Biol Invasions
https://doi.org/10.1007/s10530-020-02403-5

ORIGINAL PAPER

Fighting on the edge: reproductive effort and population structure of the invasive coral *Tubastraea coccinea* in its southern Atlantic limit of distribution following control activities

Marcelo Schuler Crivellaro · **Thiago Cesar Lima Silveira** · **Fernanda Yumi Custódio** · **Lucas Cambraia Battaglin** · **Michele de Sá Dechoum** · **Adriana Carvalhal Fonseca** · **Bárbara Segal**

Received: 6 April 2020 / Accepted: 28 October 2020

Abstract The detection and control of invasive alien species in marine ecosystems is especially challenging because it is difficult to visualize the full extension of an invasion, while control options are often limited. The invasive scleractinian coral *Tubastraea coccinea* have spread over 8000 km of the Atlantic coastline, from Florida in the USA to southern Brazil, mainly in association with oil and gas platforms. This invasive coral threatens endemic species, reduces native coral recruitment, and modifies communities and trophic interactions, posing a relevant threat to shallow Atlantic reefs. The main aim of our study was to assess the effectiveness of an ongoing *T. coccinea* control program by analyzing the results of control interventions on population structure and cover of the target species in its southern Atlantic limit of distribution. We also describe the reproductive effort of *T. coccinea* in a 12-month time span. Between 2012 and 2019, almost 14,000 colonies were removed, most of them of small size (< 5 polyps). The highest reproductive effort was observed in September. Changes in *T. coccinea* cover, population structure and potential local propagule supply were observed. Control activities led to a reduction in up to half the cover of the invaded area, maintained the majority of the population in non-reproductive sizes, and consequently lowered the potential local propagule supply. Our paper highlights the fundamental role of control in preventing the local spread of *T. coccinea*. Recommendations on management frequency and timing are also discussed in order to contribute to the improvement of management efficiency.

Keywords Marine invasion · Invasive species · Management · Mechanical control · Scleractinian corals · Propagules

---

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s10530-020-02403-5.

M. S. Crivellaro (✉) · T. C. L. Silveira ·
M. de Sá Dechoum · B. Segal
Programa de Pós-graduação em Ecologia, Universidade
Federal de Santa Catarina, Florianópolis,
SC, Brazil
e-mail: marcelocrivellaro@gmail.com

F. Y. Custódio · L. C. Battaglin
Curso de Ciências Biológicas, Universidade Federal de
Santa Catarina, Florianópolis, SC, Brazil

A. C. Fonseca
Instituto Chico Mendes de Conservação da
Biodiversidade, Florianópolis, SC, Brazil

B. Segal
Instituto Coral Vivo, Arraial D'Ajuda,
BA, Brazil

Introduction

As a consequence of the expansion of international trade, the number of translocated and introduced species is continuously increasing (Seebens et al. 2017). Most introductions in marine ecosystems are related to shipping and aquaculture (Molnar et al. 2008), while the global artificialization of habitats is the main driver of success in marine invasions (Bishop et al. 2017). Invasive non-native species can alter ecosystem functions. The introduced clam *Corbula amurensis*, for example, has caused the decline of phytoplankton biomass, consequently reducing primary production in the San Francisco Bay in California, USA (Alpine and Cloern 1992). Moreover, invasive species can impact human health and the economy, as when human pathogens are dispersed by ballast water (Ruiz et al. 2000). An estimated loss of 250 million dollars/year in the anchovy fishery industry is correlated with the introduction of the ctenophore *Mnemiopsis leidyi* (Zaitsev 1992).

Prevention is the most effective strategy to avoid future impacts of biological invasions (Bax et al. 2001). When prevention fails, early detection and rapid response are the next best opportunity for eradication (Wittenberg and Cock 2001; Simberloff et al. 2013). A well-designed eradication program must have enough funds for long-term work and monitoring possible recolonization (Wittenberg and Cock 2001). Therefore, technical and financial limitations hinder the possibilities of eradication. In such cases, control efforts to reduce species abundance or limit spread can protect the natural ecosystem on a local scale as long as they are not discontinued, and may in due course increase the feasibility of eradication (Green et al. 2014).

There are several methods to contain spread using mechanical, chemical and/or biological control (Wittenberg and Cock 2001). Biological control requires close monitoring of host specificity, which can be limited in marine ecosystems. Chemical control has been successfully used under certain conditions (Mytilopsis *sallei* in Australia—Bax et al. 2002; *Caulerpa taxifolia* in California—Anderson 2005) that weighed the benefits against potential impacts such as fast dilution in water and impact on non-target species. Although mechanical control can be labor-intensive and time-consuming, it is highly specific to the target (Wittenberg and Cock 2001). Complementarily, information on the biology of the target species is important to support the definition of control strategies (Dommisse and Hough 2004; Anderson 2005). In Australia, a “ballast window” was defined for ships to fill up with ballast water only in larvae-free periods of the *Asterina* starfish in order to reduce propagule pressure and mitigate spread (Dommisse and Hough 2004).

Two azooxanthallete scleractinian corals, *Tubastraea coccinea* Lesson 1829, native to the Indo-Pacific, and *T. tagusensis* Wells 1982, from the Galapagos Islands, successfully invaded Atlantic shallow reefs. These corals do not seem to have substrate settlement preferences (Creed and De Paula 2007). Their introduction has been mostly associated with structures translocated between regions, such as oil and gas platforms (please see Creed et al. 2017a for more details on *T. coccinea* invasion pathways). The invasion started in the Caribbean (1943) (Cairns 2000) and spread throughout the Gulf of Mexico (Fenner 2001), the Atlantic coast of Brazil (Castro and Pires 2001), Florida in the USA (Fenner and Banks 2004), and the Canary Islands of Spain (López et al. 2019). The first record of the coral on the Brazilian coast was made on an offshore oil platform in the late 1980s, in the Campos Basin, in Rio de Janeiro (RJ) state (Castro and Pires 2001). It spread fast along natural substrates on the rocky shores of Ilha Grande, also in RJ state, in the 1990s (Castro and Pires 2001; De Paula and Creed 2004). Nowadays, several disjunct populations are present along 3800 km of the Brazilian coastline, from North (Ceará state, Soares et al. 2016) to South (Santa Catarina state, Capel 2012).

These corals, popularly known as “sun corals”, are spreading fast (~ 2 km/year, Da Silva et al. 2014) mainly due to early maturity, and sexual and asexual reproduction (Glynn et al. 2008; De Paula et al. 2014; Capel et al. 2017; Luz et al. 2020). In the Ilha Grande Bay (Rio de Janeiro state), *T. coccinea* reproduces many times throughout the year. Different stages of propagule development occur simultaneously in a polyp. Two reproductive peaks were observed, one from September to December and the other one from February to May (De Paula et al. 2014). Larvae were alive for ~ 4 months in artificial conditions (Mizrahi et al. 2014; Luz et al. 2020), but they usually settle and metamorphose fast (1–3 days), displaying gregarious behavior near parental colonies (Glynn et al. 2008; De Paula et al. 2014; Luz et al. 2020). Invasive

Tubastraea corals impact trophic interactions (Miranda et al. 2018a; Silva et al. 2019), modify communities (Lages et al. 2011), damage endemic species (Creed 2006; Santos et al. 2013; Barbosa et al. 2019), and reduce native coral recruitment (Miranda et al. 2018b).

