# Coral Conservation

## Global evidence for the effects of actions

Ann Thornton, William H. Morgan, Eleanor K. Bladon,
Rebecca K. Smith, and William J. Sutherland

CONSERVATION EVIDENCE SERIES SYNOPSES

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*Conservation Evidence Series Synopses,*
*University of Cambridge, Cambridge, UK*

*Ann Thornton, William H. Morgan, Eleanor K. Bladon,*

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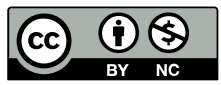





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We would like to thank all collaborators who have searched journals and reports to extract relevant studies for the Conservation Evidence project. All are listed on the following pages of our website: Conservation Evidence: Catalogue of Searched English Journals; Conservation Evidence: Catalogue of Searched Non-English Journals; Conservation Evidence: Catalogue of Searched Reports.

# 1. About this book

## The Conservation Evidence project

The Conservation Evidence project has four main parts:

1. The **synopses** of the evidence captured for the conservation of particular species groups or habitats, such as this synopsis. Synopses bring together the evidence for each possible action (also known as an intervention). They are freely available online and, in some cases, available to purchase in printed book form.
2. An ever-expanding **database of summaries** of previously published scientific papers, reports, reviews or systematic reviews that document the effects of actions. This resource comprises over 8,700 pieces of evidence, all available in a searchable database on the website https://www.conservationevidence.com.
3. ***What Works in Conservation***, which is an assessment of the effectiveness of actions by expert panels, based on the collated evidence for each action for each species group or habitat covered by our synopses. This is available as part of the searchable database and is published as an updated book edition each year (https://www.conservationevidence.com/content/page/79).
4. An online, **open access journal** *Conservation Evidence* publishes new pieces of research on the effects of conservation management actions. All our papers are written by, or in conjunction with, those who carried out the conservation work and include some monitoring of its effects (https://conservationevidencejournal.com/).

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# The purpose of Conservation Evidence synopses

| Conservation Evidence synopses **do** | Conservation Evidence synopses **do not** |
|---|---|
| • Bring together scientific evidence captured by the Conservation Evidence project (over 7,800 studies so far) on the effects of actions to conserve biodiversity | • Include evidence on the basic ecology of species or habitats, or threats to them |
| • List all realistic actions for the species group or habitat in question, regardless of how much evidence for their effects is available | • Make any attempt to weigh or prioritize actions according to their importance or the size of their effects |
| • Describe each piece of evidence, including methods, as clearly as possible, allowing readers to assess the quality of evidence | • Weigh or numerically evaluate the evidence according to its quality |
| • Work in partnership with conservation practitioners, policymakers and scientists to develop the list of actions and ensure we have covered the most important literature | • Provide recommendations for conservation problems, but instead provide scientific information to help with decision-making |

## Who this synopsis is for

If you are reading this, we hope you are someone who makes decisions about how best to support or conserve biodiversity. You might be a resource manager, a conservationist in the public or private sector, a fisher, a campaigner, an advisor or consultant, a policymaker, a researcher or someone taking action to protect your own local wildlife or reef. Our synopses summarize scientific evidence relevant to your conservation objectives and the actions you could take to achieve them.

We do not aim to make your decisions for you, but to support your decision-making by telling you what evidence there is (or isn't) about the effects that your planned actions could have.

When decisions must be made with particularly important

consequences, we recommend carrying out a systematic review, as the latter is likely to be more comprehensive than the summary of evidence presented here. Guidance on how to carry out systematic reviews can be found at the Collaboration for Environmental Evidence (Guidelines for Authors – Environmental Evidence).

## Background

Corals (Phylum: Cnidaria), including the stony corals as well as soft and cold-water species, are found in a diverse range of marine habitats in tropical, temperate, and arctic waters from shallow coasts all the way down to the deep sea. Formed over many thousands of years, corals can form reefs which develop when larvae from reef-building stony corals (also known as hard or scleractinian corals) settle on hard substrate at the edge of an island or volcano. As individual polyps grow and the hard calcium carbonate skeleton forms to support the polyps, the colony grows. Despite covering just 0.2% of the ocean floor, coral reefs are believed to support more than 25% of marine species (Souter *et al*. 2021). Corals are threatened by anthropogenic impacts, such as land-use change (urban, industrial and agricultural), nutrient enrichment, chemical and noise pollution, resource exploitation, damage from fishing gear and destructive fishing activity, invasive species, disease, and declining water quality (Souter *et al*. 2021). Climate change and increasing levels of atmospheric greenhouse gases continue to have serious direct and indirect effects, including ocean acidification, more frequent bleaching events and increasing frequency and intensity of storms (Hein *et al*. 2020). The Status of the Coral Reefs of the World 2020 report, by the Global Coral Reef Monitoring Network (GCRMN) reported that between 2009 and 2018 there was a loss of 14% of the global average cover of stony (hard) coral on the world's coral reefs (Souter *et al*. 2021).

Although all corals face broadly similar threats, actions aimed at conserving or restoring corals will vary depending on the communities/composition of species and where the reef is located. Actions can be considered within three broad themes i) protection of healthy reefs (e.g. use of Marine Protected Areas), ii) measures to reduce impact of threats (e.g. fisheries management, pollution control), and iii) active restoration (e.g. ex-situ cultivation, transplanting). As coral colonies can

take many years or even centuries to recover from damage (Boch *et al*. 2019), the priority should be effective measures to protect existing reefs, and colonies, followed by measures to mitigate anthropogenic impacts and threats. Restoration should work in conjunction with protection and mitigation (Montero-Serra *et al*. 2019; Boström-Einarsson *et al*. 2020; Souter *et al*. 2021). All these measures must be combined with global efforts to tackle climate change and reduce the devastating impact on the world's corals such as a rise in ocean temperature and increased ocean acidification, as well as the equally deleterious impacts of excess nutrient input and eutrophication (Silbiger *et al*. 2018).

Evidence-based knowledge is key for planning successful conservation and/or restoration strategies and for the cost-effective allocation of scarce resources for conservation and restoration programmes. However, there is a paucity of evidence within the literature for the effectiveness of actions aimed at coral conservation or reef restoration (Shaver *et al*. 2018). Reviews of some conservation actions (e.g. Marine Protected Areas) have been carried out (e.g. Pendleton *et al*. 2018; Montero-Serra *et al*. 2018, NASEM 2019, Boström-Einarsson *et al*. 2020). However, the evidence for the effectiveness of all actions targeting coral conservation or reef restoration has not yet been synthesized and assessed under a formal review.

Here, we use a subject-wide evidence synthesis approach (Sutherland & Wordley 2018, Sutherland *et al*. 2019) to simultaneously summarize the evidence for the wide range of practical actions dedicated to the protection, conservation and restoration of all corals (including stony, soft, and deep-water species in tropical, temperate, or deep-sea environments). By simultaneously targeting the range of potential actions, we can review the evidence for each action cost-effectively, and the resulting synopsis can be updated periodically and efficiently to incorporate new research. This synopsis is freely available at https://www.conservationevidence.com and, alongside the *Conservation Evidence* online database (comprising all summarized information from the synopsis along with expert assessment scores), should be a valuable asset to the toolkit of practitioners and policy makers seeking sound information to support coral reef conservation and restoration.

# Scope of the Coral Conservation synopsis

## *Review subject*

This synthesis collates global evidence for the effectiveness of all conservation and restoration actions for all corals (including stony, soft, and deep-water species) on reefs (fringing, barrier, patch, and atoll) in tropical, temperate, or cold-water environments. The synopsis focuses on summarizing evidence for practical actions including, but not limited to, designating Marine Protected Areas, cultivating corals both in-situ and ex-situ, transplanting coral fragments, and stabilizing damaged reefs. We include corals on reefs in shallow waters through the mesophotic zone (reduced light penetration at 30–150 m depth) down to the deep sea. This subject has not yet been covered using subject-wide evidence synthesis. This is defined as a systematic method of reviewing and synthesising evidence that covers broad subjects (in this case conservation and restoration of corals) at once, including all closed review topics within that subject at a fine scale, and analysing results through study summary and expert assessment, or through meta-analysis. The term can also refer to any product arising from this process (Sutherland *et al*. 2019). The topic is therefore a priority for the discipline-wide Conservation Evidence database.

Evidence for the effectiveness of actions targeting the conservation of other aquatic or semi-aquatic species that rely on coral reef habitats (such as fishes, invertebrates, etc.) are covered in separate synopses.

In relation to active restoration actions, we did not include evidence from the substantial literature on the commercial cultivation of coral species solely for the aquarium trade. However, we included these actions where they were relevant to the conservation of wild declining or threatened species (e.g. cultivation and propagation of coral species for the purpose of reintroductions). We also included actions designed to restore coral reef habitat by transplanting ex-situ nursery-cultivated coral on to man-made structures (e.g. frames) or natural surfaces (e.g. existing dead coral, rocks, or crevices on the sea floor); a process known as 'coral gardening'. Actions relating to enhancing artificial structures in the marine environment, other than for the purposes of coral reef restoration, are published in other synopses and will not be considered here. For this synthesis, restoration actions include those that aim to restore corals.

# Methods

## *Literature searches*

Literature was obtained from the Conservation Evidence discipline-wide literature database, and from searches of additional subject specific literature sources. The Conservation Evidence discipline-wide literature database is compiled using systematic searches of journals and organisational reports; relevant publications describing studies of conservation actions for all species groups and habitats are saved from each search and are added to the database.

***a*) *Global evidence***

Evidence from all around the world is included.

***b*) *Languages included***

A recent study on the topic of language barriers in global science indicates that approximately 35% of conservation studies may be in non-English languages (Amano *et al*. 2016). Therefore, journals published in a total of 17 languages have been searched and relevant papers extracted by Conservation Evidence:

- Arabic (11 journals)
- Chinese, simplified (61 journals)
- Chinese, traditional (14 journals)
- English (over 330 journals)
- French (13 journals)
- German (40 journals)
- Hungarian (4 journals)
- Indonesian (1 journal)
- Italian (7 journals)
- Japanese (20 journals)
- Korean (5 journals)
- Persian (9 journals)

- Polish (10 journals)
- Portuguese (29 journals)
- Russian (12 journals)
- Spanish (63 journals)
- Turkish (27 journals)
- Ukrainian (4 journals)

Journals listed as "English" are either published in English or at least carry English summaries (Appendix 1). Non-English-language journals are listed in Appendix 2. All relevant papers were added to the Conservation Evidence discipline-wide literature database (see below).

***c*) *Journals searched***

***i*) *From Conservation Evidence discipline-wide literature database***

All of the journals (and years) listed in Appendix 1 and Appendix 2 had already been searched and relevant papers added to the Conservation Evidence discipline-wide literature database. An asterisk indicates the journals most relevant to this synopsis. Others are less likely to have included papers relevant to this synopsis, but if they did, they will be summarized.

***ii*) *Update searches***

Due to time constraints, we did not update journal searches. However, we did undertake new searches of more relevant journals (see list below).

***iii*) *New searches***

In addition to the list of journals in Appendix 1, specialist journal searches were carried out for studies published from 2000–2021 in the journals *Coral Reefs* and *Bulletin of Marine Science*. There was no time to conduct focused searches of other journals relevant to the conservation or restoration of coral reefs listed below. Searches of these journals may be carried out at a later date and added to any future synopsis update.

- Ecological Engineering
- Ocean & Coastal management

- Marine Biology
- Marine Policy
- Estuarine Coastal and Shelf Science
- Western Indian Ocean Journal of Marine Science
- Deep Sea Research II
- Marine Biodiversity

*d*) ***Reports from specialist websites searched***

*i*) ***From Conservation Evidence discipline-wide literature database***

All of the report series (and years) shown in Appendix 3 had already been searched for the Conservation Evidence project. An asterisk indicates the report series most relevant to this synopsis. Others are less likely to have included reports relevant to this synopsis, but if they did, they were summarized.

*ii*) ***Update searches***

Due to time constraints, we did not update report searches. However, we undertook new searches of more relevant reports (see list below).

*iii*) ***New searches***

New searches of targeted specialist reports relevant to coral reef conservation and restoration as listed below. These searches reviewed every report title and abstract or summary within each report series (published before the end of 2021) and any relevant reports were added to the project database.

- **Global Coral Reef Monitoring network (GCRMN)**: Status of coral reefs of the world reports 1998- 2020 (https://gcrmn.net/?s=status+of+coral+reefs+of+the+world)
- International Council for the Exploration of the Sea (ICES) Expert Groups (https://www.ices.dk/community/groups/Pages/default.aspx) reports from Working Group on Deep-water Ecology (WGDEC) Volume 2: Issue 62 (2020) and Volume 1: Issue 56 (2019) (https://www.ices.dk/community/groups/Pages/WGDEC.aspx)

The following resource has published over 9,000 reports and therefore a systematic search of every title was not possible within the time frame of this project. Instead, key-word searches (for 'coral', 'reef', 'atoll', 'zooxanthellae') were carried out within the topic.

- National Academies Press Reports (https://www.nap.edu/)

*e*) ***Other literature searches***

The online database https://www.conservationevidence.com was searched for relevant publications that have already been summarized.

*f*) ***Reviews or supplementary literature identified by advisory board or relevant stakeholders***

There were no reviews identified and summarized for this synopsis. New or collective data from reviews (both systematic and non-systematic) were summarized as part of the broader synthesis covering the effectiveness of all conservation and restoration actions for coral reefs. An example of new data would be previously unpublished data from a case study, which may be used to support or illustrate points arising from the review. Examples of collective data would be a meta-analysis of results from previously published studies, a table listing the survival rate of transplanted corals in previously published studies, or a combination of multiple published studies to describe long-term changes in one restoration site. Summary paragraphs for reviews will indicate which other summarized studies they include (if any). Due to time constraints, reviews will not be used to identify further publications to summarize unless they are explicitly identified by the advisory board.

*g*) ***Search record database***

A database was created of all relevant publications found during searches. Reasons for exclusion were recorded for all those included during screening that were not summarized for this synopsis.

## *Publication screening and inclusion criteria*

### *a*) *Screening*

To ensure consistency/accuracy when screening publications for inclusion in the literature database, an initial test using the Conservation Evidence inclusion criteria (provided below) and a consistent set of references was carried out by authors, compared with the decisions of the experienced core Conservation Evidence team. Results were analysed using Cohen's Kappa test (Cohen 1960). Where initial results did not show 'substantial' (K = 0.61–0.8) or 'almost perfect' agreement (K = 0.81–1.0), authors were given further training. A second Kappa test was used to assess the consistency/accuracy of article screening for the first two years of the first journal searched by each author. Again, where results did not show 'substantial' (K = 0.61–0.8) or 'almost perfect' agreement (K = 0.81–1.0), authors received further training before carrying out further searches.

Authors of other synopses who have searched journals and added relevant publications to the Conservation Evidence literature database since 2018, and all other searchers since 2017 have undertaken the initial paper inclusion test described above; searchers prior to that have not. Kappa tests have been conducted on the first two years of searches carried out by all new contributors to the Conservation Evidence literature database since July 2018.

We acknowledge that the literature search and screening method used by Conservation Evidence, as with any method, will result in gaps in the evidence. The Conservation Evidence literature database currently includes relevant papers from over 330 English language journals as well as over 320 non-English journals. Additional journals are frequently added to those searched, and years searched are often updated. It is possible that searchers will have missed relevant papers from those journals searched. Publication bias has not been taken into account, and it is likely that additional biases will result from the evidence that is available, for example there are often geographic biases in study locations.

### *b) Inclusion criteria*

The following Conservation Evidence inclusion criteria were used.

**Criteria A: Conservation Evidence includes studies that measure the effect of an action that might be done to conserve biodiversity**

1. Does this study measure the effect of an action that is or was under the control of humans, on wild taxa (including captives), habitats, or invasive/problem taxa? If yes, go to 3. If no, go to 2.
2. Does this study measure the effect of an action that is or was under the control of humans, on human behaviour that is relevant to conserving biodiversity? If yes, go to Criteria B. If no, the study will be excluded.
3. Could the action be put in place by a conservationist/decision maker to protect, manage or restore wild taxa or habitats, reduce impacts of threats to wild taxa or habitats, or control or mitigate the impact of the invasive/problem taxon on wild taxa or habitats? If yes, the study will be included. If no, the study will be excluded.

**Explanation**:

1a. Study must have a measured outcome on wild taxa, habitats or invasive species: excludes studies on domestic/agricultural species, theoretical modelling or opinion pieces. See Criteria B for actions that have a measured outcome on human behaviour only.

1b. Action must be carried out by people: excludes impacts from natural processes (e.g. wave action, natural storms), impacts from background variation (e.g. sediment type, climate change), correlations with habitat types, where there is no test of a specific action by humans, or pure ecology (e.g. movement, distribution of species).

2. Study must test an action that could be put in place for conservation. This excludes assessing impacts of threats (actions which remove threats would be included). The test may involve comparisons between sites/factors not originally

put in place or modified for conservation, but which could be (e.g. fished vs unfished sites, dredged vs undredged sites – where the removal of fishing/dredging is as you would do for conservation, even if that was not the original intention in the study).

If the title and/or abstract are suggestive of fulfilling our criteria, but there is not sufficient information to judge whether the action was under human control, the action could be applied by a conservationist/decision maker or whether there are data quantifying the outcome, then the study was included. If the article has no abstract, but the title is suggestive, then a study was included.

We sort articles into folders by which taxon/habitat they have an outcome on. If the title/abstract does not specify which species/taxa/habitats are impacted, then the full article was searched and then assigned to folders accordingly.

The outcome for wild taxa/habitats can be negative, neutral or positive, does not have to be statistically significant but must be quantified (if hard to judge from abstract, then it will be included). It could be any outcome that has implications for the health of individuals, populations, species, communities or habitats, including, but not limited to the following:

- **Individual health, condition or behaviour, including in captivity**: e.g., growth, size, weight, stress, disease levels or immune function, movement, use of natural/artificial habitat/structure, range, or predatory or nuisance behaviour that could lead to retaliatory action by humans
- **Breeding**: egg/sperm production, sperm motility/viability after freezing, artificial fertilization success, mating success, birth rate, litter size, calf/pup condition, 'overall recruitment'
- **Genetics**: genetic diversity, genetic suitability (e.g. adaptation to local conditions, use of correct flyways for migratory species, etc.)
- **Life history**: age/size at maturity, survival, mortality
- **Population measures**: number, abundance, density, presence/absence, biomass, movement, cover, age-structure, species distributions (only in response to a human action), disease prevalence, sex ratio

- **Community/habitat measures**: species richness, diversity measures (including trait/functional diversity), community composition, community structure (e.g. trophic structure), area covered (e.g. by different habitat types), physical habitat structure (e.g. rugosity, height, basal area)

**Actions** within the scope of Conservation Evidence include:

- Clear management actions: e.g. closing an area to fishing, modifying fishing gear to reduce bycatch, controlling invasive species, creating or restoring habitats
- International or national policies
- Reintroductions or management of wild species in captivity
- Actions that reduce human-wildlife conflict
- Actions that change human behaviour, resulting in an impact on wild taxa or habitats

See https://www.conservationevidence.com/data/index for more examples of actions.

Note on study types:

Literature reviews, systematic reviews, meta-analyses or short notes that review studies that fulfil these criteria were included.

Theoretical modelling studies were excluded, as no action has been taken. However, studies that use models to analyse real-world data, or compare models to real-world situations were included (if they otherwise fulfil these criteria).

**Criteria B: Conservation Evidence includes studies that measure the effect of an action that might be done to change human behaviour for the benefit of biodiversity**

1. Does this study measure the effect of an action that is or was under human control on human behaviour (actual or intentional) which is likely to protect, manage or restore wild taxa or habitats, or reduce threats to wild taxa or habitats? If yes, go to 2. If no, the study will be excluded.

2. Could the action be put in place by a conservationist, manager or decision maker to change human behaviour? If yes, the study will be included. If no, the study will be excluded.

**Explanation**:

1a. Study must have a measured outcome on *actual or intentional human behaviour* including self-reported behaviours: excludes outcomes on human psychology (tolerance, knowledge, awareness, attitude, perceptions or beliefs).

1b. Change in human behaviour must be linked to outcomes for wild taxa and habitats, excludes changes in behaviour linked to outcomes for human benefit, even if these occurred under a conservation program (e.g. we would exclude a study demonstrating increased school attendance in villages under a community-based conservation program).

1c. Action must be under human control: excludes impacts from climatic or other natural events.

2. Study must test an action that could be put in place for conservation: excludes studies with no action, e.g. correlating human personality traits with likelihood of conservation-related behaviours.

The human behaviour outcome of the study can be negative, neutral or positive, does not have to be statistically significant but must be quantified (if hard to judge from abstract, then it was included). It could be any behaviour that is likely to have an outcome on wild taxa and habitats (including mitigating the impact of invasive/problem taxon on wild taxa or habitats). Actions include, but are not limited to the following:

- Change in adverse behaviours (which directly threaten biodiversity) e.g. unsustainable fishing (industrial, artisanal or recreational), urban encroachment, creating noise, entering sensitive areas, polluting or dumping waste, clearing or habitat destruction, introducing invasive species
- Change in positive behaviours, e.g. uptake of alternative/ sustainable livelihoods, number of households adopting sustainable practices, donations

- Change in policy or conservation methods, e.g. placement of protected areas, protection of key habitats/species
- Change in consumer or market behaviour, e.g. purchasing, consuming, buying, willingness to pay, selling, illegal trading, advertising, consumer fraud
- Behavioural intentions to do any of the above

**Actions** which are particularly likely to have a behaviour change outcome include, but are not limited to the following:

- **Enforcement**: Closed seasons, size limits, fishing gear/hunting restrictions, auditable/traceable reporting requirements, market inspections, increase number of rangers, patrols or frequency of patrols in, around or within protected areas, improve fencing/physical barriers, improve signage, improve equipment/technology used by guards
- **Behaviour Change**: promote alternative/sustainable livelihoods, payment for ecosystem services, ecotourism, poverty reduction, increased appreciation or knowledge, debunking misinformation, altering or re-enforcing local taboos, financial incentives
- **Governance**: Protect or reward whistle-blowers, increase government transparency, ensure independence of judiciary, provide legal aid
- **Market Regulation**: trade bans, taxation, supply chain transparency laws
- **Consumer Demand Reduction**: Increase awareness or knowledge, fear appeals (negative association with undesirable product), benefit appeal (positive association with desirable behaviour), worldview framing, moral framing, employing decision defaults, providing decision support tools, simplifying advice to consumers, promoting desirable social norms, legislative prohibition
- **Sustainable Alternatives**: Certification schemes, captive bred or artificial alternatives, sustainable alternatives
- **New policies** for conservation/protection

We allocated studies to folders by their outcome. All studies under Criteria B went in the 'Behaviour change' folder. They were additionally duplicated into a taxon/habitat folder if there was a specific intended outcome of the behaviour change (if none mentioned, they were filed only in Behaviour change).

***c*) *Relevant subject***

Studies relevant to the synopsis subject included those focused on the conservation or restoration of coral reefs.

***d*) *Relevant types of action***

An action has to be one that could be put in place by a manager, conservationist, policy maker, advisor or consultant to protect, manage or restore coral reefs or reduce the impacts of threats to them. Alternatively, actions may aim to change human behaviour (actual or intentional), which is likely to protect, manage or restore coral reefs or reduce threats to them. See inclusion criteria above for further details.

If the following two criteria were met, a combined action was created within the synopsis, rather than duplicating evidence under all the separate actions: a) there were five or more publications that used the same well-defined combination of actions, with very clear description of what they were, without separating the effects of each individual action, and b) the combined set of actions is a commonly used conservation strategy.

***e*) *Relevant types of comparator***

To determine the effectiveness of actions, studies must include a comparison, i.e., monitoring change over time (typically before and after the action was implemented), or for example at treatment and control sites. Alternatively, a study could compare one specific action (or implementation method) against another. For example, this could be comparing a coral reef before and after the closure of an area to bottom trawling or measuring the effectiveness of coral restoration or 'gardening' using different types of structures.

Exceptions, which may not have a control but will still be included, are for example the effectiveness of ex-situ coral cultivation.

### *f*) *Relevant types of outcome*

Below we provide a list of anticipated metrics; others were included if reported within relevant studies.

- Community response
  - Community composition
  - Richness/diversity
  - Extent/cover
- Population response
  - Abundance/Cover: number, density, presence/absence, extent
  - Reproductive success: larvae production, overall recruitment, larvae settlement
  - Survival: survival, mortality, attachment
  - Condition: growth, size, condition factors, disease levels
  - Establishment of new growth/larval settlement on restored reef
- Other
  - Reef soundscape
  - Change in human behaviour

### *g*) *Relevant types of study design*

The table below lists the study designs included. The strongest evidence comes from randomized, replicated, controlled trials with paired sites and before-and-after monitoring.

Table 1. Study designs

| Term | Meaning |
|---|---|
| Replicated | The action was repeated on more than one individual or site. In conservation and ecology, the number of replicates is much smaller than it would be for medical trials (when thousands of individuals are often tested). If the replicates are sites, pragmatism dictates that between five and ten replicates is a reasonable amount of replication, although more would be preferable. We provide the number of replicates wherever possible. Replicates should reflect the number of times an action has been independently carried out, from the perspective of the study subject. For example, 10 plots within a mown field might be independent replicates from the perspective of plants with limited dispersal, but not independent replicates for larger motile animals such as birds. In the case of translocations/ release of captive bred animals, replicates should be sites, not individuals. |
| Randomized | The action was allocated randomly to individuals or sites. This means that the initial condition of those given the action is less likely to bias the outcome. |
| Paired sites | Sites are considered in pairs, within which one was treated with the action and the other was not. Pairs, or blocks, of sites are selected with similar environmental conditions, such as water quality or adjacent land use. This approach aims to reduce environmental variation and make it easier to detect a true effect of the action. |
| Controlled* | Individuals or sites treated with the action are compared with control individuals or sites not treated with the action. (The treatment is usually allocated by the investigators (randomly or not), such that the treatment or control groups/sites could have received the treatment). |
| Before-and-after | Monitoring of effects was carried out before and after the action was imposed. |
| Site comparison* | A study that considers the effects of actions by comparing sites that historically had different actions (e.g. action vs no action) or levels of action. Unlike controlled studies, it is not clear how the actions were allocated to sites (i.e. the investigators did not allocate the treatment to some of the sites). |

| Term | Meaning |
|---|---|
| Review | A conventional review of literature. Generally, these have not used an agreed search protocol or quantitative assessment of the evidence. |
| Systematic review | A systematic review follows structured, predefined methods to comprehensively collate and synthesize existing evidence. It must weigh or evaluate studies, in some way, according to the strength of evidence they offer (e.g. sample size and rigour of design). Environmental systematic reviews are available at: https://www.environmentalevidence.org/index.htm |
| Study | If none of the above apply, for example a study looking at the number of people that were engaged in an awareness raising project. Or a study measuring change over time in only one site and only after an action. |

* *Note that 'controlled' is mutually exclusive from 'site comparison'. A comparison cannot be both controlled and a site comparison. However, one study might contain both controlled and site comparison aspects, e.g. study of bycatch by fishers using modified nets (e.g. with a smaller mesh size) and unmodified nets (controlled), and fishers using an alternative net modification, e.g. stiffened nets (site comparison).*

## *Study quality assessment & critical appraisal*

We did not quantitatively assess the evidence from each publication or weight it according to quality. However, to allow interpretation of the evidence, we made the size and design of each study we reported clear. We critically appraised each potentially relevant study and excluded those that did not provide data for a comparison to the treatment, did not statistically analyse the results (or if included this was stated in the summary paragraph) or had obvious errors in their design or analysis. A record of the reason for excluding any of the publications was included during screening and kept within the synopsis database.

