# Biological consequences of quick fixes in coral reef restoration

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**Abstract.**

Restoration projects can run the risk of simply transforming one degraded habitat into another anthropogenically altered habitat. While aesthetically different and possibly of higher socio-economic value, the new state of alteration may or may not have any added ecosystem function.

Electro-mineral accretion has been used in many restoration and artificial reef projects as both a means to secure scleractinian coral transplants to substrate and to promote growth. The applied electric current induces accretion of calcium carbonate onto a metal structure and is assumed to have a positive effect on growth of attached corals. Previous research regarding this process has dealt with quantity rather than quality of attached corals. This study attempts to determine if the higher skeletal growth rates experienced by the coral are at the expense of a trade-off with some other aspect of their life history – such as reproductive ability or fitness.

The capacity for restoration efforts on the surrounding reef is governed by the viability of the transplants. In this study the functional biology of transplants under electro-mineral accretion conditions are examined to ascertain if the corals growing on such structures are biologically viable and contribute to restoration efforts. Comparisons in the fecundity, polyp density, skeleton density, and growth rates were conducted on the Pocilloporid coral Stylophora pistillata in a field site in Lombok, Indonesia. Corals treated with electro-mineral accretion were found to have significantly lower fecundity rates as well as significant differences in polyp density, skeleton density and growth rate to the naturally growing control colonies.

**Key words:** Restoration, electric reefs, artificial reef, coral transplant

## Introduction

A large portion of the world's fringing reefs in developing countries have become anthropogenically degraded through destructive fishing and reckless marine tourism practices to a point where natural recovery under present conditions is unlikely. Many argue that the resulting unstable substrate should be left in its degraded state and allow natural processes to recover and adapt to the change. While this may be the case, the dilemma arises as to if time-frames involved in these natural recovery processes are beyond that to which the dependant coastal communities of people can afford. Social considerations must be factored into this period of reef absence. Is the resulting increase in erosion of economically valuable shoreline and collapse of localized reef dependant fishery something that coastal communities are prepared to deal with during the time of natural reef recovery? Many possible options can be assessed for restoring ecosystem services, one option being to remove all human pressures and leave the area alone for natural recovery. At the other end of the continuum from leaving the reef alone are the options of engineering artificial breakwaters to deal with the issue of shoreline erosion and an economy shift away from fisheries. However, these rather drastic options also present obstacles. The intermediary solution of restoration might better enhance the natural recovery of such reef ecosystems whilst sustaining the dependant human population's uses.

*Restoration ecology and artificial reefs*

Artificial reefs have been widely used as a tool for marine ecosystem restoration and a plethora of literature has been written in regards to methods for implementing these artificial reef structures. The structural complexity of reef through the presence of crevices and topographical relief contributes significantly to species composition and biological productivity (Kohn & Leviten 1976; Gratwicke & Speight 2005). Artificial substrate has also been found to have similar effects (Chandler et al 1985; Rilov & Benayahu 1998). The driving forces behind implementing such structures are the economic costs that have resulted from an anthropogenically denuded marine environment. Artificial reefs are therefore seen as a compensation method for restoring a devalued resource. Historically whether or not restoration efforts take place often has less to do with loss of ecosystem function but more fundamentally driven by economic losses to dependant industry. Artificial reefs are installed to perform either one or a combination of four broad functions: Restore biodiversity & ecosystem function (Clark & Edwards 1999; 1994; Pratt 1994); Fisheries enhancement or recovery (Seaman 2007; Pickering & Whitmarsh 1997; Grossman et al 1997); shoreline protection and erosion prevention (Gardner et al 1997; Ranasinghe & Turner 2006); Dive tourism (Stolk & Markwell 2007; van Treek & Schuhmacher 1998; Brock 1994). Artificial reefs are able to facilitate the recovery of denuded ecosystems by restoring topographical complexity of sites which have been altered (Fox et al 2005; Clark & Edwards 1999; Gardner et al 1997), stabilizing substrate to allow settlement of coral larvae (Clark & Edwards 1994; Fox 2004), providing a seed-bank of larvae for recruitment-limited sites (Epstein et al 2001; Morse 2000; Yap 2000), and increasing percent cover of habitat forming species (Shafir et al 2006; Sleeman et al 2005; Raymundo 2001; Edwards & Clark 1998). Although artificial reefs can aid in restoring some functions of a coral reef ecosystem, they can in no way solely recreate a healthy ecosystem. Therefore they should be used as a single tool within a broader restoration and reef management plan.

Often the inherent value of a "pristine" ecosystem over a degraded one is not case enough in itself to warrant protection or restoration - unless the loss in ecosystem function also equates to economic losses. Several economic models have attempted to place economic value pristine coral reefs in an attempt to make a case for restoration (Spurgeion & Lindahl 2000; Sumaila 2004). Human interactions create changes in the functions of ecosystems. Countless examples of changes in entire ecosystem functions have taken place due to overfishing during the last century in what Pauly et al (1998) describes as "fishing down the food webs." First large carnivores are targeted, then progressively smaller fish as higher trophic levels are over-exploited. The removal of higher trophic levels creates shifts in entire community composition and relative abundances. These biotic changes inevitably lead to altered interactions with their dependant environment. Phase shifts in coral reef ecosystems have been documented as negative human-derived impacts reduce species diversity and community structure (Bellwood et al 2004; Done 1992). For example coral reef environments which have been heavily disturbed by nutrient loading and overfishing can shift from coral to a macro-algae dominated state.