Several initiatives based on mechanical methods to control *Tubastraea* spp. have been locally implemented by environmental government agencies, non-governamental organizations, researchers and volunteers in Brazil (Creed et al. 2017b), as well as by local managers in the Flower Garden Banks Marine Sanctuary (Florida, USA) (Precht et al. 2014). Although a National Plan for Prevention, Control and Monitoring (NPPCM) for *Tubastraea* spp. was published in Brazil by the Ministry of Environment in 2018 (MMA 2018), no control actions have been implemented on a national scale so far. However, one of the goals of the NPPCM is to define priority areas for control and monitoring, as well as to eradicate small, isolated, and initial populations. Albeit manual removal using a chisel and hammer has proven effective to reduce *Tubastraea* spp. cover (De Paula et al. 2017), sun corals grow at high regeneration rates and can grow back from remaining tissue that is not completely detached from the substrate (Luz et al. 2018), which can undermine the effectiveness of control.

The main aim of our study was to assess the effectiveness of an ongoing *T. coccinea* control program by analyzing the results of control interventions on population structure and cover of the target species in its southern Atlantic limit of distribution. We also assessed the outcome of control activities conducted by local managers and volunteer researchers. In the same region, we investigated propagule production of *T. coccinea* for 13 months to determine the most appropriate time interval for control interventions. In addition, we combined information on reproduction and cover to estimate the potential of local propagule release. We assumed that control activities would reduce cover and maintain populations with small individuals, reducing the local population propagule supply. Considering the data available and the threat of invasion by *T. coccinea* in southern Atlantic ecosystems, we expect our study to contribute with relevant information to support and improve the ongoing control program.

Materials and methods

Study system

This study was carried out on rocky shores of the Arvoredo Marine Biological Reserve (Rebio Arvoredo from now on), a no-take, no-entry protected area (176 km^2) on the coast of Brazil, in the southwestern Atlantic (between 28° 36′ 16.94″ S and 28° 13′ 43.18″ S). Rebio Arvoredo comprises an archipelago formed by three islands (Arvoredo, Deserta, and Galé) of rocky shores surrounded by a highly productive pelagic ecosystem. Oceanographic conditions vary seasonally, with temperatures ranging from ~ 15 to 29 °C (Faria-Junior and Lindner 2019). In the summer, tropical hot and nutrient poor waters dominate the continental shelf. In winter, waters are cold, generating much higher primary production on the ocean surface due to the influence of the La Plata River and Patos Lagoon discharges (Freire et al. 2017). Given these conditions, Rebio Arvoredo plays a key biological role in the region. The area includes the most southern coralline algal bank on the Brazilian coast (Rocha et al. 2006), serves as refuge to commercial fish species (Anderson et al. 2019), and is a transitional area between tropical and temperate fauna for many benthic species. The Reserve is the southern distribution limit of of *T. coccinea* in the Atlantic Ocean (Capel 2012; Lindner et al. 2017). This occurrence is at least 450 km away from the nearest known population, located farther north (Alcatrazes islands, São Paulo state).

The first record of *T. coccinea* on the rocky shores of Arvoredo island was made in 2012, outside the limits of Rebio Arvoredo, in a site called Engenho (EG, Fig. 1) (Capel 2012). Subsequently, invasion patches were observed in two other sites, Saco do Farol (SF) and Saco do Vidal (SV), in 2013 and 2014, respectively (Fig. 1). Still in 2014, the invader was found within the limits of Rebio Arvoredo in Rancho Norte (RN). Judging by the large size of colonies, this seems to be the site that was invaded first in the area (Fig S1a). In 2015, two colonies were found in a shipwreck by Galé Island (GI), also within the limits of Rebio Arvoredo. Currently, invasion patches are still scattered, outside and within the limits of Rebio Arvoredo. Invasion is generally restricted to vertical surfaces, caves and crevices. These substrates are mostly occupied by calcareous and turf algae,

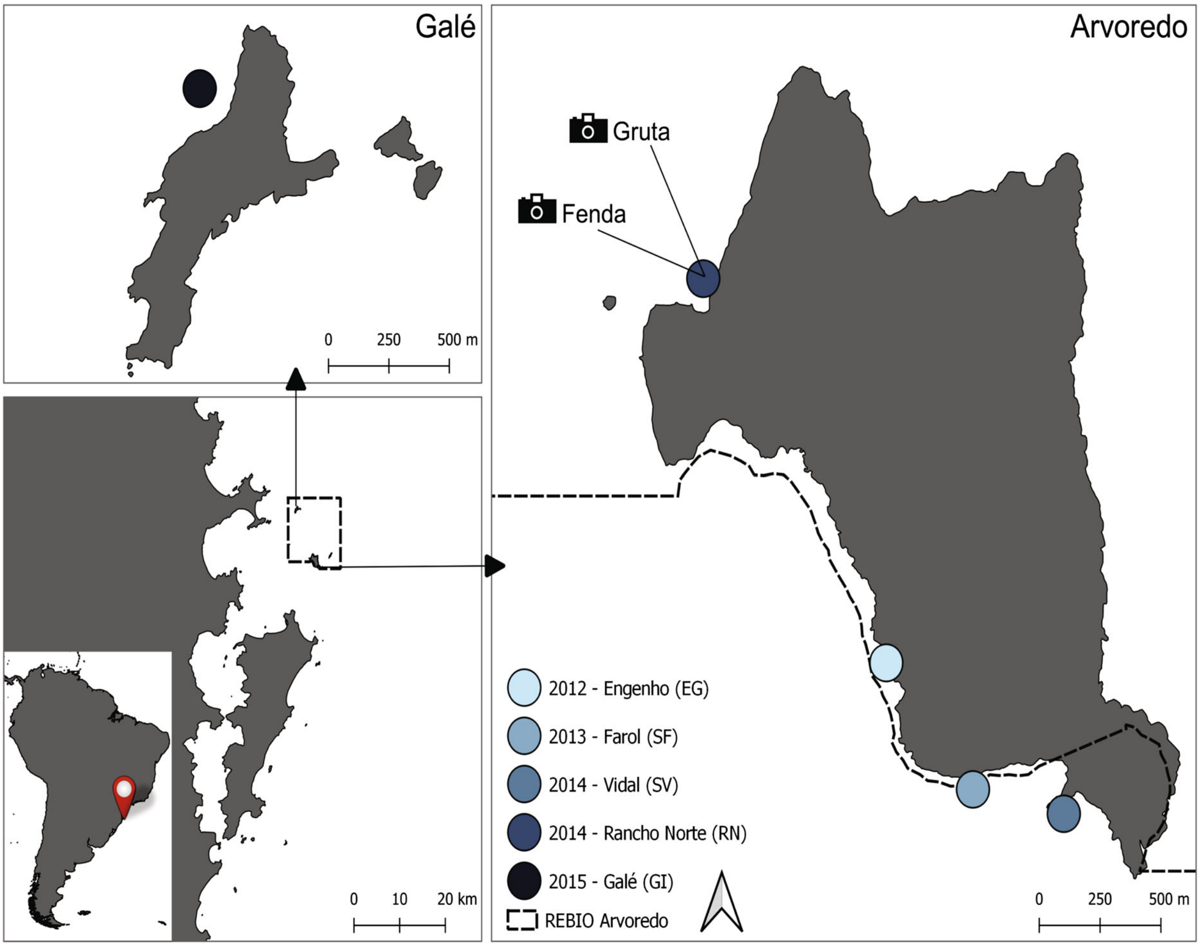

Fig. 1 Map of the study area showing sites of occurrence of *Tubastraea coccinea* and areas of photographic sampling. Sites are chronologically ordered by successive observations of the invasive coral. The dotted line shows the limits of Rebio Arvoredo

sponges, ascidians, the octocoral *Carijoa riisei*, the zoanthids *Parazoanthus swiftii* and *Palythoa caribaeorum* (only on horizontal surfaces, see Almeida Saá et al. 2019), and the scleractinians *Astrangia rathbuni, Phyllangia americana* and *Madracis decatis* (a rare species) (Capel 2012; Lindner et al. 2017).