## *Data extraction*

Data on the effectiveness of the relevant action (e.g. mean species abundance inside or outside a protected area; reduction in bycatch after installation of a bycatch reduction device) was extracted from and summarized for publications that included the relevant subject, types of action, comparator and outcomes outlined above.

In addition to ensuring consistency/accuracy when screening publications for inclusion in the discipline-wide literature database (see above), when authors first began summarizing, the first 10 publications were sent to Conservation Evidence for editing. Further to this, relevant data were extracted by a member of the core Conservation Evidence team for a set of publications as well as the synopsis author to ensure agreement on the correct data and interpretation of the results for inclusion in the synopsis. In addition, summaries were also swapped between authors on a semi-regular basis to quality control the paragraphs that were being written.

## *Evidence synthesis*

### *a*) *Summary protocol*

Each publication usually has just one paragraph for each action it tests describing the study in (usually) no more than 150 words using plain English, though more complex studies required longer summaries. Each summary is in the following format:

A [TYPE OF STUDY] in [YEARS X–Y] in [HOW MANY SITES] in/of [HABITAT] in [REGION and COUNTRY] [REFERENCE] found that [ACTION] [SUMMARY OF ALL KEY RESULTS] for [SPECIES/HABITAT TYPE]. [DETAILS OF KEY RESULTS, INCLUDING DATA]. In addition, [EXTRA RESULTS, IMPLEMENTATION OPTIONS, CONFLICTING RESULTS]. The [DETAILS OF EXPERIMENTAL DESIGN, ACTION METHODS and KEY DETAILS OF SITE CONTEXT]. Data were collected in [DETAILS OF SAMPLING METHODS].

Type of study — use terms and order in Table 1.

Site context — for the sake of brevity, only nuances essential to the interpretation of the results are included. The reader is always encouraged to read the original source to get a full understanding of the study site (e.g. history of management, physical conditions, landscape context etc.).

**For example:**

A replicated, controlled study in 2008–2009 at two coral reef sites near Guana Island, British Virgin Islands (1) found that removing macroalgae

from the transplant site for storm-generated fragments of elkhorn *Acropora palmata* coral led to a higher increase in live tissue growth but no difference in survival compared to fragments transplanted without algae removal. One year after transplanting, the increase in live tissue surface area was higher on fragments where algae had been removed (160%) than fragments transplanted without algae clearance (68%). Survival of fragments after one year did not vary significantly (algae cleared: 52%; algae not cleared: 60% survival). In July – August 2008, a total of 237 storm-generated fragments of elkhorn coral were collected from a coral reef and prepared for transplantation either at the collection site or another site 0.4–3.6 km away. Fragments were attached to the reef substrate, or dead elkhorn coral skeletons, using cable ties, marine epoxy or cement and ensuring live tissue was in contact with the substrate. Once attached, macroalgae was scraped away from a circle of 20 cm radius around 117 of the 237 fragments. Growth (surface area of live tissue) was measured after two and 12 months, and survival was recorded after 12 months using photographs.

(1) Forrester G.E., O'Connell-Rodwell C., Baily, P., Forrester L.M., Giovannini S., Harmon L., Karis R., Krumholz, J., Rodwell T. & Jarecki L. (2011), Evaluating methods for transplanting endangered Elkhorn Corals in the Virgin Islands. *Restoration Ecology*, 19, 299–306. https://doi.org/10.1111/j.1526-100X.2010.00664.x

A replicated, paired, site comparison study in 2002 of two coastal coral reefs in the Philippines (2) found that establishing a marine reserve closed to fishing resulted in higher density and biomass of species of fish taken by local fishers within the reserve compared to a fished area in one of two cases. For species taken by fishers, density and biomass inside reserve one was higher (density: 68 fish/500 $m^2$; biomass: 89 kg) than outside (27/500 $m^2$; 25 kg), but not significantly different inside and outside reserve two (density inside and outside: 41/500 $m^2$; no biomass data provided). For fish species not subject to fishing, density was higher inside both reserves compared to outside; however, statistical tests showed this was mainly due to habitat variation not protection status (reserve one: 146 fish/250 $m^2$ inside, 113/250 $m^2$ outside; reserve two: 93/250 $m^2$ inside, 32/250 $m^2$ outside). No-take reserves approximately 450 m long (protected for 20 years) and 650 m long (protected for 15 years) off two islands were each compared to fished areas approximately

500 m away. Fish were surveyed in November and December 2002. Divers surveyed fish at six (reserve one) and eight (reserve two) coral reef slope sites inside and outside each reserve. Counts were along 50 × 10 m transects for fish taken by fishers and 50 × 5 m transects for fish not fished. Transects were surveyed twice.

(2) Abesamis R.A., Russ G.A. & Alcala A.C. (2006) Gradients of abundance of fish across no-take marine reserve boundaries: Evidence from Philippine coral reefs. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 16, 349–371. https://doi.org/10.1002/aqc.730

***b*) *Terminology used to describe the evidence***

Unless specifically stated otherwise, results reflect statistical tests performed on the data, i.e. we only state that there was a difference if it was supported by the statistical test used, and otherwise state that there was no difference or that outcomes were similar. If there was a good reason to report differences between treatments and controls that were not tested for statistical significance, it was made clear within the summary that statistical tests were not carried out. Table 1 above defines the terms used to describe the study designs.

***c*) *Dealing with multiple actions within a publication***

When separate results were provided for the effects of each of the different actions tested, separate summaries were written under each action heading. However, when several actions were carried out at the same time and only the combined effect reported, the result was described with a similar paragraph under all relevant actions. In these circumstances, we clearly communicated within the summary paragraph where multiple actions were used in combination. For example, the first sentence would articulate that a combination of actions was carried out, i.e. '...(REF) found that [x action], along with [y] and [z actions] resulted in [describe effects]'.

***d*) *Dealing with multiple publications reporting the same results and reviews***

If two publications described results from the same action implemented in the same space and at the same time, we only included the most stringently peer-reviewed publication (i.e. journal of the highest impact

factor). If one included initial results (e.g. after year one) of another (e.g. after 1–3 years), we only included the publication covering the longest time span. If two publications described at least partially different results, we included both but made clear they were from the same project in the paragraph, e.g. 'A controlled study... (Gallagher *et al*. 1999; same experimental set-up as Oasis *et al*. 2001)...'.

*e*) ***Taxonomy***

Taxonomy was not updated but follows that used in the original publication. Where possible, common names and scientific names were both given the first time each species was mentioned within each summary.

*f*) ***Key messages***

Each action has a set of concise, bulleted key messages at the top, written once all the literature had been summarized. These include information such as the number, design and location of studies included. The first bullet point describes the total number of studies that tested the action and the locations of the studies, followed by key information on the relevant metrics presented under the headings and sub-headings shown below (with number of relevant studies in parentheses for each).

- **X studies** examined the effects of [ACTION] on [TARGET POPULATION]. Y studies were in [LOCATION 1][1,2] and Z studies were in [LOCATION 2][3,4].
    - *Locations will usually be countries, ordered based on chronological order of studies rather than alphabetically, i.e. 'the USA1, Australia2' rather than 'Australia2, the USA1'. However, when more than 4–5 separate countries, they may be grouped into regions to make it clearer e.g. Europe, North America. The distribution of studies amongst habitat types may also be added here if relevant.*

**COMMUNITY COMPOSITION (x STUDIES)**

- **Richness/diversity (x studies):**

**POPULATION RESPONSE (x STUDIES)**

- **Abundance/Cover (x studies):**

- **Reproductive success (x studies):**
- **Survival (x studies):**
- **Condition (x studies):**

**OTHER (x STUDIES)** (*Included only for actions/chapters where relevant*)

- **[Sub-heading(s) for the metric(s) reported will be created] (x studies)**: If no suitable studies were found for an action, the following text was added in place of the key messages above:
- We found no studies that evaluated the effects of [ACTION] on [TARGET POPULATION].

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

### *g) Background information*

Background information for an action is provided to describe the action and where we feel recent knowledge is required to interpret the evidence. This is presented after the key messages, and relevant references are included in a reference list at the end of the Background section. In some cases, where a body of literature has strong implications for coral conservation, but does not directly test actions for their effects, we may also refer the reader to this literature in the background sections.

## *Dissemination/communication of evidence synthesis*

The information from this evidence synthesis will be available in three ways:

- A synopsis pdf, downloadable from https://www.conservationevidence.com, containing the study summaries, key messages and background information on each action.
- The searchable database at https://www.conservationevidence.com containing all the summarized information from the synopsis, along with expert assessment scores.

- A chapter in What Works in Conservation, available as a pdf to download and a book from https://www.conservationevidence.com/content/page/79, containing key messages from the synopsis as well as expert assessment scores on the effectiveness and certainty of the synopsis, with links to the online database.

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# 2. Threat: Residential and commercial development

**Background**

Threats from residential and commercial development are related to the development of land adjacent to coral reefs, and the reclamation of reefs to make space for new developments. In addition, new developments can be associated with threats from extraction of aggregates (e.g. sand) or corals (e.g. coral rock or live coral) for building materials, increased pollution and sedimentation, and impacts of transportation and service corridors. Actions in response to these threats are described in the following chapters: *Habitat protection*, *Habitat restoration and creation*, *Threat: Biological resource use*, *Threat: Pollution* and *Threat: Transportation and service corridors*.

## 2.1 Prohibit or limit residential or commercial development on coasts

https://www.conservationevidence.com/actions/4018

- We found no studies that evaluated the effects on corals of prohibiting or limiting residential or commercial development on coasts.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*


https://doi.org/10.11647/OBP.0453.02

**Background**

Coastal development, including buildings and other infrastructure, poses a range of threats to corals, from the direct threats of dredging and land reclamation to more indirect threats of increased pollution and sedimentation (Burke & Maidens 2004). Prohibiting or limiting the extent of coastal developments could help reduce the impact of these threats.

Burke L.M. & Maidens J. (2004) *Reefs at Risk in the Caribbean*. World Resources Institute: Washington, DC. Available from: https://www.wri.org/research/reefs-risk-caribbean

## 2.2 Prohibit or limit landfilling of reef flats for land reclamation

https://www.conservationevidence.com/actions/4019

- We found no studies that evaluated the effects on corals of prohibiting or limiting landfilling of reef flats for land reclamation.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Landfilling of reef flats for the purposes of land reclamation can directly impact reefs by reducing the extent of coral reefs, increasing reef fragmentation, and altering coastal dynamics (e.g. long-shore drift) that can lead to increased sedimentation and burial of reefs (Valadez-Rocha *et al*. 2013). Subsequent developments on reclaimed land may lead to additional threats to reefs, including increased pollution (Burke & Maidens 2004).

Burke L.M. & Maidens J. (2004) *Reefs at Risk in the Caribbean*. World Resources Institute: Washington, DC. Available from: https://www.wri.org/research/reefs-risk-caribbean

Valadez-Rocha V. & Ortiz-Lozano L. (2013) Spatial and temporal effects of port facilities expansion on the surface area of shallow coral reefs. *Environmental Management*, 52, 250–260. https://doi.org/10.1007/s00267-013-0098-5

# 3. Threat: Aquaculture & agriculture

**Background**

Marine aquaculture (also known as mariculture) is the farming of fish, crustaceans, molluscs, algae and other organisms under controlled conditions in the marine environment. Aquaculture facilities can cause increased levels of nutrient pollution and microorganisms (Becker *et al*. 2017). Practices such as sea cage farming are often sited close to reefs (Hedberg *et al*. 2015, Hedberg *et al*. 2017) and have been associated with reduced coral cover and increased coverage of turf algae (Hedberg *et al*. 2015).

Land agriculture can lead to nutrient-rich and pesticide-rich run-offs reaching the marine environment through rivers, and negatively impacting coastal areas due to the increase in nutrients such as nitrogen and phosphorous (Falace *et al*. 2018; Gabric & Bell 1993). These increases in nutrients often lead to diminished water quality and eutrophication events including hypoxia or anoxia, creating "dead zones" (Breitburg *et al*. 2018).

Much of the conservation effort related to threats from aquaculture and agriculture has been directed at reducing the impacts of pollution and impoverished water quality, as well as reducing the threat from non-native and invasive species. Actions related to these threats are described in *Threat: Pollution* and *Threat: Non-native, invasive and problematic species*


https://doi.org/10.11647/OBP.0453.03

Becker C., Hughen K., Mincer T.J., Ossolinski J., Weber L. & Apprill A. (2017) Impact of prawn farming effluent on coral reef water nutrients and microorganisms. *Aquaculture Environment Interactions*, 9, 331–346. https://doi.org/10.3354/aei00238

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Falace A., Tamburello L., Guarnieri G., Kaleb, S., Papa L. & Fraschetti S. (2018) Effects of a glyphosate-based herbicide on *Fucus virsoides* (Fucales, Ochrophyta) photosynthetic efficiency. *Environmental Pollution*, 243, 912–918. https://doi.org/10.1016/j.envpol.2018.08.053

Gabric A.J. & Bell P.R.F. (1993) Review of the effects of non-point nutrient loading on coastal ecosystems. *Marine and Freshwater Research*, 44, 261–283. https://doi.org/10.1071/MF9930261

Hedberg N., Kautsky N., Hellström M. & Tedengren M. (2015) Spatial correlation and potential conflicts between sea cage farms and coral reefs in South East Asia. *Aquaculture*, 448, 418–426. https://doi.org/10.1016/j.aquaculture.2015.06.024

Hedberg N., Stenson I., Kautsky N., Hellström M. & Tedengren M. (2017) Causes and consequences of spatial links between sea cage aquaculture and coral reefs in Vietnam. *Aquaculture*, 481, 245–254. https://doi.org/10.1016/j.aquaculture.2017.09.009

# 4. Threat: Energy production and mining

**Background**

Energy production (renewable and non-renewable), mining (for minerals), quarrying, and aggregate extraction has the potential to significantly impact corals and the reefs they form through the direct modification of coastal and benthic habitats and through the resulting pollution caused by such activities (Martinez-Escobar & Mallela 2019). Additional threat arises from the spread of non-native and invasive species colonizing offshore infrastructures associated with these activities.

Actions related to recreating or re-establishing natural habitats following activities or related to modifying or repurposing infrastructure as artificial habitats (Langhamer 2012) are described in the chapter *Habitat restoration and creation*. Actions related to pollution emanating from energy production and mining, including noise generation, are described in *Threat: Pollution*. Actions related to the direct extraction of corals and coral rock are described in *Threat: Biological resources use*. Actions related to the introduction and spread of non-native, invasive or problematic species due to the "stepping stones" effects associated with installations and anthropogenic structures are described in *Threat: Non-native, invasive and problematic species*.


https://doi.org/10.11647/OBP.0453.04

Langhamer O. (2012) Artificial reef effect in relation to offshore renewable energy conversion: State of the art. *The Scientific World Journal*, 386713. https://doi.org/10.1100/2012/386713

Martinez-Escobar D.F. & Mallela J. (2019) Assessing the impacts of phosphate mining on coral reef communities and reef development. *Science of the Total Environment*, 692, 1257–1266. https://doi.org/10.1016/j.scitotenv.2019.07.139

# Oil and gas drilling

## 4.1 Prohibit or limit oil and gas drilling near coral reefs

https://www.conservationevidence.com/actions/4021

- We found no studies that evaluated the effects on corals of prohibiting or limiting oil and gas drilling near coral reefs.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Routine oil and gas drilling activities can impact corals and the reefs they form due to smothering and burial from drill cuttings and drill fluids, pollution from the use of chemicals and additives, physical damage, or loss of suitable natural sediment (Cordes *et al*. 2016). Establishing new drilling sites can cause permanent damage to corals through the installation of fixed infrastructure, e.g., laying pipelines or from transient disturbance such as anchor damage from transport vessels (Cordes *et al*. 2016). Ceasing on-going oil and gas drilling, for instance following protective legislation or the non-renewal of permit, can stop the threat and potentially allow corals to recover over time.

Cordes E.E., Jones D.O., Schlacher T.A., Amon D.J., Bernardino A.F., Brooke S., Carney R., DeLeo D.M., Dunlop K.M., Escobar-Briones E.G. & Gates A.R. (2016) Environmental impacts of the deep-water oil and gas industry: a review to guide management strategies. *Frontiers in Environmental Science*, 4, 58. https://doi.org/10.3389/fenvs.2016.00058

## 4.2 Prohibit or limit the deposition or disposal of drill cuttings near coral reefs

https://www.conservationevidence.com/actions/4022

- We found no studies that evaluated the effects on corals of prohibiting or limiting the deposition or disposal of drill cuttings near coral reefs.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Drill cuttings are fragments of rock produced during drilling. Usually, these are deposited on the seafloor forming a cuttings pile (Cordes *et al*. 2016). Cuttings can vary in size and can be as small as ~ 2 μm depending on the substrate being drilled. Small cuttings could be carried in the water column leading to deposition in shallower waters away from the drill site. Ensuring these cuttings are not deposited near coral reefs will reduce the potential for contamination or smothering of the reef surface.

Cordes E.E., Jones D.O., Schlacher T.A., Amon D.J., Bernardino A.F., Brooke S., Carney R., DeLeo D.M., Dunlop K.M., Escobar-Briones E.G. & Gates A.R. (2016) Environmental impacts of the deep-water oil and gas industry: A review to guide management strategies. *Frontiers in Environmental Science*, 4, 58. https://doi.org/10.3389/fenvs.2016.00058

## 4.3 Use water-based drilling fluids and recycle or repurpose drilling fluids

https://www.conservationevidence.com/actions/4024

- We found no studies that evaluated the effects on corals of using water-based drilling fluids and recycling or repurposing drilling fluids.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Fluids used in oil and gas drilling (also known as 'muds') perform a variety of functions during the drilling process including providing pressure, cooling and cleaning the drill, lubrication, transporting drill cuttings, and limiting corrosion (Cordes *et al*. 2016). Waste fluids (such as drilling mud) that may contain cement or other chemicals can cause damage to the coral reef system (Cordes *et al*. 2016). Historically, these muds were made from oil but discharging oil-based muds into the water column is prohibited in the OSPAR region (OSPAR Commission 2000). In recent years, there has been a move away from more toxic oil-based fluids to more synthetic or water-based. Although these are less toxic, they still pose a threat to coral reefs through contamination of the water column and substrate (Cordes *et al*. 2016)

Cordes E.E., Jones D.O., Schlacher T.A., Amon D.J., Bernardino A.F., Brooke S., Carney R., DeLeo D.M., Dunlop K.M., Escobar-Briones E.G. & Gates A.R. (2016) Environmental impacts of the deep-water oil and gas industry: A review to guide management strategies. *Frontiers in Environmental Science*, 4, 58. https://doi.org/10.3389/fenvs.2016.00058

OSPAR Commission (2000) *OSPAR Decision 2000/3 on the Use of Organic-phase Drilling Fluids (OPF) and the Discharge of OPF-Contaminated Cuttings*. OSPAR 00/20/1-E, Annex 18. Available from: https://www.ospar.org/work-areas/oic

## 4.4 Contain sediment during drilling

https://www.conservationevidence.com/actions/4025

- We found no studies that evaluated the effects on corals of containing sediment during drilling.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Drilling activites disturb the seabed and can lead to increased sedimentation. Sediment particles (sometimes known as 'marine snow') can fall and smother coral reefs in the immediate vicinity of drilling. Smaller particles can be suspended within the water column and transported away from the drilling site before being deposited on coral reefs (Hayward *et al*. 2016). Whether on the surface or suspended in the water column, sediment particles can impact coral reefs by smothering the delicate coral polyps or reducing light thereby reducing coral growth (Fabricius & Wolanski 2000). Actions to contain sediment and prevent its spread could include the use of 'sediment curtains' around drilling sites.

Fabricius K.E. & Wolanski E. (2000) Rapid smothering of coral reef organisms by muddy marine snow. *Estuarine, Coastal and Shelf Science*, 50, 115–120. https://doi.org/10.1006/ecss.1999.0538

Haywood M.D.E., Denis D.P., Thomson D.P. & Pillans R.D. (2016) Environmental impacts of the deep-water oil and gas industry: A review to guide management strategies. *Frontiers in Environmental Science*, 4. https://doi.org/10.3389/fenvs.2016.00058

## 4.5 Prohibit or limit or modify rock dumping

https://www.conservationevidence.com/actions/4026

- We found no studies that evaluated the effects on corals of prohibiting or limiting or modifying rock dumping.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Waste rocks from mining or drilling are often left on the seabed (dumped). Mining waste (known as tailings) from terrestrial mining activities are often dumped at sea. If these are disposed of in shallow waters, rock dumping can impact coral reefs by reducing areas of suitable habitat for larvae to settle on (Haywood *et al*. 2016) and increasing sediment in the water column that may settle on and smother corals or reduce light availablity (Fabricius & Wolanski 2000). Even if the rock dumping was carried out away from coral reefs, the sediment may be suspended in the water column and transported hundreds of kilometres away before settling on coral reefs (Fabricius & Wolanski 2000).

Fabricius K.E. & Wolanski E. (2000) Rapid smothering of coral reef organisms by muddy marine snow. *Estuarine, Coastal and Shelf Science*, 50, 115–120. https://doi.org/10.1006/ecss.1999.0538

Haywood M.D.E., Denis D.P., Thomson D.P. & Pillans R.D. (2016) Environmental impacts of the deep-water oil and gas industry: A review to guide management strategies. *Frontiers in Environmental Science*, 4. https://doi.org/10.3389/fenvs.2016.00058

## 4.6 Remove pipelines, stabilization material and infrastructure following decommissioning

https://www.conservationevidence.com/actions/4027

- We found no studies that evaluated the effects on corals of removing pipelines, stabilization material and infrastructure following decommissioning.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Removing pipelines, stabilization material and infrastructure from the vicinity of coral reefs once a drilling site has been decommissioned can enable the reef to recover over time (Cordes *et al*. 2016). Such structures will likely have been in place for some time, potentially acting as an artificial reef and being colonized by corals (Cordes *et al*. 2016). There would, however, be some initial disturbance from the removal leading to the destruction of any corals growing on the pipeline and an increase in sediment on the coral reef or in the water column.

Cordes E.E., Jones D.O., Schlacher T.A., Amon D.J., Bernardino A.F., Brooke S., Carney R., DeLeo D.M., Dunlop K.M., Escobar-Briones E.G. & Gates A.R. (2016) Environmental impacts of the deep-water oil and gas industry: A review to guide management strategies. *Frontiers in Environmental Science*, 4, 58. https://doi.org/10.3389/fenvs.2016.00058

## 4.7 Leave pipelines, stabilization material and infrastructure in place following decommissioning

https://www.conservationevidence.com/actions/4028

- We found no studies that evaluated the effects on corals of leaving pipelines, stabilization material and infrastructure in place following decommissioning.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Leaving pipelines, stabilization material and infrastructure in place once a drilling site has been decommissioned can reduce additional damage to the reef and surrounding seabed (Cordes *et al*. 2016). Such structures will likely have been in place for some time potentially acting as an artificial reef and colonized by corals (Cordes *et al*. 2016). However, leaving these man-made structures on the seabed could lead to an increase in pollution as the material degrades and prevents the natural reef re-establishing (Cordes *et al*. 2016).

Cordes E.E., Jones D.O., Schlacher T.A., Amon D.J., Bernardino A.F., Brooke S., Carney R., DeLeo D.M., Dunlop K.M., Escobar-Briones E.G. & Gates A.R. (2016) Environmental impacts of the deep-water oil and gas industry: A review to guide management strategies. *Frontiers in Environmental Science*, 4, 58. https://doi.org/10.3389/fenvs.2016.00058

# Mining and quarrying

## 4.8 Prohibit or limit or modify aggregate extraction

https://www.conservationevidence.com/actions/4029

- We found no studies that evaluated the effects on corals of prohibiting or limiting or modifying aggregate extraction.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Extraction of aggregates (including sand) for use in construction can impact coral reefs near the extraction site. Seabed disturbance caused by aggregate extraction can increase the quantity of sediment in the water column which can settle and smother corals or, if remaining in the water column, can reduce light availability to corals thereby restricting growth (Fabricus & Wolanski 2000).

This action does not refer to extraction of corals (either 'coral rock' or live corals) for use as building material. Actions relating to this threat are covered in *Threat: Biological resource use*.

Fabricius K.E. & Wolanski E. (2000) Rapid smothering of coral reef organisms by muddy marine snow. *Estuarine, Coastal and Shelf Science*, 50, 115–120. https://doi.org/10.1006/ecss.1999.0538

# 5. Threat: Transportation and service corridors

**Background**

Threats from transportation and service corridors include infrastructures such as shipping, roads, utility and service lines (e.g. communication and power cables). The greatest threats to corals from transportation and service corridors are habitat destruction, increased sediment in the water column, and pollution. Actions in response to these threats are covered in *Habitat restoration and creation*; and *Threat: Pollution*. Actions in response to threats from oil and gas pipelines are covered in *Threat: Energy production and mining: oil and gas drilling*.

## Shipping

### 5.1 Limit, cease, prohibit, or divert shipping

https://www.conservationevidence.com/actions/4030

- We found no studies that evaluated the effects on corals of ceasing, prohibiting, or diverting shipping.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*


https://doi.org/10.11647/OBP.0453.05

**Background**

Ships/boats/vessels passing near coral reefs can impact corals. Larger ships can create excessive bow waves which can lead to coral breaking and being unable to reattach to the substrate. A large ship running aground can devastate extensive areas of coral reef leaving damage that, without intervention, can take decades to recover from (Griffin *et al*. 2015). Actions in response to the threat of pollution from shipping are detailed in *Threat: Pollution*. Actions for restoring coral reefs after a ship has run aground or sunk are detailed in *Habitat restoration and creation* and *Species management*.

Griffin S.P., Nemeth M.I. Moore T.D. & Gintert B. (2015) Restoration using *Acropora cervicornis* at the T/V MARGARA grounding site. *Coral Reefs*, 34, 885–885. https://doi.org/10.1007/s00338-015-1310-2

## 5.2 Limit, cease, or prohibit anchoring, from ships/boats/vessels or change anchoring method

https://www.conservationevidence.com/actions/4031

- We found no studies that evaluated the effects on corals of limiting, ceasing, or prohibiting anchoring, or changing anchoring method.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Ships/boats/vessels anchoring near coral reefs can impact corals. The anchor itself can damage the seabed, and the chain can scour the substrate. Coral reef density, size and diversy are lower in areas with high levels of anchoring (Flynn & Forrester 2019). Creating 'no anchoring' zones and prohibiting or limiting anchoring near coral reefs could reduce the threat. Alternatively, different types of anchors could be used to reduce the threat, or chains could be modified to avoid excess chain scouring the seabed (Flynn & Forrester 2019). Technology could provide a solution with ships using computer-controlled systems to maintain the position of vessels rather than anchoring. Actions for restoring coral reefs after damage caused by anchoring are detailed in *Habitat restoration and creation* and *Species management*.

