main methodologies of the citizen science approach. The former involves direct participation of citizens and stakeholders in management decisions [17], while the latter implies the collaboration of citizens and stakeholders for data collection to monitor, track, and respond to areas of concern (e.g., environmental health) [18]. Scuba Tourism for the Environment, which was launched in 2007 (STE; www.steproject.org), provides a great example of involving large numbers of volunteers for community-based monitoring throughout much of the Red Sea [19]. Divers participating in the STE program collected ecological community data with the assistance of active dive centers and trained guides throughout Egypt. Recreational citizen science was applied throughout the project. This approach to citizen science allows participants to carry out their normal activities (volunteer behavior is unchanged throughout the survey), and to collect casually observed data. To analyze the reliability of data collected by the participants, data were compared with those collected by control divers (marine biologists of the Marine Science group at the University of Bologna) [20]. Consistency, or the similarity of data collected by individual volunteers during the same dive, was the lowest ranking parameter. The percentage recorded of the total number of taxa present (acquired from control diver data), or the Percent Identified, was the highest-ranking parameter [20]. This indicates that while divers can accurately identify most taxa there is a tendency to focus on certain organisms due to personal interest. However, with correct development to the needs of specific projects, data collected through citizen science can reliably support monitoring efforts [16, 17, 20]. The SS Thistlegorm, a world class dive site [21] and one of topmost visited wreck dives in the world [22], was one of the dive locations monitored in the STE project and is the focus of this study. The SS Thistlegorm was built in 1940 as a steal cargo steamship and was quickly taken over for the war efforts of the British military. On her final voyage from Glasgow (UK) to Alexandria (Egypt) she was met with two German Heinkel HE 111 bombers. The 128 m long and 17 m high cargo steam ship was split in two by munitions explosions and sunk within minutes [23]. Her ultimate resting ground is in the Straights of Gubal 30 meters below the sea surface [23]. In the 80 plus years since the discovery of the wreck by Jacques-Yves Cousteau in 1955, and the development of the Sharm el-Sheikh resort in the 1990s, the SS Thistlegorm has recruited up to 175,000 visiting divers per year from around the world [23]. Due to its high popularity and frequent visitors, there are rising concerns regarding the structural longevity of this historic site. Irresponsibly moored boats, inexperienced divers, and even air bubbles can cause irreparable damage to the structural integrity of underwater relics [24] as well as the biological community residing there [25]. Additionally, because the wreck is afforded no legal protection, it is prey to looting and souvenir plundering (www.thethistlegormproject.com). A dedicated team of divers, archeologists, and researchers developed the Thistlegorm Project (www.thethistlegormproject.com) to survey the site using 360 videography for the purposes of creating an accurate archeological 3D survey and to raise awareness for the protection of underwater cultural heritage sites.

Aside from the historical importance of the SS Thistlegorm wreck, the ship has recruited numerous marine life forms through the decades spent on the sandy bottom close to a natural coral reef and far from touristic resorts (70 km by boat from Sharm el Sheikh). We report here a first investigation on the temporal community trends of biological organisms at the SS Thistlegorm wreck. Scientific literature concerning long-term (more than 5 years) temporal trends and monitoring of community structure on wreck and/or artificial reefs is scarce [26]. There is extensive interest, however, in the role of artificial reefs, (including wrecks and scuttled ships) as a conservation tool, potentially serving as a compensatory habitat for anthropogenically damaged natural reefs [27–29], and as possible refugia assisting corals in the colonization of cooler waters in response to ocean warming [26]. Shipwrecks in particular have been shown to significantly increase biodiversity of both fish and benthic species on soft bottom environments [30, 31] along the coast and represent key microhabitats in the open ocean [32]. The material of the wreck, the bottom type it lies upon, as well as its structural complexity play key roles in determining marine community and have been shown to be fundamental in epibenthic colonization patterns [33].

The aim of this study was to characterize the community structure and its temporal trends of the SS Thistlegorm, over 8 years of monitoring through recreational citizen science.

Materials and methods

The study site

The SS Thistlegorm lies 27˚ 48' 30.59" N and 33˚ 55' 7.19" E at 30 m depth on sandy bottom substrate within the Straights of Gubal. The bow of the ship faces S40˚W, with tidal currents flowing from Northeast to Southwest at varying strength, from stern to bow [34]. Her final voyage entailed a resupply mission to the British Armed Forces and was destined for Alexandria (Egypt). The cargo onboard the SS Thistlegorm included Bedford trucks, BSA motorcycles, Wellington rubber boots, aircraft parts, rifles, land mines, ammunition, and other weaponry, as well as two LMS Stanier Class 8F steam locomotives among other goods [28], much of which is potentially hazardous to the marine environment.

Data collection and isolation

The Scuba Tourism for the Environment—Red Sea Biodiversity Monitoring Program (www.steproject.org),included the collection of large scale spatial-temporal biodiversity data within the Red Sea from 2007 to 2015 using a recreational citizen science approach where the behavior of the underwater tourists is unaltered during surveys [19]. Participant volunteers completed questionnaires that contained general demographic information (name, country of residence, address, diving certification, etc.), dive site and environment type (sandy bottom, wreck, blue, or coral reef), six diving parameters (date (PY), maximum depth (MD), depth where they spent most of their time (AD), temperature (T), dive time (RTD), and hour (FD)), and the sightings of 72 target taxa which were recorded on the questionnaires immediately following the dive and without the use of underwater marking materials during the dive [19]. For each of the taxa sighted, the divers also recorded an estimated Sighting Abundance according to 3 classes: 1, 2, and 3 (rare, frequent, and very frequent respectively) [19]. The classes were weighted to each taxon's individual expected occurrence [19]. The 72 faunal taxa were chosen because they are representative of the main ecosystem trophic levels within the Red Sea, they are common/abundant, and they are easily identifiable by recreational divers [19]. Data coming from the Thistlegorm wreck site were aggregated per year.

Preliminary treatment and analyses

Seven of the dives within the Thistlegorm dataset were viable for validation trials and were analyzed to measure the quality of volunteer collected data. The mean similarity index, accuracy, and correctness of abundance ratings (CAR) parameters were tested in accordance with the standard methodology for this citizen science project [20].

