INDO PAC J OCEAN LIFE
Volume 10, Number 1, June 2026
Pages: 7

P-ISSN: 2775-1961
E-ISSN: 2775-1953
https://doi.org/10.13057/oceanlife/o100102

Biological traits of corals under electro-mineral accretion at a coral restoration site in Indonesia

ANDREW C. F. TAYLOR♥
Blue Corner Marine Research. Jl. Jungutbatu, Nusa Lembongan, Klungkung 80771, Bali, Indonesia. Tel.: +62-81916231054,
♥email: andrew@bluecornerconservation.org

Manuscript received: 22 August 2025. Revision accepted: 2 January 2026.

Abstract. *Taylor ACF. 2026. Biological traits of corals under electro-mineral accretion at a coral restoration site in Indonesia. Indo Pac J Ocean Life 10 (1): o100102. https://doi.org/10.13057/oceanlife/o100102.* Coral reef restoration efforts risk creating habitats that appear functional yet lack key ecological roles. Electro-Mineral Accretion (EMA)—a method applying low-voltage current to promote calcium carbonate deposition—has been used to enhance coral growth, but its effects on other biological traits remain poorly understood. This study investigated whether corals grown under EMA exhibit trade-offs between growth, reproduction, and skeletal structure compared to natural colonies. At restoration sites in Lombok, Indonesia, colonies of *Stylophora pistillata* were assessed for fecundity, polyp density, skeletal density, and growth. Growth comparisons of *Acropora muricata* were also measured between EMA treatments and naturally growing corals. EMA-treated corals showed significantly higher skeletal growth rates but reduced reproductive output, with fecundity 40% lower than natural colonies. Skeleton density was also reduced, suggesting lighter, less robust structures, while differences in polyp density were minor. These patterns indicate that enhanced growth under EMA may occur at the expense of reproduction and structural strength. The findings highlight a key ecological trade-off: faster-growing EMA corals may contribute to short-term reef cover but potentially compromise long-term population viability and resilience. Coral restoration success depends not only on promoting rapid growth but also on preserving reproductive capacity and skeletal integrity. This study underscores the importance of evaluating multiple biological traits when assessing restoration methods and cautions against relying solely on growth enhancement as a measure of ecological success.

Keywords: Artificial reef, biorock, coral biology, coral restoration, electro-mineral accretion

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 may be unlikely (Tkachenko 2023). For coastal communities reliant on reef ecosystem services there is growing interest in active reef restoration.

Artificial reefs are widely used tools for reef rehabilitation (Bracho-Villavicencio et al. 2023). They have been implemented for diverse purposes including enhancing fisheries, preventing shoreline erosion, restoring biodiversity, and supporting tourism (Higgins et al. 2022). However, artificial reefs may not recreate all ecological functions of natural reefs, and their role should be considered as one component within broader restoration and management strategies.

Among restoration tools, Electro-Mineral Accretion (EMA) applies low-voltage direct current to submerged metal frameworks to promote calcium carbonate deposition and secure coral transplants, with reported acceleration of skeletal extension in some projects (Hilbertz and Goreau 1996; Nugroho et al. 2023). 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 $CaCO_3$, $Mg(OH)_2$, $CaSO_4$, NaCl on the iron structure (Meyer and Schuhmacher 1993; Hilbertz and Goreau 1996). Several previous studies have reported higher linear extension rates of corals when subjected to electro-mineral accretion (Sabater and Yap 2002; Cook et al. 2023; Nugroho et al 2023). This effect has been attributed to increased availability of ions for calcification near the cathode.

Research on EMA has largely focused on growth performance and survival of attached corals with much fewer studies focusing on other fundamental biological traits that underpin coral fitness and ecological function (Knoester et al 2024). However, enhancing linear growth alone may obscure biological trade-offs that determine whether restored populations persist and contribute to reef recovery. In corals, energy acquired via photosynthesis and heterotrophy is a finite budget partitioned among somatic growth, reproduction, and maintenance (Perrin and Sibly 1993). Life-history and energetic theory predict that increased allocation to one function often reduces investment in others, producing measurable trade-offs at colony and population scales (Rinkevich 1996). Recent work provides empirical support: size-structured studies show shifts from growth to reproduction as colonies mature, reflecting dynamic energy-allocation schedules across the life cycle; conversely, under resource limitation

or stress, colonies reallocate energy toward survival and recovery at the expense of fecundity (Grinblat et al. 2023). Trade-offs can also emerge after prolonged acclimation (Roik et al. 2024).