Flynn R.L. & Forrester G.E. (2019) Boat anchoring contributes substantially to coral reef degradation in the British Virgin Islands. *PeerJ* 7, e7010. https://doi.org/10.7717/peerj.7010

## 5.3 Provide fixed moorings to reduce anchoring

https://www.conservationevidence.com/actions/4033

- We found no studies that evaluated the effects on corals of providing and moving moorings.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Ships/boats/vessels anchoring near coral reefs can impact corals. The anchor itself can damage the seabed, and the chain can scour the substrate. Coral reef density, size and diversy are lower in areas with high levels of anchoring (Flynn & Forrester 2019). Providing suitable moorings could reduce the number of anchorings (Forrester 2020). Actions for restoring coral reefs after damage caused by anchoring are detailed in *Habitat restoration and creation* and *Species management*.

Flynn R.L. & Forrester G.E. (2019) Boat anchoring contributes substantially to coral reef degradation in the British Virgin Islands. *PeerJ* 7, e7010. https://doi.org/10.7717/peerj.7010

Forrester G.E. (2020) The influence of boat moorings on anchoring and potential anchor damage to coral reefs. *Ocean & Coastal Management*, 198, 105354. https://doi.org/10.1016/j.ocecoaman.2020.105354

# Roads and railroads

## 5.4 Limit development of major roads on coasts near coral reefs

https://www.conservationevidence.com/actions/4034

- We found no studies that evaluated the effects on corals of limiting development of major roads on coasts.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Coastal road building can lead to increased airborne dust. These particles can get blown into the sea, settle and smother coral reefs in the area. Smaller particles can become suspended within the water column and transported away from the development site before being deposited on coral reefs (Tuttle *et al*. 2020). Whether on the surface or suspended in the water column, dust particles can impact coral reefs by smothering the delicate coral polyps or reducing light, thereby reducing coral growth (Tuttle *et al*. 2020). Increased run-off of chemicals, nutrients, and microplastics caused by tyre wear can affect water quality around coral reefs (Tuttle *et al*. 2020). Actions in response to the threat of pollution from road development are detailed in *Threat: Pollution*.

Tuttle L.J., Johnson C., Kolinski S. Minton D. & Donahue M.J. (2020) How does sediment exposure affect corals? A systematic review protocol. *Environmental Evidence*, 9, 17. https://doi.org/10.1186/s13750-020-00200-0

# Utility and service lines

## 5.5 Prohibit or limit new utility or service lines near coral reefs

https://www.conservationevidence.com/actions/4035

- We found no studies that evaluated the effects on corals of prohibiting or limiting new utility and service lines.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

As the drive for global technological development and renewable energy increases, it is likely that there will be a need for more submarine utility and service lines. The installation of new utility and service lines on or near coral reefs can impact corals through damage and habitat loss. Installing new cables can disturb the soft substrate and lead to sediment in the water column which can smother corals or reduce light penetration to the reef (Tuttle *et al*. 2020). Cables installed on coral reefs can damage the coral colonies and reduce the available habitat for coral larvae to settle on. If the cables become damaged, actions to repair them could impact the reef by machinery or anchors damaging the substrate (Flynn & Forrester 2019). Limiting the number of new utility or service lines, prohibiting their installation on coral reefs, or diverting utility and service lines around coral reefs could reduce the potential impact.

Flynn R.L. & Forrester G.E. (2019) Boat anchoring contributes substantially to coral reef degradation in the British Virgin Islands. *PeerJ* 7, e7010. https://doi.org/10.7717/peerj.7010

Tuttle L.J., Johnson C., Kolinski S. Minton D. & Donahue M.J. (2020) How does sediment exposure affect corals? A systematic review protocol. *Environmental Evidence*, 9, 17. https://doi.org/10.1186/s13750-020-00200-0

## 5.6 Remove utility and service lines after decommissioning

https://www.conservationevidence.com/actions/4036

- We found no studies that evaluated the effects on corals of removing utility and service lines after decommissioning.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Utility and service lines can impact coral reefs through damage and habitat loss. Removing these after decommissioning can allow the coral reef to recover through natural settlement of larvae. However, utility and service lines are likely to have been in place for some time so their removal could result in some initial disturbance and damage to the reef from the removal and increases in sediment on the coral reef or in the water column (Tuttle *et al*. 2020).

Tuttle L.J., Johnson C., Kolinski S. Minton D. & Donahue M.J. (2020) How does sediment exposure affect corals? A systematic review protocol. *Environmental Evidence*, 9, 17. https://doi.org/10.1186/s13750-020-00200-0

## 5.7 Leave utility and service lines in place after decommissioning

https://www.conservationevidence.com/actions/4037

- We found no studies that evaluated the effects on corals of leaving utility and service lines in place after decommissioning.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Utility and service lines can impact coral reefs through damage and habitat loss. Leaving these in place could reduce the risk of further disturbance to the coral reef associated with removing lines and allowing the lines to be settled by coral larvae.

# 6. Threat: Biological resource use

**Background**

Biological resource use includes the deliberate extractive use of corals through collection for jewellery, trinkets, and curios, and for the aquarium trade (Bruckner 2009; Montero-Serra *et al*. 2015); excavation of coral rock for building material (Brown & Dunne 1988); and the unintentional damage caused to corals and coral reefs by fishing gear targeting other fisheries (Mangi & Roberts 2006, Althaus *et al*. 2009). The threats to corals from lost or abandoned fishing gear arise from activities targeting other fisheries; actions related to this threat are described in *Threat: Pollution*.

Althaus F., Williams A., Schlacher T.A., Kloser R.J., Green M.A., Barker B.A., Bax N.J., Brodie P. & Schlacher-Hoenlinger M.A. (2009) Impacts of bottom trawling on deep-coral ecosystems of seamounts are long-lasting. *Marine Ecology Progress Series*, 397, 279–294. https://doi.org/10.3354/meps08248

Brown B.E. & Dunne R.P. (1988) The environmental impact of coral mining on coral reefs in the Maldives. *Environmental Conservation*, 15, 159–165. https://doi.org/10.1017/s0376892900028976

Bruckner A.W. (2009) Rate and extent of decline in Corallium (pink and red coral) populations: Existing data meet the requirements for a CITES Appendix II listing. *Marine Ecology Progress Series*, 397, 319–332. https://doi.org/10.3354/meps08110

Mangi S.C. & Roberts C.M. (2006) Quantifying the environmental impacts of artisanal fishing gear on Kenya's coral reef ecosystems. *Marine Pollution Bulletin*, 52, 1646–1660. https://doi.org/10.1016/j.marpolbul.2006.06.006

 https://doi.org/10.11647/OBP.0453.06

Montero-Serra I., Linares C., García M., Pancaldi F., Frleta-Valić M., Ledoux J.B., Zuberer F., Merad D., Drap P. & Garrabou J. (2015) Harvesting effects, recovery mechanisms, and management strategies for a long-lived and structural precious coral. *PloS One*, 10, e0117250. https://doi.org/10.1371/journal.pone.0117250

## 6.1 Limit, cease or prohibit all types of fishing (outside protected areas)

https://www.conservationevidence.com/actions/4063

- We found no studies that evaluated the effects on corals of ceasing or prohibiting all types of fishing (outside protected areas).

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Fishing can have both direct and indirect impacts on corals. Fishing poses a direct threat when gear damages corals during operations (Mangi & Roberts 2006, Althaus *et al*. 2009) or is lost or abandoned (so called 'ghost' gear; Ballesteros *et al*. 2018, Figueroa-Pico *et al*. 2020). Indirect threats can emerge when overfishing reduces the diversity and abundance of herbivorous fishes, with consequences for macroalgal abundance and its potential to impact on coral growth (Burkepile *et al*. 2010).

Prohibiting all fishing has the potential to reduce these threats and allow corals and the reefs they form to recover and flourish. Restricted fishing types covered here include hook and line fishing; bottom trawling, dredging and other towed gear; static gear including traps; and dynamite and cyanide fishing.

Where fishing was stopped or prohibited within a protected area, evidence is summarized under *Habitat protection: Designate a Marine Protected Area and prohibit all types of fishing; Designate a Marine Protected Area and prohibit all types of fishing and collection* and *Designate a Marine Protected Area and prohibit all types of fishing, collecting and access*.

Althaus F., Williams A., Schlacher T.A., Kloser R.J., Green M.A., Barker B.A., Bax N.J., Brodie P. & Schlacher-Hoenlinger M.A. (2009) Impacts of bottom trawling on deep-coral ecosystems of seamounts are long-lasting. *Marine Ecology Progress Series*, 397, 279–294. https://doi.org/10.3354/meps08248

Ballesteros L. V., Matthews J. L. & Hoeksema B. W. (2018) Pollution and coral damage caused by derelict fishing gear on coral reefs around Koh Tao, Gulf of Thailand. *Marine Pollution Bulletin*, 135, 1107–1116. https://doi.org/10.1016/j.marpolbul.2018.08.033

Burkepile D.E. & Hay M.E. (2010) Impact of herbivore identity on algal succession and coral growth on a Caribbean reef. *PloS One*, 5, e8963. https://doi.org/10.1371/journal.pone.0008963

Figueroa-Pico J., Tortosa F.S. & Carpio A.J. (2020) Coral fracture by derelict fishing gear affects the sustainability of the marginal reefs of Ecuador. *Coral Reefs*, 39, 819–827. https://doi.org/10.1007/s00338-020-01926-6

Mangi S.C. & Roberts C.M. (2006) Quantifying the environmental impacts of artisanal fishing gear on Kenya's coral reef ecosystems. *Marine Pollution Bulletin*, 52, 1646–1660. https://doi.org/10.1016/j.marpolbul.2006.06.006

## 6.2 Limit, cease or prohibit commercial fishing (outside protected areas)

https://www.conservationevidence.com/actions/4064

- We found no studies that evaluated the effects on corals of ceasing or prohibiting commercial fishing (outside protected areas).

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Commercial fishing is extraction of marine organisms by any method for sale and profit. Commercial fishing gear – including bottom trawlers, longlines, crab and fish pots – can damage corals during operations, with trawling being particularly damaging (Althaus *et al*. 2009, Heifetz *et al*. 2009). Lost or abandoned fishing gear (so called 'ghost' gear) also poses a threat (Ballesteros *et al*. 2018, Figueroa-Pico *et al*. 2020). Indirect threats can emerge when overfishing reduces the diversity and abundance of herbivorous fishes, with consequences for macroalgal abundance and its potential to impact coral growth (Burkepile *et al*. 2010).

Prohibiting commercial fishing has the potential to reduce these threats and allow corals and the reefs they form to recover and flourish.

When commercial fishing was stopped or prohibited within a protected area, evidence is summarized under *Habitat protection: Designate a Marine Protected Area and prohibit some fishing and collection* (*including where restrictions are unspecified*).

Althaus F., Williams A., Schlacher T.A., Kloser R.J., Green M.A., Barker B.A., Bax N.J., Brodie P. & Schlacher-Hoenlinger M.A. (2009) Impacts of bottom trawling on deep-coral ecosystems of seamounts are long-lasting. *Marine Ecology Progress Series*, 397, 279–294. https://doi.org/10.3354/meps08248

Ballesteros L. V., Matthews J. L. & Hoeksema B. W. (2018) Pollution and coral damage caused by derelict fishing gear on coral reefs around Koh Tao, Gulf of Thailand. *Marine Pollution Bulletin*, 135, 1107–1116. https://doi.org/10.1016/j.marpolbul.2018.08.033

Burkepile D.E. & Hay M.E. (2010) Impact of herbivore identity on algal succession and coral growth on a Caribbean reef. *PloS One*, 5, e8963. https://doi.org/10.1371/journal.pone.0008963

Figueroa-Pico J., Tortosa F.S. & Carpio A.J. (2020) Coral fracture by derelict fishing gear affects the sustainability of the marginal reefs of Ecuador. *Coral Reefs*, 39, 819–827. https://doi.org/10.1007/s00338-020-01926-6

Heifetz J., Stone R.P. & Shotwell S.K. (2009) Damage and disturbance to coral and sponge habitat of the Aleutian Archipelago. *Marine Ecology Progress Series*, 397, 295–303. https://doi.org/10.3354/meps08304

## 6.3 Limit, cease or prohibit some types of fishing and collection (outside protected areas; including where restrictions are unspecified)

https://www.conservationevidence.com/actions/4065

- We found no studies that evaluated the effects on corals of ceasing or prohibiting some types of fishing and collection (outside protected areas; including where restrictions are unspecified).

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Fishing and collecting of corals and other organisms can have both direct and indirect impacts on corals. Fishing and collecting poses a direct threat when gear damages corals during operations (Mangi & Roberts 2006, Althaus *et al*. 2009) or is lost or abandoned (so called 'ghost' gear; Ballesteros *et al*. 2018, Figueroa-Pico *et al*. 2020). Indirect threats can emerge when overfishing reduces the diversity and abundance of herbivorous fishes, with consequences for macroalgal abundance and its potential to impact on coral growth (Burkepile *et al*. 2010).

Prohibiting some types of fishing and collecting has the potential to reduce these threats and allow corals and the reefs they form to recover and flourish.

Restricted fishing types covered here include hook and line fishing; bottom trawling, dredging and other towed gear; static gear including traps; and dynamite and cyanide fishing. Studies that do not specify the specific types of activities that were prohibited are also included here. When this action is undertaken within a protected area, evidence is summarized under *Habitat protection: Designate a Marine Protected Area and prohibit some fishing and collection* (*including where restrictions are unspecified*).

Althaus F., Williams A., Schlacher T.A., Kloser R.J., Green M.A., Barker B.A., Bax N.J., Brodie P. & Schlacher-Hoenlinger M.A. (2009) Impacts of bottom trawling on deep-coral ecosystems of seamounts are long-lasting. *Marine Ecology Progress Series*, 397, 279–294. https://doi.org/10.3354/meps08248

Ballesteros L. V., Matthews J. L. & Hoeksema B. W. (2018) Pollution and coral damage caused by derelict fishing gear on coral reefs around Koh Tao, Gulf of Thailand. *Marine Pollution Bulletin*, 135, 1107–1116. https://doi.org/10.1016/j.marpolbul.2018.08.033

Burkepile D.E. & Hay M.E. (2010) Impact of herbivore identity on algal succession and coral growth on a Caribbean reef. *PloS One*, 5, e8963. https://doi.org/10.1371/journal.pone.0008963

Figueroa-Pico J., Tortosa F.S. & Carpio A.J. (2020) Coral fracture by derelict fishing gear affects the sustainability of the marginal reefs of Ecuador. *Coral Reefs*, 39, 819–827. https://doi.org/10.1007/s00338-020-01926-6

Mangi S.C. & Roberts C.M. (2006) Quantifying the environmental impacts of artisanal fishing gear on Kenya's coral reef ecosystems. *Marine Pollution Bulletin*, 52, 1646–1660. https://doi.org/10.1016/j.marpolbul.2006.06.006

## 6.4 Limit, cease or prohibit commercial harvesting and/or sale of coral (outside protected areas)

https://www.conservationevidence.com/actions/4066

- We found no studies that evaluated the effects on corals of ceasing or prohibiting commercial harvesting and/or sale of coral (outside protected areas).

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Many corals are commercially valuable and can be sold as curios and jewellery, with coral jewellery in Hawaii generating around $70 million in 2006 (Grigg 2006). However, over-harvesting poses a threat to corals (Bruckner 2009) and can impact population sizes and cause shifts towards a predominance of small colonies (Montero-Serra *et al*. 2015).

Prohibiting collecting for commercial purposes has the potential to reduce these threats and allow corals and the reefs they form to recover and flourish. When commercial harvesting and/or sale of coral is ceased or prohibited within a protected area, evidence is summarized under *Habitat protection: Designate a Marine Protected Area and prohibit some fishing and collection* (*including where restrictions are unspecified*)*; Designate Marine Protected Area and prohibit all types of collection; Designate a Marine Protected Area and prohibit all types of fishing and collection* and *Designate a Marine Protected Area and prohibit all types of fishing, collecting and access*.

Bruckner A.W. (2009) Rate and extent of decline in Corallium (pink and red coral) populations: Existing data meet the requirements for a CITES Appendix II listing. *Marine Ecology Progress Series*, 397, 319–332. https://doi.org/10.3354/meps08110

Grigg R.W. (2006) History of the black coral fishery in Hawaii – 2006. Pages 9–10 in: Western Pacific Regional Fishery Management Council (Eds) *2006 Black Coral Science and Management Workshop. 18–19 Apr 2006, Honolulu, Hawaii*

Montero-Serra I., Linares C., García M., Pancaldi F., Frleta-Valić M., Ledoux J.B., Zuberer F., Merad D., Drap P. & Garrabou J. (2015) Harvesting effects, recovery mechanisms, and management strategies for a long-lived and structural precious coral. *PloS One*, 10, e0117250. https://doi.org/10.1371/journal.pone.0117250

## 6.5 Limit, cease or prohibit use of coral rock or live coral for building roads or infrastructure

https://www.conservationevidence.com/actions/4067

- We found no studies that evaluated the effects on corals of ceasing or prohibiting use of coral rock for building roads or infrastructure.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Coral rock (in the form of dead coral skeletons, also known as coral rubble), as well as live coral has historically been used as a building material (in both buildings and roads), but with increased demand, extraction of corals presents a potentially serious threat (Brown & Dunne 1988, Caras & Pasternak 2009). Removing coral rock from the reef can reduce available habitat for larvae to settle and limit the natural regeneration of degraded reefs. In addition, using live coral will result in the destruction of coral reefs. Machinery used to extract coral rubble could damage the surrounding reef or lead to an increase in sediment in the water column. Increased sediment can damage corals by smothering the delicate polyps or, if it remains in the water column, by reducing available light thereby limiting coral growth (Tuttle *et al*. 2020). Prohibiting the use of corals as building materials has the potential to reduce these threats and allow corals and the reefs they form to recover and flourish.

Brown B.E. & Dunne R.P. (1988) The environmental impact of coral mining on coral reefs in the Maldives. *Environmental Conservation*, 15, 159–165. https://doi.org/10.1017/s0376892900028976

Caras T. & Pasternak Z. (2009) Long-term environmental impact of coral mining at the Wakatobi marine park, Indonesia. *Ocean & Coastal Management*, 52, 539–544. https://doi.org/10.1016/j.ocecoaman.2009.08.006

Tuttle L.J., Johnson C., Kolinski S. Minton D. & Donahue M.J. (2020) How does sediment exposure affect corals? A systematic review protocol. *Environmental Evidence*, 9. https://doi.org/10.1186/s13750-020-00200-0

# 7. Threat: Human intrusions and disturbances

**Background**

Human intrusions and disturbances that impact coral reefs can be from a range of activities such as recreational SCUBA diving or snorkelling, and anchoring of boats, through to large-scale activities including war and military exercises (Souter *et al*. 2021).

Actions related to protecting or restoring and recreating habitats following such disturbances are described in *Habitat Protection*; and *Habitat Restoration and Creation*. Actions related to restoring coral reef populations following human intrusion and disturbance are covered in *Species Management*.

Souter D., Planes S., Wicquart J., Logan M., Obura D. & Staub F. (Eds) (2021) *Status of Coral Reefs of the World: 2020*. Global Coral Reef Monitoring Network (GCRMN). Available from: https://gcrmn.net/2020-report/

## Recreational activities

### 7.1 Prohibit or limit or modify access to coral reefs for any recreational purposes

https://www.conservationevidence.com/actions/4038

 https://doi.org/10.11647/OBP.0453.07

- We found no studies that evaluated the effects on corals of prohibiting or limiting or modifying access to coral reefs for any recreational purposes.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Coral reefs are beautiful, and it is understandable that people want to experience the unique ecosystem. SCUBA diving and snorkelling can result in damage, such as individual branches being broken off or small areas of tissue being damaged through contact. Repeated visits by divers and snorkellers can lead to a cumulative impact from hundreds of minor damages affecting large areas of the reef. As corals can take several years to recover, each individual incident of damage can reduce the coral colony's ability to withstand other environmental pressures such as climate change, bleaching, or storms (Roche *et al*. 2016). Repeated damage to the same reef area can result in the corals being unable to recover and the reef dying (Roche *et al*. 2016).

Anchoring by recreational boats (including those used by divers), near or on coral reefs can damage corals. The anchor itself can damage the seabed and the chain can scour the substrate. Coral reef density, size and diversy are lower in areas with high levels of anchoring (Flynn & Forrester 2019). Actions including providing suitable moorings that could reduce the number of anchorings (Flynn & Forrester 2019), can be found in *Threat: Transport and Service Corridors: Shipping*.

Actions including protecting corals and restoring damaged reefs are covered in *Habitat protection*, *Habitat restoration and creation*, and *Species management*.

Flynn R.L. & Forrester G.E. (2019) Boat anchoring contributes substantially to coral reef degradation in the British Virgin Islands. *PeerJ* 7, e7010. https://doi.org/10.7717/peerj.7010

Roche R.C., Harvey C.V., Harvey J.J., Kavanagh A.P., McDonald M., Stein-Rostaing V.R. & Turner J.R. (2016) Recreational diving impacts on coral reefs and the adoption of environmentally responsible practices within the SCUBA diving industry. *Environmental Management*, 58, 107–116. https://doi.org/10.1007/s00267-016-0696-0

## 7.2 Create alternative locations for recreational activities

https://www.conservationevidence.com/actions/4039

- We found no studies that evaluated the effects on corals of creating alternative locations for recreational activities.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Coral reefs are beautiful, and it is understandable that people want to experience the unique ecosystem. SCUBA diving and snorkelling can result in damage, such as individual branches being broken off or small areas of tissue being damaged through contact. Repeated visits by divers and snorkellers can lead to a cumulative impact from hundreds of minor damages affecting large areas of the reef. As corals can take several years to recover, each individual incident of damage can reduce the coral colony's ability to withstand other environmental pressures such as climate change, bleaching, or storms (Roche *et al*. 2016). Repeated damage to the same reef area can result in the corals being unable to recover and the reef dying (Roche *et al*. 2016).

One approach to reducing the pressures on reefs caused by recreational activities might be to create alternative locations for these activities to take place. For example, Leeworthy *et al*. (2006) found that after creating an artificial reef by sinking a large ship, many divers began visiting this new dive location, and the number of dives at an adjacent natural reef declined. Such reductions in diver numbers could have benefits for corals, though to assess the effectiveness of this approach, more information would be needed on the impacts of divers on corals at both newly created and adjacent natural reefs. Actions including installing artificial reefs, protecting corals and restoring damaged reefs are covered in *Habitat protection*, *Habitat restoration and creation*, and *Species management*.

Leeworthy V.R., Maher T. & Stone E.A. (2006) Can artificial reefs alter user pressure on adjacent natural reefs? *Bulletin of Marine Science*, 78, 29–38. Available from: https://www.ingentaconnect.com/content/umrsmas/bullmar/2006/00000078/00000001/art00004

Roche R.C., Harvey C.V., Harvey J.J., Kavanagh A.P., McDonald M., Stein-Rostaing V.R. & Turner J.R. (2016) Recreational diving impacts on coral reefs and the adoption of environmentally responsible practices within the SCUBA diving industry. *Environmental Management*, 58, 107–116. https://doi.org/10.1007/s00267-016-0696-0

## War, civil unrest and military activities

### 7.3 Prohibit testing of weapons (including explosive, chemical, nuclear) and military exercises (including 'live' firing) near coral reefs

https://www.conservationevidence.com/actions/4040

- We found no studies that evaluated the effects on corals of prohibiting testing of weapons near coral reefs.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

The marine environment is frequently used to test both conventional and nuclear weapons. In the 1940s and 1950s, for example, the coral reef at Bikini Atoll was the site of nuclear weapons testing. The reef was severely damaged by the blasts. Although the coral reef has recovered to about 70% of species before the testing, with the exception of some specialized stony coral species (Richards *et al*. 2008), this recovery has taken 50 years. Abandoned weapons, particularly explosives, can cause pollution across the reef. Impacts on coral reefs from conventional weapons testing are similar to those found after 'blast' fishing (using dynamite); explosions can reduce large areas of coral reef to rubble (Hampton-Smith *et al*. 2021). Prohibiting weapons testing and military exercises from the vicinity of coral reefs will remove the threat.

Actions including protecting corals and restoring damaged reefs are covered in *Habitat protection*, *Habitat restoration and creation*, and *Species management*.

Hampton-Smith M., Bower D.S. & Mika S. (2021) A review of the current global status of blast fishing: Causes, implications and solutions. *Biological Conservation*, 262, 109307. https://doi.org/10.1016/j.biocon.2021.109307

Richards Z.T., Beger M., Pinca S. & Wallace C.C. (2008) Bikini Atoll coral biodiversity resilience five decades after nuclear testing. *Marine Pollution Bulletin*, 56, 503–515. https://doi.org/10.1016/j.marpolbul.2007.11.018

# 8. Invasive alien and other problematic species

**Background**

Invasive, alien and problematic species present challenges to coral reef functioning. Species such as the crown of thorns starfish *Acanthaster planci*, predatory fishes, snails, and crabs can cause damage to the soft tissue on the outer skeleton of a stony coral that can inhibit the coral's ability to grow and survive. In addition, mobile invasive or problematic species can spread disease between coral colonies (Nicolet *et al*. 2013). If attacks occur at the same time as other environmental events, such as bleaching, the combination of stressors can further reduce the coral's ability to recover. Invasive or problematic algae can smother juvenile corals reducing their ability to grow. When breaking down, algae release nutrients into the system potentially leading to a rise in pollution (Souter *et al*. 2021). Corals left physically damaged or smothered by problematic species are more susceptible to disease such as brown-band disease (Nicolet *et al*. 2013). The bacterial stony coral tissue loss disease can also spread rapidly through water between reefs, causing widespread coral mortality (Johnston 2021).

Actions related to protecting habitats from invasive, alien, or problematic species and restoring and recreating habitats following the removal of these species are described in *Habitat Protection*; and *Habitat Restoration and Creation*. Actions related to restoring coral reef populations following the removal of invasive, alien or problematic species are covered in *Species management*.


https://doi.org/10.11647/OBP.0453.08

Johnston M.A. (2021) *Strategy for Stony Coral Tissue Loss Disease Prevention and Response at Flower Garden Banks National Marine Sanctuary*. National Marine Sanctuaries Conservation Series ONMS-21-06. National Oceanic and Atmospheric Administration: Galveston, USA. Available from: https://sanctuaries.noaa.gov/science/conservation/strategy-for-stony-coral-tissue-loss-desease-prevention-repsonse-at-fgbnms.html

Nicolet K.J., Hoogenboom M.O., Gardiner N.M. Pratchet M.S. & Willis B.L. (2013) The corallivorous invertebrate *Drupella* aids in transmission of brown band disease on the Great Barrier Reef. *Coral Reefs*, 32, 585–595. https://doi.org/10.1007/s00338-013-1010-8

Souter D., Planes S., Wicquart J., Logan M., Obura D. & Staub F. (Eds) (2021) *Status of Coral Reefs of the World: 2020*. Global Coral Reef Monitoring Network (GCRMN). Available from: https://gcrmn.net/2020-report/

# General

## 8.1 Control spread of non-native/invasive/problematic invertebrates using biological, chemical and/or mechanical methods

https://www.conservationevidence.com/actions/4197

- **Three studies** examined the effects on corals of controlling non-native/invasive/problematic invertebrates. One study was in each of the USA[1], Brazil[2], and Malaysia[3]

**COMMUNITY RESPONSE (1 STUDY)**

- **Richness/diversity (1 study)**: One replicated, controlled study in Malaysia[3] found no difference in coral community composition between sites with and without the removal of invasive crown of thorns starfish.