Thistlegorm data were then split into three data sets: 1) taxa abundance; 2) taxa presence/absence (p/a), obtained through an overall transformation of taxa abundance; and 3) diving parameters. The sighting frequency (*SF%*) and Relative Abundance (*RA*) for each taxon were calculated as such [19]:

$$SF\% = \left(\frac{number\ of\ sightings\ per\ taxon}{number\ of\ dives}\right) x\ 100$$

$$RA = (SF\%)(Average\ Sighting\ Abundance)$$

The analyses below were completed using PRIMER-E version 6 with PERMANOVA+ version 1 software (PRIMER-E, Ltd., Ivybridge, UK).

For a general picture of the trophic community composition and to identify outliers within each year, 2-dimensional plots were created using non-metric multidimensional scaling (MDS) from the Bray-Curtis resemblance matrices of both taxa data sets [35, 36]. A BVSTEP "Best" test was performed on both taxa data sets to find the subset of taxa that best represented the community of each year [37]. These subsets of taxa were later used for aggregation of the data in the temporal analysis.

A distance matrix of normalized diving parameters was created using Euclidean distances. Both taxa distance matrices were tested for a relationship with the distance matrices of the diving parameters using the Relate test to determine whether the variation between years was affected only by the change in time or also by diving parameters [38]. Additional distance based linear models [39] were ran for each year with a significant relationship between the taxa data and diving parameters, providing the individual parameters that significantly affected the variation of community structure in each year.

To test the differences in diving parameters over the years, a permutational multivariate analysis of variance (PERMANOVA, [39]) was designed with the fixed factor of "year" and run on the Euclidean resemblance matrix of the normalized aggregate diving parameters. Both main and pairwise tests were run. Additional PERMANOVAs were created for individual diving parameters to test whether each different diving parameter was significantly different among the years.

Temporal analysis

Two reduced datasets containing only the BVSTEP best subsets of taxa representing th community structure from any individual year were obtained, and resemblance matrices were created from Bray-Curtis distances of both the abundance taxa data and the overall-transformed p/a taxa data [35]. To test significant difference among years, PERMANOVAs [39] were run with the fixed factor "year". For a sharper visualization of temporal variation in community structure, distances between centroids were obtained from the resemblance matrices using "Year" as the grouping factor and visualized with 2D and 3D MDS plots. 3D plots were used to better interpret the direction of vectors in 2D plots and were not included as figures in the current manuscript. Any similar groupings of the centroidal "years" were found with a hierarchical cluster analysis [40]. The average sample taxa abundance and presence/absence from each year were calculated, and a BVSTEP Best analysis was run from the Bray-Curtis resemblance matrix to find the subset of taxa that best explained the trohic community structure variance within centroid data sets [37].

Ethics statement

Participants (or parents/guardians in case of minors) gave their informed, written consent by signing a declaration inserted in the questionnaires. STE project and its consent acquisition procedure have received the approval of Bioethics Committee of the University of Bologna (prot. 2.6).

Results

Data collection and isolation

Nine questionnaires from 2015 were removed from the analysis, as per lack of efficacy for less than 10 questionnaires per year (following 19). While Branchini and colleagues [19] examined the "coral reef" environment questionnaires, this study reports findings from an unpublished data set within the "wreck" environment type. Data from a total of 390 questionnaires between 2007 and 2014 were included in this analysis (Table 1). Questionnaire counts per year ranged from 12 to 84, and significantly decreased over time ($R^2 = 0.78$, $P < 0.05$). Over the 8 years, 71 (97%) of the 72 target taxa present on the STE questionnaire were sighted (all taxa excluding the Manta Ray).

Preliminary treatment and analysis

According to the RELATE test, years 2007 ($\rho_s = 0.162$, N = 6, $P = 0.007$), 2008 ($\rho_s = 0.157$, N = 2, $P = 0.003$), 2009 ($\rho_s = 0.111$, N = 45, $P = 0.046$), and 2012 ($\rho_s = 0.217$, $N = 9$, $P = 0.01$) showed a significant relationship between taxa abundance and dive parameters (Table 2; S1 Table). In all other years there was no significant relationship (Table 2; S1 Table). The DistLM of taxa abundance data in 2007 showed a significant relationship (P <0.05) with all 6 diving parameters: date (PY), maximum depth (MD), depth where they spent most of their time (AD),

Table 1. Yearly counts of completed questionnaires.

Year	2007	2008	2009	2010	2011	2012	2013	2014	Total
Number of Questionnaires	84	62	59	59	72	29	12	13	390

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Table 2. SS Thistlegorm dive parameters with significant ($P < 0.05$) sequential DistLM test to community structure by year for abundance and presence/absence data. PY as date (expressed as the percentage of year), MD as maximum depth, AD as depth at most time spent, T as temperature, RTD as dive duration, and FD as hour (expressed as percentage of day).

Year	Abundance	Presence/Absence
2007	PY, MD, AD, T, RTD, FD	PY, AD, T, RTD, FD
2008	PY, MD, AD, T, RTD	PY, MD, T, RTD
2009	PY, AD, FD	
2010		
2011		
2012	PY, T	PY, T, RTD
2013		
2014		

https://doi.org/10.1371/journal.pone.0282239.t002

temperature (T), dive duration (RTD), and hour (FD) (Table 2; S2 Table). In 2008 all the diving parameters except FD were significantly correlated ($P < 0.05$) to the taxa abundance (Table 2; S2 Table). PY, AD, and FD were significantly related ($P < 0.05$) to taxa abundance in 2009 (Table 2; S2 Table). In 2012 only PY and T diving parameters were significantly related to the community structure ($P < 0.05$; Table 2; S2 Table).

Regarding taxa presence/absence (p/a) data, the RELATE results indicated that in 2007 ($\rho_s = 0.18$, N = 1, $P = 0.002$), 2008 ($\rho_s = 0.136$, N = 13, $P = 0.014$), and 2012 ($\rho_s = 0.209$, N = 4, $P = 0.005$) there was a significant relationship between the community structure and dive parameters (Table 2, S1 Table). The DistLM in 2007 revealed that all dive parameters except MD had significant relationships to the taxa p/a data ($P < 0.05$; Table 2, S3 Table). In 2008 PY, MD, T, and RTD were significant ($P < 0.05$; Table 2, S3 Table), while only PY, T and RTD resulted significant in 2012 ($P < 0.05$).

Aggregated diving parameters showed significant differences among years (PERMANOVA, F = 9.25, df = 7, $P = 0.001$). PY, MD, AD, T and RTD all differed significantly among years ($P < 0.05$). FD was homogenous among years.