Control activities

In 2012, the federal environmental agency in charge of Rebio Arvoredo, Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio), in partnership with researchers of the Federal University of Santa Catarina (UFSC), established a program to control and monitor invasions by *T. coccinea*. Current control activities consist in two 50-min dives performed by two pairs of divers, each pair working on a site. While one diver manually removes colonies with a chisel and hammer, the other inserts them in a plastic bag to prevent dispersal of larvae or colony fragments. The removal of larger colonies was prioritized because of the larger number of polyps and likely higher reproductive potential. Control activities were undertaken outside (Engenho, Saco do Farol and Saco do Vidal) and within the limits of Rebio Arvoredo (Galé Island and Rancho Norte). The numbers of control actions in each site and of colonies removed since the beginning of the control program were recorded and analyzed. All colonies were classified in five size classes according to the number of polyps: I (1–5 polyps), II (6–10 polyps), III (11–20 polyps), IV (21–40 polyps), and V (41 or more polyps). Figures were produced using the package *ggplot2* (Wickham 2016) in RStudio software (2020).

Reproductive effort

To describe the yearly reproductive effort of sun corals, we collected five colonies of similar sizes in each campaign between January, 2018, and June, 2019 (January to April, and July to November, 2018, and January, February, April and June, 2019) Collections were made by scuba divers at 2–8 m depth on rocky shores of Arvoredo Island. Colonies were fixed in 4% formalin solution, decalcified in 10% formic acid and 2% formalin for 48 h, then cleaned up in running tap water for 24 h (Glynn et al. 2008; De Paula et al. 2014). We measured the diameter, maximum oral disc diameter and distance between the oral and aboral ends of polyps to evaluate colony and polyp area (circle formula) and polyp volume (cylinder formula). We dissected two central polyps (N = 126) in each colony, and, when present, we counted all propagules under a stereomicroscope. As the development stages of embryos can only be differentiated through histological analysis (Glynn et al. 2008), we used the term propagule to refer to all stages. We only selected central polyps because larvae are more abundant in central polyps than on colony edges (Chornesky and Peters 1987). We extrapolated the number of propagules observed per polyp area to total colony area (propagules cm^2. $colony^{-1}$) in order to establish a correlation with cover (explained in the next section). We considered solely the central (fertile) part of the colonies for the extrapolation. Descriptive analyses were performed in RStudio software (2020) and figures were produced using the package *ggplot2* (Wickham 2016).

T. coccinea cover, population size structure and propagule supply

Between January, 2018, and June, 2019, we monitored two invaded sites where frequent control actions were conducted in REBIO Arvoredo (Rancho Norte, sites “Fenda” and “Gruta”) approximately every four months (5 sampling efforts in Gruta and 6 in Fenda, Table S1). Both sites occupy an area of approximately 5 m in length by one and a half in width at a depth of three meters. Accessibility to invasion patches for mechanical control differed between sites. Although a crevice in “Fenda” is inaccessible (Fig. S1c), a vertical surface in this site is easier to access than in “Gruta”, where sun corals were growing on the negative surface of a cave. The invaded surfaces were photographed in the format of photo quadrats (25 × 25 cm). To evaluate cover and assess population size structure, we outlined the area of each colony in ImageJ Software (Schindelin et al. 2012) on five photographs taken in each site (Ntotal = 55). For this evaluation, we classified each colony according to five size classes based on surface area (< 0.3 cm^2; 0.3–1 cm^2; 1–5 cm^2; 5–10 cm^2; >10 cm^2) and determined their relative frequency (%). Classes < 0.3 cm^2 (recruit size 4 mm, see Mizrahi 2008) and 0.3–1 cm^2 represent non-reproductive individuals, since propagules were observed in colonies larger than 1.2 cm^2 or more (De Paula 2007). Descriptive statistics were performed in RStudio software (2020) and figures were produced using the package ggplot2 (Wickham 2016).

To estimate the local potential propagule supply of sun coral populations in our samples, we combined the data obtained on propagule production (propagules cm^2.$colony^{-1}$) and coral cover (cm^2). We used the median of propagules per cm^2 from the month of September (200 propagules cm^2 $colony^{-1}$) to extrapolate to the coral cover (cm^2) measured in each sample. Data from the month of September were used because it was the period of highest reproductive effort observed in our analysis. In the absence of control actions, all colonies would probably have reached an equivalent reproductive condition. Polyps located on the edge of colonies produce less larvae due to potential defense activity that is common along the edges (Chornesky and Peters 1987). Therefore, we measured the total area of 30 colonies on photographic samples and excluded the area of the peripheral (infertile) polyps from each of the colonies. We considered peripheral polyps as those not entirely inside the colony (with a free border). As a result, we obtained a fraction of 44.37% of potentially reproductive tissue in relation to total colony area. The cover of non-reproductive colonies/recruits (< 0.3 cm^2 and 0.3–1 cm^2 classes) was excluded from the extrapolation.

Results

Control activities

Between February, 2012, and October, 2019, 59 control actions were conducted in Rebio Arvoredo, resulting in the removal of 13,986 colonies (Fig. 2). The majority of these colonies belonged to the smallest size classes I (6872; 49.1%) and II (4611; 33%), with less colonies in classes III (1865; 13.3%), IV (468; 3.4%), and V (170; 1.2%). A considerable number of large colonies was eliminated shortly after discovery (Rancho Norte, site "Fenda", 2014), with the removal of 108 and 75 colonies in classes IV and V, respectively. Only small colonies remain since then. Another large group of class V colonies was eliminated in Saco do Farol 2019 (43 colonies), and a few more colonies of size V (maximum 7) were removed in other 20 control efforts from scattered areas.

Rancho Norte was the site where more control actions (21) were conducted and the highest number of colonies was removed (6239), mainly because it contained the two largest invasion patches (in "Fenda" and "Gruta"). The highest number of colonies eliminated in a single control action was 876 (class I; 437; II: 264; III: 128; IV: 40; V: 7) at Engenho (2019). The smallest number of colonies eliminated refers to only two colonies immediately removed when discovered in a shipwreck by Galé island in 2015. Despite this early detection and rapid response effort, 23 colonies were found and eliminated in the same shipwreck in 2019. The years with the largest numbers of colonies removed were 2018 (4595) and 2019 (3805); conversely, 2012 (324) and 2015 (137) were the years with the lowest number of colonies removed. The years 2018 (15) and 2019 (11) were also those with more control campaigns, while 2013 (3) and 2015 (2) were the years with less control campaigns.

Reproductive effort

Samples were collected during 13 months, between January, 2018, and June, 2019 (Appendix Table 1). Colony diameter ranged between 2.1 and 7.3 cm (mean = 4 cm ± 1.2), while polyp diameter ranged between 0.6 and 1.6 cm (mean = 1 cm ± 0.16). Polyp volume ranged between 250 and 5069 mm^3 (mean = 1205 ± 757 mm^3). The number of propagules per polyp varied widely (mean = 153.8 ± 165.6 SD). Propagules were present throughout the year except in January, 2018, and were observed in low numbers in August, 2018 (44 ± 56.3 SD propagules.polyp^{-1}), January, 2019 (39.8 ± 42.5 SD propagules.polyp^{-1}), February, 2019 (37.6 ± 26.8 SD propagules.polyp^{-1}) and June, 2019 (78.6 ± 47.7 SD propagules.polyp^{-1}) (Fig. 3,