Human uses must be incorporated into restoration and management plans. Segregating the issues of ecosystem health and human resource use will continue to perpetuate the current situation of the earth's natural areas – wherein areas excluding human uses are patch-worked haphazardly amid expanses of varying levels of resource over-exploitation. Establishing networks of protected areas where anthropogenic activities and impacts are excluded is important (Halpern & Warner 2002; Soule 1991). However advocating this method as the only means to conservation is impractical because of the vast majority of areas laying beyond protection. Additionally there are ethical problems associated with excluding resource use from communities dependant on their local marine environment. Such a polarized approach has the potential to cause communities to either not adopt or not adhere to management protocols within these restricted areas. Many protected areas in developing nations undergo a functional collapse after the founding organization (NGO) leaves because of the inability of the newly re-managed marine resource to fulfill the nutritional and economic requirements to the community. Often unrestricted activity simply continues outside the boundaries of the protected area and the same anthropogenic pressures are translocated and even intensified in adjacent areas. The majority of coral reefs are located in coastal areas of the developing world. In such areas, an ever increasing coastal population drives intensified resource use due to increasing demands on a scarce resource. This phenomenon has led to Malthusian overfishing in many areas (Russ 1991; Pauly 1990). Destructive fishing practices, although technically illegal in almost all countries, is currently causing irreversible damage to coral reefs (Fox 2004, McManus et al 1997) as individual fishermen/fleets race for a bigger slice of a diminishing resource. The economic doctrine of scarcity dictates that a resource becomes increasingly valuable (and therefore less likely to be conserved) as supply diminishes and demand increases. When resources are depleted and the dependant communities can no longer be supported economically, communities turn to quick fixes for restoring resource capital in their over-exploited ecosystems. Restoration projects that are implemented by such means mistakingly view healthy ecosystems as replaceable (as in the case of Yeemin et al 2006) rather than managed for conservation. A coral reef ecosystem therefore runs the risk of being seen as a renewable resource which can be exploited and rebuilt. Clearly coastal management that either allows unregulated use or completely excludes use, rather than sustainably manages resource use within the area, is destined for failure.

*The study site*

This paper analyses a coral reef restoration project on the small island community of Gili Trawangan in Lombok, Indonesia. The community whose roots lie traditionally in fishing now supports an ever growing tourist industry, with one of the primary facets being dive tourism. The Gili Islands are labeled as a Marine Natural Park under the Directorate of Forest Protection and Nature Conservation by the Government of Indonesia, however, there is little local community acceptance or acknowledgment of marine park management protocols (pers. obs.; Satria et al 2006; Graci 2006). The exponentially growing human population in both Lombok and the Gili islands has led to greater and greater strains on coastal resources. Although officially illegal as of 1985 [Indonesia Fisheries Law no. 9/1985] the use of destructive dynamite fishing, muroami-netting (dragging nets across reef flat to herd fish) and chemical-poison fishing are common practice in the area. Interestingly, dynamite fishing has recently been greatly reduced in the local area surrounding Gili Trawangan not because of adherhance to marine park management protocols, but instead because of enforcement by members of the now more economically dominant dive tourism industry. A local security task force (SATGAS) have been employed by the island's Eco-Trust (an organization of tourism entrepreneurs and local village figure heads) to enforce fishing agreements in the local area. While this may at first glance appear as a victory to conservation driven by the ecotourism industry, one must look at a broader view to assess the validity of that claim. The more complex the system of resources, the more difficult it is for resource users to agree on rules addressing these externalities (Dolsak & Ostrom 2003). The not-in-my-backyard approach to marine conservation is destined to have catastrophic effects on both the environmental condition and social interactions of neighboring communities. Not only is fishing pressure intensified in neighboring areas due to more fishers using a smaller area, but demand for fish products also increases from tourism services. Additionally, marginalization of the fishing community has taken place as fishers feel that their property rights are being undermined due to loses in operational area (Satria et al 2006).

The underwater fields of unconsolidated rubble along the reef slope and reef flats which are remnants of destructive fishing practices still remain a dominant feature of the fringing reefs around the Gili islands. Although blast fishing has essentially been eliminated since the late 1990's in the local area, a lack of reef recovery with continued widespread coral mortality has happened since that time. In a study by Fox (2004) on blast-fished coral reef sites in Komodo, Indonesia, the failure of corals to recover from the rubble is due to post settlement mortality. The unstable nature of the substratum in blasted sites was attributed as the fundamental reason for mortality in coral recruits. Brown and Dunne (1988) in a study on areas of mined coral reefs in the Maldives found that such areas showed no recovery in a time period of over 25 years. The lack of coral recovery was attributed to limited topographical relief and the highly mobile sediment reducing settlement and smothering juvenile corals. The findings from both of these studies are reinforced by the highly cited experimental study by Clarke & Edwards in the Maldives on restoration of blasted and mined coral reef flats. Similarly to both Fox(2004) and Brown & Dunne (1998), Clarke & Edwards findings indicated that such blasted reefs will not recover without stable substrate (Clarke & Edwards 1995).