Of the 7 validation trial dives, the average mean similarity index score was 98.8%, the average mean accuracy score was 57.1% and ranged from 40.4 to 70.0%, and the mean average CAR was73.8%.

Temporal analysis

According to the PERMANOVA from both the taxa abundance and p/a data, the community structure was significantly different between each pair of years (F = 4.88, df = 7, $P = 0.001$ and F = 4.88, df = 7, $P = 0.001$ respectively). The variation in community structure of taxon abundance and p/a through years 2007–2014 was represented as an MDS (Figs 1A and 2A). Given the high stress value (> 0.3) of these MDS plots, further centroidal MDS plots of both taxa data sets were generated to have a clearer, even if more general, visualization highlighting the similar groupings of years within their respective distances resulting from the hierarchical cluster analysis (Figs 1B and 2B).

The BVSTEP subset of taxa that explained the distances between yearly centroids of community structure for the taxa abundance data were the Soft Tree Coral (*Dendronephthya spp*.,), Giant Moray (*Gymnothorax javanicus*), Red Sea Clownfish (*Amphiprion bicinctus*), Napoleon Wrasse (*Cheilinus undulates*), and Caranxes (Carangidae). The BVSTEP best subset of taxa explaining the distances between yearly centroids of community structure for the taxa p/a data were the Soft Tree Coral, Giant Moray, Squirrel Fish (*Sargocentron spp*.), Humpback Batfish

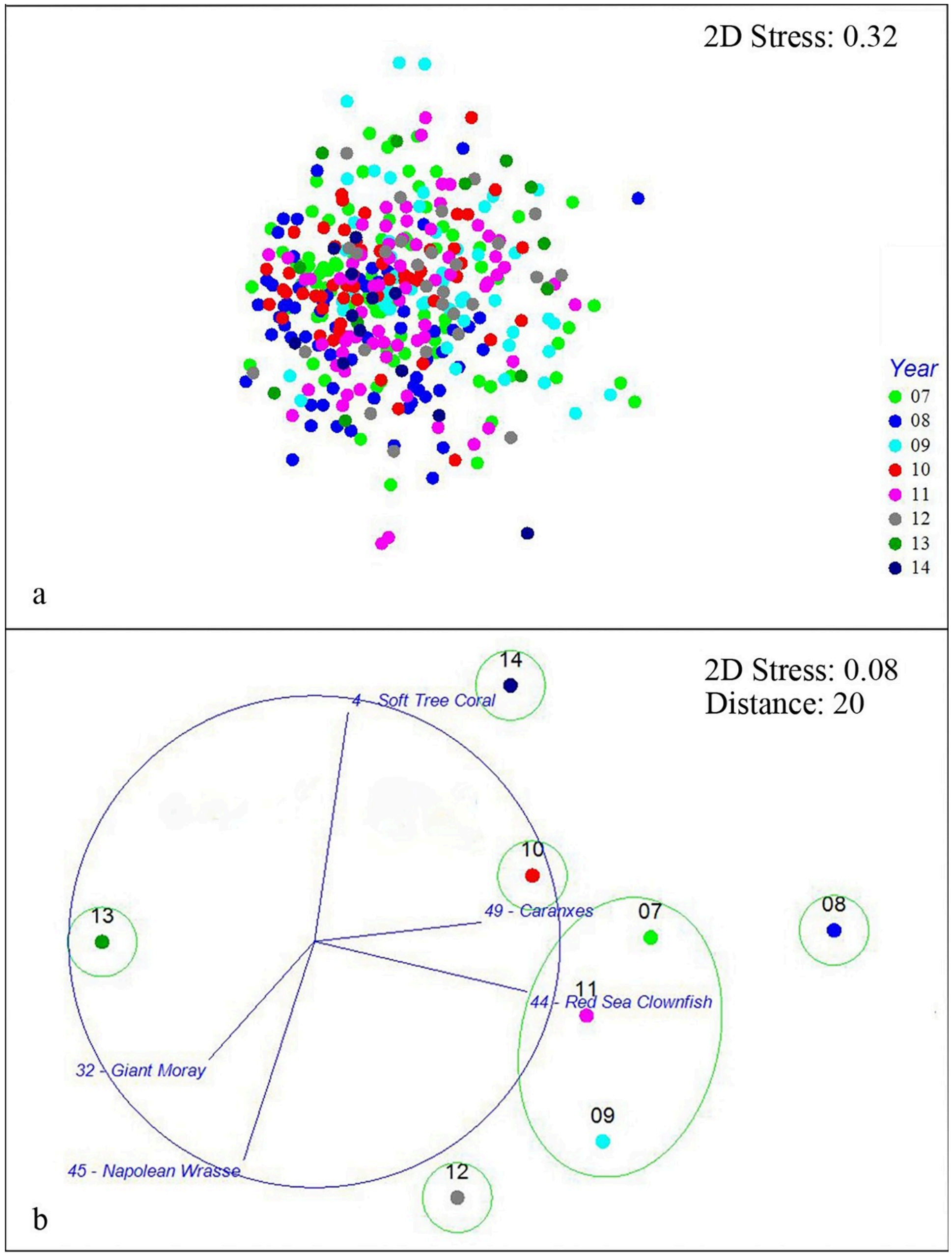

Fig 1. a) MDS plot of the SS Thistlegorm taxa abundance community structure over 8 years. Each point represents an individual questionnaire while the different shapes/colors indicate the different years. **b) Centroidal MDS of SS Thistlegorm taxa abundance community structure variation over 8 years with overlayed vectors of aggregated best taxa and the indicated trajectories from 2007–2014.** Each point represents the central location of the groups "Year". The green circles surround the groups resulting from the hierarchical cluster analysis.

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(*Platax batavianus*), and Caranxes. Tables 3 and 4 show the relative abundance and SF% of the best taxa through the 8 years.