Within this framework, EMA could theoretically shift energetic partitioning. By increasing local carbonate availability at the cathode, EMA may favor rapid accretion and linear extension. Yet if overall energy or dissolved inorganic carbon supply to the holobiont is finite, elevated calcification could draw on reserves otherwise available for gametogenesis or maintenance, producing reductions in fecundity or in skeletal densification. Reproductive capacity is critical for restoration success - restored populations must contribute larvae to local and regional recruitment to support demographic persistence and genetic diversity, especially under rapid environmental change (Schopmeyer et al. 2024).

This study whether EMA alters energy allocation among growth, reproduction, and maintenance-related structural traits. This study compares EMA-treated colonies with naturally growing conspecifics at a Lombok, Indonesia, restoration site, focusing on fecundity, polyp density, skeletal density, and growth. By integrating reproductive and structural metrics with growth measurements, we evaluate whether apparent gains in extension under EMA coincide with reductions in fecundity or skeletal robustness. Such trade-offs may diminish long-term ecological function despite short-term increases in cover.

MATERIALS AND METHODS

This study took place in the Gili Islands in North Lombok, West Nusa Tenggara, Indonesia. The community whose roots lie traditionally in fishing now supports a growing tourist industry (Satria et al. 2006). For several years the community-based organization Gili Eco Trust has overseen a coral restoration project using electro-mineral accretion reef structures upon degraded rubble reef areas.

The study site was located directly in front of the Gili Meno Village, North Lombok, along the algal and rubble dominated reef slope. The EMA treated corals were located between 5-7 m depth, and naturally growing control colonies were located in the surrounding area located 20-100m away from structures on same reef slope, exposure, and depth. All parameters were measured on *Stylophora pistillata*, and coral growth tests were also replicated on *Acropora muricata.* For each species, four replicate colonies were identified and tagged for each treatment. All colonies used were of similar size (~30 cm in diameter) to remove possible variations in fecundity associated with colony size (Hall and Hughes 1996). From each of the colonies, 4 replicate branches were sampled. Branches were taken from similar spots on each colony.

EMA treatment [4 x colonies (4 x branches)]

Natural control [4 x colonies (4 x branches)]

Part A: Fecundity

The tests for fecundity were performed under the following experimental design with samples collected from Gili Meno Island, North Lombok, over a 3 day period in January; EMA treatments received constant electricity over the previous 12 month period. Samples of *S. pistillata* were collected from the field and decalcified using hydrochloric acid, formaldehyde and water. Then stored in ethanol for observational analysis (McLachlan et al. 2021).

Tissue samples were removed to count number of polyps with brooded larvae (Rinkevich and Loya 1979). Polyps were examined in the area between 1-2 cm 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 1).

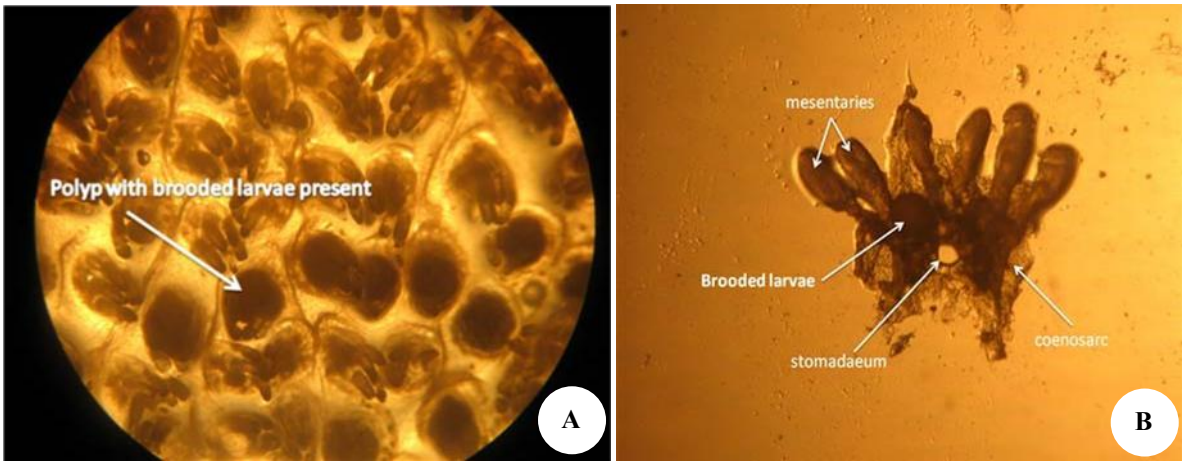

Figure 1. A. Tissue from decalcified *Stylophora pistillata* sample under 10× dissecting microscope. B. *Stylophora pistillata* polyp with brooded larvae highlighted

Part B: Polyp density (# polyps / area)

To determine density of polyps on the *S. pistillata* samples, the number of coralites within a standardized square 4×4 mm (16 mm^2) clear plastic overlay was done on samples before skeletons were decalcified. Counts were replicated in 5 randomly placed spots on branches within 0.5-1.5 cm from branch tip.