The second major impact on the reefs surrounding Gili Trawangan is the rampant coastal development and population increase due to a growing tourism industry. Although the tourism industry has arguably brought about positive benefits to the area by halting destructive fishing practices and bringing economic prosperity to an area of traditional sustenance fishing. Tourism must be viewed as a double edge sword. Beach front development and building of disjointed retaining walls is altering the natural coastline and sediment transport regimes. A lack of regulation and infrastructure has meant an inability to deal with issues of waste disposal and sewage treatment. High levels of algae growth associated with nutrient loading are seen on the reefs surrounding Gili Trawangan.

Like many tourist destinations in developing countries, the high level of foreign investment into the tourism industry coupled with Indonesia's unstable economic climate creates a lack of community cohesion. As with any business venture in high risk economic environments, tourism operators strive to maximize their returns in the short-term while minimizing investment. Therefore long-term sustainability of both the industry and environment upon which it is ultimately dependant takes a lower precedence to quick-fixes to immediate problems.

Any attempt to restore a coral reef ecosystem needs specifically defined management goals. In order to determine if a reef community has been successfully restored, aims must be set for a target endpoint measurable by some means of indicator (Edwards & Gomez 2007). For example if a coral reef community is to be "restored" then some historical baseline measurement must be set as the target for which the community will be restored to (Hawkins et al 1999). Whereas if a community is to be "rehabilitated" then the aim is to increase the ecological value by enhancing structural and functional characteristics of the degraded reef (Edwards & Gomez 2007). The target endpoint in the management plan of a rehabilitation project may not be the complete restoration to an exact historical state, but rather some defined state which has an improved ecosystem health to its current status. Management efforts of both terrestrial and marine restoration projects are increasingly looking at restoring ecosystem functions rather than states (Palmer et al 2004). While historically restoration efforts primarily focused on restoring the assemblage of foundation species, emerging practices now incorporate dynamic physical, chemical, and biological processes – including restoring complexity of food webs, habitat heterogeneity, and environmental health. The historical bottom up approach of recreating all the pieces of an ecosystem is far less effectual than the top down approach of restoring ecosystem health and biological processes. It is therefore important that the organisms used as foundation species and the techniques employed restore the natural ecosystem functions rather than simply quickly recreate the aesthetics a coral reef. In a review by Baine (2001) of artificial reefs being used for reef restoration projects, only 50% of projects were found to be meeting the objectives for which they were implemented. It is therefore important that management targets must be set from which performance indicators can be drawn. In Gili Trawangan, electo-mineral accretion artificial reefs were deployed by the Gili

Ecotrust (funded by tourism operators). The primary purpose of deploying these structures was for restoring degraded reef areas to a natural state (Appendix: Interview A – F. Perry), and attempting to recreate the historical live fringing reef to protect against shoreline erosion (Appendix: Interview B - A. Walker).

*Electro-mineral accretion process*

The electro-mineral accretion (EMA) process as applied to artificial reefs is outlined by Hilbertz & Goreau (1996) and Schumacher & Schillak (1994). This process has been utilized in several coral reef restoration projects (Van Treek & Schuhmacher 1997; Schuhmacher et al 2000; Henderson 2002; Sabater & Yap 2002) as well as tested in experimental aquaria environments (Eggeling 2006; Charko 2005; Lacharmois 2005; Egan 2004). Electro-mineral accretion has been used as both a means to secure scleractinian coral transplants to substrate and to promote growth. The artificial reef structures act as an electrolytic cell. They consist of an iron frame acting as a cathode upon which nursery corals are attached. An anode is placed in the direct vicinity of the frame, with a weak DC electric current running to the set up. Seawater acts as the conductive electrolyte solution between the anode and cathode. The applied electric current causes the deposition of minerals such as CaCO3 , Mg(OH)2, CaSO4, NaCl on the iron structure (Meyer & Schuhmacher 1993; Hilbertz et al 1996). This is due to an oxidation reaction taking place on the negatively charged cathode of the electrolytic cell. The product of the oxidation reaction which is of benefit to corals is CaCO3 as this is the compound which forms the skeletons of scleractinian corals. It accretes through the following redox reaction at the cathode:

Oxidation: $CO_3^{2-} \rightarrow CO_3 + 2\,e^-$

$$Ca^{2+}_{(aq)} + CO_3^{2-}{}_{(aq)} \rightarrow CaCO_{3\,(s)}$$

The EMA process is claimed to have a positive effect on growth of attached corals (Hilbertz & Goreau 1996). Research about these EMA artificial reef structures so far has primarily dealt with only the ability and speed with which attached corals can grow (Sabater & Yap 2002, 2004; Van Treek & Schumacher 1999). Previous studies have shown that scleractinian corals have significantly higher linear extension rates when subjected to electro-mineral accretion (Sabater & Yap 2002; 2004; Eggeling 2006). Sabater & Yap (2004) found significantly higher longitudinal growth rate rates of their test subject Porites cylindrical on EMA treatments compared to controls. They attributing the difference to an increase in mineral ion concentration within the vicinity of transplants attached to the cathode causing a diffusional influx of ions into the coral polyp's coelenterons, increasing the availability of ions for calcification.