**POPULATION RESPONSE (3 STUDIES)**

- **Abundance/Cover (2 studies)**: Two replicated, controlled studies (including one before-and-after) in Brazil[2] and Malaysia[3], found that live soft or stony coral cover increased after hand removal of invasive corals[2] or crown of thorns starfish[3].

- **Survival (1 study)**: One replicated, controlled study in Malaysia[3] found that mortality was higher for one stony coral species (*Porites*) after hand removal of invasive crown of thorns starfish.
- **Condition (1 study)**: One replicated, controlled study in the USA[1] found that removing coral-eating snails by hand from branches of stony coral led to a reduction in live coral tissue loss compared to branches where snails remained.

**Background**

Non-native/invasive or problematic invertebrates present challenges to coral reef functioning (Souter *et al*. 2021). Species such as the crown of thorns starfish *Acanthaster planci*, and drupella snails *Drupella* spp. can devastate large areas of coral reefs (Souter *et al*. 2021). Invasions of crown of thorns starfish have been found to be controlled through culling by injecting bile into individual starfish (Rivera-Posada *et al*. 2021), or by removing the starfish and destroying them (Kroon *et al*. 2021). Snails can be removed from corals by hand (Miller 2001). Invasive/non-native corals such as the orange cup coral *Tubastraea coccinea* and sun coral *Tubastraea tagusensis* can impact native soft coral species (Creed *et al*. 2021; De Paula *et al*. 2017). Control measures usually involve manual removal of the invasive coral (Creed *et al*. 2021).

Actions related to protecting habitats from invasive, alien, or problematic species and restoring and recreating habitats following the removal of these species are described in *Habitat Protection*; and *Habitat Restoration and Creation*. Actions related to restoring coral reef populations following the removal of invasive, alien or problematic species are covered in *Species management*.

Creed J.C., Casares F.A., Oigman-Pszczol S.S & Masi B.P. (2021) Multi-site experiments demonstrate that control of invasive corals (*Tubastraea* spp.) by manual removal is effective. *Ocean & Coastal Management*, 207, 105616. https://doi.org/10.1016/j.ocecoaman.2021.105616

De Paula A.F., Fleury B.G., Lages B.G. & Creed J.C. (2017) Experimental evaluation of the effects of management of invasive corals on native communities. *Marine Ecology Progress Series*, 572, 141–154. https://doi.org/10.3354/meps12131

Kroon F.J., Barneche D.R. & Emslie M.J. (2021) Fish predators control outbreaks of Crown-of-Thorns Starfish. *Nature Communications*, 12, 6986. https://doi.org/10.1038/s41467-021-26786-8

Miller, M. (2001) Corallivorous snail removal: evaluation of impact on *Acropora palmata. Coral Reefs* 19, 293–295. https://doi.org/10.1007/PL00006963

Rivera-Posada J., Pratchett M.S., Aguilar C., Grand A. & Caballes C.F. (2014) Bile salts and the single-shot lethal injection method for killing crown-of-thorns sea stars (*Acanthaster planci*). *Ocean & Coastal Management*, 102, 383–390. https://doi.org/10.1016/j.ocecoaman.2014.08.014

Souter D., Planes S., Wicquart J., Logan M., Obura D. & Staub F. (Eds) (2021) *Status of Coral Reefs of the World: 2020*. Global Coral Reef Monitoring Network (GCRMN). Available from: https://gcrmn.net/2020-report/

A replicated, controlled study in 1999 at two coral reefs in the Florida Keys National Marine Sanctuary, USA, (1) found that removing coral-eating snails *Coralliophilia abbreviata* from branches of elkhorn coral *Acropora palmata* colonies led to a reduction in live tissue loss compared to branches with snails left in place, but no difference compared to branches without snails. After two months, average tissue loss was lower on branches with snails removed (0.80 $cm^2$/day) and control branches with no snails (0.001 $cm^2$/day) compared to branches with snails (3.37 $cm^2$/day). There was an average of 81% fewer incidences of snails observed on branches with snails removed compared to branches with snails remaining. In June 1999, individual elkhorn coral colonies with snails attached were selected at two reefs (French and Pickles). Snails were removed from one branch from each colony, snails were left in place on another branch and a third branch (with no snails) was left snail-free. Where possible, different treatments were applied to branches from the same colony, however, on two occasions, treatments were applied to branches from different, adjacent, colonies. Sampling was carried out every 4–11 days for two months and tissue loss recorded using photographs. Snails were removed or replaced as necessary to maintain the treatment levels.

A replicated, controlled, before-and-after study in 2004–2006 at a rocky reef at Macacos Island, Brazil (2) found that a year after removing invasive

orange cup *Tubastraea coccinea* and sun *Tubastraea tagusensis* corals, the change in cover of native soft coral *Palythoa caribaeorum* varied depending on which removal method was used. Soft coral cover was lower in plots where invasive corals were removed once (single-removal before: 11%, after: 10%) compared to plots where no invasive corals were removed (no-removal before: 15%, after: 22%). Soft coral never recolonized plots where the whole seabed community was removed (community-removal before: 8%, after: 0%). No soft corals were recorded in plots before invasive corals were removed multiple times and did not colonize over time (multiple-removal: 0%). After one year, invasive corals had recolonized all removal plots (single-removal: 14%; multiple-removal: 3%; community-removal: 14%; no removal: 27%). In December 2004, twenty 0.16 $m^2$ plots, all with ≥20% cover of invasive corals, were selected and native soft coral cover was recorded. Four treatments were used (5 plots/treatment): a single removal of invasive corals (December 2004), multiple removals of invasive corals (December 2004–January 2006), a single removal of the whole seabed community (December 2004), and no removal. Removal was done manually by divers. Invasive and soft corals were counted before, immediately after first removal (December 2004), then monthly or quarterly until January 2006. Invasive corals in the multiple-removal plots were removed on each occasion.

A replicated, controlled study in 2009–2010 at three coral reef sites in Malaysia (3) found that seasonal removal of invasive crown of thorns starfish *Acanthaster planci* led to higher live stony coral cover, but increased mortality for one stony coral species and had no effect on community composition, compared to sites where no starfish were removed. For six months after the removal season, live coral cover increased in sites where starfish were removed (0 months: 31–78%; 6 months: 44–78%) but decreased in sites with no starfish removal (0 months: 40–69%; 6 months: 7–52%). Starfish removal had no effect on mortality for *Acropora* (removal: 27–71%; no removal: 29–64%), *Montipora* (removal: 2–17%; no removal: 0–25%), and *Pavona* (removal: 2.3–100%; no removal: 0–100%), but increased *Porites* mortality (removal: 0–38%; no removal: 50–95%). Starfish removal had no effect on the coral community (data reported as graphical analysis). From March–September each year since 1998, invasive crown of thorns starfish were removed manually by volunteers and through clean ups organized by dive shops. Three sites

were surveyed after the 2009 removal season: twice in August 2009 and twice in April 2010. They were compared with three control sites, where no removal had taken place for 10 years. At each reef site, 20 photo-quadrats (50 $cm^2$) were taken in each of eight belt transects (50 × 2 m) at 2–10 m depth during daytime. Coral composition and percent cover were estimated from 25 random points/5 × 5 grid cell/photo-quadrat. Mortality was measured as the relative percent of dead corals.

(1) Miller M. (2001) Corallivorous snail removal: Evaluation of impact on *Acropora palmata*. *Coral Reefs*, 19, 293–295. https://doi.org/10.1007/PL00006963

(2) De Paula A.F., Fleury B.G., Lages B.G. & Creed J.C. (2017) Experimental evaluation of the effects of management of invasive corals on native communities. *Marine Ecology Progress Series*, 572, 141–154. https://doi.org/10.3354/meps12131

(3) Chak S.T.C, Dumont C.P., Adzis K-A.Abd, Yewdall K. (2018) Effectiveness of the removal of coral-eating predator *Acanthaster planci* in Pulau Tioman Marine Park, Malaysia. *Journal of the Marine Biological Association of the United Kingdom*, 98, 183–189. https://doi.org/10.1017/S002531541600117X

## 8.2 Control spread of non-native/invasive/problematic plants/algae using biological, chemical and/or mechanical methods

https://www.conservationevidence.com/actions/4198

- **Two studies** examined the effects on corals of controlling non-native/invasive/problematic plants/algae. One study was in each of Belize[1], and French Polynesia[2].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (2 STUDIES)**

- **Abundance/Cover (1 study)**: One replicated, controlled, before-and-after study in Belize[1] found no difference in stony coral or soft gorgonian coral cover at sites where nuisance algae were removed than where algae remained.
- **Reproductive success (1 study)**: One replicated, controlled study in French Polynesia[2] found that removing all erect macroalgae from stony coral 'bommies' led to higher recruitment than on partially or uncleared bommies.

**Background**

Non-native/invasive or problematic plants or algae present challenges to coral reef functioning (Souter *et al*. 2021). Invasive or problematic algae can smother juvenile corals reducing their ability to grow, and, when breaking down, algae release nutrients into the system potentially leading to a rise in pollution (Souter *et al*. 2021). Removing problematic algae can be time consuming and risks damaging the surface of the coral if the algae are attached.

Actions related to protecting habitats from invasive, alien, or problematic species and restoring and recreating habitats following the removal of these species are described in *Habitat Protection*; and *Habitat Restoration and Creation*. Actions related to restoring coral reef populations following the removal of invasive, alien or problematic species are covered in *Species management*.

Souter D., Planes S., Wicquart J., Logan M., Obura D. & Staub F. (Eds) (2021) *Status of Coral Reefs of the World: 2020*. Global Coral Reef Monitoring Network (GCRMN). Available from: https://gcrmn.net/2020-report/

A replicated, controlled, before-and-after study in 1998–1999 at 16 coral reef sites off the coast of Belize (1) found that removing problematic algae did not lead to an increase in stony coral or soft gorgonian coral cover. There was no difference in stony coral cover in algae removal sites (after: 10–34%, before: 20–29%) compared to sites with no removal (after: 11–22%, before: 19–22%) and no difference for gorgonian cover in removal (after: 2–9%, before: 7–8%) compared to non-removal sites (after: 3–7%, before: 5–7%). A year after algae removal, composition of the benthic community (including corals) had returned to its pre-removal state (data reported as graphical analysis). Overall, cover of coral declined through the course of the study (stony coral: 10% decline, gorgonian: 3% decline). Sixteen patch reefs sites were selected (average 1,000 $m^2$, eight in each of a wilderness and general use area). Four patches from each area had algae removed with hedge trimmers and wire brushes. Coral cover was assessed in September 1998 before algae removal, and then in October 1998 (for removal patches only),

December 1998, and April and September 1999. Cover was assessed along three 10 m transects on each patch reef, and all organisms over 3 cm in size were recorded.

A replicated, randomized, controlled study in 2014–2015 at a lagoon in Taareu, Moorea, French Polynesia (2), found that clearing all macroalgae species from the tops of massive coral *Porites porites* colonies (known as 'bommies') led to higher coral recruitment compared to partially cleared or uncleared bommies. Eight months after clearance, a total of 54 coral recruits were found on bommies with a higher number on fully cleared (total: 40, average 9.5/bommie) compared to partially cleared and uncleared (each total: seven, average 1.6) bommies. In November 2014, fifteen bommies (2–3 m diameter) colonized with macroalgae (*Turbinaria ornata*) were randomly selected from the lagoon. Five bommies were randomly assigned to one of three treatments (full clearance: removal of all macroalgae, including understorey species and holdfasts; partial clearance: removal of fronds of canopy-forming macroalgae including *Turbinaria ornata*; uncleared: nothing removed). Coral recruits (colonies ≤ 1 cm diameter) were counted after eight months.

(1) McClanahan T., McField M., Huitric M., Bergman K., Sala E., Nyström M., Nordemar I., Elfwing T. & Muthiga N. (2001) Responses of algae, corals and fish to the reduction of macroalgae in fished and unfished patch reefs of Glovers Reef Atoll, Belize. *Coral Reefs*, 19, 367–379. https://doi.org/10.1007/s003380000131

(2) Bulleri F., Thiault L., Mills S.C., Nugues M.M., Eckert E.M., Corno G. & Claudet J. (2018) Erect macroalgae influence epilithic bacterial assemblages and reduce coral recruitment *Marine Ecology Progress Series*, 597, 65–77. https://doi.org/10.3354/meps12583

## 8.3 Control spread of disease using biological, chemical and/or mechanical methods

https://www.conservationevidence.com/actions/4199

- **Four studies** examined the effects on corals of controlling the spread of diseases. Two studies were in Israel[2a,b], and one study was in each of the British Virgin Islands[1], and Puerto Rico[3].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (4 STUDIES)**

- **Condition (4 studies)**: Two of three replicated, controlled studies in the British Virgin Islands[1], and Israel[2a,b] found that white plague disease progression was slowed within stony coral colonies when corals were shaded[1], or treated with biological methods (phage therapy)[2a]. The third study found that transmission of the disease to adjacent healthy colonies was reduced when diseased corals were treated with biological methods (phage therapy[2b]). One replicated, before-and-after study in Puerto Rico[3] found that scraping or chopping lesions off diseased sea fans led to live tissue regrowth.

**Background**

Non-native/invasive or problematic species present challenges to coral reef functioning (Souter *et al*. 2021). Corals attacked and left damaged or smothered by problematic species are more susceptible to disease such as brown-band disease (Nicolet *et al*. 2013). The bacterial stony coral tissue loss disease can also spread rapidly through water between reefs, causing widespread coral mortality (Johnston 2021). Actions aimed at controlling the spread could include removing diseased tissue (Ruiz-Diaz *et al*. 2016), removing smaller (<30 cm) infected coral colonies, administering antibiotics (Johnston, 2021) or using phage therapy (a virus that targets the disease bacteria) (Atad *et al*. 2012). Prevention methods, such as shading colonies, aim to reduce thermal stress and increase colony resilience which could, in turn, reduce the likelihood of infection (Muller & van Woesik 2009).

Atad I., Zvuloni A., Loya Y. & Rosenberg E. (2012) Phage therapy of the white plague-like disease of *Favia favus* in the Red Sea. *Coral Reefs*, 31, 665–670. https://doi.org/10.1007/s00338-012-0900-5

Johnston M.A. (2021) *Strategy for Stony Coral Tissue Loss Disease Prevention and Response at Flower Garden Banks National Marine Sanctuary*. National Marine Sanctuaries Conservation Series ONMS-21-06. National Oceanic and Atmospheric Administration: Galveston, USA. Available from: https://sanctuaries.noaa.gov/science/conservation/strategy-for-stony-coral-tissue-loss-desease-prevention-repsonse-at-fgbnms.html

Muller E.M. & van Woesik R. (2009) Shading reduces coral-disease progression. *Coral Reefs*, 28, 757–760. https://doi.org/10.1007/s00338-009-0504-x

Nicolet K.J., Hoogenboom M.O., Gardiner N.M. Pratchet M.S. & Willis B.L. (2013) The corallivorous invertebrate *Drupella* aids in transmission of brown band disease on the Great Barrier Reef. *Coral Reefs*, 32, 585–595. https://doi.org/10.1007/s00338-013-1010-8

Ruiz-Diaz C.P., Toledo-Hernández C., Mercado-Molina A.E. & Sabat A.M. (2016) Scraping and extirpating: two strategies to induce recovery of diseased *Gorgonia ventalina* sea fans. *Marine Ecology*, 37, 336–343. https://doi.org/10.1111/maec.12283

Souter D., Planes S., Wicquart J., Logan M., Obura D. & Staub F. (Eds) (2021) *Status of Coral Reefs of the World: 2020*. Global Coral Reef Monitoring Network (GCRMN). Available from: https://gcrmn.net/2020-report/

A replicated, controlled study in 2008 at a coral reef off Guana Island, British Virgin Islands (1) found that shading wild-growing colonies of massive coral *Colpophyllia natans* from sunlight reduced the progression of white-plague disease compared to unshaded colonies. After 10 days, average rate of disease progression in shaded colonies decreased by ~0.15 cm/day (progression before shading: 0.37 cm/day, after shading: 0.22 cm/day) whereas the average daily disease progression rate for unshaded colonies increased by ~0.07 cm/day (from 0.41 cm–0.48 cm/day). Disease progression did not stop completely on any individual colony. In August 2008, eighteen *Colpophyllia natans* colonies, growing ~7 m deep, were monitored for four days to measure the rate of white-plague disease progression. After this initial assessment, shading, comprising smoked plastic sheets attached to plastic frames and placed ~10 cm above the coral, was installed over nine coral colonies with the remaining nine colonies left unshaded. Disease progression was recorded daily for 10 days using scaled photographs.

A replicated, controlled study in 2009 and 2011 at a reef in the Gulf of Aqaba, Eilat, Israel (2a) found that using biological 'phage' therapy (a virus that attacks the disease bacteria) to treat massive coral *Favia favus* infected with white plague-like disease *Thalassomonas loyana* slowed the disease progression compared to untreated coral. In 2009, average tissue loss after 28 days following treatment was significantly lower for treated coral (5%) compared to untreated (65%). In 2011, after 47 days there was no live tissue loss in two of three treated corals but a 60% loss in the third treated coral (overall average 13% loss) compared to an average

of 57% loss for untreated corals. In September 2009 and September 2011, healthy *Favia favus* colonies (16–20/year) and four to six colonies showing signs of white plague-like disease were collected from a reef near Eilat. Eight to 10 healthy colonies were fixed in a circle to each of two nets on a plastic frame placed on the seabed and two (2009) or three (2011) diseased colonies fixed in the centre of each frame. Each frame was covered by a bottomless three-sided clear plastic box. Phages were injected into one of the two boxes each year, the second box was a no-phage control. Boxes were removed after 48 h. Corals were monitored visually and photographically for signs of disease progression at intervals for 28 days (2009) and 47 days (2011).

A replicated, controlled study in 2009 and 2011 at a reef Gulf of Aqaba, Eilat, Israel (2b) found that using biological 'phage' therapy (a virus that attacks the disease bacteria) to treat massive coral *Favia favus* infected with white plague-like disease *Thalassomonas loyana* reduced the spread of disease to neighbouring, healthy coral. In 2009, after 20 days, only one of nine healthly corals surrounding the treated diseased coral showed signs of disease compared to five of eight healthy corals surrounding untreated diseased coral. In 2011, after 47 days, none of the 10 healthy corals near treated corals showed signs of disease whereas six of 10 corals near untreated diseased corals showed disease. In September 2009 and September 2011, healthy *Favia favus* colonies (16–20/year) and four to six colonies showing signs of white plague-like disease were collected from a reef near Eilat. Eight to 10 healthy colonies were fixed in a circle to each of two nets on a plastic frame placed on the seabed and two (2009) or three (2011) diseased colonies fixed in the centre of each frame. Each frame was covered by a bottomless three-sided clear plastic box. Phages were injected into one of the two boxes each year, the second box was a no-phage control. Boxes were removed after 48 h. Healthy corals were monitored visually and photographically for signs of disease at intervals for 28 days (2009) and 47 days (2011).

A replicated, before-and-after study in 2011–2013 at two nature reserves in Puerto Rico (3), found that scraping or chopping lesions off diseased sea fan *Gorgonia ventalina* colonies led to most colonies regrowing live tissue or growing new branches, but recovery depended on the amount of initial lesion coverage. After 16 months, 50% of coral colonies with lesions scraped off had regrown between 80–100%

of the lost live tissue but 7–10% of colonies had further tissue loss. After chopping off lesions, there was no difference in regrowth of live tissue between diseased (11%, 0.09 mm/day) and healthy coral (19%, 0.14 mm/day). Seventy-five percent of colonies fully recovered when the area of the colony initially covered by lesions was low (<5%), compared to 42% of colonies with a high proportion of initial lesion coverage (≥ 10%). In July 2011, a total of 60 diseased sea fan colonies (with lesions or purpling tissue), and 29 healthy colonies were identified in two Natural Reserves and diseased tissue photographed. All lesions and surrounding purpling tissue on diseased colonies and ~10% of the surface area from the healthy colonies were scraped using metal-bristle brushes. A further 27 colonies (17 diseased, 10 healthy) were identified in one of the reserves and branches with lesions were cut from the diseased colonies and ~10% of surface area cut from healthy colonies. Recovery was monitored monthly using photographs for 16 months (scraped) and 12 months (chopped) colonies.

(1) Muller E.M. & van Woesik R. (2009) Shading reduces coral-disease progression. *Coral Reefs*, 28, 757–760. https://doi.org/10.1007/s00338-009-0504-x

(2) Atad I., Zvuloni A., Loya Y. & Rosenberg E. (2012) Phage therapy of the white plague-like disease of *Favia favus* in the Red Sea. *Coral Reefs*, 31, 665–670. https://doi.org/10.1007/s00338-012-0900-5

(3) Ruiz-Diaz C.P., Toledo-Hernández C., Mercado-Molina A.E. & Sabat A.M. (2016) Scraping and extirpating: Two strategies to induce recovery of diseased *Gorgonia ventalina* sea fans. *Marine Ecology*, 37, 336–343. https://doi.org/10.1111/maec.12283

# Aquaculture

## 8.4 Control spread of non-native/invasive/problematic species in aquaculture

https://www.conservationevidence.com/actions/4041

- We found no studies that evaluated the effects on corals of controlling the spread of non-native/invasive/problematic species in aquaculture.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Invasive, alien and problematic species present challenges to coral reef functioning. (Souter *et al*. 2021). Corals can be cultivated in ex-situ tanks or in-situ to provide colonies for use in restoration, research, or, more commonly, for the aquarium trade. Invasive, alien and problematic species, such as disease or pathogens, can be spread through seawater flowing through tanks or from the open environment. These can be inadvertently transferred to other coral reefs when cultivated corals are used for restoration projects. Actions aimed at controlling the spread could include removing smaller (<30 cm) infected coral colonies or administering antibiotic treatment (Johnston, 2021).

This action refers to the spread of invasive, alien and problematic species within aquaculture. Actions for cultivating corals in in-situ or ex-situ nurseries are provided in *Cultivate coral fragments in an artificial nursery located in a natural habitat; Cultivate coral larvae in an artificial nursery located in a natural habitat*; and *Cultivate corals in an ex-situ nursery*.

Johnston M.A. (2021) *Strategy for Stony Coral Tissue Loss Disease Prevention and Response at Flower Garden Banks National Marine Sanctuary*. National Marine Sanctuaries Conservation Series ONMS-21-06. National Oceanic and Atmospheric Administration: Galveston, USA. Available from: https://sanctuaries.noaa.gov/science/conservation/strategy-for-stony-coral-tissue-loss-desease-prevention-repsonse-at-fgbnms.html

Souter D., Planes S., Wicquart J., Logan M., Obura D. & Staub F. (Eds) (2021) *Status of Coral Reefs of the World: 2020*. Global Coral Reef Monitoring Network (GCRMN). Available from: https://gcrmn.net/2020-report/

# Shipping, transportation, and anthropogenic structures

## 8.5 Control spread of non-native/invasive/problematic species and diseases via shipping, transportation, and anthropogenic structures

https://www.conservationevidence.com/actions/4042

- We found no studies that evaluated the effects on corals of controlling the spread of non-native/invasive species via shipping, transportation, and anthropogenic structures.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Invasive, alien and problematic species present challenges to coral reef functioning. (Souter *et al*. 2021). Shipping provides an ideal opportunity for species to be taken to a new location where they can colonize and damage the native ecosystem. Non-native/invasive species (e.g. lionfish *Pterois* spp.) or disease (e.g. stony-coral-tissue-loss disease) can be inadvertently spread around the world through ballast-water transport (MacIsaac *et al*. 2016) and present a threat to corals (Aguirre-Macedo *et al*. 2008). Legislation was initially introduced that required ships to exchange ballast water in the open ocean before arriving at a port. It is now a requirement that ships have a treatment system for ballast water onboard to minimize the risk (IMO 2019). Anthropogenic structures such as sea walls, permanent anchor points and even shipwrecks can provide areas on which invasive species can settle which could act as 'stepping stones' to move these problematic species to different reefs. Actions aimed at controlling the spread could include removing smaller (<30 cm) infected coral colonies or administering antibiotic treatment (Johnston, 2021).

Aguirre-Macedo M.L., Vidal-Martinez V.M., Herrera-Silveira J.A., Valdés-Lozano D.S., Herrera-Rodríguez M. & Olvera-Novoa M.A. (2008) Ballast water as a vector of coral pathogens in the Gulf of Mexico: The case of the Cayo Arcas coral reef. *Marine Pollution Bulletin*, 56, 1570–1577. https://doi.org/10.1016/j.marpolbul.2008.05.022

de Oliveira Soares M., Salani S., Paiva S.V. & Braga M.D.A. (2020) Shipwrecks help invasive coral to expand range in the Atlantic Ocean. *Marine Pollution Bulletin*, 158, 111394. https://doi.org/10.1016/j.marpolbul.2020.111394

International Maritime Organisation (IMO) (2019) *Ballast water management*. Available from: https://www.imo.org/en/OurWork/Environment/Pages/BallastWaterManagement.aspx (Accessed July 2024).

Johnston M.A. (2021) *Strategy for Stony Coral Tissue Loss Disease Prevention and Response at Flower Garden Banks National Marine Sanctuary*. National Marine Sanctuaries Conservation Series ONMS-21-06. National Oceanic and Atmospheric Administration: Galveston, USA. Available from: https://sanctuaries.noaa.gov/science/conservation/strategy-for-stony-coral-tissue-loss-desease-prevention-repsonse-at-fgbnms.html

MacIsaac H.J., De Roy E.M., Leung B., Grgicak-Mannion A. & Ruiz GM (2016) Possible ballast water transfer of Lionfish to the Eastern Pacific Ocean. *PLoS One*, 11, e0165584. https://doi.org/10.1371/journal.pone.0165584

Souter D., Planes S., Wicquart J., Logan M., Obura D. & Staub F. (Eds) (2021) *Status of Coral Reefs of the World: 2020*. Global Coral Reef Monitoring Network (GCRMN). Available from: https://gcrmn.net/2020-report/

# 9. Threat: Pollution

**Background**

Pollution of the marine environment and, in particular, coral reefs can have a major impact on biodiversity and habitat function (Souter *et al*. 2021). Sources of pollution into the coral reef habitat include domestic wastewaters, industrial and military effluents, intensive aquaculture systems, and run-offs from land agriculture, garbage and solid wastes, and pollution from excess energy such as noise, light, and thermal pollution (Souter *et al*. 2021).

Souter D., Planes S., Wicquart J., Logan M., Obura D. & Staub F. (Eds) (2021) *Status of Coral Reefs of the World: 2020*. Global Coral Reef Monitoring Network (GCRMN). Available from: https://gcrmn.net/2020-report/https://gcrmn.net/2020-report/

## General

### 9.1 Use chemicals or minerals to neutralize or remove pollutants

https://www.conservationevidence.com/actions/4043

- We found no studies that evaluated the effects on corals of using chemicals or minerals to neutralize or remove pollutants.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

 https://doi.org/10.11647/OBP.0453.09

**Background**

A range of marine pollutants, such as those leaching from aquaculture systems, sewage outfalls, or nearby agriculture fields, can accumulate in corals, coral reefs, and marine sediments, particularly in coastal areas (Miraji *et al*. 2021). Chemicals or minerals can be added to sediments to reduce or remove pollutants within the sediments (Shin & Kim 2016; Yamamoto *et al*. 2013). For example, granulated coal ash can be used with the aim of reducing concentrations of phosphates and hydrogen sulphide (Kim *et al*. 2014).