Part C: Skeleton density

Samples of *S. pistillata* were treated in hydrogen peroxide solution until was tissue dissolved and removed from skeletons. Samples were then rinsed in fresh water and let dry for 2 days. Dry weight measured using electronic scale graduated to 0.01 g. Volume measured using a graduated cylinder recording water displaced by skeleton sample. Density measurement calculated as dry weight of skeleton over volume of water displacement (g/mL).

Part D: Growth measurement

Coral growth was determined by staining the skeleton with Alizarin Red dye (Holcomb et al. 2013) at one point in time, then measuring growth of new white skeleton after a 5 week period (Figure 2). After 5 weeks the tagged branches which had been stained weeks earlier were removed and placed into hydrogen peroxide 50% solution to dissolve tissue for 24 hours. Growth over 5 week time period since staining was represented by white area of new skeleton extension past purple stained skeleton (Figure 2.B). Linear extension of skeleton was measured using fine scale vernier calipers. During the course of this study the *S. pistillata* colonies experienced high mortality, so *A. muricata* was used instead for growth analysis.

Data analysis

Differences in fecundity, polyp density, skeletal density, and growth between EMA-treated and naturally growing colonies were tested using nested Analysis of Variance (ANOVA), with treatment as a fixed factor and colony nested within treatment as a random factor. This structure accounted for pseudoreplication due to multiple branches sampled from each colony. Effect sizes were calculated to quantify the magnitude of treatment effects. Significance thresholds were set at $p<0.05$ for all analyses. For graphical representation, means were reported ± Standard Error (SE).

RESULTS AND DISCUSSION

Part A: Fecundity

To test fecundity rates on naturally growing corals with EMA treated corals, samples of *S. pistillata* from the Gili Meno study site were analysed and a nested ANOVA performed on the data from (Table 1). The difference in fecundity between the treatment and control was found to be strongly significant (p-level=0.0028); there was not a significant amount of variability between colony within treatment. Natural control colonies of *S. pistillata* had a mean fecundity of 66±10% while the colonies growing on the EMA artificial reef structure had a lower fecundity of 38±5% (Figure 3).

Table 1. Nested-ANOVA results for fecundity comparison of *Stylophora pistillata* on Gili Meno EMA structures in North Lombok, North Lombok, West Nusa Tenggara, Indonesia

Nested ANOVA for fecundity at Gili Meno study site	Effect	SS	df	MS	F	p
Treatment	fixed	0.66125	1	0.66125	23.6866	0.002803
Colony (Treatment)	random	0.16750	6	0.02792	2.0938	0.091704
error		0.32000	24	0.01333		