Typically in any organism, a finite amount of energy is available which must be allocated in such a way that gives the organism the highest fitness in the particular set of environmental conditions which are exerted upon it. Energy must be allocated by the competing demands of a colony which include: growth, reproduction, and maintenance (Perrin & Sibly 1993). Previous studies on EMA have only looked at the effects on growth; but not at the effects on reproduction. Perrin & Sibly summarize in their model of energy allocation that selection pressure of which process the organism invests energy into is dependant on the pressures exerted on it. If all pressures are equal energy allocation is optimized in what they refer to as "balanced growth." Variations in pressures then change the proportions of energy investment into growth vs. reproduction (Perrin & Sibly 1993). The trade off between allocating resources between growth and reproduction in order to optimize an organism's fitness can be expressed in terms of basic microeconomics. The Opportunity Cost Doctrine put forward originally by Friedrich Von Wieser in 1876 states that there are limits to production, and that efficiency must be optimized by allocating resources between a combination of alternative products (Buchanan 1979). If resources are spent on the production of 'Product A' then there are fewer resources available for the alternative production of 'Product B'. Analogous to this economic model, modular organisms such as coral also face a production trade-off between allocating resources into colony growth in terms of linear extension of branches and polyp replication, or allocating resources into production of reproductive cells (Rinkevich 1996).

If corals are experiencing higher growth rates, then the logical hypothesis would be that colony fecundity would be compromised resulting in a lower reproductive output. No research yet has looked at if the higher skeletal growth rates experienced by corals subjected to EMA result in the biological expense of a trade-off with some other aspect of their life history – such as reproductive ability or fitness. This research project attempts to fill this research gap by determining the effects of EMA on coral reproduction. Increasing percent cover of foundation species or habitat forming species (coral) is often seen as a primary goal in restoration efforts as they provide key habitat and facilitate colonization by conspecifics (Temperton & Hobbs 2004) Although much debate has taken place in terrestrial systems about acceleration of restoration processes through non-natural means (Hilderbrand et al 2005), little research thus far has looked at ecological consequences of quick-fixes in coral reef restoration. The capacity for restoration efforts on the surrounding reef is governed by the viability of the transplants.

*Research Objectives*

In this study the functional biology of transplants under electro-mineral accretion conditions are examined to ascertain if the corals growing on such structures are biologically viable and contribute to restoration efforts. Comparisons were made on the physiology of Stylophora pistillata growing under electro-mineral accretion treatments and under natural conditions in a field site in Lombok, Indonesia to determine the effect of the electro-mineral accretion process on sexual reproduction and asexual budding of polyps.

The overall goal of this study is to determine if electro-mineral accretion artificial reefs are and effective method for coral reef restoration in Gili Trawangan. To determine this, the following question was posed: Is there a significant difference biologically between corals (Stylophora pistillata) growing on EMA structures and those found growing under natural conditions, in terms of fecundity, polyp density, skeletal density, and growth.

**Material and Methods**

*Description of Study Site*

The Gili islands (Gili Trawangan and Gili Meno) are located off the coast of Lombok in the province of Nusa Tenggara in Indonesia (figure 1). The islands are a popular tourist destination with the tourism industry focusing around scuba diving and snorkeling on the fringing reefs. Both of the islands in this study have a small village with boat anchorage areas in front, and have been experiencing rapid development.


*Figure 1:* Aerial photo of Gili islands with arrows indicating location of study sites.

The study site on Gili Trawangan was located directly in front of the village along the algal and rubble dominated reef slope. The five EMA structures at the study site were located at similar depths between 5-7m and staggered every 50-100m with the electric cables supporting the structures running directly out from shore across the reef flat to each structure. The structures had been installed in November 2006 and had been operating for just over one year. One EMA structure was used as a non-electric treatment control supplied with no electricity, as the cable had been broken shortly after installation one year ago. The naturally growing control colonies were located in the surrounding area located 20-100m away from structures on same reef slope and exposure at depths between 5-7m (see figure 2). The study site at Gili Meno was similarly located in front of the village at a depth of 5-7m with a single EMA structure used and a naturally growing control area located approximately 30m away.