Kim K., Hibino T., Yamamoto T., Hayakawa S., Mito Y., Nakamoto K. & Lee I.C. (2014) Field experiments on remediation of coastal sediments using granulated coal ash. *Marine Pollution Bulletin*, 83, 132–137. https://doi.org/10.1016/j.marpolbul.2014.04.008

Miraji H., Ripanda A. & Moto E. (2021) A review on the occurrences of persistent organic pollutants in corals, sediments, fish and waters of the Western Indian Ocean. *The Egyptian Journal of Aquatic Research*, 47, 373–379. https://doi.org/10.1016/j.ejar.2021.08.003

Shin W. & Kim Y.K. (2016) Stabilization of heavy metal contaminated marine sediments with red mud and apatite composite. *Journal of Soils and Sediments*, 16, 726–735. https://doi.org/10.1007/s11368-015-1279-z

Yamamoto T., Harada K., Kim K.H., Asaoka S., & Yoshioka I. (2013) Suppression of phosphate release from coastal sediments using granulated coal ash. *Estuarine, Coastal and Shelf Science*, 116, 41–49. https://doi.org/10.1016/j.ecss.2012.06.010

# Domestic and urban wastewater

## 9.2 Reduce pollution from domestic and urban wastewater

https://www.conservationevidence.com/actions/4044

- We found no studies that evaluated the effects on corals of reducing pollution from domestic and urban wastewater.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Untreated wastewater and sewage reaching the marine environment can impact corals through introducing excess freshwater, sediments, nutrients, pathogens, endocrine disrupters, heavy metals and other toxins (Wear & Thurber 2015). These pollutants have been linked with a range of negative consequences for corals, including reduced coral cover and species richness, increased bleaching and mortality, reduced fecundity and lower condition, as well as a range of coral diseases and algal overgrowth (Wear & Thurber 2015). It is therefore likely that reducing pollution from domestic and urban wastewater will have benefits for corals and the reefs they form.

This action includes studies that report the effects of reducing the amount of domestic and urban wastewater pollution that is produced. Studies that report the effect of removing pollutants from existing levels of domestic and urban wastewater are described in *Use biological, chemical or mechanical methods to manage excess pollution from domestic and urban wastewater*.

Wear S.L. & Thurber R.V. (2015) Sewage pollution: Mitigation is key for coral reef stewardship. *Annals of the New York Academy of Sciences*, 1355, 15–30. https://doi.org/10.1111/nyas.12785

## 9.3 Use biological, chemical or mechanical methods to manage excess pollution from domestic and urban wastewater

https://www.conservationevidence.com/actions/4045

- We found no studies that evaluated the effects on corals of using biological, chemical or mechanical methods to manage excess pollution from domestic and urban wastewater.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Allowing wastewater and sewage to reach the marine environment can impact on corals through introducing excess freshwater, sediments, nutrients, pathogens, endocrine disrupters, heavy metals and other toxins (Wear & Thurber 2015). These pollutants have been linked with a range of negative consequences for corals, including reduced coral cover and species richness, increased bleaching and mortality, and reduced fecundity, as well as a range of coral diseases and algal overgrowth (Wear & Thurber 2015). Barriers may be used to prevent pollutants from entering waterways, and a range of options for both temporary and permanent structures have been suggested, including silt fences and gabions (Botting & Bellette 1998). Other approaches include conventional wastewater treatment or the use of bioremediating organisms (e.g. microalgae; Geremia *et al*. 2021) and constructed wetlands (Biswal & Balasubramanian 2022).

This action includes studies that report the effects of using biological, chemical or mechanical methods to manage excess pollution from domestic and urban wastewater. Studies that report the effect of actions aimed at reducing pollutants are described in *Reduce pollution from domestic and urban wastewater*.

Biswal B.K. & Balasubramanian R. (2022) Constructed wetlands for reclamation and reuse of wastewater and urban stormwater: A review. *Frontiers in Environmental Science*, 10, 836289. https://doi.org/10.3389/fenvs.2022.836289

Botting J. & Bellette K. (1998) *Stormwater pollution prevention: Code of practice for local, state and federal government*. Environment Protection Authority: South Australia. Available from: https://www.epa.sa.gov.au/files/47791_govcop1.pdf

Geremia E., Ripa M., Catone C.M. & Ulgiati S. (2021) A review about microalgae wastewater treatment for bioremediation and biomass production—a new challenge for Europe. *Environments*, 8, 136. https://doi.org/10.3390/environments8120136

Wear S.L. & Thurber R.V. (2015) Sewage pollution: Mitigation is key for coral reef stewardship. *Annals of the New York Academy of Sciences*, 1355, 15–30. https://doi.org/10.1111/nyas.12785

# Industrial and military activities

## 9.4 Reduce pollution from industrial and military activities

https://www.conservationevidence.com/actions/4046

- We found no studies that evaluated the effects on corals of reducing pollution from industrial and military activities.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Pollutants from industrial and military activities include mining waste (tailings), drill cutting from oil and gas drilling activities, fly-ash from coal combustion and a range of other persistent environmental pollutants such as the pesticide dichlorodiphenyltrichloroethane (DDT) and nuclear and radioactive waste. These pollutants can impact corals when they are disposed of at sea or in watercourses that flow into the sea. For example, disposal of radioactive waste at sea was practiced by 13 countries from 1946 until 1993. Ceasing or prohibiting such practices should reduce the threat posed by these pollutants to corals. A range of national laws and international treatise have sought to end their disposal into the marine environment, though enforcement is lacking in parts of the world, where illegal dumping is reported to occur.

Oil spills also pose a serious threat to corals (White *et al*. 2012), and coral reef sites may be particularly at risk due to the potential for oil tankers to run aground in these locations (Fragoso ados Santos *et al*. 2015). Again, national laws and international treatise may play a role in reducing the threat posed by oil spills. This action includes studies that report the effects of reducing pollution from industrial and military activities. Studies that report the effect of actions aimed at removing or cleaning up pollution are described in *Remove or clean-up oil pollution following a spill*.

Fragoso ados Santos H., Duarte G.A.S., Rachid C.T.D.C., Chaloub R.M., Calderon E.N., Marangoni L.F.D.B., Bianchini A., Nudi A.H., do Carmo F.L., van Elsas J.D., Rosado A.S., Castro C.B. & Peixoto, R. S. (2015) Impact of oil spills on coral reefs can be reduced by bioremediation using probiotic microbiota. *Scientific reports* 5, 18268. https://doi.org/10.1038/srep18268

White H.K., Hsing P.Y., Cho W., Shank T.M., Cordes E.E., Quattrini A.M., Nelson R.K., Camilli R., Demopoulos A.W., German C.R. & Brooks J.M. (2012) Impact of the Deepwater Horizon oil spill on a deep-water coral community in the Gulf of Mexico. *Proceedings of the National Academy of Sciences*, 109, 20303–20308. https://doi.org/10.1073/pnas.1118029109

## 9.5 Remove or clean-up oil pollution following a spill

https://www.conservationevidence.com/actions/4047

- We found no studies that evaluated the effects on corals of removing or cleaning-up oil pollution following a spill.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Oil spills pose a serious threat to corals (White *et al*. 2012) and a range of approaches have been developed for dealing with oil in the environment. For example, the use of mechanical methods devices such as booms (floating barriers that contain a spill to a delimited zone) or skimmers (devices that collect and remove oil), chemical dispersants and sorbents, bioremediating organisms such as bacteria or fungi, or using controlled burning of the oil (Al-Majed *et al*. 2012). It is important to also consider the potential harms that such methods may pose to corals. This action includes studies that report the effects of removing or cleaning up oil pollution following a spill. Studies that report the effect of actions aimed at reducing pollution from industrial and military activities are described in *Reduce pollution from industrial and military activities*.

Al-Majed A.A., Adebayo A.R. & Hossain M.E. (2012) A sustainable approach to controlling oil spills. *Journal of Environmental Management*, 113, 213–227. https://doi.org/10.1016/j.jenvman.2012.07.034

White H.K., Hsing P.Y., Cho W., Shank T.M., Cordes E.E., Quattrini A.M., Nelson R.K., Camilli R., Demopoulos A.W., German C.R. & Brooks J.M. (2012) Impact of the Deepwater Horizon oil spill on a deep-water coral community in the Gulf of Mexico. *Proceedings of the National Academy of Sciences*, 109, 20303–20308. https://doi.org/10.1073/pnas.1118029109

# Aquaculture effluents

## 9.6 Reduce pollution from aquaculture effluents

https://www.conservationevidence.com/actions/4048

- We found no studies that evaluated the effects on corals of reducing pollution from aquaculture effluents.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Aquaculture systems may discharge a range of pollutants into the marine environment, including faeces, excess feed and nutrients, and chemicals, such as disinfectants, antifoulants, pesticides, herbicides, and drugs for disease control. Effluents from aquaculture facilities can cause changes in nutrient levels and microbial communities in the marine environment (Becker *et al*. 2017) and increased nutrient loads in coastal waters have the potential to negatively impact corals (Zhao *et al*. 2021). Reducing pollution from aquaculture facilities is therefore likely to benefit corals. Reductions might be achieved through a range of different regulatory mechanisms, such as regulating water quality, aquaculture practices (e.g., stocking densities, pesticide and antibiotic use), or by restricting operations in certain sensitive areas.

This action includes studies that report the effects of reducing the amount of aquaculture pollution that is produced. Studies that report the effect of removing pollutants from existing levels of aquaculture effluent are described in *Use biological, chemical or mechanical methods to manage excess pollution from aquaculture effluents*. Aquaculture systems also pose serious environment risks by promoting the spread of non-native, invasive, and pest species and diseases. Evidence for actions related to non-native, invasive and pest species is summarized in *Threat: Invasive and other problematic species, genes and diseases - Aquaculture*.

Becker C., Hughen K., Mincer T.J., Ossolinski J., Weber L. & Apprill A. (2017) Impact of prawn farming effluent on coral reef water nutrients and microorganisms. *Aquaculture Environment Interactions*, 9, 331–346. https://doi.org/10.3354/aei00238

Zhao H., Yuan M., Strokal M., Wu H.C., Liu X., Murk A., Kroeze C. & Osinga R. (2021) Impacts of nitrogen pollution on corals in the context of global climate change and potential strategies to conserve coral reefs. *Science of the Total Environment*, 774, 145017. https://doi.org/10.1016/j.scitotenv.2021.145017

## 9.7 Use biological, chemical or mechanical methods to manage excess pollution from aquaculture effluents

https://www.conservationevidence.com/actions/4049

- We found no studies that evaluated the effects on corals of using biological, chemical or mechanical methods to manage excess pollution from aquaculture effluents.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Aquaculture systems may discharge a range of pollutants into the marine environment, including faeces, excess feed and nutrients, and chemicals, such as disinfectants, antifoulants, pesticides, herbicides, and drugs for disease control. Effluents from aquaculture facilities can cause changes in nutrient levels and microbial communities in the marine environment (Becker *et al*. 2017) and increased nutrient loads in coastal waters have the potential to negatively impact corals (Zhao *et al*. 2021). A range of biological, chemical or mechanical methods may be employed to manage excess pollutants from aquaculture facilities (Turcios *et al*. 2014), including the use of reagents (Karia *et al*. 2022), certain invertebrates (Gómez *et al*. 2019) or algae (Lugo *et al*. 2020), or constructed wetlands (Huang *et al*. 2019).

This action includes studies that report the effects of removing pollutants from existing levels of aquaculture effluent. Studies that report the effect of reducing the amount of aquculture pollution that is produced are described in *Reduce pollution from aquaculture effluents*. Aquaculture systems also pose serious environment risks by promoting the spread of non-native, invasive, and pest species and diseases. Evidence for actions related to non-native, invasive and pest species is summarized in *Threat: Invasive and other problematic species, genes and diseases - Aquaculture*.

Becker C., Hughen K., Mincer T.J., Ossolinski J., Weber L. & Apprill A. (2017) Impact of prawn farming effluent on coral reef water nutrients and microorganisms. *Aquaculture Environment Interactions*, 9, 331–346. https://doi.org/10.3354/aei00238

Gómez S., Hurtado C.F. & Orellana J. (2019) Bioremediation of organic sludge from a marine recirculating aquaculture system using the polychaete *Abarenicola pusilla* (Quatrefages, 1866). *Aquaculture*, 507, 377–384. https://doi.org/10.1016/j.aquaculture.2019.04.033

Huang X.F., Ye G.Y., Yi N.K., Lu L.J., Zhang L., Yang L.Y., Xiao L. & Liu J. (2019) Effect of plant physiological characteristics on the removal of conventional and emerging pollutants from aquaculture wastewater by constructed wetlands. *Ecological Engineering*, 135, 45–53. https://doi.org/10.1016/j.ecoleng.2019.05.017

Karia M.T., Haziq A.H., Ramli N.M., Zuhan M.K.N.M. & Razali M.N. (2022) Remediation of aquaculture effluents using physical treatment. *Materials Today: Proceedings*, 57, 1196–1201. https://doi.org/10.1016/j.matpr.2021.10.386

Lugo L.A., Thorarinsdottir R.I., Bjornsson S., Palsson O.P., Skulason H., Johannsson S. & Brynjolfsson S. (2020) Remediation of aquaculture wastewater using the microalga *Chlorella sorokiniana. Water*, 12, 3144. https://doi.org/10.3390/w12113144

Turcios A.E. & Papenbrock J. (2014) Sustainable treatment of aquaculture effluents—what can we learn from the past for the future? *Sustainability*, 6, 836–856. https://doi.org/10.3390/su6020836

Zhao H., Yuan M., Strokal M., Wu H.C., Liu X., Murk A., Kroeze C. & Osinga R. (2021) Impacts of nitrogen pollution on corals in the context of global climate change and potential strategies to conserve coral reefs. *Science of the Total Environment*, 774, 145017. https://doi.org/10.1016/j.scitotenv.2021.145017

# Agriculture and forestry effluents

## 9.8 Reduce pollution from agriculture and forestry effluents

https://www.conservationevidence.com/actions/4050

- We found no studies that evaluated the effects on corals of reducing pollution from agriculture and forestry effluents.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Land used for agriculture and forestry often receives high chemical inputs to control pests, weeds and pathogens, and can also be a source of increased levels of nutrients and sediment (Berger *et al*. 2022, Kroon *et al*. 2014). On entering the marine environment, these pollutants all have the potential to negatively impact corals. A range of methods may be used to reduce the levels of pollution that are produced in agricultural and forestry lands, including regulating the usage and dosage of agrichemicals or switching to production systems with lower chemical inputs.

This action includes studies that report the effects of reducing the amount of agricultural and forestry pollution that is produced. Studies that report the effect of removing pollutants from existing levels of agricultural and forestry effluent are described in *Use biological, chemical or mechanical methods to manage excess pollution from agriculture and forestry effluents*.

Berger M., Canty S.W., Tuholske C. & Halpern B.S. (2022) Sources and discharge of nitrogen pollution from agriculture and wastewater in the Mesoamerican Reef region. *Ocean & Coastal Management*, 227, 106269. https://doi.org/10.1016/j.ocecoaman.2022.106269

Kroon F.J., Schaffelke B. & Bartley R. (2014) Informing policy to protect coastal coral reefs: Insight from a global review of reducing agricultural pollution to coastal ecosystems. *Marine Pollution Bulletin*, 85, 33–41. https://doi.org/10.1016/j.marpolbul.2014.06.003

## 9.9 Use biological, chemical or mechanical methods to manage excess pollution from agriculture and forestry effluents

https://www.conservationevidence.com/actions/4051

- We found no studies that evaluated the effects on corals of using biological, chemical or mechanical methods to manage excess pollution from agriculture and forestry effluents.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Land used for agriculture and forestry often receives high chemical inputs to control pests, weeds and pathogens, and can also be a source of increased levels of nutrients and sediment (Berger *et al*. 2022, Kroon *et al*. 2014). On entering the marine environment, these pollutants all have the potential to negatively impact corals. A range of biological, chemical or mechanical methods may be employed to manage excess pollutants from agriculture and forestry lands, including treating wastewater from livestock holdings, retaining or establishing riparian buffer strips, creating artificial (constructed) wetlands (Tournebize *et al*, 2017), or establishing aquaculture facilities (He *et al*. 2008) to reduce the amount of pollutant reaching the marine environment.

This action includes studies that report the effects of removing pollutants from existing levels of agricultural and forestry effluent. Studies that report the effect of reducing the amount of agricultural and forestry pollution that is produced are described in *Reduce pollution from agriculture and forestry effluents*.

Berger M., Canty S.W., Tuholske C. & Halpern B.S. (2022) Sources and discharge of nitrogen pollution from agriculture and wastewater in the Mesoamerican Reef region. *Ocean & Coastal Management*, 227, 106269. https://doi.org/10.1016/j.ocecoaman.2022.106269

He P., Xu S., Zhang H., Wen S., Dai Y., Lin S. & Yarish C. (2008) Bioremediation efficiency in the removal of dissolved inorganic nutrients by the red seaweed, *Porphyra yezoensis*, cultivated in the open sea. *Water Research*, 42, 1281–1289. https://doi.org/10.1016/j.watres.2007.09.023

Kroon F.J., Schaffelke B. & Bartley R. (2014) Informing policy to protect coastal coral reefs: Insight from a global review of reducing agricultural pollution to coastal ecosystems. *Marine Pollution Bulletin*, 85, 33–41. https://doi.org/10.1016/j.marpolbul.2014.06.003

Tournebize J., Chaumont C. & Mander Ü. (2017) Implications for constructed wetlands to mitigate nitrate and pesticide pollution in agricultural drained watershEds *Ecological Engineering*, 103, 415–425. https://doi.org/10.1016/j.ecoleng.2016.02.014

# Garbage and solid waste

## 9.10 Prevent garbage and solid waste from reaching the marine environment

https://www.conservationevidence.com/actions/4052

- We found no studies that evaluated the effects on corals of preventing garbage and solid waste from reaching the marine environment.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Garbage (litter) and solid waste can enter the marine environment through a multitude of pathways, for instance vessels, rivers, storms, beaches, fishing activities. Once in the marine environment, garbage can accumulate and subsist for a long time due to very slow degradation (Andrady 2015; Connan *et al*. 2021; Pham *et al*. 2014) and in some cases (e.g. derelict fishing gear, so called 'ghost' gear) cause direct harm to corals (Ballesteros *et al*. 2018, Figueroa-Pico *et al*. 2020).

A range of options exist for reducing the amount of garbage and solid waste that reaches the marine environment. For example, installing stormwater traps or grids to prevent garbage from entering stormwaters (Armitage & Rooseboom 2000), enforcing regulations around discarding of waste in coastal and aquatic environments, providing facilities for easy disposal of waste (e.g. fishing gear) or offering incentives for reusing or recycling fishing gear. This action includes studies that report the effects of preventing garbage and solid waste from reaching the marine environment. Studies that report the effect of actions aimed at removing garbage and solid waste are described in *Remove garbage and solid waste from the marine environment*.

Andrady A.L. (2015) Persistence of plastic litter in the oceans. Pages 57–72 in: *Marine Anthropogenic Litter*. Springer: Cham. https://doi.org/10.1007/978-3-319-16510-3_3

Armitage N. & Rooseboom A. (2000) The removal of urban litter from stormwater conduits and streams: Paper 1- The quantities involved and catchment litter management options. *Water SA*, 26, 181–188. https://hdl.handle.net/10520/AJA03784738_2350

Ballesteros, L.V., Matthews, J.L. & Hoeksema, B.W. (2018) Pollution and coral damage caused by derelict fishing gear on coral reefs around Koh Tao, Gulf of Thailand. *Marine Pollution Bulletin*, 135, 1107–1116. https://doi.org/10.1016/j.marpolbul.2018.08.033

Connan M., Perold V., Dilley B.J., Barbraud C., Cherel Y. & Ryan P.G. (2021) The Indian Ocean 'garbage patch': Empirical evidence from floating macro-litter. *Marine Pollution Bulletin*, 169, 112559. https://doi.org/10.1016/j.marpolbul.2021.112559

Figueroa-Pico J., Tortosa F.S. & Carpio A.J. (2020) Coral fracture by derelict fishing gear affects the sustainability of the marginal reefs of Ecuador. *Coral Reefs*, 39, 819–827. https://doi.org/10.1007/s00338-020-01926-6

Pham C.K., Ramirez-Llodra E., Alt C.H., Amaro T., Bergmann M., Canals M., Davies J., Duineveld G., Galgani F., Howell K.L. & Huvenne V.A. (2014) Marine litter distribution and density in European seas, from the shelves to deep basins. *PloS One*, 9, e95839. https://doi.org/10.1371/journal.pone.0095839

## 9.11 Remove garbage and solid waste from the marine environment

https://www.conservationevidence.com/actions/4053

- We found no studies that evaluated the effects on corals of removing garbage and solid waste from the marine environment.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Garbage (litter) and solid waste can enter the marine environment through a multitude of pathways, for instance vessels, rivers, storms, beaches, fishing activities. Once in the marine environment, garbage can accumulate and subsist for a long time due to very slow degradation (Andrady 2015; Connan *et al*. 2021; Pham *et al*. 2014) and in some cases (e.g. derelict fishing gear, so called 'ghost' gear) cause direct harm to corals (Ballesteros *et al*. 2018, Figueroa-Pico *et al*. 2020). Removing this solid waste from the environment may reduce the threat posed to corals and allow for the recovery of corals and the reefs they form.

This action includes studies that report the effects of removing garbage and solid waste from the marine environment. Studies that report the effect of actions aimed at preventing garbage and solid waste from reaching the marine environment are described in *Prevent garbage and solid waste from reaching the marine environment*.

Andrady A.L. (2015) Persistence of plastic litter in the oceans. Pages 57–72 in: *Marine Anthropogenic Litter*. Springer: Cham. https://doi.org/10.1007/978-3-319-16510-3_3

Ballesteros L.V., Matthews J.L. & Hoeksema B.W. (2018) Pollution and coral damage caused by derelict fishing gear on coral reefs around Koh Tao, Gulf of Thailand. *Marine Pollution Bulletin*, 135, 1107–1116. https://doi.org/10.1016/j.marpolbul.2018.08.033

Connan M., Perold V., Dilley B.J., Barbraud C., Cherel Y. & Ryan P.G. (2021) The Indian Ocean 'garbage patch': Empirical evidence from floating macro-litter. *Marine Pollution Bulletin*, 169, 112559. https://doi.org/10.1016/j.marpolbul.2021.112559

Figueroa-Pico J., Tortosa F.S. & Carpio A.J. (2020) Coral fracture by derelict fishing gear affects the sustainability of the marginal reefs of Ecuador. *Coral Reefs*, 39, 819–827. https://doi.org/10.1007/s00338-020-01926-6

Pham C.K., Ramirez-Llodra E., Alt C.H., Amaro T., Bergmann M., Canals M., Davies J., Duineveld G., Galgani F., Howell K.L. & Huvenne V.A. (2014) Marine litter distribution and density in European seas, from the shelves to deep basins. *PloS One*, 9, e95839. https://doi.org/10.1371/journal.pone.0095839

## 9.12 Change to fishing gear made from biodegradable materials

https://www.conservationevidence.com/actions/4054

- We found no studies that evaluated the effects on corals of changing to fishing gear made from biodegradable materials.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore,*

*we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Abandoned, lost or otherwise discarded fishing gear (or 'ghost' gear) causes direct harm to corals (Ballesteros *et al*. 2018, Figueroa-Pico *et al*. 2020). Synthetic materials used for fishing gear, such as nylon, may persist for decades, leading to an accumulation of 'ghost' gear in marine and freshwater environments. Biodegradable fishing gear, which is naturally broken down by microbes or ultraviolet light, may offer an alternative to traditional materials (Kim *et al*. 2016) and help to reduce the impact of discarded fishing gear on corals. Careful consideration of the materials used is important, ensuring that the degraded products of such materials also have no negative impact on corals and the marine environment.

Ballesteros, L.V., Matthews, J.L., & Hoeksema, B.W. (2018). Pollution and coral damage caused by derelict fishing gear on coral reefs around Koh Tao, Gulf of Thailand. *Marine Pollution Bulletin*, 135, 1107–1116. https://doi.org/10.1016/j.marpolbul.2018.08.033

Figueroa-Pico J., Tortosa F.S. & Carpio A.J. (2020) Coral fracture by derelict fishing gear affects the sustainability of the marginal reefs of Ecuador. *Coral Reefs*, 39, 819–827. https://doi.org/10.1007/s00338-020-01926-6

Kim S., Kim P., Lim J., An H. & Suuronen P. (2016) Use of biodegradable driftnets to prevent ghost fishing: physical properties and fishing performance for yellow croaker. *Animal conservation*, 19, 309–319. https://doi.org/10.1111/acv.12256

# Excess energy: light and noise pollution

## 9.13 Prohibit or reduce light pollution near coral reefs

https://www.conservationevidence.com/actions/4055

- We found no studies that evaluated the effects on corals of prohibiting or reducing light pollution near coral reefs.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Artificial lighting in coastal and offshore developments has been shown to influence the timing of coral spawning, with the potential to negatively impact coral gamete (eggs and sperm) fertilization and survival (Davies *et al*. 2023). A range of options could be employed to potentially reduce the threat of artificial light on corals, including regulating artificial light during vulnerable periods, regulating the timing of nighttime lighting (e.g. switching on one hour after sunset; Davies *et al*. 2023), and enforcing those regulations.

Davies T.W., Levy O., Tidau S., de Barros Marangoni L.F., Wiedenmann J., D'Angelo C. & Smyth T. (2023) Global disruption of coral broadcast spawning associated with artificial light at night. *Nature Communications*, 14, 2511. https://doi.org/10.1038/s41467-023-38070-y

## 9.14 Prohibit or reduce noise pollution near coral reefs

https://www.conservationevidence.com/actions/4056

- We found no studies that evaluated the effects on corals of prohibiting or reducing noise pollution near coral reefs.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

There is a growing understanding of the role of acoustic cues in coral settlement (Lillis *et al*. 2016), with larvae responding to the 'soundscapes' typical of healthy reef habitat with higher settlement rates (Lillis *et al*. 2018). Noise pollution poses a threat to corals by masking these natural coral reef soundscapes (Ferrier-Pagès *et al*. 2021) and disrupting coral settlement cues (Lecchini *et al*. 2018). A range of options could be employed to potentially reduce the threat of noise pollution on corals, including imposing and enforcing noise restrictions in proximity to coral reefs or restricting noise during particularly sensitive time periods.