Years 2007, 2009, and 2011 cluster around a general community structure in the taxa abundance data (Fig 1B). The Red Sea Clownfish (*Amphiprion bicinctus*) was overrepresented in 2008 (98 in 2008 *vs* 49–80 in 2007, 2009, and 2011; Table 3). The year 2010 was overrepresented in Soft Tree Coral (*Dendronephthya spp*.,) (98 in 2010 *vs* 41–77 in 2007, 2009, and 2011;

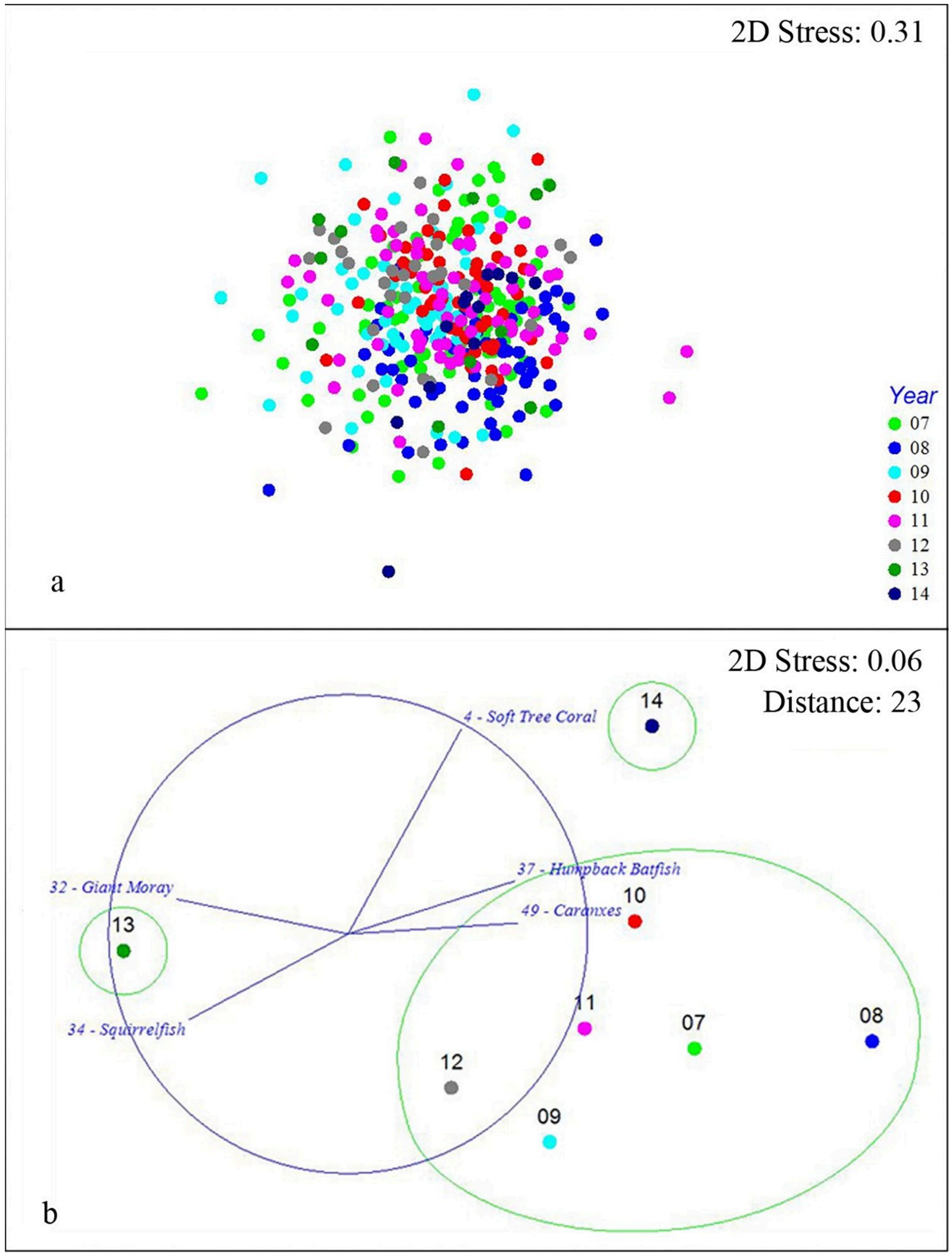

Fig 2. a) MDS plot of the SS Thistlegorm taxa presence/absence community structure over 8 years. Each point represents an individual questionnaire while the different shapes/colors indicate the different years. **b) Centroidal MDS of SS Thistlegorm taxa presence/absence community structure variation over 8 years with overlayed vectors of aggregated best taxa and the indicated trajectories from 2007–2014.** Each point represents the central location of the groups "Year". The green circles surround the groups resulting from the hierarchical cluster analysis.

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Table 3), and Giant Moray (*Gymnothorax javanicus*) (93 in 2010 *vs* 48–78 in 2007, 2009, and 2011; Table 3) and underrepresented in Red Sea Clownfish (44 in 2010 *vs* 49–80 in 2007,2009, and 2011; Table 3). In 2012 there was an overrepresentation of Giant Moray (121 in 2012 *vs* 48–78 in 2007, 2009 and 2011; Table 3) and Napoleon Wrasse (*Cheilinus undulatus*) (55 in 2012 *vs* 26–59 in 2007,2009, and 2011; Table 3) with an underrepresentation of Soft Tree Coral (31 in 2012 *vs* 41–77 in 2007, 2009 and 2011; Table 3) and Red Sea Clownfish (35 in 2012 *vs*

Table 3. Yearly relative abundance values of BVSTEP best species subset from centroid aggregated taxa abundance data.

Year	Soft Tree Coral	Giant Moray	Red Sea Clownfish	Napoleon Wrasse	Caranxes
2007	77	48	64	26	117
2008	69	26	98	27	94
2009	41	78	80	59	125
2010	98	93	44	37	154
2011	61	63	49	47	114
2012	31	121	35	55	93
2013	50	67	17	50	8
2014	154	62	31	0	115

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49–80 in 2007, 2009 and 2011; Table 3). In 2013 there was an underrepresentation of Caranxes (Carangidae) (8 in 2013 *vs* 114–125 in 2007, 2009, and 2011; Table 3) and Red Sea Clownfish (17 in 2013 *vs* 49–80 in 2007, 2009, and 2011; Table 3). 2014 was overrepresented by Soft Tree Coral (154 in 2014 *vs* 41–77 in 2007, 2009 and 2011; Table 3) and underrepresented by Red Sea Clownfish (31 in 2014 *vs* 49–80 in 2007, 2009 and 2011; Table 3) and Napoleon Wrasse (0 in 2014 *vs* 26–59 in 2007,2009, and 2011; Table 3).