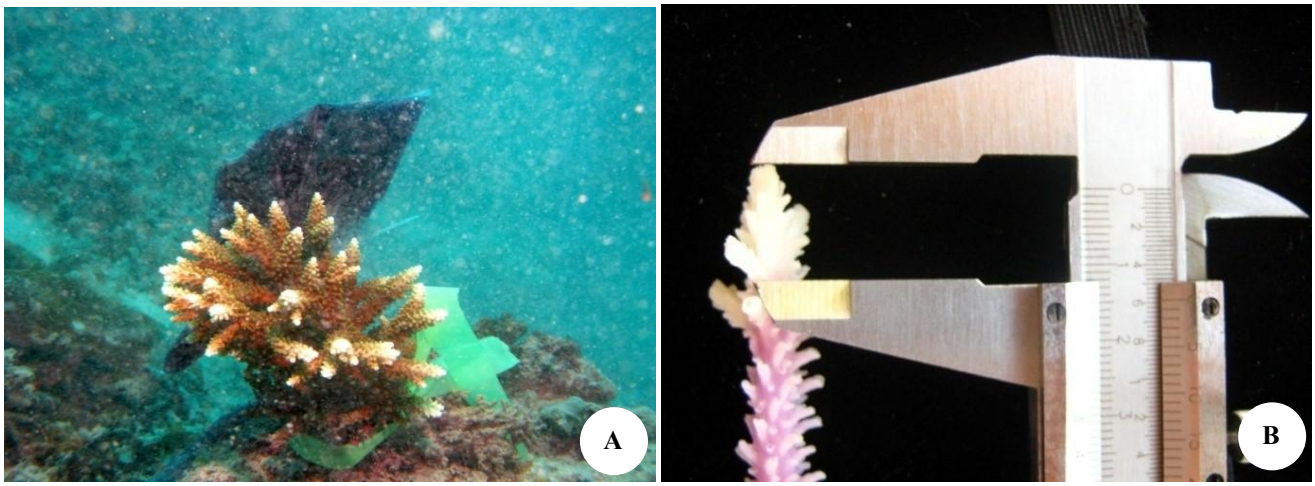

Figure 2. A. Naturally growing colony of *Acropora muricata* with plastic bag containing Alizarin red stain. B. Growth measurements taken after a 5 week period of EMA treatment sites and naturally growing control sites

Part B: Polyp density

Polyp density (number of polyps within a given surface area on coral branches) had no statistically significant difference between treatments. The average density of polyps on EMA treated corals were 47.87±0.73 polys per cm^2, compared to the naturally growing controls with 46.02±0.79 polyps per cm^2 (Figure 4).

Part C: Skeleton density

The density of *S. pistillata* skeleton as measured by dry skeleton weight per unit volume was significantly different (p=0.001109) between the corals on EMA structures compared to the natural coral controls (Figure 5). The corals growing on each of the EMA structures had average skeletal densities lower than the naturally growing coral. This indicates that skeletons of corals growing under EMA treatment have lighter, less dense skeletons than those growing under natural conditions. The average skeleton density of natural corals was 1.89±0.07 g/mL, while the average of EMA treated corals was less dense at 1.63±0.04 g/mL.

Part D: Growth measurement

Comparisons of growth rates were made on two species of coral (*A. muricata* and *S. pistillata*). However results were not able to be obtained from *S. pistillata* colonies because of high mortality of natural colonies during the period of study. The results displayed here are from *A. muricata* only.

A significantly different (p=0.00448) skeletal growth rate was found between the EMA treatment and control colonies (Figure 6). However there was also significant location within treatment effect (p=0.0149) indicating a high level of variability in growth rates among EMA treatments (Table 2). 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 mm while the grouped average of the EMA structures was 9.92±0.39 mm.

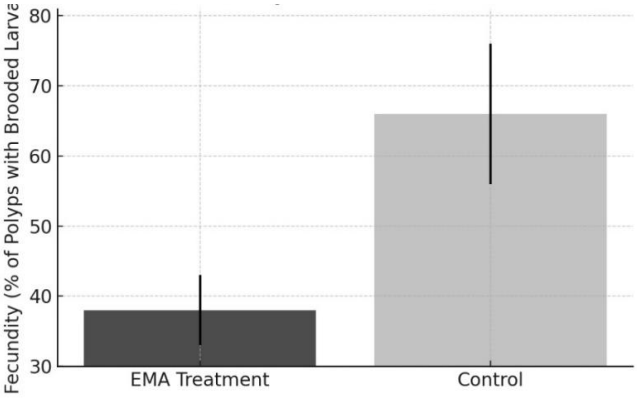

Figure 3. Fecundity comparison of *Stylophora pistillata* EMA treatments compared to naturally growing control corals. Error bars represent standard error of samples within colonies at each treatment. Significant between treatment effect p=0.0028

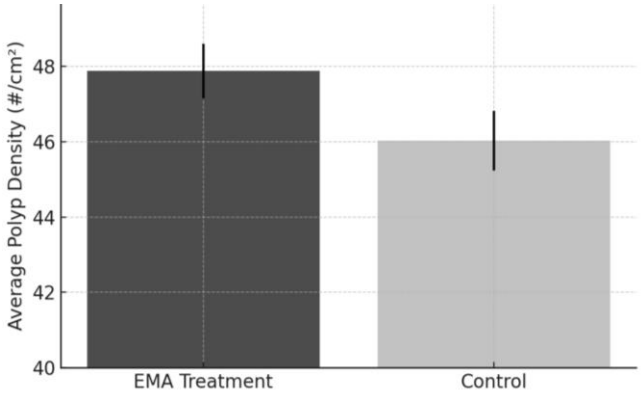

Figure 4. Polyp density comparison of *Stylophora pistillata* with EMA treatment compared to naturally growing control corals. Error bars represent standard error of samples within colonies at each treatment. Significance p=0.3863