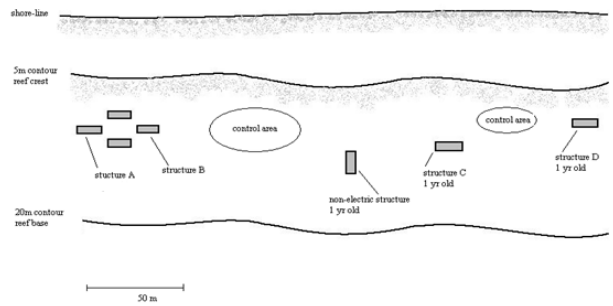



*Figure 2:* Diagram of Gili Trawangan study site outlining locations of EMA structures and control areas.

*Data collection and experimental methods:*

Stylophora Pistillata colonies were identified and tagged on each of the electric reef structures and in surrounding area to be used. All colonies used were of similar size (~30cm in diameter) because previous studies have found variations in fecundity of S. pistillata according to colony size (Hall & Hughes 1996)

<u>On Gili Trawangan site:</u>

The "electro-mineral accretion" treatment variable was replicated 4 times as represented by each of the structures. The naturally growing control colonies were used to make comparisons between treatment conditions and non-treatment conditions. Additionally a non-electric control structure was used to test for error due to transplantation stress of corals growing on the artificial reef structures. Therefore any differences can be attributed to presence or absence of electro-mineral accretion treatment. Four replicate colonies were used on each of the 4 EMA structures, 12 naturally growing control colonies, and 4 colonies on the non-electric control structure.

<u>On Gili Meno site:</u>

Four replicate colonies from the EMA structure and 4 replicate naturally growing colonies used. From each of the colonies, 4 replicate branches sampled. Branches were taken from similar spots on each colony -periphery of colony close to cathode/structure.

*Part A: Fecundity*

The tests for fecundity were performed under the following experimental designs:

<u>Test #1</u> (samples collected from Gili Trawangan island 16-18 Dec 2007; structures A-D have received intermittent electricity since installation):

4 x EMA structures [4 x colonies (4 x branches)]
1 x non-electric structure [4 x colonies (4 x branches)]
1 x natural control [12 x colonies (4 x branches)]

Test #2 (samples collected from Gili Meno island 11-13 Jan 2008; structures received constant electricity since installation):

1 x EMA structure [4 x colonies (4 x branches)]
1 x natural control [4 x colonies (4 x branches)]

i) Decalcification of coral samples

Samples of Stylophora pistillata were collected from the field and decalcified according to the technique outlined by Willis (2007). Specimens fixed in seawater-formaldehyde (formaldehyde 37%, 5mL; seawater, 45mL) for 2 days in 50mL sample jars. Specimens were then transferred into weak decalcifying solution (hydrochloric acid 30%, 10mL; formaldehyde 37%, 100mL; water 1000mL). Low acid concentration used initially to minimize tissue disruption due to effervescence as the skeleton dissolves. Containers were loosely capped to avoid pressure build-up. After 2 days decalcifying solution was replaced with a stronger solution (hydrochloric acid 30%, 50mL; formaldehyde 37%, 100mL; water 1000mL). Then after an additional 2 days decalcifying solution replaced again with increased acid strength (hydrochloric acid 30%, 100mL; formaldehyde 37%, 100mL; water 1000mL). Decalcification process deemed complete when all skeletal matter was dissolved and tissue floated in solution. After 2 weeks in decalcification solution, specimens rinsed in fresh water to remove acid, then stored in 70% ethanol.

ii) Counts of brooded larvae

Tissue samples were removed to count number of polyps with brooded larvae (Rinkevich & Loya 1979). Tissue was dissected and opened flat on petri dish, then polyp were examined in the area between 1-2cm from branch tip in order to maintain consistency and eliminate error due to examining pre-reproductive polyps located in growth region of tip. Ten polyps were randomly chosen within this region and examined for presence or absence of brooded larvae to give a fecundity measurement as percentage of fecund polyps. (see figure 3)

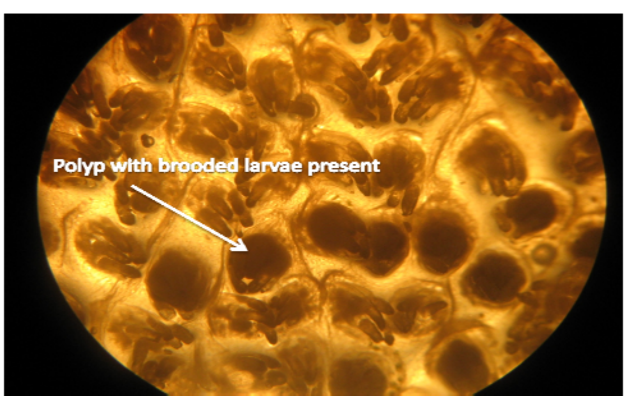



*Figure 3a:* Tissue from decalcified Stylophora pistillata sample under 10 x dissecting microscope.