The taxa p/a centroidal data (Fig 2B) displayed a general cluster in community structure, like the abundance data, but this general cluster was larger and represented by the years 2007 through 2012. With respect to this general cluster the year 2013 exhibited an underrepresentation in Humpback batfish (*Platax bativianus*) (33% in 2013 vs 56–86% in 2007–2012; Table 4) and Caranxes (8% in 2013 vs 52–75% in 2007–2012; Table 4). The following year showed a community similar to the general cluster but with an overrepresentation of Soft Tree Coral (92% in 2014 vs 28–54% in 2007–2012; Table 4) and an underrepresentation of Squirrel Fish (*Sargocentron spp*.) (8% in 2014 *vs* 16–52% in 2007–2012; Table 4).

Discussion

We have described here the first investigation of temporal community trends of an historical relic, and world class dive site within the Red Sea, The SS Thistlegorm. Notwithstanding these merits, The SS Thistlegorm can also be regarded as an artificial reef community, though "accidental" in nature. Since the 1960's, artificial reefs have been used for a variety of purposes including biodiversity conservation [41], fisheries management [42], and tourist locations [43]. One study in Eilat (Red Sea) designed and submerged multiple concrete structures to test whether artificial reefs could decrease diving pressures on the surrounding natural reef areas, concluding that small scale structures, knowledge of such structures within the local diving community, and education of divers could be sufficient [44]. The provision of migratory

Table 4. Yearly sighting frequencies of BVSTEP best species subset from centroid aggregated taxa p/a data.

Year	Soft Tree Coral	Giant Moray	Squirrelfish	Humpback Batfish	Caranxes
2007	46%	38%	33%	70%	55%
2008	53%	21%	16%	61%	53%
2009	34%	61%	24%	56%	71%
2010	54%	70%	42%	86%	75%
2011	44%	47%	38%	61%	65%
2012	28%	66%	52%	62%	52%
2013	33%	67%	50%	33%	8%
2014	92%	54%	8%	70%	69%

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networks for coral movement and resettlement within changing oceans [45] is another application for artificial reefs, though highly debated and rather unexplored. Organismal community recruitment and interchange is an evident and well observed occurrence when any new colonizable object, be it an island or a plastic bottle, appears near a source population. In accordance with island biogeography theory [46], the evolution of this community is much dependent on the size of the object in question, the time it is in existence, and its distance from the source. Sunken warships have been shown to serve as exceptionally good artificial substrates for coral reefs as their size and complexity offer a multitude of opportunities for microhabitats, and in cases where they extend down to 30–40 m, the cooler waters can provide respite to corals from warming oceans but still sufficient light for their symbiotic zooxanthellae [23]. Many shipwrecks in the North Sea have supported hard substrate communities throughout the Belgian Continental Shelf which has otherwise been transformed into a soft-bottom environment due to anthropogenic activity [47, 48]. Thanks to the steel structure of the SS Thistlegorm, recruitment of various tolerant sessile species typical of hard substrates was enabled in the sandy bottom environment within the Straights of Gubal and a well-established coral community can now be witnessed, but no information is available in the scientific literature.

Much of the work regarding artificial coral reefs examine differences between colonization patterns on various substratum material or recently scuttled ships (less than 3 years) and natural reefs, which are largely short-term studies [24–26, 49, 50]. The few studies that have investigated community structure change over the long term (more than 5 years) have primarily been snapshot comparisons between the virgin and mature communities [51, 52] or follow succession patterns of virgin artificial reefs [53].

The present study begins to fill this gap through an 8-year monitoring of a well-established wreck community (i.e. > 80 years since the sinking of the SS Thistlegorm in 1941). Despite the internal fluctuation among years (50% in 2013, 93% in 2007), all target taxa, apart from the Manta Ray, were sighted throughout the duration of this study. These findings indicate a well-developed, if variable in time, community structure at the Thistlegorm and are in line with the monitoring of natural reefs in the region where reef sites far from touristic facilities show higher biodiversity [54]. The Kingston, a 119-year-old shipwreck 10 km east of the Thistlegorm, mimics the surrounding natural reef communities due to its similarities in structural complexity [55]. Using a visual census technique and the recordings of all species seen, oil jetties in Eilat (northern Red Sea) were studied as artificial reef proxies and were shown to have even higher species richness in fish assemblages than neighboring natural reefs [50] due to increased structural complexity. Studies of artificial reefs and wrecks in other regions throughout the world have shown similarly promising results with respect to community development. In the Florida Keys, artificial reef fish communities mirror those of natural reefs in terms of fish numbers and species composition [56] and two thirds of genera from neighboring natural reefs can thrive on sunken shipwrecks within the Caribbean [35]. With the use of SCUBA diving video sampling, microbenthic species abundances on eight shipwrecks and one airplane of varying construction and bottom type were compared to those of sampling on nearby natural substrates in the Ligurian seabed (Mediterranean Sea) [27]. The findings of this study exhibited not only a strong correlation of wreck material and bottom type to the epibenthic community present, but also revealed that some species abundances were greater on the wrecks than those of nearby natural rocky bottom substrates [27]. Even in temperate seas, such as the North Sea along the Belgian Continental Shelf, shipwrecks are described as biodiversity hotspots of hard substrate communities in an otherwise sandy bottom environment while protecting nearby soft-bottom communities from fishing activities [47, 48]. The steel structured SS Thistlegorm lies upon a sandy bottom marine environment and while this study does not compare its taxa sightings to natural reefs nearby, it does provide key insights into the community dynamics of a well-established artificial reef.

The number of observations collected in this study through recreational citizen science methods decreased in time (Table 1). This is ascribable to the reduction in overall tourism due to political upheaval consuming Egypt during the revolution and the following transitional period of the "Arab Spring" [57, 58]. As tourists seek to vacation in more stable and safe countries, participation in citizen science is inherently affected. This temporal non-homogeneity is an evident and acknowledged limitation of citizen science, and thus it should not outweigh the benefits (e.g., increasing participant education and awareness, reduction of economic and time costs associated with environmental monitoring, and acceptable level of accuracy in data collection) of such a monitoring approach [19, 20, 54, 59].

Diving parameters were significantly correlated with the taxa data in some of the years. PY was the most frequently related dive parameter, appearing in all years with a significant relationship to diving parameters for both the taxa abundance and taxa p/a data sets (Table 2). The yearly average PY spans about half a year, ranging from early April in 2014 to late September in 2009, with little internal variation (S4 Table). Some of the variation in community structure over time in this study could thus be due to the differing seasonal environmental conditions in which questionnaires were collected. Total macroalgal biomass and community structure on coral reefs are strongly linked to seasonality and temperature variation in the Red Sea [60], where variation in benthic communities also affect the distribution of fish species [61]. Species richness, diversity, and evenness of reef fish assemblages vary significantly between summer and winter in the Red Sea [62], and these seasonal variations may be mirrored on wrecks and artificial reefs close to natural coral reefs.