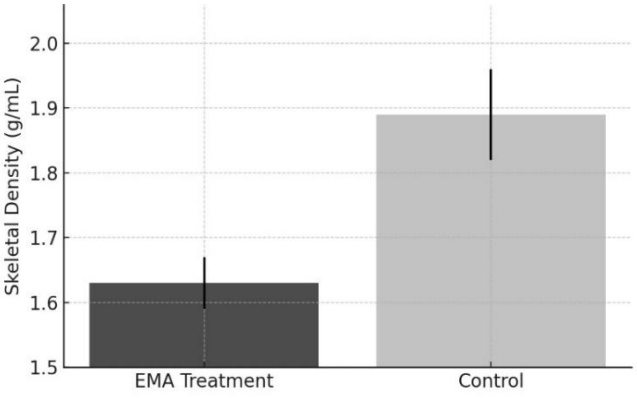

Figure 5. Skeletal density comparison of *Stylophora pistillata* with EMA treatment compared to naturally growing controls. Error bars represent standard error of samples within colonies at each treatment. Significance p=0.001109

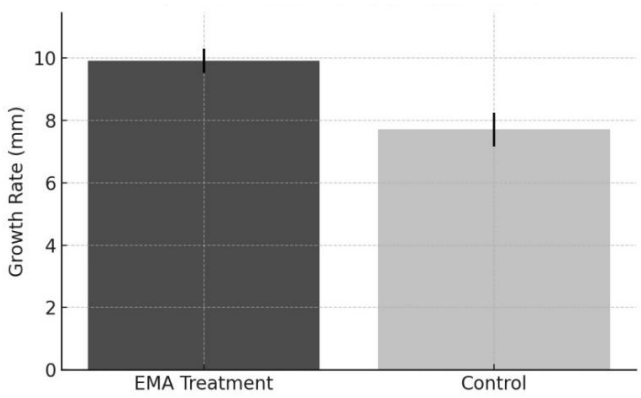

Figure 6. Growth comparison of *Acropora muricata* on EMA structures compared to naturally growing controls over 5 weeks. Error bars represent standard error of samples within colonies at each treatment. Significance p=0.00448

Table 2. Nested-ANOVA results for skeleton growth comparing treatments and location within treatments. Planned comparison at the location level for natural control versus EMA treatment

Nested ANOVA for growth of *Acropora muricata*	**Effect**	**SS**	**df**	**MS**	**F**	**p**
Location	fixed	50.718	3	16.906	3.9494	0.01494
Control or treatment		38.968	1	38.968	9.103	0.00448
error		166.945	39	4.281		

Discussion

Trade-offs between growth, reproduction, and skeletal structure

This study provides evidence that corals exposed to Electro-Mineral Accretion (EMA) exhibit enhanced skeletal growth but reduced fecundity and skeletal density compared with naturally growing colonies. These results support the hypothesis that EMA may alter coral energy allocation, favoring somatic growth over reproductive and structural investment. Similar physiological trade-offs are well documented in coral biology, where finite energy reserves are partitioned among growth, reproduction, and maintenance (Rinkevich 1996; Perrin and Sibly 1993). Several studies confirm that energy allocation in corals is dynamic and sensitive to environmental conditions, resource availability, and physiological stressors (Leuzinger et al. 2012; Hoogenboom et al. 2015).

Enhanced calcification under EMA likely increases energetic demand, diverting resources from gametogenesis and skeletal consolidation. This finding aligns with the principle that rapid skeletal extension does not necessarily equate to overall fitness. Experimental work by Knoester et al. (2024) similarly found that mineral accretion treatments improved coral growth but decreased thermal resilience and recruitment, suggesting physiological trade-offs extend across multiple life-history traits. Naturally growing corals were observed to have significantly denser skeletons than those growing under EMA treatment. This difference may be due to changes in calcium carbonate crystal structure – electrochemically precipitated calcium carbonate has been found to exhibit unique microstructures and variations in pore size of corals (Margheritini et al. 2021; Li et al. 2025). Skeleton density could affect the coral's ability to withstand physical pressures of waves and currents. Lighter, less dense skeletons may leave colonies prone to breakage and forming rubble. Skeletal growth traits include both linear extension and skeletal densification; the latter underpins mechanical strength and resistance to physical breakage. Recent records and syntheses indicate that density and calcification can diverge under contemporary stressors, with notable density declines that suggest structural weakening even when extension persists (Deng et al. 2024). Such decoupling may have significant effects on restoration outcomes: colonies with rapid extension but low structural density may add short-term cover yet contribute less to reef framework stability over time.