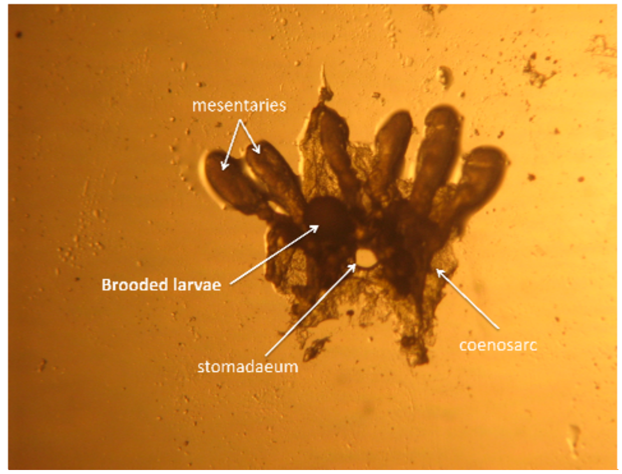



*Figure 3b*: Stylophora pistillata polyp with brooded larvae highlighted.

*Part B: Polyp density (# polyps / area)*

To determine density of polyps, the number of coralites within a standardized square 4mm x 4mm (16mm2) clear plastic overlay was done on samples before skeletons were decalcified. A pilot study was conducted with plastic overlays measuring both 1cm x 1cm and 4mm x 4mm. The larger square was found to be too large and inaccurate due to small diameter of coral branches. To allow for increased precision in counts, polyps which intersected right & bottom sides of the square overlay were counted, while polyps intersecting left & top sides of the square were not counted. Counts were replicated in 5 randomly placed spots on branches within 0.5 – 1.5 cm from branch tip. Data from structures on Gili Trawangan were grouped together into an "intermittent electricity treatment" and the structure on Gili Meno was treated separately as a "constant electricity treatment" for statistical analysis.

3 x Intermittent electricity EMA structures
[4 x colonies (4 x branches)]
1 x constant electricity EMA structure
[4 x colonies (4 x branches)]
1 x control
[16 x natural colonies (4 x branches)]

*Part C: Skeleton density*

Skeleton density measurements were made on samples from Gili Trawangan EMA treatments and natural controls. Samples of Stylophora pistillata were left overnight in 50% hydrogen peroxide solution until was tissue dissolved and removed from skeletons. Samples were then rinsed in fresh water and let dry in the sun for 2 days. Dry weight measured using electronic scale graduated to 0.01g. Volume measured using a 25mL graduated cylinder recording water displaced by skeleton sample. Density measurement calculated as dry weight of skeleton over volume of water displacement (g/mL).

4 x EMA structures (4 x colonies)
1 x control (12 x natural colonies)

*Part D: Growth measurement*

Coral growth was determined by staining the skeleton with a dye at one point in time, then measuring growth of new white skeleton after a period of time. A pilot study was conducted in early December 2007 using both whole colonies and branches of Stylophora pistillata to determine the best method for staining skeletons. These results indicated that a method similar to the one used by Rinkevich & Loya (1983) in their study on Stylophora pistillata would work best for this project. A solution of Alizarin Red stain was put into plastic wash-bottles. Single branches on a colony were stained in late afternoon by: placing plastic bag over the branch and filling bag with alizarin red using wash bottle, then securing plastic bag to coral branch using plastic zip-tie to ensure dye does not diffuse out of bag. Bags attached to coral branch were left over night to allow time for dye to be absorbed into tissue and deposited on skeleton. The following morning, bags were carefully removed avoiding tissue abrasion and plastic zip-ties were left attached to tag branch.

In week 1 (Dec 19-21) replicate control & treatment colonies were stained with alizarin red. Two species chosen for growth measurements: Stylophora pistillata and Acropora formosa.

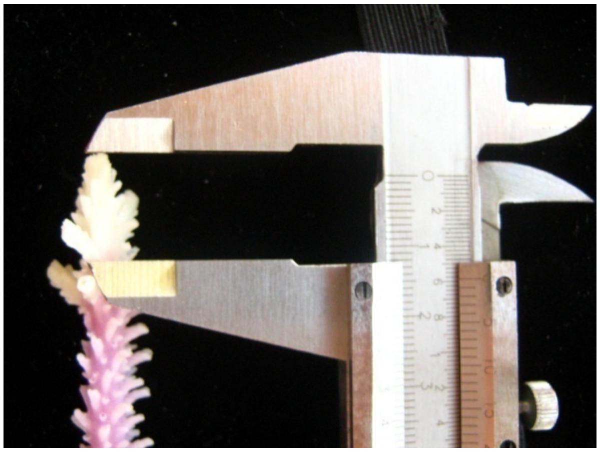
Acropora formosa:
- Naturally growing controls – 6 x colonies
- Structure A – 3 x colonies
- Structure B – 3 x colonies
- Structure C – 3 x colonies

Stylophora pistillata:
- Naturally growing controls – 4 colonies
- Structure C – 3 x colonies
- Structure D – 3 x colonies

*Figure 4a*: Staining a naturally growing colony with Alizarin red for growth measurements.

*Figure 4b:* Naturally growing colony of Acropora formosa with plastic bag containing Alizarin red stain.