Temperature, the next most linked dive parameter (Table 2) is generally influenced by both seasonality and depth. In this study, T is more likely a function of the seasonal variation because the range in average depth (AD) and Maximum Depth (MD) throughout the years is quite small (19–22 m and 25–28 m respectively) (S4 Table). In fact, coral species that have narrower depth ranges tend to occupy shallower depths, while species that can colonize deeper in the water column have wider depth ranges [63]. This may suggest that the small range of average depth in surveys at the Thistlegorm, even if statistically significant, is trivial in terms of its effect on the community structure variation given the greater depth at which the surveys were conducted. This is especially true when considering non-sessile benthic and/or free-swimming taxa. The yearly average dive duration ranged from 41 to 55 minutes (S4 Table) and presented a significant correlation in 2007 and 2008 for both data sets and in 2012 for the taxa p/a data (Table 2). Dive duration certainly plays a role in the number and amount of target taxa sighted. In validation trials of the STE project [20], volunteer scores of mean accuracies, mean consistency, and mean percent identified are all positively correlated with dive time.

Following the temporal sequence of centroids of taxa abundance (Fig 1B), no clear trend emerged. Years 2007, 2009, and 2011 cluster around what could be interpreted as a general (even if variable) community composition. In the p/a centroidal data (Fig 2B) the years 2007 through 2012 displayed a relatively larger general cluster in community structure when compared with the abundance data.

The taxa abundance data revealed more variance among years than taxa p/a data, and even if the general trend is quite similar, two of the five best species best explaining community structure variations among years are different in the analysis of the two data sets (the Red Sea Clownfish and Napoleon Wrasse for p/a data *vs*. the Squirrel Fish and Humpback Batfish for abundance data). This partial inconsistency may depend on the different reliabilities of volunteer collected data between presence/absence and abundance data sets. In fact, the correctness of abundance parameter ratings ranges from 41 to 82 percent in validation trials of the entire

STE project (and from 64 to 82 percent in the present analysis) while groups of volunteers show better performances in correctly identifying the presence of taxa [19, 20]. In 7 validation trials from the SS Thistlegorm, the parameters of data quality measurements were in line with previous analyses on coral reef environment types within the STE project [20]. The SS Thistlegorm average mean accuracy scored 57.1 percent within a range of 46 to 70 percent, coinciding with the results from [20] where 94 percent of trials scored a mean accuracy between 40 and 70 percent. From the similarity index (SI) parameter all 7 of the dives analyzed here resulted in high levels of precision (SI, 95% CI lower bound $> 75\% \leq 100\%$), whereas only 0.4 percent of trials in [20] obtained these similarity index scores. Notwithstanding the small number of validation trials ran on the SS Thistlegorm data set with respect to the entire STE project [20], the results highly resemble those of the more rigorous trials analyzed in [20] and can therefore provide enhanced reliability to the data that has been used for evaluating the community structure variance over time at this dive site.

In conclusion, there was no clear sequential shift of the SS Thistlegorm community structure over the eight years of monitoring, but some fluctuation around a general cluster characterized by a well-developed community, mainly driven by yearly relative changes in the frequency of a few species. The temporal analysis may have been slightly biased by the different average PY (i.e., seasonality) of collected data, but this bias is likely to be minimal, as the years in which PY was significantly related to taxa data are mainly included in the general cluster and do not display unusual patterns. The community structure at the SS Thistlegorm showed relative stability over time, making this artificial reef a possible and promising refugia for Red Sea communities. Further investigation on the influence of artificial reefs as an auxiliary tool for human-influenced decline could entail the comparison of multiple wreck sites within the northern Red Sea to nearby natural coral reef dive sites through species abundance analyses between and among groupings of sites, possibly by latitudinal and/or temperature gradients.

Supporting information

S1 Table. Yearly sample statistics of the relate test and significance of the relation between taxa data sets (abundance and presence/absence) and diving parameters. * Indicate significant differences ($P < 0.05$).
(DOCX)

S2 Table. DistLM sequential test table of results for each year with a significant relate test between taxa abundance data and diving parameters data. * Indicates significant relationship ($P < 0.05$).
(DOCX)

S3 Table. DistLM sequential test table of results for each year with a significant relate test between taxa presence/absence data and diving parameters data. * Indicates significant relationship ($P < 0.05$).
(DOCX)

S4 Table. Yearly averages of all diving parameters with confidence intervals. PY as date of dive (expressed as percentage of year), MD as maximum depth, AD as average depth, T as temperature, RTD as dive time, and FD as hour (expressed as percentage of day).
(DOCX)