Ferrier-Pagès C., Leal M.C., Calado R., Schmid D.W., Bertucci F., Lecchini D. & Allemand D. (2021) Noise pollution on coral reefs? — A yet underestimated threat to coral reef communities. *Marine Pollution Bulletin*, 165, 112129. https://doi.org/10.1016/j.marpolbul.2021.112129

Lecchini D., Bertucci F., Gache C., Khalife A., Besson M., Roux N., Berthe C., Singh S., Parmentier E., Nugues M.M., Brooker R.M., Dixson D.L. & Hédouin L. (2018) Boat noise prevents soundscape-based habitat selection by coral planulae. *Scientific reports*, 8, 9283. https://doi.org/10.1038/s41598-018-27674-w

Lillis A., Bohnenstiehl D., Peters J.W. & Eggleston D. (2016) Variation in habitat soundscape characteristics influences settlement of a reef-building coral. *PeerJ*, 4, e2557. https://doi.org/10.7717/peerj.2557

Lillis A., Apprill A., Suca J.J., Becker C., Llopiz J.K. & Mooney T.A. (2018) Soundscapes influence the settlement of the common Caribbean coral *Porites astreoides* irrespective of light conditions. *Royal Society Open Science*, 5, 181358. https://doi.org/10.1098/rsos.181358

# Excess energy: thermal pollution

## 9.15 Limit, cease or prohibit pollution caused by excess thermal energy

https://www.conservationevidence.com/actions/4057

- We found no studies that evaluated the effects on corals of limiting, ceasing or prohibiting pollution caused by excess thermal energy.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Cooling effluent from power stations is a source of excess thermal energy that has the potential to impact corals when discharged into the ocean (Fan 1992). Limiting, ceasing or prohibiting discharge of this excess thermal energy may reduce this threat to corals and allow for their recovery.

Fan K.L. (1992) The thermal discharges from nuclear power plants in Taiwan. *Chemistry and Ecology*, 6, 213–224. https://doi.org/10.1080/02757549208035273

# Other pollution

## 9.16 Reduce pollution from toxic antifouling coatings

https://www.conservationevidence.com/actions/4058

- We found no studies that evaluated the effects on corals of reducing pollution from toxic antifouling coatings.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Antifouling paints and coatings are commonly used to manage biofouling (organisms that can attach to hard surfaces) on vessels and other hard anthropogenic structures. However, some of the antifouling paints and coatings (e.g. tributyltin, also known as TBT) are toxic to corals and other marine organisms, and are detected in even remote and "pristine" habitats such as the Antartic and the Great Barrier Reef (Negri & Marshall 2009). A range of options could be employed to potentially reduce the threat of toxic antifouling paints and coatings, including restricting their use through bans or regulations, or developing non-toxic alternatives, thereby allowing coral populations to recover.

Negri A. & Marshall P. (2009) TBT contamination of remote marine environments: Ship groundings and ice-breakers as sources of organotins in the Great Barrier Reef and Antarctica. *Journal of Environmental Management*, 90, S31–S40. https://doi.org/10.1016/j.jenvman.2008.06.009

# 10. Threat: Climate change and severe weather

**Background**

Climate change and severe weather are having a catastrophic impact on coral reefs with increasing evidence of widespread bleaching caused by rising sea temperatures (Hoegh-Guldberg *et al*. 2017). Changes in coral reef ecosystems can be caused by changes in coastal water quality and rainfall intensity. Damage and destruction of reefs is occurring through increased wave energy brought on by ever more intense storms (Hoegh-Guldberg *et al*. 2017). However, these are very large-scale threats so actions that could be carried out in response to them need to be on a global scale.

Chapters describing conservation actions aimed at protecting habitats, restoring coral reefs and coral species are described in *Habitat protection; Habitat restoration and creation*; and *Species management*.

Hoegh-Guldberg, O., Poloczanska, E.S., Skirving, W. & Dove, S. (2017) Coral reef ecosystems under climate change and ocean acidification. *Frontiers in Marine Science*, 4, Article 158. https://doi.org/10.3389/fmars.2017.00158

 https://doi.org/10.11647/OBP.0453.10

# 11. Habitat protection

**Background**

Habitat destruction is the largest threat to biodiversity worldwide. Habitat fragmentation and degradation reduces both the amount and quality of remaining habitat, and as such, habitat protection may play a vital role in global conservation efforts. Habitat can be protected through the designation of legally protected areas using national or local area legislations. It can range from entire habitat protection (e.g. EU Habitats Directive 1992, USA Habitat Conservation Plans under the Endangered Species Act of 1973, and Environment Canada Protected Areas Strategy 2011) to community conservation with no formal protection or designation schemes. It can be difficult, if not impossible, to measure the effectiveness of legal protection on an area as there are usually no suitable comparisons. For example, monitoring generally only begins once the designation to a protected area comes into effect, meaning pre-designation data often do not exist, and the best quality habitats are often those selected for protection, meaning a similar unprotected habitat is not available as a comparison.

## 11.1 Designate a Marine Protected Area and prohibit all types of fishing, collecting and access

https://www.conservationevidence.com/actions/3999

- **One study** examined the effects on corals of designating a Marine Protected Area and prohibiting all types of fishing, collecting and access. This study was in the Mediterranean[1].


https://doi.org/10.11647/OBP.0453.11

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (1 STUDY)**

- **Condition (1 study)**: One replicated, site comparison study in the Mediterranean[1] found that in protected areas, stony coral *Corallium rubrum* colonies were larger than in unprotected areas according to some diameter and height measurements, but similar according to others.

**Background**

Fishing, collecting of corals and other organisms, and a range of other recreational and non-extractive activities can have both direct and indirect impact on corals. Prohibiting some or all of these activities within Marine Protected Areas is a widely used conservation action (Kriegl *et al*. 2021).

Fishing and collecting poses a direct threat when gear damages corals during operations (Mangi & Roberts 2006, Althaus *et al*. 2009) or is lost or abandoned (so called 'ghost' gear; Ballesteros *et al*. 2018, Figueroa-Pico *et al*. 2020). Indirect threats can emerge when overfishing reduces the diversity and abundance of herbivorous fishes, with consequences for macroalgal abundance and its potential to impact coral growth (Burkepile *et al*. 2010). Recreational diving and anchoring (often associated with diving) can also cause physical damage to corals (Hasler & Ott 2008, Giglio *et al*. 2017), and damage caused by high levels of tourism has been associated with greater prevalence of coral disease (Lamb *et al*. 2014).

Prohibiting all fishing and collecting (so called no-take areas/zones) and access in protected areas has the potential to reduce all these threats and allow corals and the reefs they form to recover and flourish.

Restricted fishing types covered here include hook and line fishing; bottom trawling, dredging and other towed gear; static gear including traps; and dynamite and cyanide fishing.

This action specifically refers to studies that report the effect of prohibiting fishing, collecting and access. Studies that report the effect of prohibiting a subset of fishing, collecting and access are described in the following actions: *Designate a Marine Protected Area and prohibit all types of fishing and collection; Designate a Marine Protected Area and prohibit all types of fishing; Designate a Marine Protected Area and prohibit all types of collection; Designate a Marine Protected Area and prohibit some fishing and collection* (*including where restrictions are unspecified*) and *Designate a Marine Protected Area and prohibit/limit recreational activities* (*including anchoring*).

Althaus F., Williams A., Schlacher T.A., Kloser R.J., Green M.A., Barker B.A., Bax N.J., Brodie P. & Schlacher-Hoenlinger M.A. (2009) Impacts of bottom trawling on deep-coral ecosystems of seamounts are long-lasting. *Marine Ecology Progress Series*, 397, 279–294. https://doi.org/10.3354/meps08248

Ballesteros L.V., Matthews J.L. & Hoeksema, B.W. (2018). Pollution and coral damage caused by derelict fishing gear on coral reefs around Koh Tao, Gulf of Thailand. *Marine Pollution Bulletin*, 135, 1107–1116. https://doi.org/10.1016/j.marpolbul.2018.08.033

Burkepile D.E. & Hay M.E. (2010) Impact of herbivore identity on algal succession and coral growth on a Caribbean reef. *PloS One*, 5, e8963. https://doi.org/10.1371/journal.pone.0008963

Figueroa-Pico J., Tortosa F.S. & Carpio A.J. (2020) Coral fracture by derelict fishing gear affects the sustainability of the marginal reefs of Ecuador. *Coral Reefs*, 39, 819–827. https://doi.org/10.1007/s00338-020-01926-6

Giglio V.J., Ternes M.L., Mendes T.C., Cordeiro C.A. & Ferreira C.E. (2017) Anchoring damages to benthic organisms in a subtropical scuba dive hotspot. *Journal of Coastal Conservation*, 21, 311–316. https://doi.org/10.1007/s11852-017-0507-7

Hasler H. & Ott J.A. (2008) Diving down the reefs? Intensive diving tourism threatens the reefs of the northern Red Sea. *Marine Pollution Bulletin*, 56, 1788–1794. https://doi.org/10.1016/j.marpolbul.2008.06.002

Kriegl M., Elías Ilosvay X.E., von Dorrien C. & Oesterwind D. (2021) Marine protected areas: at the crossroads of nature conservation and fisheries management. *Frontiers in Marine Science*, 8, Article 676264. https://doi.org/10.3389/fmars.2021.676264

Lamb J.B., True J.D., Piromvaragorn S. & Willis B.L. (2014) Scuba diving damage and intensity of tourist activities increases coral disease prevalence. *Biological Conservation*, 178, 88–96. https://doi.org/10.1016/j.biocon.2014.06.027

Mangi S.C. & Roberts C.M. (2006) Quantifying the environmental impacts of artisanal fishing gear on Kenya's coral reef ecosystems. *Marine Pollution Bulletin*, 52, 1646–1660. https://doi.org/10.1016/j.marpolbul.2006.06.006

A replicated, site comparison study in 2003–2005 in three sites, with data from an additional 11 sites, off the coasts of France, Spain and Italy (1) found that in protected areas that prohibited all fishing, collecting and access, stony coral *Corallium rubrum* colonies were larger than in unprotected areas according to some diameter and height measurements, but similar according to others. Results were not tested for statistical significance. Average basal diameter of stony coral colonies in three protected areas (6–9 mm) was larger than in eight unprotected areas in all comparisons (4–5 mm), and average height (39–74 mm) was larger than in 10 unprotected areas (20–69 mm) in 28 of 30 comparisons. Maximum diameter of colonies was similar in protected (17–30 mm) and unprotected areas (7–32 mm), as was maximum height (protected: 150–190 mm, unprotected: 52–200 mm). For the 30 largest colonies at each site, height was larger in protected than in unprotected areas, but diameter was similar, and the percentage of colonies over a given size threshold tended to be larger in protected compared to unprotected areas (see paper for data). Three protected areas were selected, and areas where all activity was prohibited were chosen for sampling. Several transects/area were selected (2 $m^2$ total area) and photographs were taken along transects, placing a 20 × 20 cm quadrat sequentially along the transect (two photographs/quadrat placement). Coral measurements were compared with values from previously published studies from 10 unprotected sites.

(1) Linares C., Bianchimani O., Torrents O., Marschal C., Drap P., & Garrabou J. (2010) Marine Protected Areas and the conservation of long-lived marine invertebrates: the Mediterranean red coral. *Marine Ecology Progress Series*, 402, 69–79. https://doi.org/10.3354/meps08436.

## 11.2 Designate a Marine Protected Area and prohibit all types of fishing and collection

https://www.conservationevidence.com/actions/4007

- **Twenty-six studies** evaluated the effects on corals of designating a Marine Protected Area and prohibiting all types of fishing and collection. Seven studies were in Kenya[2,4,6,9,16,25], or Kenya and Tanzania[1], four were in Australia[5,14,20,26], four were in the Philippines[12,15,21,24], three were in each of the USA[8,17,18], and Belize[3,13,22], and one study was in each of the Cayman Islands[7], China[23], Tanzania[11], Thailand[19] or had global coverage[10].

**COMMUNITY RESPONSE (7 STUDIES)**

- **Richness/diversity (7 studies)**: Two of three replicated, site comparison studies (including one before-and-after study) in Kenya[9,25] and the USA[8] found that coral richness was higher in protected than in unprotected areas[9,25]. The other study found that richness was lower in protected than in unprotected areas[8]. One replicated, site comparison study in Tanzania[11] found more coral species groups in protected areas than areas without formal management. Two of three studies (including one replicated, before-and-after, site comparison study) in Australia[14], Kenya[25] and the Philippines[24] found that diversity of species between sites was lower for protected than unprotected areas[25] or that differences in community composition between protected and unprotected areas were dependent on proximity to mangrove habitat[14]. The other study found that overall, community traits were similar in protected and unprotected areas[24]. One replicated, before-and-after, site comparison study in Kenya[4] found that coral diversity was higher in protected than in unprotected areas.

**POPULATION RESPONSE (25 STUDIES)**

- **Abundance/Cover (25 studies)**: Twelve of 19 studies (including four replicated, before-and-after, site comparison studies) in Australia[5,19,26], Belize[3,13,22], China[23], Kenya[2,6,9,16],

Kenya and Tanzania[1], the Philippines[15,21,24], Tanzania[11], Thailand[19] and the USA[8,17] found that coral cover was similar in protected and unprotected areas[1,3,5,13,15,16,17,19,20,22,24] or cover of sessile invertebrates (including soft coral) was similar[3,26]. Four studies found that coral cover was higher in protected areas[2,9] or in most protected areas[21] compared to unprotected areas or higher prior to a bleaching event but similar four years after bleaching[6]. One study found that coral cover was lower in protected than in unprotected areas[8], and two studies found that differences in cover were mixed[11,23]. Two of four studies (including one replicated, site comparison study) in Australia[14], Belize[13], the USA[18] and a global review[10] found that density of corals[10] or juvenile corals[13] was similar in protected and unprotected areas. The other two studies found that either density of coral recruits was higher in protected than in unprotected areas at two of three depths[18], or that close to mangroves, density of recruits was higher in protected than unprotected sites, but densities were similar in protected and unprotected sites more distant from mangroves[14]. Four of nine studies (including three replicated, paired, site comparison studies) found mixed trends in coral cover in protected compared to unprotected areas for hard corals[21], hard vs soft corals[3,16] or broadcasting vs brooding corals[13]. Three studies found similar declines in coral cover in protected and unprotected areas[7,17] or no relationship between cover and duration of protection[12]. One study found positive changes in cover in protected areas and no change in unprotected areas[15], and one study found positive changes in cover in one of four protected areas and no change in the others[24].

- **Reproductive success (1 study)**: One replicated, before-and-after, site comparison study in Kenya[6] found that coral recruitment was similar in protected and unprotected areas.
- **Condition (8 studies)**: Three of four studies (including one replicated, before-and-after site comparison study) in Belize[13], China[23], Kenya[9] and a global review[10] found that coral size[10] or growth[9,23] was similar in protected and unprotected areas. The other study[13] found that size differences were

mixed for broadcasting and brooding corals in protected and unprotected areas. Two of four replicated, site comparison studies in Australia[20], Cayman Islands[7], Kenya[2] and Thailand[19] found that disease[7,19] and a range of other health indicators[19] were similar in protected and unprotected areas. One of the studies[7] also found that the extent of bleaching was mixed in protected and unprotected areas. Two studies found that disease, damage and bleaching[20] or erosion caused by urchins[2] was lower in protected areas (or older protected areas[2]) than in unprotected areas.

**OTHER (1 STUDY)**

- **Structural complexity (1 study)**: One randomized, replicated, site comparison study in Australia[5] found that coral structural complexity was similar in protected and unprotected areas.

**Background**

Fishing and collecting of corals and other organisms can have both direct and indirect impacts on corals. Prohibiting some or all of these activities within Marine Protected Areas is a widely used conservation action (Kriegl *et al*. 2021).

Fishing and collecting poses a direct threat when gear damages corals during operations (Mangi & Roberts 2006, Althaus *et al*. 2009) or is lost or abandoned (so called 'ghost' gear; Ballesteros *et al*. 2018, Figueroa-Pico *et al*. 2020). Indirect threats can emerge when overfishing reduces the diversity and abundance of herbivorous fishes, with consequences for macroalgal abundance and its potential to impact coral growth (Burkepile *et al*. 2010).

Prohibiting all fishing and collecting (so called no-take areas/zones) has the potential to reduce these threats and allow corals and the reefs they form to recover and flourish.

Restricted fishing types covered here include hook and line fishing; bottom trawling, dredging and other towed gear; static gear including traps; and dynamite and cyanide fishing.

Studies that report the effect of prohibiting fishing and collection in addition to access are described in *Designate a Marine Protected Area and prohibit all types of fishing, collecting and access*, and those that prohibit a subset of fishing or collection activities are described in the following actions: *Designate a Marine Protected Area and prohibit all types of fishing; Designate a Marine Protected Area and prohibit all types of collection; Designate a Marine Protected Area and prohibit some fishing and collection* (*including where restrictions are unspecified*).

Althaus F., Williams A., Schlacher T.A., Kloser R.J., Green M.A., Barker B.A., Bax N.J., Brodie P. & Schlacher-Hoenlinger M.A. (2009) Impacts of bottom trawling on deep-coral ecosystems of seamounts are long-lasting. *Marine Ecology Progress Series*, 397, 279–294. https://doi.org/10.3354/meps08248

Ballesteros L.V., Matthews J.L. & Hoeksema, B.W. (2018). Pollution and coral damage caused by derelict fishing gear on coral reefs around Koh Tao, Gulf of Thailand. *Marine Pollution Bulletin*, 135, 1107–1116. https://doi.org/10.1016/j.marpolbul.2018.08.033

Burkepile D.E. & Hay M.E. (2010) Impact of herbivore identity on algal succession and coral growth on a Caribbean reef. *PloS One*, 5, e8963. https://doi.org/10.1371/journal.pone.0008963

Figueroa-Pico J., Tortosa F.S. & Carpio A.J. (2020) Coral fracture by derelict fishing gear affects the sustainability of the marginal reefs of Ecuador. *Coral Reefs*, 39, 819–827. https://doi.org/10.1007/s00338-020-01926-6

Kriegl M., Elías Ilosvay X.E., von Dorrien C. & Oesterwind D. (2021) Marine protected areas: at the crossroads of nature conservation and fisheries management. *Frontiers in Marine Science*, 8, Article 676264. https://doi.org/10.3389/fmars.2021.676264

Mangi S.C. & Roberts C.M. (2006) Quantifying the environmental impacts of artisanal fishing gear on Kenya's coral reef ecosystems. *Marine Pollution Bulletin*, 52, 1646–1660. https://doi.org/10.1016/j.marpolbul.2006.06.006

A replicated, before-and-after, site comparison study in 1974 and 1996 in 15 marine sites in Kenya and Tanzania (1) found that protected areas that prohibited all types of fishing and collection did not have a higher cover of corals compared to unprotected sites and the dominance of corals varied between sites and was not related to protection level. In five marine protected sites, hard coral cover (33%) was similar compared to

10 unprotected sites (average 40%), all surveyed in 1996, while the cover of sand was higher in protected sites (average 12%) than in unprotected sites (3%). The dominance of corals varied widely between sites, while fish abundance (kg/ha) was 350% higher and sea urchin abundance was 600% lower in protected sites compared to unprotected sites (see paper for details). Coral abundance and cover varied widely between surveys at the four unprotected sites surveyed both 1974 and 1996 (see paper for details). Five protected sites, regularly patrolled to exclude fishing (two in the 10km$^2$ Kisite Marine National Park, on the Kenyan–Tanzanian border and three in the 500 m long Chumbe Island Coral Park, Zanzibar) and 10 unprotected sites used for fishing were selected. Attached benthic communities including corals were studied. Four of the unprotected sites surveyed in 1974 were resurveyed in 1996.

A replicated, site comparison study in 1995–1998 in 18 marine sites in coastal Kenya (2) found that protected areas that prohibited all types of fishing and collection had higher coral cover and lower coral bioerosion from sea urchins compared to unprotected reefs. Live coral cover was higher in older protected (42%) and newly protected areas (44%) compared to unprotected reefs (18%). Coral bioerosion rates from sea urchins were 20 times lower in older protected areas (50 $CaCO_3$/m$^2$/year) compared to unprotected reefs (1180 g $CaCO_3$/m$^2$/year), and intermediate in newly protected reefs (711 g $CaCO_3$ /m$^2$/year). Total sea urchin density and biomass were 10 times lower in older protected areas (density: 0.6 urchins/10 m$^2$; biomass: 134g/10 m$^2$) compared to unprotected sites (62 urchins/10 m$^2$; biomass: 3182 g/10 m$^2$), while values were intermediate in newly protected areas (12 urchins/10 m$^2$; biomass: 2032 g/10 m$^2$). Unless specified, differences were not statistically assessed. Coral reefs within each category (five sites in older protected areas, three in newly protected areas and 10 in unprotected sites) were each separated by >20 km and distributed along ~390 km of the coastline. Sites included Malindi and Watamu National Parks, which have been protected from fishing and shell collection since 1968, and Kisite Marine National Park, similarly protected since 1972. Mombasa Marine National Park was established in 1989 but received effective protection from fishing only in 1991.

A site comparison study in 1998–1999 in a marine reserve off the coast of Belize (3) found that in an area where all fishing and collection

was prohibited, stony coral cover was lower and soft gorgonian coral cover was similar compared to a general use area where some fishing was allowed. Stony coral cover was lower where fishing and collection was prohibited (10–26%) than in the general use area (15–34%), but gorgonian cover was similar in both areas (no fishing: 3–7%, some fishing: 2–9%). Across all areas, both stony coral and gorgonian cover declined over the course of the study (stony corals: 10% reduction, gorgonian corals: 3% reduction). Sixteen patch reefs sites were selected (average 1,000 $m^2$): eight from the area where all fishing and collection was prohibited, and eight from the area with some fishing allowed. Coral cover was assessed in September, October (half of patches) and December of 1998, and in April and September 1999 along three 10 m transects on each patch reef. All organisms over 3 cm in size were recorded. After the first sampling event, algae were removed from half of the patches.

A replicated, before-and-after, site comparison study in 1997–1999 in 16 marine sites in coastal Kenya (4) found that protected areas that prohibited all types of fishing and collection had higher coral diversity compared to unprotected, heavily fished areas before a major beaching event, but there were widespread live coral declines in both protected and unprotected reefs after the strong 1998 El Niño event. Coral diversity was higher in protected areas compared to unprotected areas before the 1998 bleaching (reported as diversity index values) and was similar between protected and unprotected areas four months post-bleaching but the decline was higher in the protected sites (77% decline from 40% to 11% cover of benthic substrate) than in unprotected sites (44% decline from 21% to 11%). Soft coral decline was lower in protected sites (65% decline from 4% to 1%) than in unprotected sites (86% decline from 3% to 0.4%). Nine coral reef sites in four Marine Protected Areas (Malindi, Watamu National Parks, Mombasa and Kisite Marine National Parks) and seven sites in four non-protected reef sites, were selected. Sites were separated by 20–100 m within each reef and 3–50 km between reefs and were distributed along ~150 km of the coastline. Before and post-bleaching recovery was monitored in the 16 sites in 1997–1999 using 10 m long transects.

A randomized, replicated, site comparison study in 2001–2002 in 32 marine sites across two groups of islands in the Great Barrier Reef,

Australia (5) found that protected areas that prohibited all types of fishing and collection had similar cover and structural complexity of live corals compared to sites where some fishing was allowed. Live coral cover and structural complexity were similar between protected and unprotected sites (data reported as statistical model results). Densities of individual prey fish species varied between areas but leopard coral grouper and spotted coral grouper (*Plectropomus leopardus* and *Plectropomus maculatus*) had three times higher biomass in the protected sites compared to the unprotected, fished sites (see paper for details). Five 50 m long and 6 m wide replicate belt transects were surveyed per site at 16 sites in Whitsunday Islands (8 fished and 8 protected as Marine Protected Areas for 14 years) and 16 sites at Palm Islands (8 in Marine Protected areas and 8 in areas with recreational and commercial line and trawling fishing but away from the reef). The two island groups are 315 km apart. Transects (minimum 5 m apart) were laid haphazardly in April 2002 along the reef slope at 7–11 m depth and percentage live coral cover was estimated every 2 m along the transects.

A replicated, before-and-after, site comparison study in 1995–2002 in 16 sites across six marine areas in coastal Kenya (6) found twice as much coral cover in protected sites compared to unprotected sites but after the mass coral bleaching event in 1998, protected areas that prohibited all types of fishing and collection had similar coral recruitment levels compared to marine areas where fishing was allowed, and coral mortality and post-bleaching recruitment varied widely between sites. Before bleaching, coral cover was double in sites with no fishing (48–52% cover) compared to sites where fishing was allowed (19–27%). However, four years after bleaching, coral cover varied between sites but was similar between non-fished sites (14–25%) and fished sites (6–27%). Coral recruitment was similar in terms of recruit abundance for each of the 23 coral genera included between protected sites with no fishing and sites where fishing was allowed (see paper for details). All protected sites had higher abundance and biomass of fish compared to heavily-fished reefs (see paper for details). Coral reef sites of similar aspects were selected, three sites in Marine Protected Areas with no fishing (Malindi, Watamu and Mombasa National Parks MNPs) and three sites in four partly protected sites where fishing was allowed. Coral recruits were sampled annually in 1999–2001 within 16 sites (seven in the three

unfished MPAs and nine in fished sites). In each site 5 × 0.25 m quadrats were randomly placed at least 5 m apart and >70% on hard substratum. Corals 0.5–5.0 cm in diameter were classified as recruits.

A replicated, site comparison study in 1999–2004 in nine marine sites around the coast of Little Cayman Island in the Caribbean Sea (7), found that during a period of hurricane events, protected areas that prohibited all types of fishing and collection showed similar decreases in live coral cover compared to unprotected sites, while coral bleaching and coral disease similarly affected both protected and unprotected sites. Average live coral cover decreased at similar levels both inside the four marine no-take reserves (from 1999: 29% to 2004: 19%) and the five non-protected marine sites (from 1999: 24% to 2004: 14%). Coral disease prevalence was similar inside protected and unprotected sites each year (see paper for details). Bleaching in protected areas was lower in comparison to non-protected areas in 1999, but the reverse was observed in 2002 (see paper for details). Nine sites were selected, four marine no-take reserves (protected since the mid-1980s) and five non-protected marine sites. In summer months in 1999–2004 randomly placed 10 m transects (8–15) were surveyed per site (9–13 m deep) to measure hard coral cover, diversity, size and disease presence. In 2002–2004 Little Cayman was impacted by multiple hurricanes and tropical storms.

A replicated, site comparison study in 2003–2004 in five marine reef areas at Florida Keys, USA (8) found that protected areas that prohibited all types of fishing and collection (including physical contact with organisms) had lower coral cover and diversity, but similar juvenile coral abundance compared to sites where fishing was permitted. Across the two survey years coral cover was lower in protected sites (2003: 9, 17, 26%; 2004: 13, 22, 25%) compared to the non-protected sites (2003: 42, 49%; 2004: 39, 55%). Coral species richness was also lower in protected sites, but juvenile abundance and levels of bleaching did not differ in protected and unprotected sites (data as statistical results only). Predatory fish biomass was higher in protected sites compared to non-protected sites while the opposite was true for juvenile parrotfish (see paper for details). In 1997 a total of 24 non-consumptive zones (Special Protection Areas) were created in Florida Keys where all fishing types and all physical contact with organisms were prohibited. Three protected and two non-protected reef sites were selected by matching depth range,

reef size, complexity and distance from shore. In June 2003–2004, four 20 m transect lines were randomly placed and surveyed on each reef, and 5 quadrat locations (1 $m^2$) were randomly selected on either side to quantify benthic cover, juvenile coral abundance and coral health.