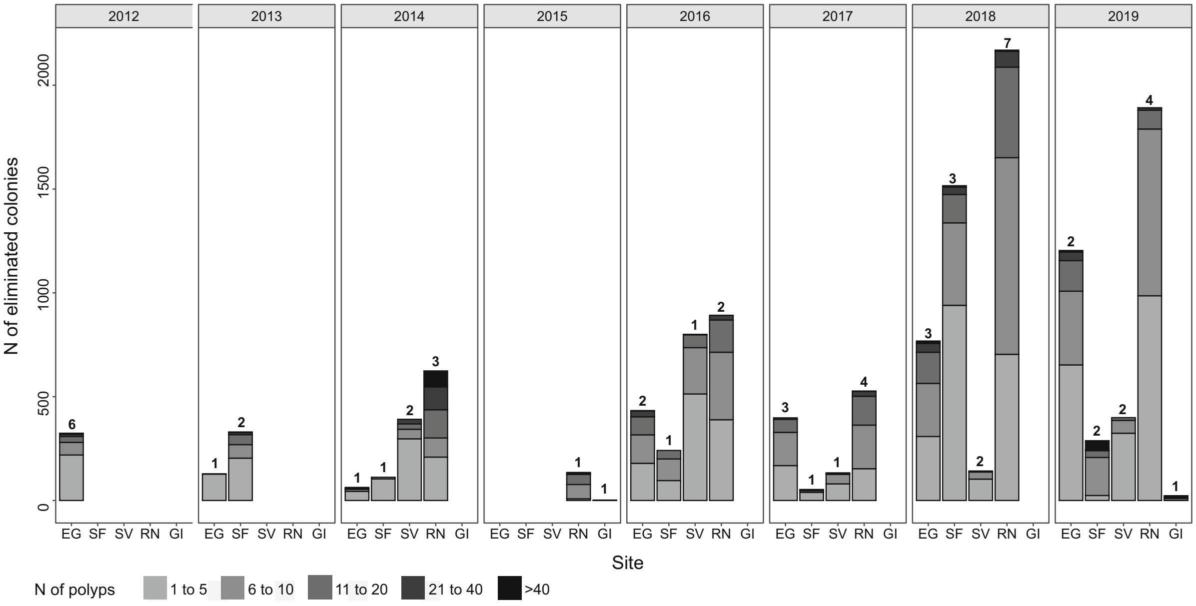

Fig. 2 Number of control activities (labels on top of the bars) and colonies eliminated in each class of polyp numbers (vertical axis on the left) in each site. EG = Engenho; GI = Galé Island; RN = Rancho Norte; SF = Saco do Farol; SV = Saco do Vidal. Zero indicates the absence of control activities

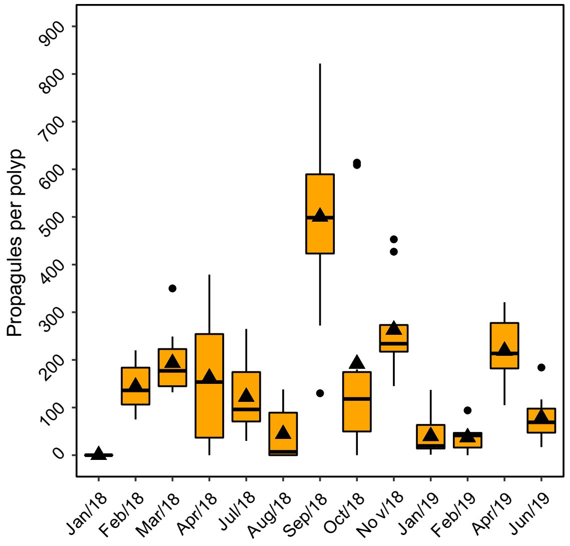

Fig. 3 Variation and mean (black triangle) of *T. coccinea* propagules per month (2 polyps each, 5colonies/month). The horizontal black line inside box-plots represents the median

Appendix Table 1). The absence and low number of propagules in summer (January and February) may indicate events of larvae release, as propagules were clearly observed in an advanced larvae development stage only in January, 2019, which coincides with the peak reproductive period observed in Ilha Grande Bay, in Rio de Janeiro (600 km north of Santa Catarina). The highest numbers of propagules per polyp were observed in September (822 max; 500 ± 205 SD) and November, 2018 (453 max; 263 ± 102.3 SD). Polyps in October, 2018, also contained a large number of propagules (614 max; 191 ± 229.3 SD), with the exception of one colony in which no propagules were observed.

Cover, population size structure and propagule supply

In both invaded areas, our first sampling effort (January 2018) was conducted 167 days after a control action. During the period of sampling, both areas were submitted to control actions six times, with similar total numbers of colonies removed (Fenda—1673; Gruta—1614). Most colonies eliminated (83%) contained between 1 and 10 polyps (classes I and II), with only three of them having more than 41 polyps.

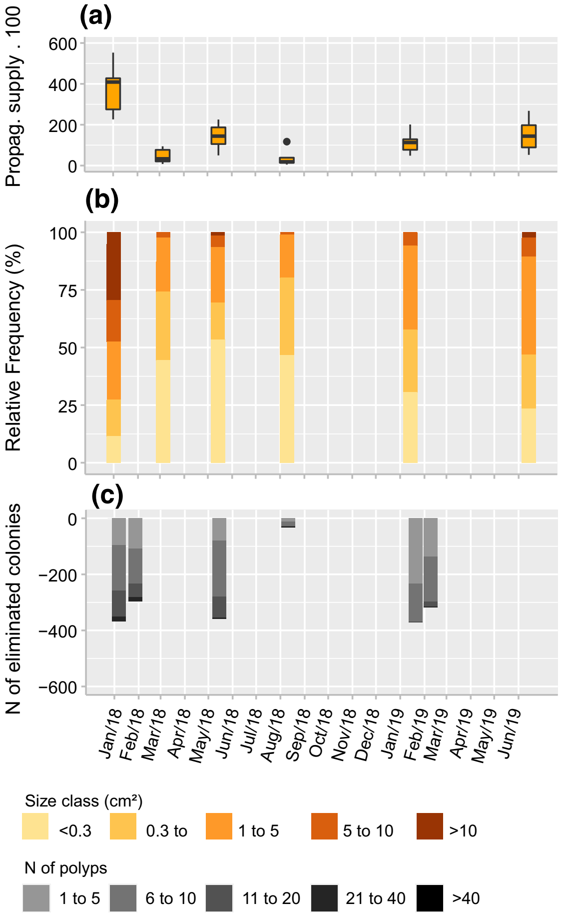

Fig. 4 Changes in population structure after control actions in Fenda. **a** Estimated propagule supply (value of y axis was divided by 100 to adjust figure dimensions; the horizontal black line inside box-plots represents the median); **b** relative frequency of colonies in each size class, and **c** number of colonies eliminated in each class of polyp numbers per control action

Figure 4 shows the estimated propagule supply (4a), population structure (4b), and the number of eliminated colonies in each control action (4c) in Fenda. The highest mean cover 30.6 ± 10.35 SD % (Fig S2a) and mean estimated propagule supply of 37,913 ± 12,811 SD propagules per 0.06m^2 were observed in the first sampling effort (January, 2018). Population structure was almost evenly distributed ($<$ 0.3 cm^2—11.5%; 0.3–1 cm^2—15.8%; 1–5 cm^2—25.1%; 5–10 cm^2—17.9%; and $>$ 10 cm^2—29.4%), with by far the highest proportion of reproductive colonies observed. By January, 2018, 664 colonies had been removed in two control actions (classes I, 204; II, 285; III, 142; IV: 33). These efforts led to a reduction in cover from 30.6 ± 10.35 SD % to 5 ± 3 SD % (Mar 2018), or almost eight times less potential production of propagules (from 37,913 ± 12,811 SD to 5047 ± 3758 SD propagules per 0.06 m^2). Only 2.3% of colonies were larger than 5 cm^2. In August, 2018, mean cover was 4.9 ± 3.2 SD %, and propagule supply, 4438 ± 4545 SD propagules per 0.06 m^2, with 80.4% of colonies probably non-reproductive and only 0.9% of colonies larger than 5 cm^2. A low proportion of large colonies was also observed in the following control actions. Only 9 colonies with more than 20 polyps (class IV) were removed in control actions in May (5 colonies) and August (2 colonies), 2018, and in two actions in January, 2019 (2 colonies). In June, 2019, cover was 12.9 ± 7 SD %, propagule supply, 15,273 ± 8,639 SD propagules per 0.06 m^2, and the proportion of reproductive colonies, 52.8%.