Such trade-offs have profound implications for coral persistence. High growth rates may benefit early colonization and short-term habitat stabilization, yet diminished fecundity limits larval output and genetic replenishment, reducing population resilience. Moreover, lighter skeletons may increase susceptibility to breakage, bioerosion, and storm damage—particularly under increasing hydrodynamic stress predicted for tropical reefs.

Implications for coral restoration practice

The ecological effectiveness of coral restoration depends not only on increasing coral cover but also on ensuring that restored colonies can reproduce, persist, and maintain structural integrity. EMA has gained popularity for its apparent ability to accelerate coral growth (Sabater and Yap 2002; Nugroho et al. 2023). However, the results here caution against interpreting faster extension rates as unequivocal indicators of restoration success.

A growing body of literature emphasizes that restoration metrics must include functional and reproductive outcomes, not just morphological ones (Randall et al. 2020; Harrison 2024). Projects that optimize early growth at the cost of fecundity risk creating coral assemblages that appear healthy but lack self-sustaining population dynamics. This mirror concerns raised in other restoration contexts—such as coral gardening and microfragmentation—where growth-promoting conditions can mask long-term declines in reproductive capacity and genetic diversity (Boström-Einarsson et al. 2020; Knapp et al. 2022).

The present findings suggest that EMA may be best used as a short-term tool within adaptive restoration frameworks. For example, practitioners might employ EMA to stabilize rubble fields and accelerate early coverage, but then deactivate electrical current once colonies reach sufficient size to resume natural energy allocation. This staged approach could combine the short-term structural benefits of EMA with the long-term biological resilience of natural growth.

Furthermore, trait-based evaluation should be incorporated into monitoring programs. Restoration projects increasingly assess traits such as skeletal density, reproductive output, and heat tolerance to evaluate ecological function (Baums et al. 2022; Quigley et al. 2022). Including these measures in EMA projects would enable managers to quantify potential trade-offs and refine intervention duration or intensity to minimize adverse effects. In particular, understanding how EMA influences energy allocation under thermal stress will be critical for predicting outcomes under climate change.

Uncertainties and future research needs

While this study documents significant physiological differences between EMA and natural corals, several uncertainties remain. First, sample size was limited due to colony mortality, which prevented direct pairing of growth and fecundity data within individuals. Unfortunately, 4 of the 6 naturally growing control colonies of *S. pistillata*

which were being monitored for growth died during the course of this experiment, therefore growth measurements were not able to be taken on the corresponding colonies to which fecundity measurements were made. Therefore a correlation showing trade-offs between growth and reproduction cannot be adequately drawn from these results. Future studies should include larger sample sets and extended monitoring to confirm whether growth–reproduction trade-offs exist on the same individuals and whether those effects persist over time or diminish once electrical current is discontinued.

Second, the mechanistic basis of observed differences remains unresolved. It is unclear whether reduced fecundity results primarily from energetic constraints, altered ion balance, or changes in skeletal microstructure that affect polyp physiology. Advanced histological and metabolomic analyses could clarify whether EMA induces changes in energy metabolism, gametogenic tissue development, or skeletal mineralogy.

Third, long-term ecological outcomes of EMA remain largely untested. Most existing studies, including this one, span months to a few years, whereas restoration success requires multi-decadal persistence. Longitudinal tracking of EMA-treated reefs would help determine whether reproductive performance and skeletal integrity recover after treatment cessation or whether early trade-offs have lasting demographic consequences. Additionally, studies should consider potential species-specific responses, as energy allocation strategies differ among branching, massive, and encrusting coral morphotypes (Darling et al. 2012).

Finally, restoration science increasingly integrates climate resilience into project design. Recent frameworks advocate combining active methods like EMA with selective propagation of thermally tolerant genotypes or assisted gene flow (Baums et al. 2022). EMA could be evaluated not as a stand-alone technique but as one component of multifaceted strategies that enhance early survival while maintaining adaptive capacity.