In week 5 (Jan 17) the tagged branches which had been stained weeks earlier were removed and placed into hydrogen peroxide 50% solution to dissolve tissue. After 24 hours specimens were rinsed in water to remove remaining tissue and hydrogen peroxide. Growth over 5 week time period since staining was represented by white area of new skeleton extension past purple stained skeleton (see image 5). Linear extension of skeleton was measured using fine scale vernier calipers. Acropora samples measured from purple line in skeleton to tip of axial polyp. Stylophora samples measured from purple line in skeleton to branch tip.



## Results

*Part A: Fecundity (Comparisons of sexually formed brooded larvae)*

(i) Gili Trawangan – intermittent electricity supply:

There was no significant difference in fecundity of *Stylophora pistillata* between the locations of naturally growing colonies, colonies growing on non-electric structure, and each of the four electric structures (Fig. 1). Significant (p=0.00002) within location between colonies indicates that there was a high level of variability in colony fecundity.

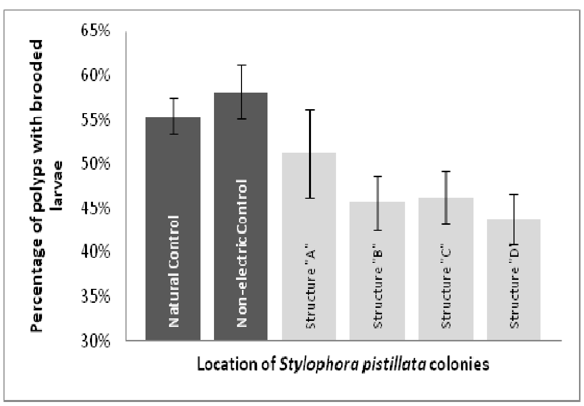



Fig. 1: Fecundity comparison of *Stylophora pistillata* at Gili Trawangan study site. Each structure analyzed as an independent location. Error bars represent standard error.

Although no significant difference was seen at a location level (p=0.185), both the natural control and the non-electric control had higher average fecundities than all four EMA structures. To test this two planned comparisons were conducted. First the two controls were grouped and compared against the four structures grouped together. This test provided a significant p-value of 0.01557 between the two groups, indicating that the average fecundity of *Stylophora pistillata* colonies growing on artificial reefs under electro-mineral accretion treatment (structures A-D) is significantly lower than colonies growing without this treatment (natural and non-electric controls).

A second independent planned comparison was done to look at the difference between the two controls. No significant difference (p=0.648) was found between the fecundity of naturally growing colonies of *Stylophora pistillata* and colonies growing on the non-electric EMA structure.

(ii) Gili Meno– continuous electricity supply:

There was a significant difference in fecundity between treatment and control (p=0.0028). Natural control colonies of *Stylophora pistillata* had a mean fecundity of 66% (+/- 10% SE) while the colonies growing on the EMA artificial reef structure had a lower fecundity of 38% (+/- 5% SE) (Fig. 2). These results indicate that fecundity is lower under treatment conditions of continuous electricity supply than naturally growing colonies.

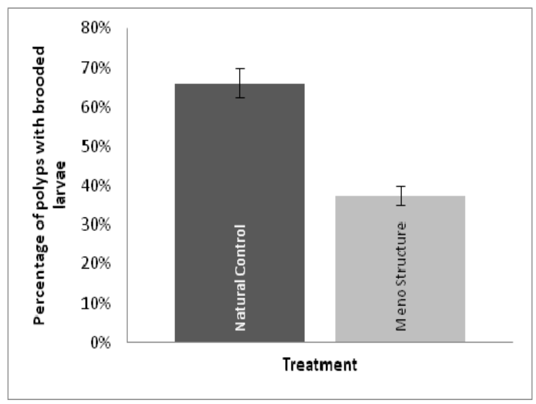



Fig. 2: Fecundity comparison of *Stylophora pistillata* on Gili Meno EMA structures. Error bars represent standard error.

*Part B: Growth Measurement*

Comparisons between growth rates in nature and under EMA treatment were made on two species of coral (*Acropora muricata* and *Stylophora pistillata*). However results were not able to be obtained from measurements of growth rates on *Stylophora pistillata* colonies because of high mortality of natural colonies. The results displayed here are from *Acropora muricata* only.

A significantly different skeletal growth rate was found between the EMA treatment and control colonies (p=0.015). However there was a significant location within treatment effect to a level of (p=0.004) indicating a high level of variability in growth rates among EMA structures.

Over the 5 week time period when growth measurements were made, naturally growing control colonies had an average linear skeleton extension of 7.71 (+/-0.54 SE) mm while the grouped average of the EMA structures was 9.92 (+/-0.39 SE) mm (Fig. 3).