A replicated, before-and-after, site comparison study in 1992–2004 in 12 sites across seven marine areas in coastal Kenya (9) found that protected areas that prohibited all types of fishing and collection had higher coral cover and richness compared to areas where fishing was allowed, but after the mass coral mortality event in 1998, coral recovery rates were low and similar between protected and unprotected sites. In sites with no fishing, coral cover (29 % cover) and diversity were greater (15 coral genera) compared to sites where fishing was allowed (16 % cover; 10 genera). However, coral growth was similar between protected sites with no fishing (2%/year) and unprotected sites with fishing (2%/year). Twelve coral reef sites were selected, five in three Marine Protected Areas with no fishing and seven in three unprotected sites with heavy fishing. All forms of fishing were excluded within Malindi and Watamu National Parks since 1968 and in Mombasa Marine Protected Area since 1991. In each site there was a permanently marked 30 × 30 m area where 9–12 randomly-placed 10 m benthic line-intercept transects were surveyed annually in 1992–2004. The cover of major benthic substrate groups was measured on each transect and hard corals were identified to species group.

A review of 31 studies of global protected areas (10) found that in protected areas that prohibited all types of fishing and collection, hard and soft coral density and size was similar to in unprotected areas. Density and size were not significantly different in protected compared to unprotected areas for hard corals (density: 120% higher in protected than unprotected, based on 22 studies; size: 102% higher in protected than unprotected, 1 study), soft corals (density: 14% lower in protected than unprotected, 3 studies; size: 52% higher in protected than unprotected, 1 study), or hard and soft corals combined (density: 2% lower in protected than unprotected, 4 studies, size: no data reported). In addition, when data on all species groups were included (fish, invertebrates, algae), there was no difference in biomass, density, size or richness inside and outside reserves before protection was implemented (see paper for details). The peer reviewed literature was searched for studies on fully

protected, no-take marine reserves, with only those with comparisons to unprotected areas, comparisons to areas before protection, or both being included in analysis. A total of 221 studies from 1977–2006 from 124 marine reserves were retained for analysis, although only 31 of those studies included results for corals. For comparisons of inside and outside reserves before protection, 23 studies were used.

A replicated, site comparison study in 1996–2005 in one area off Zanzibar, Tanzania (11) reported that sites that prohibited all types of fishing and collection had more coral species groups and higher coverage of hard corals but lower soft coral coverage than sites in areas with no formal management restrictions. Results were not tested for statistical significance. Sites with restrictions had a total of 44 coral species groups, 62–63% cover of hard corals and 0–1% cover of soft corals. Sites with no formal management had 23–25 species groups, 45–52% cover of hard corals and 4–17% of soft coral. When all areas with restrictions (including three areas that prohibited only some fishing and collection) and areas without restrictions were analysed together, no significant differences were found in the abundance of different coral species groups over time. Two sites were selected where all fishing and collection were prohibited, and two were selected where there was no formal management. Each site was sampled using nine 10 m transects and all organisms 3 cm or larger were recorded.

A replicated, paired, site comparison study in 2006–2008 in 30 coral reef sites in central Philippines (12) found that in protected areas that prohibited all types of fishing and collection, coral cover did not increase with duration of protection. There was no relationship between hard coral cover and duration of protection in protected areas (16–68% cover, 0.5–11 years of protection) or in adjacent unprotected areas (6–54% cover, 0.5–11 years since protection started in adjacent areas). Soft corals similarly showed no relationship (no data reported). Fifteen protected areas, managed by local communities, were selected with duration of protection ranging from six months to 11 years. Fifteen unprotected (fished) sites were also selected, 11 adjacent to the protected areas (<500 m away), and four <2 km away. In 2006–2008, surveys were conducted along 50 × 10 m transects (six transects/site), and benthic cover was recorded every 0.5 m (100 points/transect).

A site comparison study in 1998–2009 at 87 patch reef sites off the coast

of Belize (13) found that in a protected area that prohibited all fishing and collection, change in coral cover over 10 years was similar compared to the unprotected area, but the protected area lost more broadcast-spawning coral species and tended to have smaller colonies than the unprotected area. Coral cover remained similar in both the protected and unprotected areas (1998–1999: 8%, 2008–2009: 7%). Authors reported that two previous studies estimated coral cover at this site of 20% in 1996–1997 and 82% in 1970–1971. Cover of broadcast-spawning corals declined more in the protected area (–5%) than the unprotected area (–2%), cover of corals that reproduce by brooding stayed similar in both (0% change). Estimated juvenile coral density was similar in the protected (0.5 individuals/$m^2$) and unprotected area (0.6 individuals/$m^2$). Two broadcasting species tended to be smaller and one brooding species larger in the protected area compared to unprotected area (see paper for details). A no-take protected area was established in 1998. In 2008–2009, a total of 87 patch reef sites were selected, 51 of which had previously been sampled in 1998–1999 (1 year after protection). In 1998–1999, four 10 m transects were surveyed/site, and in 2008–2009, surveys were carried out using photographs taken at 2 m intervals. Colony size was assessed with two 10 m transects/site in 2008–2009 only.

A site comparison study in 2010–2011 at five coral reef sites in Moreton Bay, eastern Australia (14) found that in a protected area that prohibited all types of fishing and collection, density of coral recruits was higher than in unprotected areas for sites close to mangroves, but at sites more distant from mangroves, protected and unprotected areas had similar densities. Close to mangroves, density of coral recruits on settlement tiles was higher at sites in protected areas (6 colonies/tile) than at sites in unprotected areas (1 colony/tile), but further away from mangroves densities were similar at sites in protected (3 colonies/tile) and unprotected areas (1 colony/tile). Benthic communities on settlement tiles had higher coral cover in protected sites close to mangroves (11% cover) compared to protected sites more distant from mangroves and unprotected sites close to, and distant from, mangroves (1%). Two sites in a protected area where fishing and collection was prohibited were selected: one close to mangroves (<250 m away) and one distant from mangroves (>500 m away). In addition, at each of four unprotected locations, a site close to mangroves and a site distant from

mangroves was selected. Twelve settlement tiles were deployed at each site, and half of the tiles were covered with a cage (results from caging experiment not reported here). In September–December 2010, benthic communities were recorded on the tiles using photographs, and tiles were removed in January 2011 to assess stony coral recruits.

A review of 37 studies of coral reefs in the Philippines (15) found that in protected areas that prohibited all fishing and collection, annual change in coral cover was similar compared to partially protected areas, and change was higher in all protected areas compared to unprotected areas. Annual change in cover was not significantly different in fully protected areas (3% increase/year) and partially protected areas (6% increase/year). Across all protected areas, coral cover was similar in protected (19–48%) and unprotected areas (18–48%), but on average, annual change in cover increased in protected areas (3% increase/year) and stayed the same in unprotected areas (0% change/year). In addition, protected areas that were older and larger had annual increases in coral cover (≥6 years: 2–3% increase/year; >10 ha: 2–4% increase/year), while there was no significant annual change in younger and smaller (≤5 years: 6%; ≤10 ha: 3%) protected areas. Peer reviewed publications and grey literature was searched online and through personal communications, and studies identified that recorded hard coral cover had surveys from two or more years, and reported the number and length of transects. Protected areas were classified as fully protected (no extractive activities) or partially protected (activities and fishing gear regulated). Data from 1,096 surveys from 317 sites (155 protected, 162 unprotected) from 36 studies and one monitoring program were retained. Most sites (83% of 317) were surveyed with 50 m transects at depths of 2–20 m.

A replicated, site comparison study in 1987–2010 at eight coral reef sites along the coast of Kenya (16) found that in protected areas that prohibited all types of fishing and collection, hard and soft coral cover was not higher compared to unprotected areas and did not increase with time since protection. For hard and soft corals, cover was not higher in protected areas (hard: 9–50%, soft: 0–7%) compared to fished areas (hard: 9–22%, soft: 1–9%). For hard corals there was no trend with time since protection. For soft corals, cover was 0–7% after 35–40 years of closure and 1–9% when open to fishing, with authors suggesting some evidence for a decline with time since closure. Five areas were selected that excluded all

fishing and collection: four were protected areas and one was a community enforced closure. Three areas open to fishing were also selected. The five sites closed to fishing had closure dates of 1968, 1972, 1973, 1991 and 2005. Areas were surveyed 18 times at 1–3-year intervals in 1987–2010, with 1–4 sampling sites selected/area, and 9–12 transects (10 m long) surveyed at each site. Coral cover was compared across different sites in terms of the numbers of years since closure (with zero years = open to fishing).

A replicated, paired, site comparison study in 1998–2011 at six coral reef sites in the Florida Keys National Marine Sanctuary, USA (17) found that in protected areas that prohibited all types of fishing and collection, coral cover declined, and changes were no different compared to unprotected areas. Changes in hard coral cover declined similarly in protected (1998: 6–7%, 2011: 3%) and unprotected areas (1998: 3–5%, 2011: 2–3%). Cover of species of *Orbicella annularis, Siderastrea* spp., *Millepora* spp., *Porites astreoides, Montastraea cavernosa* and gorgonians were similar in protected and unprotected areas (data reported as statistical model results). Change in relative cover of *Agaricia* spp. over time was greater in protected (3–10%) than unprotected areas ((1–3%). In 1998–2011, changes in coral cover were assessed in three no-take sites (established in 1997) and three fished sites. Fished sites were adjacent to, and comparable in size to no-take sites. At each site, a rectangular plot (25 × 80 m) was established at two depths (7–9 m and 15–18 m, total of 12 plots), and 10–12 transects (25 m long) were surveyed in each plot, once each summer from 1998–2011 (excluding 2006 and 2009). Surveys were conducted by a diver with a video camera.

A replicated, site comparison study in 2011 in eight coral reef sites in the Florida Keys, USA (18) found that protected areas where all fishing and collection was prohibited had higher coral recruitment than fished areas at two of three water depths. Density of coral recruits was higher in protected areas (0.4–0.6 recruits/tile) than in fished areas at 2–5 m (0.05 recruits/tile) and 7–10 m (0.1 recruits/tile), but similar at 14–17 m (0.6 recruits/tile). In 2011, eight sites were selected, four on unfished reefs (no-take zones established in 1997) and four on fished reefs. At each site, 30 terracotta tiles (10 × 10 cm) were deployed, with 10 placed at each of three depths: 2–5 m, 7–10 m, and 14–17 m (total of 240 tiles). Tiles were deployed in May 2011, retrieved in September 2011 (133–141 days underwater), and coral recruits were counted.

A replicated, site comparison study in 2013 at six coral reef sites off Koh Tao, Thailand (19) found that protected areas that prohibit all types of fishing and collection had similar hard coral cover, disease prevalence and coral health compared to unprotected areas one year after protection was established. Hard coral cover was similar in protected (47–75%) and unprotected areas (43–89%), and cover by different coral families was also similar in protected vs unprotected areas (Acroporidae: 7 vs 18%, Poritidae: 19 vs 18%, Agariciidae: 15 vs 22%, others: 16 vs 26%). Overall disease prevalence was also similar in protected and unprotected areas (0.9 vs 0.5%), as was prevalence of nine other indicators of coral health (26 vs 14 % of corals showing any indicator). A protected area prohibiting all fishing and collection was established in 2012. Surveys were conducted in early August to late September 2013 at six sites, three within the protected area, and three outside. Sites were also classified based on historic use of human activities (including diving and snorkelling, boat traffic, waste-water run-off and sedimentation) as high or low-use (protected sites: two high, one low-use; unprotected sites: one high, two low-use). Each site was surveyed via three belt transects (15 × 2 m) to estimate coral cover, disease, and health.

A replicated, site comparison study in 2012 at 41 coral reef sites in the Whitsunday Islands, Australia (20) found that protected areas that prohibited all fishing and collection had similar coral cover as sites open to some fishing and collection, but lower disease prevalence and fewer signs of disturbance and damage. Coral cover was similar in sites that prohibited all fishing (25%) compared to those that allowed some fishing or collection activity (23–25%). Prevalence of coral disease was four times lower at sites that prohibited all fishing (1% of 272 colonies with disease) compared to those with some fishing (4% of 848 colonies with disease). Sites that prohibited all fishing and collection also had better coral health than fished areas in terms of coral damage (protected: 1%, fished: 4%) and bleaching (protected: 0.6%, fished 1%). In October and November 2012, surveys were conducted at 21 sites where all fishing and collection was prohibited (63 transects) and 20 sites where some fishing and collection was allowed (60 transects). Eleven fished sites allowed hook and line fishing, spear fishing, and collecting, and nine fished sites limited the amount of hook and line fishing and prohibited spear fishing and collecting. Surveys were conducted along 15 × 2 m belt transects.

A replicated, paired, site comparison study 1983–2013 at eight coral reef sites in central Philippines (21) found that in protected areas that prohibited all types of fishing and collection, hard coral cover did not consistently increase with time since protection, and three of four protected areas had higher coral cover than unprotected areas. Coral cover varied over time in both protected (48–2% over 20 years, 30–37% over 20 years, 10–23% over 10 years, 27–37% over 8 years) and unprotected areas (2–59% over 20 years, 19–20% over 20 years, 11–18% over 10 years, 29–10% over 8 years). When averaged across the whole sampling period, three of four protected areas had higher cover than unprotected areas (50 vs 33%, 32 vs 17%, 37 vs 20%) with the fourth having similar cover (15 vs 14%). In addition, there were differences in benthic habitat composition (including coral cover) between protected and unprotected areas (data reported as statistical model results). Four locally manged no-take marine reserves were selected and paired with similar unprotected areas that were fished. All sites were heavily fished before protection was established and compliance after protection was high. Six transects (50 × 20 m) were surveyed at each site in November or December, and the number of years sampled at each site varied from 4–25. At three sites, surveys began in the year before protection was established.

A replicated, site comparison study in 2009–2013 in 16 coral reef sites along the Belize Barrier Reef, Belize (22) found that protected areas that prohibited all types of fishing and collection had similar coral cover as areas limiting some fishing activities and unprotected areas. Coral cover was similar in fully protected areas (20%), areas with some fishing restrictions (18%) and unprotected areas (21%). In addition, cover was similar across different enforcement levels (good: 21%, moderate: 15%, inadequate: 19%, absent: 20%) and did not change due to time since protection started (see paper for details). Sixteen sites were selected (15−18 m depth): four that were fully protected (only non-extractive activities allowed), four with some restrictions (limited fishing licenses and banned use of traps, nets and longlines), and four with no protection (although fishing of herbivorous fish and Nassau groupers *Epinephelus striatus* was restricted at all sites). Each site was monitored in May and June in 2009, 2010, 2012 and 2013 along six 10 m transects, spaced around 10 m apart. Coral cover was recorded, and corals were identified to species level.

A replicated, site comparison study in 2014 at nine coral reef sites in Sanya Bay, Hainan, China (23) found that protected areas that prohibited all types of fishing and collection did not have higher coral cover than areas that were not protected. Coral cover in protected areas (10%, both privately managed and non-privately managed areas) was similar to cover in unprotected, privately managed areas (8%), but was lower than cover in an unprotected, not privately managed area (36%). Coral growth was similar in protected and unprotected areas (no data reported). In 2014, nine sites were selected that varied in protection status (protected vs unprotected) and management (privately managed or not). Protected areas were established in 1990, and privately managed areas were managed by three different companies for tourism (including diving, snorkelling and other water sports). Fishing restrictions were well enforced in privately managed sites, but enforcement was lacking in protected areas without private management (see paper for details). At each site, three 50 m transects were surveyed in 2014 at each of two depths (2–3 m and 6–8 m), with photographs taken 25 times along each transect using evenly spaced quadrats (50 × 50 cm).

A replicated, before-and-after, site comparison study in 2000–2011 at eight coral reef sites on the Danajon bank reef system off Bohol, Philippines (24) found that protected areas that prohibited all types of fishing and collection had similar coral cover compared to unprotected areas. Coral cover was similar inside and outside protected areas 7–16 years after protection was established (inside: 8–58%, outside: 10–47%), and from two years before to two years after protection was established (inside: 12–44%, outside: 8–52%). Change in coral cover over time did not differ between protected and unprotected areas at seven of eight sites, but for one site cover increased inside the protected area (from 39% to 58% cover over 11 years) and decreased outside (from 48% to 25% cover). In addition, overall composition of community traits was similar for protected and unprotected areas, but at each site there were small differences between the protected and unprotected areas (data reported as statistical model result). Surveys were carried out inside and outside eight community-led, "well enforced" Marine Protected Areas (10–50 ha). Unprotected areas were subject to fishing pressure from multiple gear types, including blast fishing. From 2000–2011, coral cover was surveyed each year via 2–3 transects inside and outside protected areas in the wet and dry season.

A replicated, site comparison study in 1991–2018 at 12 coral reef sites off the coast of Kenya (25) found that protected areas that prohibited all types of fishing and collection had more coral species, lower diversity between sites, and similar turnover of species over time compared to fished reefs. Protected areas had a higher number of coral species (15 species/site) than fished reefs (10 species/site) and overall, diversity of species between sites was lower for protected areas than for fished reefs (reported as diversity index). Protected areas contained relatively more *Acropora, Echinopora, Montipora* and massive *Porites* corals, whereas fished reefs had more branching *Porites, Stylophora*, and *Pavona* corals. In addition, turnover of species groups over time was similar in protected areas and fished reefs (see paper for details). Five sites (30 × 30 m) were established in protected areas with permanent fishing and collection closures, and seven were established in reefs with high fishing intensity. All sites were shallow back-reef lagoons. In December–March 1991–2018, all 12 sites were sampled 19 times (nine 10 m transects/site). Corals >3 cm were identified to species group or species or classed as branching or massive (for *Porites* corals).

A replicated, site comparison study over six years [years unknown] at 56 sites along the Great Southern Reef off Australia (26) found that in protected areas that prohibited all types of fishing and collection, diversity and cover of sessile invertebrates (reported as "sponges, soft corals, ascidians, etc.") was similar compared to unprotected areas. See original paper for data. In addition, partially protected areas that restricted only some activities had lower diversity and cover of sessile invertebrates than unprotected areas. Sites either fully restricted all fishing and collecting (19 sites in 10 areas), restricted some types of fishing (18 sites in 11 areas) or were outside of a protected area (19 sites). Using data from an online database, a total of 1,971 photo quadrats (46% from fully protected, 33% from partially protected, 21% from unprotected) taken along 50 m transects were used to quantify diversity and cover of sessile invertebrates using the Collaborative and Annotation Tools for Analysis of Marine Imagery.

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(16) McClanahan T.R. (2014) Recovery of functional groups and trophic relationships in tropical fisheries closures. *Marine Ecology Progress Series*, 497, 13–23. https://doi.org/10.3354/meps10605

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(25) McClanahan T.R. (2020) Decadal turnover of thermally stressed coral taxa support a risk-spreading approach to marine reserve design. *Coral Reefs*, 39, 1549–1563. https://doi.org/10.1007/s00338-020-01984-w

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## 11.3 Designate a Marine Protected Area and prohibit all types of fishing

https://www.conservationevidence.com/actions/4000

- **Six studies** examined the effects on corals of designating a Marine Protected Area and prohibiting all types of fishing. Two studies were in Kenya[5,6], and one was in each of the Bahamas and Turks and Caicos Islands[1], Belize[3], Guadeloupe[2] and Spain[4]

**COMMUNITY RESPONSE (1 STUDY)**

- **Richness/diversity (1 study)**: One replicated, site comparison in the Bahamas and Turks and Caicos Islands[1] found differences in community composition between protected and unprotected areas.

**POPULATION RESPONSE (6 STUDIES)**

- **Abundance (4 studies)**: Two of three replicated, site comparison studies in the Bahamas and Turks and Caicos Islands[1], Guadeloupe[2] and Kenya[6] found that coral cover was similar in protected and unprotected areas[1,6]. The other study[2] found that cover was higher in protected than in unprotected areas. One of the studies[1] also found that three coral species were found more frequently in protected than in unprotected areas. One replicated, site comparison in Kenya[5]

found that density of coral recruits was similar in protected and unprotected areas. This study also found that caging settlement tiles resulted in fewer coral recruits in protected areas (with fish grazers) and more recruits in unprotected areas (with urchin grazers).

- **Survival (1 study)**: One replicated, controlled study in Belize[3] found that coral mortality was higher in protected than in unprotected areas for one of two transplanted coral species.
- **Condition (2 studies)**: One replicated, controlled study in Belize[3] found lower growth rates of two transplanted coral species and higher rates of bleaching for one of the two species in protected areas compared to unprotected areas. One site comparison in Spain[4] found that fewer corals had been colonized by other organisms in protected than in unprotected areas.

**Background**

Fishing can have both direct and indirect impacts on corals. Prohibiting some or all of these activities within Marine Protected Areas is a widely used conservation action (Kriegl *et al*. 2021).

Fishing poses a direct threat when gear damages corals during operations (Mangi & Roberts 2006, Althaus *et al*. 2009) or is lost or abandoned (so called 'ghost' gear; Ballesteros *et al*. 2018, Figueroa-Pico *et al*. 2020). Indirect threats can emerge when overfishing reduces the diversity and abundance of herbivorous fishes, with consequences for macroalgal abundance and potential impact on coral growth (Burkepile *et al*. 2010).

Prohibiting all fishing has the potential to reduce these threats and allow corals and the reefs they form to recover and flourish.

Restricted fishing types covered here include hook and line fishing; bottom trawling, dredging and other towed gear; static gear including traps; and dynamite and cyanide fishing.

Studies were included here if they mentioned restrictions on fishing but made no mention of restrictions on collection or other extractive activities.

Althaus F., Williams A., Schlacher T.A., Kloser R.J., Green M.A., Barker B.A., Bax N.J., Brodie P. & Schlacher-Hoenlinger M.A. (2009) Impacts of bottom trawling on deep-coral ecosystems of seamounts are long-lasting. *Marine Ecology Progress Series*, 397, 279–294. https://doi.org/10.3354/meps08248

Ballesteros L.V., Matthews J.L. & Hoeksema, B.W. (2018). Pollution and coral damage caused by derelict fishing gear on coral reefs around Koh Tao, Gulf of Thailand. *Marine Pollution Bulletin*, 135, 1107–1116. https://doi.org/10.1016/j.marpolbul.2018.08.033

Burkepile D.E. & Hay M.E. (2010) Impact of herbivore identity on algal succession and coral growth on a Caribbean reef. *PloS One*, 5, e8963. https://doi.org/10.1371/journal.pone.0008963

Figueroa-Pico J., Tortosa F.S. & Carpio A.J. (2020) Coral fracture by derelict fishing gear affects the sustainability of the marginal reefs of Ecuador. *Coral Reefs*, 39, 819–827. https://doi.org/10.1007/s00338-020-01926-6

Kriegl M., Elías Ilosvay X.E., von Dorrien C. & Oesterwind D. (2021) Marine protected areas: at the crossroads of nature conservation and fisheries management. *Frontiers in Marine Science*, 8, Article 676264. https://doi.org/10.3389/fmars.2021.676264

Mangi S.C. & Roberts C.M. (2006) Quantifying the environmental impacts of artisanal fishing gear on Kenya's coral reef ecosystems. *Marine Pollution Bulletin*, 52, 1646–1660. https://doi.org/10.1016/j.marpolbul.2006.06.006

A replicated, site comparison study in 2004 in marine sites within Exuma Cays Land and Sea Park in the Bahamas and Turks and Caicos Islands (1) found that in protected areas that prohibited all fishing, there was similar coral cover but higher occurrence of three coral species compared to sites where fishing was allowed. Three species of corals had higher frequency of occurrences within the Exuma Cays reserve compared to outside the reserve (*Montastraea franksi* inside reserve 50%; outside: 13%; *Agaricia agaricites*: 74% and 40%; *Millepora alcicornis*: 47% and 38%). In protected *Montastraea* reefs the benthic community structure was different compared to similar reefs outside the reserve while fish diversity and abundance of some large-bodied fish were higher in the reserve (see paper for details and data for fish). The Park is a large

(442 km$^2$) reserve established in 1958, with fishing bans enforced by warden patrols since 1986. Overall, 21 protected reef sites (150 × 150 m in the centre, on the edge of the reserve and separated between forereef and plain) and five non-protected reef sites near other islands were selected by matching to reduce variability. In 2004, at each site 30–40 randomly placed quadrats (1 m$^2$) were used to quantify the benthic community with coral and macroalgal cover in each quadrat recorded as the average of five randomly sub-sampled areas of 20 × 20 cm.

A replicated, site comparison study in 2004 in 10 coral reef sites off Guadeloupe in the eastern Caribbean Sea (2) found that in protected areas that prohibited all fishing, coral cover was higher than in unprotected areas. Coral cover was higher in the protected (26%) than unprotected areas (18%). Five reef sites in protected areas were selected (fishing prohibited since 1979 and 1987), along with five unprotected reef sites. In 2004, every site was sampled in the dry and rainy season (May and November respectively). Visual surveys along a 150 × 2 m transect were carried out by divers, with each transect surveyed twice/season. Benthic organisms (including corals) were recorded every meter along the transect.

A replicated, controlled study in 2003–2004 of 16 patch reefs in Belize (3) found that in protected areas that prohibited all fishing there was lower growth rates for transplanted corals *Siderastrea siderea* and *Porites astreoides*, and higher rates of mortality and bleaching for *S. siderea*, compared to areas in which fishing was not prohibited. Eighteen months after transplanting, average growth rates were lower in areas where fishing was prohibited compared to areas with fishing for *S. siderea* (13 vs 28%) and *P. astreoides* (2 vs 24%). Average bleaching and mortality rates were higher in unfished than fished areas for *S. siderea* (bleaching: 13 vs 6%; mortality: 15 vs 10%), but there was no significant effect on *P. astreoides* (data not reported). In January 2003, six 'fist-sized' *S. siderea* and *P. astreoides* were transplanted onto each of 16 patch reefs (each 25–50 m$^2$) in a marine reserve. Half were in an area in which fishing had been prohibited for eight years, and the other half in an area still fished. Corals were collected from 1–3 km away and attached using masonry cement. Bleached corals were counted monthly, and surviving corals measured every three months, until August 2004.

A site comparison study in 2010–2011 at nine coral reef sites in Cap de Creus and Medes Islands, off Spain in the northern Mediterranean

(4) found that in a protected area that prohibited all fishing and diving, fewer coral *Paramuricea clavata* colonies had other organisms growing on them (likely due to injury/damage) than in areas where fishing and/or diving was permitted. In the protected area, 4–10% of colonies had other organisms growing on them, compared to 10–33% in unprotected areas. Colonies with organisms growing on them had fewer reproductive cells (5–13 gonads/coral polyp) than those without (10–25 gonads/coral polyp) and differences in concentrations of lipids, carbohydrates and proteins in coral branches (see paper for details). One area of a marine park (established in 1996) where both fishing and diving was prohibited was selected, along with six other sites in the same area (with a mix of recreational fishing and diving) and two sites in different area (with some diving permitted but no fishing). In June 2010 and January 2011, a total of 15 surveys across the nine locations were carried out (4 in the fully protected area) by divers along transects (6–20 m long, 16–38 m deep).

