Journal for Nature Conservation 24 (2015) 17–23

Contents lists available at ScienceDirect

Journal for Nature Conservation

journal homepage: www.elsevier.de/jnc

Demographics and dynamics of two restored populations of the threatened reef-building coral *Acropora cervicornis*

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ARTICLE INFO

Article history:
Received 23 September 2014
Received in revised form
22 December 2014
Accepted 9 January 2015

Keywords:
Acropora cervicornis
Coral transplantation
Coral reefs
Fragment stabilization
Reef restoration

ABSTRACT

Acropora cervicornis is one of the principal reef-building organisms in the Caribbean; it is also considered one of the most threatened coral species. Due to its ecological importance and critical status it is the focus of many restoration and management initiatives. However, studies that quantitatively measure the efficacy or feasibility of these efforts are mostly lacking. In this study, nursery-reared fragments of *A. cervicornis* were transplanted to two reefs in Puerto Rico as part of a reef rehabilitation program, and their survival, growth, and branch production were measured for a year. We also evaluated the effect of this restoration on the dynamics and viability of the fragment populations by means of a simple model. Survival of outplanted fragments surpassed 60%. Colony growth rate varied between 0.20 ± 0.18 and 0.29 ± 0.21 cm d^{-1} (mean ± SD) whereas branch production ranged between 7.02 ± 5.72 and 11.86 ± 7.06 (mean ± SD) branches per fragment per year. Survival did not vary considerably with respect to fragment size. In contrast, large fragments (≥25 cm) grew faster and tended to produce more branches than smaller ones. Model simulations indicate that (1) in the absence of recruitment, and without any subsequent human intervention, restored populations will decrease below a quasi-extinction level of 25% of the initial population size after just 3 years and (2) transplanting at least 20 colony fragments per year (12% of initial population) is sufficient to keep the restored populations above the 25% threshold. We conclude that *A. cervicornis* may be a feasible species for restoration projects given sustained human intervention and that transplanting fragments of at least 25 cm to reefs is an effective restoration protocol that requires minimum effort to maintain a viable restored population of this key reef-building coral.

Introduction

Since the late 1970s Caribbean coral reefs have lost approximately 80% of their coral cover (Gardner et al. 2003); putting at risk not only the functioning of this ecosystem, but also the ecological services that coral reefs provide to millions of people around the region. Scientists, managers, and local stakeholders have been looking for alternatives to prevent and/or reduce coral reef degradation by restoring depleted populations of major reef-building corals. *Acropora cervicornis* is one of the most important reef-forming species in the region not only because it is among the fastest growing corals (Tunnicliffe 1981; Knowlton 1992) but also because it has the capacity to bind carbonate rubble and other reef materials (e.g. clastic sediment) that contribute to the stabilization of the reef framework (Gilmore & Hall 1976). In addition, because of its branching morphology, *A. cervicornis* provides recruitment habitat for many commercial fishes (e.g. groupers and snappers). This species, however, has collapsed along its geographical range due to many biotic stressors (e.g. diseases, predator outbreaks), physical disturbances (e.g. hurricanes, increase in sea water temperature), and anthropogenic impacts (e.g. water pollution, ship groundings) (Miller et al. 2002). As a response to this situation, *A. cervicornis* has been listed as a threatened species under the United States Endangered Species Act (NMFS 2006). Since then it has been the focus of several population restoration projects (reviewed by Young et al. 2012).

An example of the efforts to promote the recovery of this coral has been the development of different fragment-based aquaculture techniques (e.g. A-frames, hanging ropes; reviewed by Young et al. 2012). The ultimate goal of such actions is to enhance the survival and growth of colony fragments (collected from wild populations) that later will be transplanted into selected sites (Edwards 2010; Griffin et al. 2012). This human-assisted asexual propagation

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http://dx.doi.org/10.1016/j.jnc.2015.01.001
1617-1381/

has been put into practice in at least 12 localities in the Wider-Caribbean (Young et al. 2012).

Although culturing has been successful in increasing the number of colonies available for restoration programs (Hernández-Delgado & Suleimán-Ramos 2014; Hernández-Delgado et al. 2014) little is known about the demographic performance of the fragments once they have been transplanted into reefs. Moreover, the few published studies (e.g. Shinn 1966; Garrison & Ward 2008; Hollarsmith et al. 2012) have been modest in that (1) they only considered one aspect of the life-history transitions of coral transplants (growth or survivorship), (2) the sample sizes have been small (i.e. $n \leq 45$; Hollarsmith et al. 2012) and (3) none of them considered the effect of these efforts at the population level (i.e. population growth rates, recruitment). This lack of knowledge has generated debate as to the suitability of *A. cervicornis* for reef restoration. For example, while the study by Hollarsmith et al. (2012) concludes that the species is a good candidate for restoration projects because none of the coral outplants ($n = 45$) died during their 1 year study; Garrison and Ward (2008), who followed the fate of storm-generated fragments transplanted to a reef in Saint John, US Virgin Islands, argue otherwise because all transplants ($n = 15$) died within 5 years.