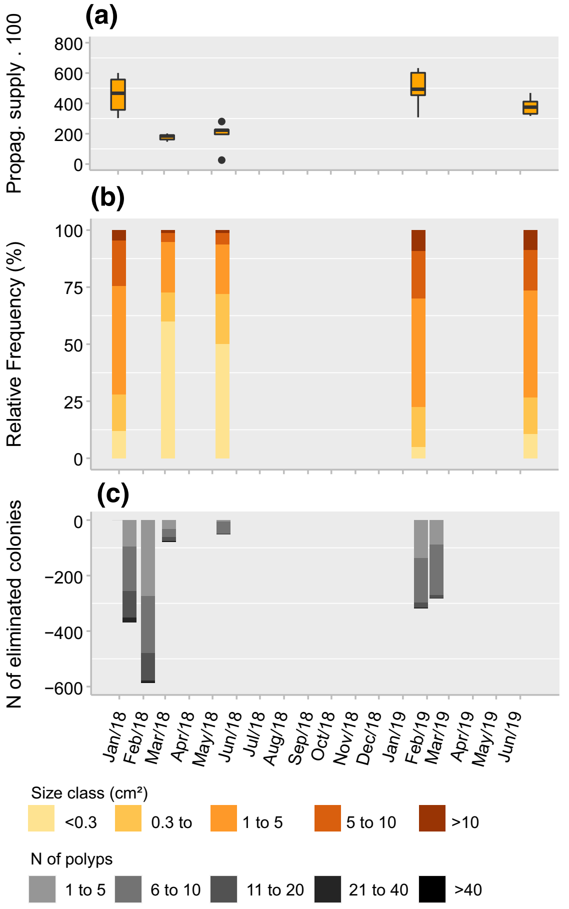

Fig. 5 Changes in population structure after control actions in Gruta. **a** Estimated propagule supply (value of y axis was divided by 100 to adjust figure dimensions; the horizontal black line inside box-plots represents the median); **b** relative frequency of colonies of each size class, and **c** number of colonies eliminated in each class of polyp numbers per control action

Figure 5 shows the estimated propagule supply (5a), population structure (5b), and the number of colonies eliminated in each control action (5c) in Gruta. In January, 2018, mean cover was 37.9 ± 10.6 SD % (Fig S2b), and propagule supply, 46,019 ± 12,811 SD propagules per 0.06 m^2 (Fig. 5b). These were smaller compared with colonies in Fenda (< 0.3 cm^2—12%; 0.3–1 cm^2—15.9%; 1–5 cm^2—47.6%; 5–10 cm^2—19.9%; and > 10 cm^2—4.5%). Afterwards, 955 colonies were removed in two control actions (classes I, 370; II, 366; III, 193; IV, 25; V, 1). In March, 2018, cover and propagule supply were 15.9 ± 2.6 SD % and 18,254 ± 2,342 SD propagules per 0.06 m^2, respectively, with 72.8% of colonies non-reproductive. In the 2018 control actions (March and May), only 130 colonies were removed (classes I, 38; II, 71; III, 17; IV, 3; V, 1). There was an interval of 252 days between control efforts in 2018 (May) and 2019 (January), when the highest cover (41 ± 10.4 SD %), propagule supply (49,912 ± 12,947 SD propagules per 0.06 m^2), proportion of reproductive colonies (77.3%) and colonies larger than 10 cm^2 (9%) were observed. Shortly after, 599 colonies (classes I, 226; II, 342; III, 28; IV, 3) were removed in two control actions conducted in January, 2019. In June, 2019, cover was 31.2 ± 5 SD %, propagule supply, 38,247 ± 6,130 SD propagules per 0.06 m^2, and 73.3% of the colonies were reproductive.

Discussion

Our results support the importance of ongoing control actions to slow the spread of *T. coccinea*. Even if manual removal is not the ideal control method because of the high regenerative capacity of *T. coccinea* (Luz et al. 2018), the current advantage of manual removal is to restrain populations to a large proportion of small, non-reproductive individuals, therefore reducing propagule supply, and consequently restricting spread. The current status of relatively small invasion patches in the Arvoredo marine reserve and surroundings is due to the results of ongoing control. Although eradication is not feasible with the control methods currently in use, the maintenance of populations in restrained sizes with individuals in small size classes increases future possibilities of eradication (Myers et al. 2000), buying time until better solutions for more effective control are identified.

Some biological traits of *T. coccinea* explain its high invasiveness. The species reproduces mostly asexually (Capel et al. 2017), and colonies as small as two polyps are already mature (Glynn et al. 2008; De Paula et al. 2014). In artificial conditions, larvae can metamorphose in the water column and form clusters that settle and start benthic life as a colony (Mizrahi et al. 2014; Luz et al. 2020). Although larvae of large colonies may be viable for a long time (~ 90 days, Luz et al. 2020), they usually show gregarious behavior and settle fast (1–3 days, Glynn et al. 2008; De Paula et al. 2014). The complexity of recruitment in corals is well known. Estimates show that only 1% of marine benthic invertebrates successfully survive the several bottlenecks in this early stage of life (Gosselin and Qian 1997). In this sense, considering our extrapolation of the propagule supply at the beginning of sampling, populations of *T. coccinea* in "Fenda" and "Gruta" may potentially produce approximately 80 thousand propagules. Considering a survival rate of ca. 1%, this means that a release event at that moment may generate almost 1000 new settled individuals with capacity to establish new invasion patches on their own. The reproductive data generated in our study are extremely important for planning the timing of control activities, which should be conducted before reproductive peaks and before recruits or regenerative colonies reach maturity. Based on these data, we recommend that control activities are preferentially carried out between May and September in the southern Atlantic coast of Brazil. Although we cannot precise the relative contribution of larval recruitment versus regeneration for the maintenance of the *T. coccinea* population in the Rebio Arvoredo, it is important to highlight that limiting local production of propagules is key to contain spread to new sites and the establishment of new invasion *foci*.

Despite the high number of colonies removed (~ 14 thousand), the invasion observed in our study sites can be considered small compared to those in the Brazilian southeastern coast. In the Ilha Grande Bay (Rio de Janeiro), for instance, more than ~ 220,000 colonies had been removed in more than 150 control actions by 2017 (Creed et al. 2017b). Furthermore, a *T. coccinea* distribution and abundance survey in a port area influenced by upwelling in Arraial do Cabo (Rio de Janeiro) fifteen years after its first record showed that more than 50% of the total population was formed by colonies larger than 30 cm^2 (Batista et al. 2017). During the 18 months in which we conducted sampling work, colonies larger than 5 cm^2 never made up more than 50% of the populations. Sun coral is assumed to change communities completely when cover reaches 45% (Lages et al. 2011). Even with inaccessible crevices in Rebio Arvoredo (Fig S1c), there are not any known invasion patches in the region of our study with such levels of cover, which emphasizes the relevance of the ongoing control program to contain the invasion and consequential impacts on local communities and ecosystems.