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References

1. Bellwood DR, Hughes TP. Regional-scale assembly rules and biodiversity of coral reefs. Science. 2001 May 25; 292(5521):1532–5. https://doi.org/10.1126/science.1058635 PMID: 11375488
2. Woodhead AJ, Hicks CC, Norström AV, Williams GJ, Graham NAJ. Coral reef ecosystem services in the Anthropocene. Fox C, editor. Functional Ecology. 2019 Mar 28; 33(6).
3. Aronson RB, Macintyre IG, Lewis SA, Hilbun NL. Emergent zonation and geographic convergence of coral reefs. Ecology. 2005 Oct; 86(10):2586–600.
4. Carpenter KE, Abrar M, Aeby G, Aronson RB, Banks S, Bruckner A, et al. One-third of reef-building corals face elevated extinction risk from climate change and local impacts. Science [Internet]. 2008 Jul 25; 321(5888):560–3. https://doi.org/10.1126/science.1159196 PMID: 18653892
5. Kubicek A, Breckling B, Hoegh-Guldberg O, Reuter H. Climate change drives trait-shifts in coral reef communities. Scientific Reports [Internet]. 2019 Mar 6; 9(1):1–10.
6. Bindoff NL, Cheung WW, Kairo JG, Arístegui J, Guinder VA, Hallberg R, et al. Changing ocean, marine ecosystems, and dependent communities. IPCC special report on the ocean and cryosphere in a changing climate. 2019:477–587.
7. Fine M, Cinar M, Voolstra CR, Safa A, Rinkevich B, Laffoley D, et al. Coral reefs of the Red Sea — Challenges and potential solutions. Regional Studies in Marine Science. 2019 Jan; 25:100498.
8. DiBattista JD, Roberts MB, Bouwmeester J, Bowen BW, Coker DJ, Lozano-Cortés DF, et al. A review of contemporary patterns of endemism for shallow water reef fauna in the Red Sea. Journal of Biogeography. 2015 Nov 3; 43(3):423–39.
9. Kleinhaus K, Al-Sawalmih A, Barshis DJ, Genin A, Grace LN, Hoegh-Guldberg O, et al. Science, diplomacy, and the Red Sea's unique coral reef: it's time for action. Frontiers in Marine Science. 2020 Feb 26.
10. Riegl BM, Bruckner AW, Rowlands GP, Purkis SJ, Renaud P. Red Sea coral reef trajectories over 2 decades suggest increasing community homogenization and decline in coral size. Roberts JM, editor. PLoS ONE. 2012 May 31; 7(5):e38396. https://doi.org/10.1371/journal.pone.0038396 PMID: 22693620
11. Tesfamichael D, Pitcher TJ. Multidisciplinary evaluation of the sustainability of Red Sea fisheries using Rapfish. Fisheries Research. 2006 May 1; 78(2–3):227–35.
12. Gladstone W, Curley B, Shokri MR. Environmental impacts of tourism in the Gulf and the Red Sea. Marine Pollution Bulletin. 2013 Jul; 72(2):375–88. https://doi.org/10.1016/j.marpolbul.2012.09.017 PMID: 23079700
13. Berumen ML, Roberts MB, Sinclair-Taylor TH, DiBattista JD, Saenz-Agudelo P, Isari S, et al. Fishes and Connectivity of Red Sea Coral Reefs. Coral Reefs of the Red Sea. 2019; 11:157–79.

14. Sauermann H, Franzoni C. Crowd science user contribution patterns and their implications. Proceedings of the National Academy of Sciences. 2015 Jan 5; 112(3):679–84. https://doi.org/10.1073/pnas.1408907112 PMID: 25561529

15. Silvertown J. A new dawn for citizen science. Trends in Ecology & Evolution. 2009 Sep; 24(9):467–71. https://doi.org/10.1016/j.tree.2009.03.017 PMID: 19586682

16. Kobori H, Dickinson JL, Washitani I, Sakurai R, Amano T, Komatsu N, et al. Citizen science: a new approach to advance ecology, education, and conservation. Ecological Research [Internet]. 2015 Nov 17; 31(1):1–19.

17. Keough HL, Blahna DJ. Achieving integrative, collaborative ecosystem management. Conservation Biology. 2006 Oct; 20(5):1373–82. https://doi.org/10.1111/j.1523-1739.2006.00445.x PMID: 17002755

18. Whitelaw G, Vaughan H, Craig B, Atkinson D. Establishing the Canadian Community Monitoring Network. Environmental Monitoring and Assessment. 2003; 88(1/3):409–18. https://doi.org/10.1023/a:1025545813057 PMID: 14570426

19. Branchini S, Meschini M, Covi C, Piccinetti C, Zaccanti F, Goffredo S. Participating in a Citizen Science Monitoring Program: Implications for Environmental Education. Bernardi G, editor. PLOS ONE. 2015 Jul 22; 10(7):e0131812. https://doi.org/10.1371/journal.pone.0131812 PMID: 26200660

20. Meschini M, Machado Toffolo M, Marchini C, Caroselli E, Prada F, Mancuso A, et al. Reliability of Data Collected by Volunteers: A Nine-Year Citizen Science Study in the Red Sea. Frontiers in Ecology and Evolution. 2021 Jun 24;9.

21. El Gohary G. Wet landscape: using sculptures to form underwater landscape. Democratic Transition and Sustainable Communities. 2013 Nov 6:351.

22. Motawea Hussein SHAIKHON A. Managing underwater heritage in Egypt. International Journal of Advanced Studies in World Archaeology. 2019 Jun 1; 2(1):1–13.

23. Brown S, Henderson J, Mustard A, Postons M. Diving the Thistlegorm: the ultimate guide to a world war II shipwreck. S.L.: Dived Up Publications; 2021.

24. Edney J, Boyd WE. Diving under the radar: divers and submerged aircraft. Journal of Heritage Tourism. 2020 May 28; 16(1):100–17

25. Maynard JA. Severe anchor damage to Lobophyllia variegata colonies on the Fujikawa Maru, Truk Lagoon, Micronesia. Coral Reefs. 2008 Oct 31; 27(2):273–3.

26. Asner GP, Giardina SF, Balzotti C, Drury C, Hopson S, Martin RE. Are Sunken Warships Biodiversity Havens for Corals? Diversity. 2022 Feb 16; 14(2):139.

27. Higgins E, Scheibling RE, Desilets KM, Metaxas A. Benthic community succession on artificial and natural coral reefs in the northern Gulf of Aqaba, Red Sea. Ferse SCA, editor. PLOS ONE. 2019 Feb 27; 14(2):e0212842. https://doi.org/10.1371/journal.pone.0212842 PMID: 30811459

28. Perkol-Finkel S, Benayahu Y. Community structure of stony and soft corals on vertical unplanned artificial reefs in Eilat (Red Sea): comparison to natural reefs. Coral Reefs. 2004 May 14; 23(2).

29. Walker SJ, Schlacher TA. Limited habitat and conservation value of a young artificial reef. Biodiversity and Conservation. 2014 Jan 4; 23(2):433–47.

30. Renzi M, Romeo T, Guerranti C, Perra G, Canese S, Consoli P et al. Are shipwrecks a real hazards for ecosystems in the Mediterranean Sea? Marine Pollution Bulletin. 2017 Nov 15; 124(1):21–32.

31. Arreola-Robles J.L., Elorduy-Garay J.F. Reef fish diversity in the region of La Paz, Baja California Sur, Mexico. Bull. Mar. Sci. 2002; 70 (1):1–18.

32. Ogden JC, Ebersole JP. Scale and community structure of coral reef fishes:A long-term study of a large artifical reef. Mar. Ecol. Prog. Ser. 1981; 4:97–103.

33. Peirano A. Wrecks on the bottom: useful ecological sentinels? Marine Technology Society Journal. 2013; 47(3):118–127.

34. Thistlegorm Siliotti A. In: Sinai Diving Guide Sharm el-Sheikh, Ras Mohammed, Tiran, Gubal, Dahab. Verona, Italy: Geodia; 2005.