In conclusion, this study reveals that electro-mineral accretion can enhance coral growth but may compromise reproduction and skeletal robustness, illustrating classic energy allocation trade-offs in coral biology. The results reinforce that coral restoration should prioritize functional recovery and long-term sustainability over short-term structural gain. As restoration goals evolve to include genetic diversity, ecosystem function, and climate resilience, tools like EMA must be critically evaluated within these broader ecological frameworks. Addressing the identified uncertainties through integrative, long-term, and multi-trait studies will be essential for determining when and how EMA can effectively contribute to sustainable reef restoration.

REFERENCES

Baums IB, Chamberland VF, Locatelli NS, Conn T. 2022. Maximizing genetic diversity in coral restoration projects. In: van Oppen MJH, Lastra MA (eds). Coral reef Conservation and Restoration in the Omics Age. Springer International Publishing, Cham. https://doi.org/10.1007/978-3-031-07055-6_3.

Boström-Einarsson L, Babcock RC, Bayraktarov E, Ceccarelli D, Cook N, Ferse SC, Hancock B, Harrison P, Hein M, Shaver E, Smith A, Suggett D, Stewart-Sinclair PJ, Vardi T, McLeod IM. 2020. Coral restoration – A systematic review of current methods, successes, failures and future directions. PLoS One 15 (1): e0226631. https://doi.org/10.1371/journal.pone.0226631.

Bracho-Villavicencio C, Matthews-Cascon H, Rossi S. 2023. Artificial reefs around the world: A review of the state of the art and a meta-analysis of its effectiveness for the restoration of marine ecosystems. Environments 10 (7): 121. https://doi.org/10.3390/environments10070121.

Cook N, Cook K, Harris KJ, Songcuan A, Smith AK. 2023. Lessons learned implementing mineral accretion and coral gardening at Agincourt Reef, Great Barrier Reef. Ecol Manag Restor 24 (2-3): 107-118. https://doi.org/10.1111/emr.12585.

Darling ES, Alvarez-Filip L, Oliver TA, McClanahan TR, Côté IM. 2012. Evaluating life-history strategies of reef corals from species traits. Ecol Lett 15 (12): 1378-1386. https://doi.org/10.1111/j.1461-0248.2012.01861.x.

Deng G, Chen X, Kang H, Zhao JX, Wei G. 2024. Millennial calcification trends in Porites corals: Resilient skeletal density and vulnerable linear extension in response to environmental change. Glob Planet Chang 237: 104456. https://doi.org/10.1016/j.gloplacha.2024.104456.

Grinblat M, Eyal-Shaham L, Eyal G, Ben-Zvi O, Harii S, Morita M, Sakai K, Hirose M, Miller DJ, Loya Y. 2023. Energy allocation trade-offs as a function of age in fungiid corals. Front Mar Sci 10: 1113987. https://doi.org/10.3389/fmars.2023.1113987.

Hall VR, Hughes TP. 1996. Reproductive strategies of modular organisms: Comparative studies of reef-building corals. Ecology 77 (3): 950-963. https://doi.org/10.2307/2265510.

Harrison PL. 2024. Sexual reproduction of reef corals and application to coral restoration. In: Wolanski E, Kingsford MJ (eds). Oceanographic Processes of Coral Reefs. CRC Press, Boca Raton. https://doi.org/10.1201/9781003320425-32.

Higgins E, Metaxas A, Scheibling RE. 2022. A systematic review of artificial reefs as platforms for coral reef research and conservation. PLoS One 17 (1): e0261964. https://doi.org/10.1371/journal.pone.0261964.

Hilbertz W, Goreau T. 1996. Method of Enhancing the Growth of Aquatic Organisms and Structures Created Thereby. U.S. Patent No. 5,543,034. U.S. Patent and Trademark Office, Washington DC.

Holcomb M, Cohen AL, McCorkle DC. 2013. An evaluation of staining techniques for marking daily growth in scleractinian corals. J Exp Mar Biol Ecol 440: 126–131. https://doi.org/10.1016/j.jembe.2012.12.003.

Hoogenboom M, Rottier C, Sikorski S, Ferrier-Pagès C. 2015. Among-species variation in the energy budgets of reef-building corals: Scaling from coral polyps to communities. J Exp Biol 218 (24): 3866–3877. https://doi.org/10.1242/jeb.124396.

Knapp ISS, Forsman ZH, Greene A, Johnston EC, Bardin CE, Chan N, Wolke C, Gulko D, Toonen RJ. 2022. Coral micro-fragmentation assays for optimizing active reef restoration efforts. PeerJ 10: e13653. https://doi.org/10.7717/peerj.13653.