De Paula et al. (2017) suggested that, after the first control effort of *T. coccinea*, follow-up within a 6–12-month interval may be sufficient. Our results corroborate this indication. From the discovery of invasion in Fenda (2014, Fig S1a) until now (Fig S1b), the effectiveness of control activities undertaken so far is palpable. Besides, immediately after control actions, regenerating colonies are flattened and virtually lack the calcareous skeleton (Fig S3b, S3e), becoming very difficult to detach from the substrate (Luz et al. 2018). Thus, the energy used on mechanical control should be invested when colonies have regenerated enough to be effectively removed, before reaching maturity. The combination of initial efforts of manual removal with a revisit to scratch/suction the remaining regenerating tissue may produce interesting results. Other methods to eliminate sun corals include the use of acetic acid (Creed et al. 2018), sodium hypochlorite (Altvater et al. 2017), freshwater (Moreira et al. 2014), and wrapping (Mantelatto et al. 2015), the last of which requires specific conditions or isolation of colonies for application and is extremely difficult to implement with success in tridimensional natural rocky reefs.

Regulatory frameworks for preventing the introduction and spread of sun corals must be established on all scales (global, regional and local) (Hewitt et al. 2008). There are 158 active offshore oil platforms, 23 drill ships, and expectations that 18 new structures will be installed by 2022 (Mafra 2018) in Brazil alone. A large part of these oil platforms (42%) are 20–25 years old, which means they must be decommissioned soon (Mafra 2018). Additionally, projections indicate that oil production will double in the next decade, generating more than 250 billion dollars in investments (MME 2018). The relevance of the expansion of these activities to the Brazilian economy is undeniable, but they are also a threat to marine ecosystems. While specific regulations on the decommissioning of such structures do not exist at present, the oil industry claims that having to clean all structures offshore would render the business unviable. The complexity of the issue must be acknowledged, but it is essential to prevent the transit of contaminated structures. There is an urgent need for the development of specific regulatory frameworks by policy makers. The commitment of the governments of Australia and New Zealand are good examples to follow (see Hewitt et al. 2008). In a similar way to using the "ballast window" proposed to reduce the spread of the starfish *Asterina* in Australia (Dommisse and Hough 2004), structures potentially contaminated with *Tubastraea* corals should be moved only in periods of lowest reproductive potential (winter in the southern hemisphere). Furthermore, it is very important that these structures are immediately sent for clean-up (exposure to air or freshwater), avoiding docking for long periods in sheltered environments suitable to sun coral development.

In order to generate an effective response to marine invasions, different technical components must be integrated, such as data on the biology of the target species, agencies in charge of invaded areas, field expertise, and financial and human resources (Anderson 2005). Successful marine eradications have been achieved based on these premises (*Mytilopsis sallei* in Australia—Bax et al. 2002; *Caulerpa taxifolia* in California—Anderson 2005). However, the early detection and rapid response of environmental agencies and scientists in the southern distribution limit of *T. coccinea* were insufficient for eradication to be achieved. This is explained mainly because sun corals have occupied crevices inaccessible to manual removal (Fig S1c), and most importantly, because a formal eradication program with sufficient funding to eliminate all individuals and monitor recolonization in the following years (such as recommended by Simberloff 2003) has not been implemented. The recently published National Plan for the Prevention, Control and Monitoring for *Tubastraea* spp. (MMA 2018) must be put into action. The small, isolated invasion in the southern Atlantic limit of distribution in Rebio Arvoredo, an important Marine Reserve, must be prioritized for eradication. We agree with Oigman-Pszczol et al. (2017) on the claim that invasion by *T. coccinea* is not a lost cause.

Citizen-science may be helpful to increase *T. coccinea* control efforts. Positive results were obtained by citizen engagement in the detection and control of lionfish (*Pterois volitans*) in the Gulf of Mexico (see Scyphers et al. 2015). *Tubastraea coccinea* was actually first reported for Arvoredo island by recreational divers (Capel 2012). In 2013, the federal environmental agency (ICMBio), in partnership with the Marine Biodiversity Lab at UFSC, organized a workshop with staff from dive schools in the region to explain the threats and impacts of invasion by *T. coccinea* and provided training for the identification of colonies. Since then, some invasion patches around Arvoredo island were detected and reported by trained divers. However, controlling *T. coccinea* is not technically simple, requiring well-trained and qualified scuba divers to prevent the propagation of fragments. Creed et al. (2017b) involved local communities to collect corals by snorkeling and sell coral skeletons as a craftwork on a highly invaded site (Ilha Grande, RJ). Because invasion is restricted in Rebio Arvoredo, citizen engagement would be best used for the early detection of new invasion patches.

Given the challenges of controlling invasive marine species (Williams and Grosholz 2008) and global predictions of increase in the number of non-native species (Seebens et al. 2017), our work provides relevant information to support and improve ongoing control of *Tubastraea* spp. Our data show that control efforts reduced the population propagule supply of *T. coccinea* by maintaining a large portion of the population in small, non-reproductive sizes. We also highlight the urgent need for new techniques that might lead to eradication. Furthermore, the NPPCM for *Tubastraea* spp. must be implemented and prioritize the southern Atlantic limit of distribution including Rebio Arvoredo. Finally, even in the lack of a formal eradication program with ensured funding, control activities have successfully slowed the spread of *T. coccinea.* With the use of more effective techniques and the commitment of federal policies, eradication can be feasible.

Acknowledgements We thank Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio) for the research permits (Document Number 57,757) and support in fieldwork. Special thanks to the editor and to two anonymous reviewers, and also to Alberto Lindner, Joel Creed, Nivaldo Peroni and Sílvia Ziller for providing helpful comments and suggestions on the manuscript. We thank "Projeto Invasores Marinhos" (FAPESC 4302/2010-8), and Bruna Gregolleto and Alberto Lindner for the support in sampling. Scholarships were provided by Coordenação de Aperfeiçoamento de Pessoal de Nível Superior—Brasil (CAPES)—Finance Code 001 for Marcelo Crivellaro and for Fernanda Yumi Custódio by UFSC (Programa Institucional de Bolsas de Iniciação Científica—PIBIC/CNPq 2018/2019).

Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

References

Almeida Saá AC, Crivellaro MS, Winter BB, Pereira GR, Bercovich V, Horta PA, Schubert N (2019) Unraveling interactions: do temperature and competition with native species affect the performance of the non-indigenous sun coral *Tubastraea coccinea*? Coral Reefs 39:99–117. https://doi.org/10.1007/s00338-019-01870-0

Alpine AE, Cloern JE (1992) Trophic interactions and direct physical effects control phytoplankton biomass and production in an estuary. Limnol Oceanogr 37:946–955

Altvater L, de Messano LVR, Andrade M, Apolinário M, Coutinho R (2017) Use of sodium hypochlorite as a control method for the non-indigenous coral species *Tubastraea coccinea* Lesson, 1829. Manag Biol Invas 8:197–204. https://doi.org/10.3391/mbi.2017.8.2.07

Anderson LWJ (2005) California's reaction to *Caulerpa taxifolia*: a model for invasive species rapid response. Biol Invas 7:1003–1016. https://doi.org/10.1007/s10530-004-3123-z

Anderson AB, Batista MB, Gibran FZ, Félix-Hackradt FC, Hackradt CW, García-Charton JA, Floeter SR (2019) Habitat use of five key species of reef fish in rocky reef systems of southern Brazil: evidences of MPA effectiveness. Marine Biodivers 49:1027–1036

Barbosa A, Vinagre C, Mizrahi D, Duarte R, Fores A (2019) Invasive sun corals and warming pose independent threats to the brain coral *Mussismilia hispida* in the Southwestern Atlantic. Mar Ecol Prog Ser 629:43–54. https://doi.org/10.3354/meps13110