The scarcity of information on the demographic success of transplanted fragments, both at the colony and population level, limits our assessment of the efficacy of transplantation as an effective restoration tool. In this study, we quantified the survival, growth, and rate of branch production of *A. cervicornis* fragments transplanted to two reefs in Puerto Rico. We considered fragments of different sizes to measure the effect of size on the demographic performance of transplants. In addition, we evaluated the effect of this restoration on the dynamics and viability of the fragment population by (1) projecting the restored population into the future, and (2) determining the number of transplants necessary to maintain the restored population over a threshold of 25% of the original population size. Results of this study will help coral ecologists and managers in determining whether coral transplantation is a feasible approach for reef restoration.

Methods

Study sites

The study was carried out at Punta Soldado (PSOL) and Tamarindo (TAM) reefs, Culebra Island, Puerto Rico (Fig. 1). Both sites have an active nursery and restoration project run by the local NGO – Sociedad Ambiente Marino (SAM). The sites are characterized by low wave energy, and relatively good water quality (mean horizontal water transparency 7.7 m, Ruiz-Ramos et al. 2011). No differences in temperature or water transparency are evident between sites (Hernández-Delgado & Suleimán-Ramos, Sociedad Ambiente Marino, unpublished data). The area of transplantation is of low topographic relief (Mercado-Molina et al. 2014a) with a consolidated bottom where macrofauna is visually dominated by *Millepora* spp., *Diploria* spp. and *Porites* spp. However, macroalgae cover is higher at PSOL (≈37%) compared to TAM (≈17%).

Fragment transplantation

A total of 170 fragments were transplanted at each site by stabilizing them directly to the substrate using concrete nails of 8 cm (length) and plastic cable ties (see Garrison & Ward 2008; Hollarsmith et al. 2012). All fragments were collected haphazardly from healthy colonies growing in situ nurseries by clipping at the base of branches projecting out of the principal branch. No more than two fragments were collected from the same donor colony. Transplantation took place at a depth of 3–4 m during May 2011. All fragments were identified with a numbered tag tied to their respective nails (no tag was in direct contact to the transplant). Fragments were categorized in two size classes (small <25 cm and large ≥25 cm) following Mercado-Molina et al. (2014b).

Survival

Survival (live tissue >0%) was monitored 1 month after transplantation and every 3 months thereafter for 1 year (2011–2012). Kaplan–Meier Survival Analysis was used to compare fragment survival schedule between the studied sites and size classes.

Growth and branch production

Growth rate of fragments was measured as the change in daily linear extension (final length – initial length/total number of days) and expressed as cm/day. Initial and final sizes were measured as the sum of the linear lengths of the live tissue portions of all branches, subtracting partial mortality from the total size when appropriate. Length of all branches was obtained by analyzing photographs (taken in situ scale-by-side) using the software Coral Point Count with Excel extensions (CPCe) (Kohler & Gill 2006). Fragments were photographed from different angles to ensure that all branches could be appreciated in their full extension. The suitability of this approach (image analysis) as estimator of actual fragment size for *A. cervicornis* was demonstrated by Mercado-Molina et al. (2014b). Growth rates were calculated for those fragments that survived to the end of the study, 104 and 112 at PSOL and TAM, respectively. Mann–Whitney *U*-test was used to compare overall growth rates between sites as well as to compare growth between size classes within each site because data were not normally distributed. Branch production, calculated as the number of new branches produced by fragments, was analyzed as explained earlier for rates of growth.

Sexual recruitment and natural fragmentation (asexual recruitment)

Asexual and sexual recruitment were monitored at a quarterly basis for 1 year in thirty 1 m^2 permanent quadrats randomly placed along three 10 m × 1 m transects separated 10 meters from each other. Following Tunnicliffe (1981) and Knowlton et al. (1990) sexual recruits were defined (a priori) as a small crust showing a round or ellipsoidal morphology and measuring less than 10 cm in height. Asexual recruits were differentiated from sexual recruits by looking at (1) its orientation (horizontal vs. vertical), (2) signs of obvious fragmentation, and (3) fragment size (>10 cm in total linear length) (Tunnicliffe 1981; Knowlton et al. 1990). Rates of natural fragmentation were also measured in situ by identifying scars within each colony fragment (Tunnicliffe 1981) as well as by counting the number of broken (missing) branches when comparing images between surveys.

Population modeling

The restored populations were projected for 15 years by iterating Expression (1), where N is the number of fragments at time t and 3 months into the future ($t+1$), and λ is the growth rate of the population. During the iterations, λ_t was randomly drawn from the set of λ's that were calculated for each of the 3 months intervals (four in total) using the expression N_{t+1}/N_t. Initial population size was set as 170 fragments. To simulate population trajectories under different outplanting scenarios, we followed the same procedure as above using Expression (2), where R represents the input of new fragments to the population every 3 months. Simulations started with the equivalent of 10 fragments per year, increasing