Knoester EG, Sanders R, Durden D, Masiga BO, Murk AJ, Osinga R. 2024. Negative effects by mineral accretion technique on the heat resilience, growth and recruitment of corals. PLoS One 19 (12): e0315475. https://doi.org/10.1371/journal.pone.0315475.

Leuzinger S, Willis BL, Anthony KRN. 2012. Energy allocation in a reef coral under varying resource availability. Mar Biol 159 (1): 177-186. https://doi.org/10.1007/s00227-011-1797-1.

Li Y, Zhao H, Zhao Y, Liao X, Chen JY, Qin Y, Lu Z, Zhang Y, He C. 2025. Skeleton-forming responses of reef-building corals under ocean acidification. Research (Washington DC) 8: 0736. https://doi.org/10.34133/research.0736.

Margheritini L, Møldrup P, Jensen RL, Frandsen KM, Antonov YI, Kawamoto K, de Jonge LW, Vaccarella R, Bjørgård TL, Simonsen ME. 2021. Innovative material can mimic coral and boulder reefs properties. Front Mar Sci 8: 652986. https://doi.org/10.3389/fmars.2021.652986.

McLachlan RH, Dobson KL, Schmeltzer ER, Thurber RV, Grottoli AG. 2021. A review of coral bleaching specimen collection, preservation, and laboratory processing methods. PeerJ 9: e11763. https://doi.org/10.7717/peerj.11763.

Meyer D, Schuhmacher H. 1993. Ökologisch verträgliche Bauprozesse im Meerwasser. Geowissenschaften 11: 408–412.

Nugroho BS, Zuhry N, Kusnandar K, Simanjuntak SW, Alamsyah HK, Karissa PT. 2023. Biorock® technology application to the growth and Acropora corals growth rate. IOP Conf Ser Earth Environ Sci 1147 (1): 012003. https://doi.org/10.1088/1755-1315/1147/1/012003.

Perrin N, Sibly RM. 1993. Dynamic models of energy allocation and investment. Ann Rev Ecol Syst 24: 379–410. https://doi.org/10.1146/annurev.es.24.110193.002115.

Quigley KM, Hein M, Suggett DJ. 2022. Translating the 10 golden rules of reforestation for coral reef restoration. Conserv Biol 36 (4): e13890. https://doi.org/10.1111/cobi.13890.

Randall CJ, Negri AP, Quigley KM, Foster T, Ricardo GF, Webster NS, Bay LK, Harrison PL, Babcock RC, Heyward AJ. 2020. Sexual production of corals for reef restoration in the Anthropocene. Mar Ecol Prog Ser 635: 203-232. https://doi.org/10.3354/meps13208.

Rinkevich B, Loya Y. 1989. Reproduction in regenerating colonies of the coral *Stylophora pistillata*. Environ Qual Ecosyst Stabil 4: 257-265.

Rinkevich B. 1996. Do reproduction and regeneration in damaged corals compete for energy allocation? Mar Ecol Prog Ser 143: 297-302. https://doi.org/10.3354/meps143297.

Roik A, Wall M, Dobelmann M, Nietzer S, Brefeld D, Fiesinger A, Reverter M, Schupp PJ, Jackson M, Rutsch M, Strahl J. 2024. Trade-offs in a reef-building coral after six years of thermal acclimation. Sci Total Environ 949: 174589. https://doi.org/10.1016/j.scitotenv.2024.174589.

Sabater M, Yap H. 2002. Growth and survival of coral transplants with and without electrochemical deposition of CaCO3. J Exp Mar Biol Ecol 272 (1): 131–146. https://doi.org/10.1016/S0022-0981(02)00118-7.

Satria A, Matsuda Y, Sano M. 2006. Questioning community-based coral reef management systems: Case study of Awig-Awig in Gili Indah, Indonesia. Environ Dev Sustain 8 (1): 99-118. https://doi.org/10.1007/s10668-005-0909-9

Schopmeyer S, Galvan V, Hernandez-Delgado EA, Nava G, D'Alessandro M, Carne L, Goergen E, Viehman S, Moulding A, Lirman D. 2024. An evaluation tool for assessing coral restoration efforts. Front Mar Sci 11: 1404336. https://doi.org/10.3389/fmars.2024.1404336.

Tkachenko KS. 2023. Degradation of coral reefs under complex impact of natural and anthropogenic factors with Nha Trang Bay (Vietnam) as an example. Biol Bull Rev 13 (5): 442-459. https://doi.org/10.1134/S2079086423050113.