Batista D, Gonçalves JEA, Messano HF, Altvater L, Candella R, Elias LMC, de Messano LVR, Apolinário M, Coutinho R (2017) Distribution of the invasive orange cup coral *Tubastraea coccinea* lesson, 1829 in an upwelling area in the South Atlantic Ocean fifteen years after its first record. Aquat Invas 12:23–32. https://doi.org/10.3391/ai.2017.12.1.03

Bax N, Carlton JT, Mathews-Amos A, Haedrich RL, Howarth FG, Purcell JE, Rieser A, Gray A (2001) The control of biological invasions in the world's oceans. Conserv Biol 15:1234–1246. https://doi.org/10.1111/j.1523-1739.2001.99487.x

Bax N, Hayes K, Marshall A, Parry D, Thresher R (2002) Man-made marinas as sheltered islands for alien marine organisms: establishment and eradication of an alien invasive marine species. In: Veitch CR, Clout MN (eds) Turning the tide: the eradication of invasive species, IUCN SSC Invasive Species Specialist Group. IUCN , Cambridge, pp 26–39

Bishop MJ, Mayer-Pinto M, Airoldi L, Firth LB, Morris RL, Loke LHL, Hawkins SJ, Naylor LA, Coleman RA, Su YC, Dafforn KA (2017) Effects of ocean sprawl on ecological connectivity: impacts and solutions. J Exp Mar Biol Ecol 492:7–30. https://doi.org/10.1016/j.jembe.2017.01.021

Cairns S (2000) Studies on the natural history of the Caribbean region. In: Studies on the natural history of the Caribbean Region, pp 134–136

Capel KCC (2012) Scleractinia (Cnidaria: Anthozoa) da reserva biológica marinha do arvoredo (sc), com ênfase na estrutura espaço-temporal da formação mais meridional de corais recifais no oceano Atlântico. Dissertation, Universidade Federal de Santa Catarina

Capel KCC, Toonen RJ, Rachid TC, Creed JC, Kitahara MV, Forsman Z, Zilberberg C (2017) Clore wars: asexual reproduction dominates in the invasive range of *Tubastraea* spp. (Anthozoa:Scleractinia) in the South-Atlantic. PeerJ 5:e3873

Castro CB, Pires DO (2001) Brazilian coral reefs: what we already know and what is still missing. Bull Mar Sci 69:357–371

Chornesky EA, Peters EC (1987) Sexual reproduction and colony growth in the scleractinian coral *Porites astreoides*. Biol Bull 172:161–177. https://doi.org/10.2307/1541790

Creed JC (2006) Two invasive alien azooxanthellate corals, *Tubastraea coccinea* and *Tubastraea tagusensis*, dominate the native zooxanthellate *Mussismilia hispida* in Brazil. Coral Reefs 25:350. https://doi.org/10.1007/s00338-006-0105-x

Creed JC, Fenner D, Sammarco P, Cairns S, Capel KCC, Junqueira AOR, Cruz I, Miranda RJ, Carlos-Junior L, Mantelatto MC, Oigman-Pszczol S (2017a) The invasion of the azooxanthellate coral *Tubastraea* (Scleractinia: Dendrophylliidae) throughout the world: history, pathways and vectors. Biol Invas 19:283–305. https://doi.org/10.1007/s10530-016-1279-y

Creed JC, Junqueira AOR, Fleury BG, Mantelatto MC, Oigman-Pszczol SS (2017b) The Sun-Coral project: the first social-environmental initiative to manage the biological invasion of *Tubastraea* spp. in Brazil. Manag Biol Invas 8:181–195. https://doi.org/10.3391/mbi.2017.8.2.06

Creed JC, Pereira B, Mantelatto MC (2018) Experimental evaluation of vinegar (acetic acid) for control of invasive corals (*Tubastraea* spp.) and a review of knowledge for other aquatic pests. Biol Invas 21:1227–1247. https://doi.org/10.1007/s10530-018-1895-9

Da Silva AG, de Paula AF, Fleury BG, Creed JC (2014) Eleven years of range expansion of two invasive corals (*Tubastraea coccinea* and *Tubastraea tagusensis*) through the southwest Atlantic (Brazil). Estuar Coast Shelf Sci 141:9–16

De Paula AF (2007) Biologia reprodutiva, crescimento e competição dos corais invasores *Tubastraea coccinea* e *Tubastraea tagusensis* (Scleractinia: Dendrophyllidae) com espécies nativas. Thesis, Universidade Federal do Rio de Janeiro

De Paula A, Creed JC (2004) Two species of the coral *Tubastraea* (Cnidaria, Scleractinia) in Brazil: a case of accidental introduction. Bull Mar Sci 74:175–183

De Paula AF, de Oliveira Pires D, Creed JC (2014) Reproductive strategies of two invasive sun corals (*Tubastraea* spp.) in the southwestern Atlantic. J Mar Biol Assoc U K 94:481–492. https://doi.org/10.1017/S0025315413001446

De Paula AF, Fleury BG, Lages BG, Creed JC (2017) Experimental evaluation of the effects of management of invasive corals on native communities. Mar Ecol Prog Ser 572:141–154. https://doi.org/10.3354/meps12131

do Santos LAH, Ribeir FV, Creed JC (2013) Antagonism between invasive pest corals *Tubastraea* spp. and the native reef-builder *Mussismilia hispida* in the southwest Atlantic. J Exp Mar Biol Ecol 449:69–76. https://doi.org/10.1016/j.jembe.2013.08.017

Dommisse M, Hough D (2004) Controlling the Northern Pacific Seastar (*Asteria amurensis*) in Australia. Final Report, Australian Government Department of the Environment and Heritage. State of Victoria, Department of Sustainability and Environment

Faria-Junior E, Lindner A (2019) An underwater temperature dataset from coastal islands in Santa Catarina, southern Brazil: high accuracy data from different depths. SEANOE. https://doi.org/10.17882/62120

Fenner D (2001) Biogeography of three Caribbean corals (Scleractinia) and the invasion of *Tubastraea coccinea* into the Gulf of Mexico. Bull Mar Sci 69:1175–1189

Fenner D, Banks K (2004) Orange Cup Coral *Tubastraea coccinea* invades Florida and the Flower Garden Banks, Northwestern Gulf of Mexico. Coral Reefs 23:505–507. https://doi.org/10.1007/s00338-004-0422-x

Freire AS, Varela ARD et al (2017) O Ambiente Oceanográfico. In: UFSC (ed) MAArE—Monitoramento Ambiental da Reserva Biológica Marinha do Arvoredo e Entorno, Florianópolis, pp 159–199

Glynn PW, Colley SB, Maté JL, Cortés J, Guzman HM, Bailey RL, Feingold JS, Enochs IC (2008) Reproductive ecology of the azooxanthellate coral *Tubastraea coccinea* in the Equatorial Eastern Pacific: part V. Dendrophylliidae. Mar Biol 153:529–544. https://doi.org/10.1007/s00227-007-0827-5

Gosselin LA, Qian PY (1997) Juvenile mortality in benthic marine invertebrates. Mar Ecol Prog Ser 146:265–282. doi:https://doi.org/10.3354/meps146265

Green SJ, Dulvy NK, Brooks AML, Akins JL, Cooper AB, Miller S, Côté IM (2014) Linking removal targets to the ecological effects of invaders: a predictive model and field test. Ecol Appl 24:1311–1322. https://doi.org/10.1890/13-0979.1

Hewitt CL, Everett RA, Parker N (2008) Examples of current international, regional and national regulatory frameworks for preventing and managing marine bioinvasions. In: Rilov G, Jeffrey AC (eds) Biological invasions in marine ecosystems, pp 335–352. https://doi.org/10.1007/978-3-540-79236-9_19