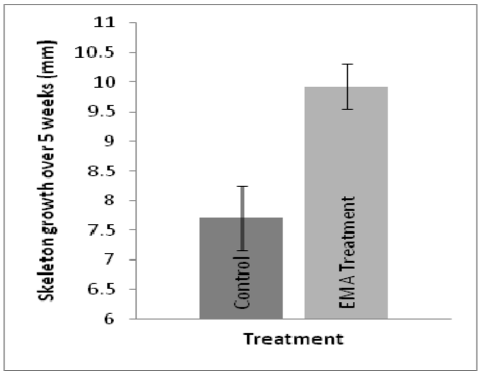



Fig. 3: Comparison of growth rates (linear extension of skeleton) for *Acropora muricata* at treatment level between naturally growing control colonies and grouped EMA treatment structures. Error bars represent standard error.

**Discussion**

These results indicate that exposure to electro-mineral accretion treatment does have an effect on the physiology of corals. Treatments in Gili Meno which had been exposed to constant DC electric current for one year had a significantly lower fecundity to neighboring naturally growing colonies, while the treatments on Gili Trawangan which had been exposed to an intermittent electric current for the past year also differed to neighboring controls (albeit with a large degree of variation). The intermittent nature of the electric current on the Gili Trawangan treatments was due to an unpredictable electricity source on the island and a lack of maintenance or continued management of projects after implementation. Such conditions accurately reflect the obstacles in implementing reef restoration projects in developing nations, and are therefore an important consideration. However, this also allowed for conclusions to be drawn about how the nature of electrical treatments affects coral fecundity.

Under constant electric treatment at Gili Meno, colonies of *Stylophora pistillata* were found to be less fecund – had lower percentages of brooded larvae present in polyps, than the naturally growing controls. Possible explanations are that either exposure to electricity is having a negative effect on the production of reproductive tissue, or that the higher skeletal growth rates under electric treatment is resulting in a trade-off with energy allocated to somatic growth rather than reproduction. Rinkevich and Loya (1989) found that fecundity in regenerating colonies of *Stylophora pistillata* was significantly lower than undamaged colonies. Later Rinkevich (1996) concluded that the trade-off for stem cells between tissue repair and sexual reproduction determines reproductive activities in regenerating corals. A similar conclusion was also made by Yap et al (1998), finding that transplantation of corals causes physiological stress resulting in compromised growth and survivorship. To isolate electrical treatment as the fundamental cause of lower fecundity, the natural controls were compared against the non-electric transplanted controls. The results from the Gili Trawangan treatments indicated that there was no significant difference in fecundity between *Stylophora pistillata* samples taken from the artificial reef structure with a broken electricity cable (non-electric control) and samples from naturally growing control colonies. This indicates that after one year (age of transplants on artificial reef structures) differences in fecundity were no longer being affected by transplantation stress but rather from electric treatment.

Unfortunately, 4 of the 6 naturally growing control colonies of *Stylophora pistillata* being monitored for growth died during the course of this experiment. Therefore growth and fecundity measurements were unable to be conducted on the same colonies, and a conclusive reproduction-growth trade-off cannot be drawn. However measurements taken during this study conducted on *Acropora muricata* (Fig. 3) demonstrated significantly higher growth under EMA treatment. Previous studies have shown similar findings of elevated growth rates of scleractinian corals on EMA structures (Sabater & Yap 2002; Eggeling 2006.

Electro-mineral accretion treatment was found to negatively affect reproductive output of *Stylophora pistillata*. Sabater & Yap (2004) found that the physiological effects from EMA treatment were only temporary and negated over time after treatment was stopped, however these physiological effects are referring to only skeletal girth and polyp density, and not reproduction. Similar effects may possibly be found in terms of fecundity. The treatments on Gili Trawangan which were exposed to intermittent electricity had no significant difference in fecundity to the controls. There is therefore a great deal of potential for using the electro-mineral accretion process in conjunction with other restoration methods. A coral reef restoration management plan must balance the restoration goals of increasing percent cover of foundation species, and restoring ecological function in the community. Previous studies (Sabater & Yap 2002; Eggeling 2006) as well as the results from this study have indicated that electro-mineral accretion is an effective method of increasing growth and therefore percent cover of hermatypic reef building corals. More research must be done to determine the best exposure duration for growing out coral colonies before allowing them to restore natural reproductive function with terminated treatment. A study by Schumacher & Schillak (1994) found that under EMA treatment simultaneous electrochemical and biogenic deposition of hard material is not possible. Therefore coral larvae will only settle on the artificial structures if an electric current is not present. If artificial reefs are installed to act as stable substrate for coral larvae to settle upon and grow - as is necessary when restoring denuded reef areas (Fox et al 2005), then EMA structures must have a period of time with no electric current for both settlement and reproduction. Effective reef restoration must apply the principles of adaptive management, wherein success of the project can be ensured under changing conditions (Thom 1997). Supplying continuous electricity to artificial reef structures may be an unsustainable means to restore an ecosystem. Whether such structures perpetuate after electric treatment is ceased or become an eroding and ultimately destructive force in themselves needs further research before this method can be adopted as a sustainable approach to reef restoration. Previous restoration efforts in Gili Trawangan have shown that after the initial project effort, support diminishes and frustration ensues as stakeholders see continued degradation of reef after quick fixes were implemented.

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