35. Shepard RN. The analysis of proximities: Multidimensional scaling with an unknown distance function. II. Psychometrika. 1962 Sep; 27(3):219–46.

36. Kruskal JB. Multidimensional scaling by optimizing goodness of fit to a nonmetric hypothesis. Psychometrika. 1964 Mar; 29(1):1–27.

37. Clarke K, Ainsworth M. A method of linking multivariate community structure to environmental variables. Marine Ecology Progress Series. 1993; 92:205–19.

38. Clarke KR, Gorley RN. Primer v6: user manual—tutorial. Plymouth Plymouth Marine Laboratory; 2006.

39. McArdle BH, Anderson MJ. Fitting multivariate models to community data: a comment on distance-based redundancy analysis. Ecology. 2001 Jan; 82(1):290–7.

40. Kaufman L, Rousseeuw PJ. Finding Groups in Data [Internet]. Kaufman L, Rousseeuw PJ, editors. Wiley Series in Probability and Statistics. Hoboken, NJ, USA: John Wiley & Sons, Inc.; 1990.
41. Bombace G. Artificial reefs in the Mediterranean Sea. Bulletin of marine Science. 1989 Mar 1; 44 (2):1023–32
42. Seixas LB, Barreto NR, Santos LN dos. Artificial reefs for marine and freshwater fish management in Brazil: Researchers profile and academic production over the 1990–2010 period. Oecologia Australis. 2013 Sep; 17(3):374–85.
43. Tessier A, Dalias N, Lenfant P. Expectations of professional and recreational users of artificial reefs in the Gulf of Lion, France. Journal of Applied Ichthyology. 2015 Dec; 31:60–73.
44. Polak O, Shashar N. Can a small artificial reef reduce diving pressure from a natural coral reef? Lessons learned from Eilat, Red Sea. Ocean & Coastal Management. 2012 Jan; 55:94–100.
45. Pinheiro HT, Bernardi G, Simon T, Joyeux J-C, Macieira RM, Gasparini JL, et al. Island biogeography of marine organisms. Nature [Internet]. 2017; 549(7670):82–5. https://doi.org/10.1038/nature23680 PMID: 28854164
46. Macarthur RH, Wilson EO. The theory of island biogeography. Princeton, N.J: Princeton University Press, Tr; 1967.
47. Lengkeek W, Coolen JW, Gittenberger A, Schrieken N. Ecological relevance of shipwrecks in the North Sea. Nederlandse Faunistische Mededelingen. 2013 Jan 1; 41:49–57.
48. Mallefet J, Zintzen V, Massin C, Norro A, Vincx M, De Maersschalck V, et al. Belgian shipwreck: hotspots for marine biodiversity (BEWREMABI). Final Scientific Report. Belgian Science Policy. 155pp. 2008 May.
49. Monchanin C, Mehrotra R, Haskin E, Scott CM, Urgell Plaza P, Allchurch A, et al. Contrasting coral community structures between natural and artificial substrates at Koh Tao, Gulf of Thailand. Marine Environmental Research. 2021 Dec; 172:105505. https://doi.org/10.1016/j.marenvres.2021.105505 PMID: 34717128
50. Rilov G, Benayahu Y. Fish assemblage on natural versus vertical artificial reefs: the rehabilitation perspective. Marine Biology. 2000 Jun 16; 136(5):931–42.
51. Abelson A, Shlesinger Y. Comparison of the development of coral and fish communities on rock-aggregated artificial reefs in Eilat, Red Sea. ICES Journal of Marine Science. 2002 Oct; 59:S122–6.
52. Kotb M. Coral colonization and fish assemblage on an artificial reef off Hurghada, Red Sea, Egypt. Egyptian Journal of Aquatic Biology and Fisheries. 2013 Dec 1; 17(4):71–81.
53. Nicoletti L, Marzialetti S, Paganelli D, Ardizzone GD. Long-term changes in a benthic assemblage associated with artificial reefs. Hydrobiologia. 2007 Apr; 580(1):233–40.
54. Branchini S, Pensa F, Neri P, Tonucci BM, Mattielli L, Collavo A, et al. Using a citizen science program to monitor coral reef biodiversity through space and time. Biodiversity and Conservation. 2015 Oct 7; 24 (2):319–36.
55. Perkol-Finkel S, Shashar N, Benayahu Y. Can artificial reefs mimic natural reef communities? The roles of structural features and age. Marine Environmental Research. 2006 Mar; 61(2):121–35. https://doi.org/10.1016/j.marenvres.2005.08.001 PMID: 16198411
56. Stone RB, Pratt HL, Parker RO Jr, Davis GE. A comparison of fish populations on an artificial and natural reef in the Florida Keys. Mar. Fish. Rev. 1979 Sep 1; 41(9):1–1.
57. Esmail HA. Impact of Terrorism and instability on the tourism industry in Egypt and Tunisia after Revolution. The Business & Management Review. 2016 Jun 1; 7(5):469.
58. Wendt JA. Comparison of the impact of the Arab Spring and terrorist attacks on the decline in tourism in Egypt and Tunisia (2010–2015). GeoJournal of Tourism and Geosites. 2019 Dec 31; 27(4):1367–76.
59. Goffredo S, Pensa F, Neri P, Orlandi A, Gagliardi MS, Velardi A, et al. Unite research with what citizens do for fun: "recreational monitoring" of marine biodiversity. Ecological Applications. 2010 Dec; 20 (8):2170–87. https://doi.org/10.1890/09-1546.1 PMID: 21265450
60. Ateweberhan M, Bruggemann JH, Breeman AM. Effects of extreme seasonality on community structure and functional group dynamics of coral reef algae in the southern Red Sea (Eritrea). Coral Reefs. 2006 Apr 14; 25(3):391–406.
61. Eyal G, Tamir R, Kramer N, Eyal-Shaham L, Loya Y. The red sea: Israel. InMesophotic coral ecosystems 2019 (pp. 199–214). Springer, Cham.
62. Megdadi JM, Khalaf MA, Al-Horani FA, Manasrah RS. Community structure of coral reef fishes in relation to habitat and depth in the northern Gulf of Aqaba, Red Sea. Fresenius Environ. Bull. 2017 Jan 1; 26:1824–34.
63. Chow GSE, Chan YKS, Jain SS, Huang D. Light limitation selects for depth generalists in urbanized reef coral communities. Marine Environmental Research. 2019 May; 147:101–1.