Lages BG, Fleury B, Menegola C, Creed JC (2011) Change in tropical rocky shore communities due to an alien coral invasion. Mar Ecol Prog Ser 438:85–96. https://doi.org/10.3354/meps09290

Lindner A, Sarti AF et al (2017) A biodiversidade marinha das ilhas da Rebio Arvoredo e entorno. In: UFSC (ed) MAArE—Monitoramento Ambiental da Reserva Biológica Marinha do Arvoredo e Entorno, Florianópolis, pp 159–199

López C, Clemente S, Moreno S, Ocaña O, Herrera R, Moro L, Monterroso O, Rodríguez A, Brito A (2019) Invasive Tubastraea spp. and Oculina patagonica and other introduced scleractinians corals in the Santa Cruz de Tenerife (Canary Islands) harbor: Ecology and potential risks. Reg Stud Mar Sci 29:100713. https://doi.org/10.1016/j.rsma.2019.100713

Luz BLP, Capel KCC, Zilberberg C, Flores AAV, Migotto AE, Kitahara MV (2018) A polyp from nothing: the extreme regeneration capacity of the Atlantic invasive sun corals *Tubastraea coccinea* and *T. tagusensis* (Anthozoa, Scleractinia). J Exp Mar Biol Ecol 503:60–65. https://doi.org/10.1016/j.jembe.2018.02.002

Luz BLP, Di Domenico M, Migotto AE, Kitahara MV (2020) Life-history traits of *Tubastraea coccinea*: reproduction, development, and larval competence. Ecol Evol 10:6223–6238

Mafra MBM (2018) Descomissionamento de Instalações Marítimas: Perspectivas para o Brasil. Agência Nacional do Petróleo, Gás Natural e Biocombustíveis. http://www.anp.gov.br/images/Palestras/redepetro_mafra_12_07_2018. Accessed 30 Mar 2020

Mantelatto MC, Pires LM, De Oliveira GJ, Creed JC (2015) A test of the efficacy of wrapping to manage the invasive corals *Tubastraea tagusensis* and *T. coccinea*. Manag Biol Invas 6:367–374. https://doi.org/10.3391/mbi.2015.6.4.05

Miranda RJ, Nunes JACC, Mariano-Neto E, Sippo JZ, Barros F (2018a) Do invasive corals alter coral reef processes? An empirical approach evaluating reef fish trophic interactions. Mar Environ Res 138:19–27. https://doi.org/10.1016/j.marenvres.2018.03.013

Miranda R, Tagliafico A, Kelaher B, Mariano-Neto E, Barros F (2018b) Impact of invasive corals *Tubastrea* spp. on native coral recruitment. Mar Ecol Prog Ser 605:125–133. https://doi.org/10.3354/meps12731

Mizrahi D (2008) Influência da temperatura e luminosidade na distribuição da espécie invasora Tubastraea coccinea na região de ressurgência de Arraial do Cabo-RJ. Thesis, Universidade Federal do Rio de Janeiro

Mizrahi D, Navarrete SA, Flores AAV (2014) Groups travel further: pelagic metamorphosis and polyp clustering allow higher dispersal potential in sun coral propagules. Coral Reefs 33(2):443–448

MMA (2018) Plano Nacional de Prevenção, Controle e Monitoramento do Coral-Sol (Tubastraea spp.) no Brasil. https://www.icmbio.gov.br/esectamoios/images/stories/2019-08-02-Plano-Nacional-de-Prevencao-Controle-e-Monitoramento-do-Coral-sol-Tubastraea-spp-no-Brasil. Accessed 30 Mar 2020

MME (2018) Plano Decenal de Expansão de Energia 2027. http://www.epe.gov.br/pt/publicacoes-dados-abertos/publicacoes/plano-decenal-de-expansao-de-energia-2027. Accessed 30 Mar 2020

Molnar JL, Gamboa RL, Revenga C, Spalding MD (2008) Assessing the global threat of invasive species to marine biodiversity. Front Ecol Environ 6:485–492. https://doi.org/10.1890/070064

Moreira PL, Ribeiro FV, Creed JC (2014) Control of invasive marine invertebrates: an experimental evaluation of the use of low salinity for managing pest corals (*Tubastraea* spp.). Biofouling 30:639–650. https://doi.org/10.1080/08927014.2014.906583

Myers JH, Simberloff D, Kuris AM, Carey JR (2000) Eradication revisited: dealing with exotic species. Trends Ecol Evol 15:316–320. https://doi.org/10.1016/S0169-5347(00)01914-5

Oigman-Pszczol S, Creed JC, Fleury B, Mantelatto MC, Capel KCC, Meireles C, Cabral D, Masi B, Junqueira A (2017) O controle da invasão do coral-sol no Brasil não é uma causa perdida. Ciência e Cultura 69:56–59

Precht WF, Hickerson EL, Schmahl GP, Aronson RB (2014) The invasive coral *Tubastraea coccinea* (Lesson, 1829): implications for natural habitats in the Gulf of Mexico and the Florida Keys. Gulf Mex Sci 32:1

Rocha RM, Metri R, Omuro JY (2006) Spatial distribution and abundance of ascidians in a bank of coralline algae at Porto Norte, Arvoredo Island, Santa Catarina. J Coast Res III:1676–1679

RStudio T (2020) RStudio: integrated development for R. RStudio, PBC, Boston, MA. http://www.rstudio.com/

Ruiz GM, Rawlings TK, Dobbs FC, Drake LA, Mullady T, Huq A, Colwell RR (2000) Global spread of microorganisms by ships. Nature 408:49–50

Schindelin J, Arganda-Carreras I et al (2012) Fiji: an open-source platform for biological-image analysis. Nat Methods. https://doi.org/10.1038/nmeth.2019

Scyphers SB, Powers SP, Akins JL, Drymon JM, Martin CW, Schobernd ZH, Schofield PJ, Shipp RL, Switzer TS (2015) The role of citizens in detecting and responding to a rapid marine invasion. Conserv Lett 8:242–250

Seebens H, Blackburn TM et al (2017) No saturation in the accumulation of alien species worldwide. Nat Commun 8:14435. https://doi.org/10.1038/ncomms14435

Silva R, Vinagre C, Kitahara MV, Acorsi IV, Mizrahi D, Flores AAV (2019) Sun coral invasion of shallow rocky reefs: effects on mobile invertebrate assemblages in Southeastern Brazil. Biol Invas 21:1339–1350. https://doi.org/10.1007/s10530-018-1903-0

Simberloff D (2003) How much information on population biology is needed to manage introduced species? Conserv Biol 17:83–92. https://doi.org/10.1046/j.1523-1739.2003.02028.x

Simberloff D, Martin JL et al (2013) Impacts of biological invasions: what's what and the way forward. Trends Ecol Evol 28:58–66. https://doi.org/10.1016/j.tree.2012.07.013

Soares M, Davis M, Bastos P, Carneiro M (2016) Northward range expansion of the invasive coral (*Tubastraea tagusensis*) in the southwestern Atlantic. Mar Biodivers 48:1651–1654. https://doi.org/10.1007/s12526-016-0623-x

Wickham H (2016) ggplot2: elegant graphics for data analysis. Springer, New York. ISBN: 978-3-319-24277-4. https://ggplot2.tidyverse.org

Williams SL, Grosholz ED (2008) The invasive species challenge in estuarine and coastal environments: marrying management and science. Estuaries Coasts 31:3–20. https://doi.org/10.1007/s12237-007-9031-6

Wittenberg R, Cock MJ (2001) Invasive alien species. A toolkit of best prevention and management practices. Publication of GISP, CAB International, Wallingford

Zaitsev YP (1992) Recent changes in the trophic structure of the Black Sea. Fish Oceanogr 1:180–189