A replicated, site comparison study in 2007–2008 in six coral reef sites off the coast of Kenya (5) found that in protected areas that prohibited all types of fishing, the number of coral recruits was similar compared to the number of recruits in fished areas. Overall, average density of coral recruits (live and covered) was not significantly different between protected areas (32 recruits/$m^2$) and fished areas (149 recruits/$m^2$). Authors also reported that caging settlement tiles to exclude grazers after six months had different effects on the number of live recruits in protected areas with fish grazers (caged lower with 70 recruits/$m^2$, uncaged: 140 recruits/$m^2$) and fished areas with urchin grazers (caged higher with 750 recruits/$m^2$, uncaged: 450 recruits/$m^2$). Three well-enforced protected areas were selected where all fishing was prohibited for >15 years, along with three nearby fished reefs. Settlement tiles were deployed in cement blocks at all six sites (4 tiles/block, 16 blocks/site). Two tiles on each block were caged to exclude grazers and two were left uncaged for six months, after which time half of the treatments were switched for a further six months. All coral recruits were counted, including those that were alive and those covered by other organisms.

A replicated, site comparison study in 2011 at six sites off the coast of Kenya (6) found that protected areas that prohibited all types of fishing had similar coral cover compared to community managed fishery closures and fished areas. Coral cover did not vary based on

management type and was 20 and 27% in protected areas, 26 and 46% in community closures, and 7 and 35% in fished areas. Two government closures were protected since 1968 and 1991. Two community managed areas were closed to fishing in 2005 and 2010. Two fished areas were fished intensively with a range of gear (including spearguns, nets, traps). In 2011, coral cover was surveyed using randomly placed 10 m transects (9 transects/site).

(1) Harborne A.R., Mumby P.J., Kappel C.V., Dahlgren C.P., Micheli F., Holmes K.E., Sanchirico J.N., Broad K., Elliott I.A. & Brumbaugh D.R. (2008) Reserve effects and natural variation in coral reef communities. *Journal of Applied Ecology*, 45, 1010–1018. https://doi.org/10.1111/j.1365-2664.2008.01490.x

(2) Kopp D., Bouchon-Navaro Y., Louis M., Mouillot D., & Bouchon C. (2010) Herbivorous fishes and the potential of Caribbean marine reserves to preserve coral reef ecosystems. *Aquatic Conservation – Marine and Freshwater Ecosystems*, 20, 516–524. https://doi.org/10.1002/aqc.1118

(3) McClanahan T.R., Huntington B.E., Cokos B. (2011) Coral responses to macroalgal reduction and fisheries closure on Caribbean patch reefs. *Marine Ecology Progress Series*, 437, 89–102. https://doi.org/10.3354/meps09285

(4) Tsounis G., Martinez L., Bramanti L., Viladrich N., Gili J.M., Martinez Á. & Rossi S. (2012) Anthropogenic effects on reproductive effort and allocation of energy reserves in the Mediterranean octocoral *Paramuricea clavata*. *Marine Ecology Progress Series*, 449, 161–172. https://doi.org/10.3354/meps09521

(5) O'Leary J.K., Potts D., Schoenrock K.M. & McClahanan T.R. (2013) Fish and sea urchin grazing opens settlement space equally but urchins reduce survival of coral recruits. *Marine Ecology Progress Series*, 493, 165–177. https://doi.org/10.3354/meps10510

(6) Humphries A.T., McClanahan T.R. & McQuaid C.D. (2014) Differential impacts of coral reef herbivores on algal succession in Kenya. *Marine Ecology Progress Series*, 504, 119–132. https://doi.org/10.3354/meps10744

## 11.4 Designate a Marine Protected Area and prohibit all types of collection

https://www.conservationevidence.com/actions/4069

- We found no studies that evaluated the effects on corals of designating a Marine Protected Area and prohibiting all types of collection.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Corals and the reefs they form are threatened by global trade in ornamental species, where they, and a range of other species, are targeted for use in aquaria and for jewellery and curios (Dee *et al*. 2014). In addition to the damage caused to corals during collection of other species, direct collection of corals can lead to higher mortality and reduced local populations (Knittweis & Wolff 2010).

Prohibiting the collection of corals and other reef wildlife has the potential to reduce these threats and allow corals and the reefs they form to recover and flourish.

Dee L.E., Horii S.S. & Thornhill D.J. (2014) Conservation and management of ornamental coral reef wildlife: Successes, shortcomings, and future directions. *Biological Conservation*, 169, 225–237. https://doi.org/10.1016/j.biocon.2013.11.025

Knittweis L. & Wolff M. (2010) Live coral trade impacts on the mushroom coral *Heliofungia actiniformis* in Indonesia: Potential future management approaches. *Biological Conservation*, 143, 2722–2729. https://doi.org/10.1016/j.biocon.2010.07.019

## 11.5 Designate a Marine Protected Area and prohibit some fishing and collection (including where restrictions are unspecified)

https://www.conservationevidence.com/actions/4006

- **Twenty-one studies** examined the effects on corals of designating a Marine Protected Area and prohibiting some fishing and collection (including where restrictions are unspecified). Three studies were in Australia[7,16,21], two were in

each of Italy[14,18], the UK[17,20], and the Caribbean[8,10], and one was in each of Belize[19], Colombia[4], Egypt[15], Grenada[12], Kenya and Tanzania[5], Mexico[1], Papua New Guinea[2], the Philippines[13], Solomon Islands[11], US Virgin Islands[6], Western Indian Ocean[9] and Vietnam[4].

**COMMUNITY RESPONSE (7 STUDIES)**

- **Richness/diversity (7 studies)**: One of three studies (including two replicated, site comparison studies) in Mexico[1], Vietnam[4], and Kenya and Tanzania[5] found more coral species groups in protected areas than areas without formal management[5], and another[1] that species richness was similar in protected and unprotected areas[1]. The third study[4] found a small decrease in species richness over three years since protection was designated, and that community composition differed after three years. A replicated, site comparison study in Italy[18] found that overall community composition was similar in protected and unprotected areas, though communities varied differently at different spatial scales. One of two replicated, site comparison studies (including one paired study) in Papua New Guinea[2] and Australia[21] found that coral diversity was similar in protected and unprotected areas[2]. The other study[21] found that diversity of sessile invertebrates (including soft corals) was lower in protected than in unprotected areas. One replicated, site comparison study in Egypt[15] found that a biodiversity index was higher in protected than in unprotected areas in two of six comparisons, but similar in other comparisons.

**POPULATION RESPONSE (18 STUDIES)**

- **Abundance/Cover (18 studies)**: Seven of 14 studies (including three replicated, paired, site comparison studies) in Australia[7,16,21], Belize[19], Colombia[3], Grenada[12], Kenya and Tanzania[5], Papua New Guinea[2], Solomon Islands[11], the UK[17], US Virgin Islands[6] and reviews in the western Indian Ocean[9], the Phippines[13] and Caribbean[10] found mixed results of protection on coral cover[6,7,9,10,11,12,17], including mixed results for hard versus

soft corals[6,7,12] or for lagoon versus barrier reef sites[11]. One study[5] found that hard coral cover was higher in protected than unprotected areas. Five studies found that coral cover was similar in protected and unprotected areas[2,3,13,16,19]. Two studies[5,21] found that cover of soft coral[5], and other sessile invertebrates (including soft corals)[21] was lower in protected than in unprotected areas. Four of five studies (including two replicated, site comparison studies) in Australia[7], Egypt[15], the UK[20], US Virgin Islands[6] and a review in the Philippines[13] found mixed results on trends in coral cover[6,7], or that one species increased while one or multiple other species stayed the same[15,20]. The other study[13] found that coral cover increased over time in protected areas but not in unprotected areas, and increased in older, larger protected areas but not in younger, smaller ones. One of two studies (one replicated, site comparison and one before-and-after study) in Italy[14] and Mexico[1] found that density of juvenile corals was similar in protected and unprotected areas[1]. The other study[14] found that densities were lower in a protected area compared to nine years before protection, but higher compared to 40 years before protection. One of two studies (one review and one before-and-after study) found mixed results on biomass in protected and unprotected areas[10]. The other study[14] found higher biomass in a protected area compared to nine years before protection.

- **Condition (5 studies)**: Two of three studies (including one replicated, before-and-after and one replicated, site comparison study) in Colombia[3], Italy[14] and the UK[20] found that corals were larger in protected than in unprotected areas depending on the species[20] or time period[14] for which comparisons were made. The other study[3] found that coral population size structure was similar in protected and unprotected areas. One replicated, site comparison study in Australia[16] found that coral disease and damage was higher in areas with more restrictions. One review in the Caribbean[8] found that estimates of negative change in structural complexity were similar in protected and unprotected areas, though the trend was significant in protected areas, but not in unprotected areas.

**Background**

Fishing and collecting of corals and other organisms can have both direct and indirect impacts on corals. Prohibiting some or all of these activities within Marine Protected Areas is a widely used conservation action (Kriegl *et al*. 2021).

Fishing and collecting poses a direct threat when gear damages corals during operations (Mangi & Roberts 2006, Althaus *et al*. 2009) or is lost or abandoned (so called 'ghost' gear; Ballesteros *et al*. 2018, Figueroa-Pico *et al*. 2020). Indirect threats can emerge when overfishing reduces the diversity and abundance of herbivorous fishes, with consequences for macroalgal abundance and its potential to impact on coral growth (Burkepile *et al*. 2010).

Prohibiting all fishing and collecting (so called no-take areas/zones) has the potential to reduce these threats and allow corals and the reefs they form to recover and flourish.

Restricted fishing types covered here include hook and line fishing; bottom trawling, dredging and other towed gear; static gear including traps; and dynamite and cyanide fishing.

Studies that do not specify the specific types of activities that were prohibited in protected areas are also included here. For studies that restrict specific fishing and collection activities see: *Designate a Marine Protected Area and prohibit all types of fishing, collecting and access; Designate a Marine Protected Area and prohibit all types of fishing and collection; Designate a Marine Protected Area and prohibit all types of fishing* and *Designate a Marine Protected Area and prohibit all types of collection*.

Althaus F., Williams A., Schlacher T.A., Kloser R.J., Green M.A., Barker B.A., Bax N.J., Brodie P. & Schlacher-Hoenlinger M.A. (2009) Impacts of bottom trawling on deep-coral ecosystems of seamounts are long-lasting. *Marine Ecology Progress Series*, 397, 279–294. https://doi.org/10.3354/meps08248

Ballesteros L.V., Matthews J.L. & Hoeksema, B.W. (2018). Pollution and coral damage caused by derelict fishing gear on coral reefs around Koh Tao, Gulf of Thailand. *Marine Pollution Bulletin*, 135, 1107–1116. https://doi.org/10.1016/j.marpolbul.2018.08.033

Burkepile D.E. & Hay M.E. (2010) Impact of herbivore identity on algal succession and coral growth on a Caribbean reef. *PloS One*, 5, e8963. https://doi.org/10.1371/journal.pone.0008963

Figueroa-Pico J., Tortosa F.S. & Carpio A.J. (2020) Coral fracture by derelict fishing gear affects the sustainability of the marginal reefs of Ecuador. *Coral Reefs*, 39, 819–827. https://doi.org/10.1007/s00338-020-01926-6

Kriegl M., Elías Ilosvay X.E., von Dorrien C. & Oesterwind D. (2021) Marine protected areas: at the crossroads of nature conservation and fisheries management. *Frontiers in Marine Science*, 8, Article 676264. https://doi.org/10.3389/fmars.2021.676264

Mangi S.C. & Roberts C.M. (2006) Quantifying the environmental impacts of artisanal fishing gear on Kenya's coral reef ecosystems. *Marine Pollution Bulletin*, 52, 1646–1660. https://doi.org/10.1016/j.marpolbul.2006.06.006

A replicated, site comparison study in 1999 in 16 marine sites across eight reef areas in two regions of Yucatan, Mexico (1) found that protected areas that prohibited some fishing and collection had similar diversity and density of juvenile stony coral cover compared to areas where fishing was unrestricted. Coral diversity was similar in the protected areas (17 coral species, with 7–13 species/reef) compared to unprotected areas (16 species, 7–13 species/site) and was similarly dominated by *Agaricia* spp., *Siderastrea* spp., and *Porites* spp. (40–100% abundance at the site level) but with differences between specific sites (see paper for details). Juvenile coral density was similar in protected reefs (3 juveniles/$m^2$) and the reefs in unprotected areas (3 juveniles/$m^2$) but with sometimes large differences between specific sites (see paper for details). The Sian Ka'an Biosphere Reserve has been protected since 1986, with a closed season fishing and additional fishing restrictions. The southern part was unprotected with unrestricted fishing activity. Within each reef, three sites (1 km apart) were selected north to south. In 1999, at each site, 10–18 transects (10 m long) were positioned 1–5 m apart 8–13 m deep. Five, 25 × 25 cm quadrats were placed every 2 m along each transect. Every quadrat (total 1,747 quadrats on 360 transects) was surveyed by a diver counting juvenile stony corals (under 2 cm diameter).

A replicated, paired, site comparison study in 2002 in six marine reef areas at Ahus and Onetta islands, Papua New Guinea (2), found that protected areas that prohibited some fishing and collection had similar diversity and live coral cover compared to sites where fishing was unrestricted. Coral diversity and live coral cover were similar in protected, traditionally managed community (tambu) sites compared to unprotected sites (data as model outputs). Protected tambu sites had 60% higher fish biomass (205 kg/ha) compared to unprotected sites (127 kg/ha) and fewer fishing gear discards (data as models). Three traditionally managed (tambu) sites were compared to three sites of similar reef profile, current regimes, and wave exposure, which had no protective management or fishing restrictions. At each site, 10 m long transects (18/site) were positioned to cover the same aspect areas of reefs at 6 –8 m depth, and hard corals were identified to genus. Management and effectiveness were assessed via interviews and the recording of discarded fishing gear in transects. With 2–3 exceptions/year to fish for ceremonial food at tambu sites, spear and net fishing were prohibited and harvesting of invertebrates was severely limited, but line fishing was unregulated. Tambu areas (six sites; total 33ha) represented 6% of the available fishing area.

A site comparison study in 2006 at 24 coral reef sites in National Natural Park Rosario and San Bernardo Corals, Colombia (3) found that sites in a protected area that prohibited some fishing and collection had similar coral cover and densities of coral species compared to sites outside the protected area. Coral cover was similar for sites in the protected area (hard corals: 27%, gorgonians: 2%) and unprotected sites (hard corals: 21%, gorgonians: 3%). Counts of four major coral species tended to be higher in the protected area than outside (result not tested for statistical significance: *Acropora cervicornis*: 56 inside vs 54 outside; *Acropora palmata*: 118 vs 32; *Diploria labyrinthiformis*: 109 vs 102; *Siderastrea siderea*: 427 vs 156), but densities and size structure of populations were similar inside and outside the protected area (data reported as statistical model results). Sixteen sites within a protected area and eight sites outside of the protected area were selected. In May 2006, sixteen sites were surveyed (8 protected, 8 unprotected), and in September 2006 an additional eight protected sites were surveyed. Surveys consisted of two transects/site (30 × 2 m), with sixty 1 $m^2$ photographs taken along each transect.

A study in 2002–2005 in four sites within one protected area in Nha Trang Bay, Vietnam (4) found that in a protected area that prohibited some fishing and collection, there was a slight decrease in the number of coral species found three years after designation compared to in the year of designation. Three years after designation, 256 species were detected (92–177 species/site) compared to 274 (122–187 species/site) in the year of designation (result not tested for statistical significance). There were differences in coral community composition three years after designation compared to the year of designation (data reported as statistical model results). Authors reported increases in aquaculture and presence of fishing boats within the protected area between the two time periods (see paper for details). A Marine Protected Area was established in 2002, with surveys conducted in August 2002 and March–April 2005. Surveys were conducted at one location in the core of the protected area, and three locations within the buffer zone, with two surveys/site at different depths along a 250 $m^2$ area.

A replicated, site comparison study in 1996–2005 in four areas off the coast of Tanzania and Kenya (5) reported that sites in areas where some fishing and collection was prohibited had more coral species groups, mixed results for hard coral cover but lower soft coral cover than sites in areas with no formal management restrictions. Results were not tested for statistical significance. Two areas that restricted some fishing and collection both had 39 groups (genera) of coral species, 22–43% cover of hard corals and 3–8% of soft corals. A further area with some restrictions was reported as having an "intermediate" number of species groups, 32–73% cover of hard corals and 1–5% cover of soft corals. An area with no formal management had 23–25 species groups, 45–52% cover of hard corals and 4–17% of soft coral. When all areas with restrictions (including one area that prohibited all fishing and collections) and areas without restrictions were analysed together, no significant differences were found in the abundance of different coral species groups over time. Two to four sites were selected from each of three areas with restrictions, and two sites were selected from an area with no formal management. Each site was sampled using nine 10 m transects and all organisms 3 cm or larger were recorded.

A replicated, site comparison study in 2003–2008 at two coral reef sites off St John, US Virgin Islands (6) found that in protected areas

that prohibited some fishing and collection, hard coral cover was lower in the protected areas compared to outside, and soft coral cover was higher in one of two protected areas compared to outside but similar in the other. Hard coral cover was lower in the protected areas (4% and 4%) compared to the unprotected areas (15% and 8%), and soft coral cover was higher in one comparison (protected: 22%, unprotected: 13%) and similar in a second (protected: 12%, unprotected: 11%). In one location, coral cover declined in the protected area (2003: 7%, 2008: 3%), but declined more dramatically in the unprotected area (2003: 26%, 2008: 6%). In the second location, cover in the protected area was 4% in 2003 and 2% in 2008, and in the unprotected area cover was 10% in 2003 and 6% in 2008 (results were not tested for statistical significance). Two protected areas were selected, one on the mid-shelf reef and one in a bay. In addition to fishing restrictions (apart from for blue runner *Caranx crysos*), anchoring was also prohibited in the protected areas. Sites in the protected areas (18–30 sites/year) and in adjacent unprotected areas (15–25 sites/year) were surveyed annually from 2003–2008. Coral cover was assessed at one location/site within a 15 m diameter area.

A replicated, paired, site comparison study in 1992–2002 of 15 pairs of coral reef sites in the Great Barrier Reef, Australia (7) found that in protected areas that prohibited some fishing and collection (a range of different restrictions), hard coral cover was similar but soft coral cover lower than in unprotected areas. When pooling data across all reefs, hard coral cover was similar in protected (39%) and unprotected reefs (31%), but soft coral cover was lower (protected: 12%, unprotected: 19%). For undisturbed reefs, hard coral cover was higher in protected reefs (46%) than in unprotected (25%), and soft coral cover was lower (protected: 18%, unprotected: 36%), but there were no overall differences for disturbed reefs (hard: 34–36%, soft: 8–10%). Annual increases in hard coral cover were higher in protected than in unprotected areas in two of five comparisons at undisturbed reefs, but at disturbed reefs neither protected or unprotected areas consistently had higher increases in cover. For soft corals, there was very little change in cover in protected and unprotected areas over time on both undisturbed and disturbed reefs (see paper for details). Fifteen pairs of sites, one protected and one unprotected, were selected across 23 reefs. The protected sites included three levels of protection: no fishing, collecting or diving; no fishing or

collecting, but diving permitted; and, no trawling or collection, limited line fishing, and other fishing and diving permitted. Disturbances included cyclones, storms and crown-of-thorns *Acanthaster planci* invasions. Annual surveys in 1992–2002 were conducted at each site, with three survey locations selected per site, and five transects (50 m) surveyed/location.

A review of 27 studies from 1978–2008 of coral reefs in the Caribbean (8) found that in protected areas that prohibited some fishing and collection (a range of different restrictions), changes in reef structural complexity was similar compared to unprotected areas. Changes in reef structural complexity were similar in protected (2% decrease/year) and unprotected sites (3% decrease/year), though the decline in protected areas was significant whereas in unprotected areas it was not. Declines in complexity in protected areas were present in both areas impacted by hurricanes (2% decrease, 7 sites) and in unimpacted areas (3% decrease, 17 sites). Changes in complexity in protected areas were not linked to duration of protection (18% decrease to 4% increase, 3–47 years of protection). Relevant journals for Caribbean reef studies were searched, and researchers and reef managers were also contacted directly to obtain data. A total of 27 studies with data from 1978–2008 from 49 reef sites were obtained (2–17 years of data/study) that compared reef complexity inside and outside a protected area. Two thirds of the studies included protected areas classified in IUCN protection category II. Reef structural complexity was measured by comparing contoured length with straight-line length along a section of reef.

A review of studies of coral reefs in the western Indian Ocean (9) found that in protected areas that prohibited some fishing and collection, coral cover was higher in two time periods compared to fished reefs, lower in one and similar in a fourth. In the first two time periods (1977–1993 and 1994–1997), cover was higher on protected reefs with fishery closures (42–47%) than on fished reefs (16–33%). In the third period (1999–2000), immediately after a climatic disturbance in 1998, cover was lower on protected reefs (17%) than on fished reefs (24%), and by the final period (2001–2005) cover was similar in both reef types (protected: 26%, fished: 26%). Information was collated for the western Indian Ocean (region west of the 90°E meridian) for areas with fisheries closures (including reserves and no-take areas) and explicitly fished

reefs. Samples came from peer reviewed publications, grey literature, Reef Check surveys and regional monitoring programs. The number of samples for each period ranged from 23–115 (16, 52, 55, 77 samples for fishery closures, 7, 31, 28, 38 for fished reefs). Most field sampling was based on haphazard or permanent line intercept transects.

A review of studies from six protected areas in the Caribbean (10) found that in protected areas that prohibited some fishing and collection, responses of corals were mixed. Studies from six reserves found that for a range of measures (including density, biomass and percent cover), coral responses to protection ranged from positive to negative (data reported as log response ratios, result not tested for statistical significance). Four databases were searched for studies published 1970–2007, and references from those studies were also searched. Studies were included from Latin American and Caribbean countries where protected areas were established for at least three years, were at least moderately enforced, and included comparisons with fished areas or comparisons to before protection was established. A total of 32 studies from 23 reserves were found, though studies from only six reserves in the Caribbean contained data on coral.

A replicated, paired, site comparison study in 2005–2010 in six coral reef sites in a lagoon and three on a barrier reef off New Georgia Island, the Solomon Islands (11) found that in protected areas that prohibited some fishing and collection, coral cover in a lagoon was similar in the protected areas compared to adjacent fished areas, but lower than on a fished barrier reef. Live coral cover was similar in protected areas in a lagoon (26–27%) and adjacent fished locations (23–30%), but lower than in fished areas out on the barrier reef (70%). At the level of individual protected areas, the same trend was seen at two of three sites. At the third, coral cover was higher in 2005 in the protected than fished area (protected: 30%, fished: 19%) but lower in 2010 (protected: 39%, fished: 48%), but again, cover was highest on the fished barrier reef (65%). Three protected areas within a lagoon were selected, with protection in place for 6–10 years. Areas were managed by Resource Management Committees from local villages, which included chiefs and elders, church authorities, and women representatives. Three paired sites within the lagoon that were fished were selected, along with three fished sites on the barrier reef. Six transects (30 × 4 m) were sampled at each site, and coral cover was sampled with 1 $m^2$ quadrats. Lagoon sites were sampled in 2005 and 2010, and barrier reef sites in 2010 only.

A replicated, before-and-after, site comparison study in 2008–2012 at five coral reef sites off the southwest coast of Grenada (12) found that in protected areas that prohibited some fishing and collection, hard coral cover was similar and soft coral cover higher than outside protected areas and neither changed following protection. Hard coral cover did not change following protection (after: 10–15%, before: 12–19%), and was similar to cover in the unprotected areas (after: 9–13% and before: 12–16%). Soft coral cover did not change following protection (after: 3%, before: 5%), but was higher in protected areas than in unprotected areas before protection in two of two cases (protected: 5%, unprotected: 1–2%) and higher after protection in one comparison (protected: 3%, unprotected: 1%) but similar in a second (protected: 3%, unprotected: 2%). Authors also reported cover by different coral forms (massive, branching and encrusting, see paper for details). Fishing restrictions were implemented at two sites in 2010. An additional three sites with no restrictions were selected. Annual surveys were carried out at all sites in 2008–2012. Four 30 m parallel transects were surveyed at each of the five sites, with substrate type recorded at 50 cm intervals. This was combined with photograph surveys along the transects to give two assessments of coral cover.

A review of 37 studies of coral reefs in the Philippines (13) found that in protected areas that prohibited some fishing and collection, coral cover was similar and annual change in cover higher compared to unprotected areas. Coral cover was similar in protected (19–48%) and unprotected areas (18–48%), but the annual change in cover increased in protected areas (3% increase/year) and did not change in unprotected areas (0% change/year). Annual change in cover was similar in partially protected areas (6% increase/year) and fully protected areas (3% increase/year). In addition, there was some evidence that coral cover increased annually in older (≥6 years: 2–3%) and larger (>10 ha: 2–4%) protected areas, while there was not a significant annual change in younger (≤5 years: 6%) and smaller (≤10 ha: 3%) protected areas. Peer reviewed publications and grey literature was searched online and through personal communications, and studies were retained that recorded hard coral cover, had surveys from two or more years, and reported the number and length of transects surveyed. Protected areas were classified in terms of level of protection (partially protected: activities and fishing

gear regulated, fully protected: no extractive activities). Data from 1,096 surveys from 317 sites (155 protected, 162 unprotected) from 36 studies and one monitoring program were retained. Most sites (83% of 317) were surveyed with 50 m transects at depths of 2–20 m.

A before-and-after study in 1955–2012 at 14 rocky reef sites on the Portofino Promontory, Ligurian Sea, Italy (14) found that in a protected area that prohibited some fishing and collection, red coral *Corallium rubrum* density was lower after protection that a few years before, but coral biomass increased. Density of corals was lower in 2012 after protection (227 colonies/$m^2$) than in 1990 before protection (378 colonies/$m^2$), but higher than before protection in 1964 (93 colonies/$m^2$). Coral biomass was higher after protection (1,505 g/$m^2$) than before (1990: 809 g/$m^2$, 1964: 302 g/$m^2$). Coral height, weight, apex number and basal diameter were all similar after protection in 2012 and before in 1955, but weight and basal diameter were higher in 2012 than before protection in 1990 (see paper for details). In 1999, an area was designated as a protected area. In 2012, coral samples were collected at 12 locations, from three 400 $cm^2$ surfaces at each location (total of 36 replicates, 368 colonies collected). Data were compared with those collected in two previous studies published in 1965 and 1993. For the 1965 study, forty location were visited, with coral samples collected from 14, and for the 1993 study, four of those 14 locations were resampled. Overall coral metrics from each sampling year were compared.

A replicated, site comparison study in 2007–2010 in five coral reef sites in the Red Sea off Egypt (15) found that in protected areas that prohibited some fishing and collection and also introduced mooring buoys for dive boats, a biodiversity index (which included corals) was higher in two of three protected areas compared to one of two unprotected areas. There was no difference between sites in other comparisons (data reported as statistical model results). For fire corals *Millepora* sp., frequency of sightings increased over time in one protected area (2007: 85% of surveys, 2010: 90%). Other coral species did not show trends over time (data reported as statistical model results). Three protected areas in Sharm el-Sheikh where commercial and sport fishing were prohibited, and two unprotected areas were selected. In 2007–2010, over 7,000 volunteer divers carried out surveys at 100 locations across the five sites (17,900 surveys, 14,500 hours of survey time). Divers completed

a questionnaire where they recorded species that they had seen (14 named coral species and option to report other corals) and estimated the number of individuals. Volunteer surveys were validated against surveys carried out by experts.

A replicated, site comparison study in 2012 at 41 coral reef sites in the Whitsunday Islands, Australia (16) found that protected areas that prohibited some fishing and collection had similar coral cover, but higher prevalence of coral disease and damage compared to areas that did not limit those fishing activities. Coral cover was similar in sites that limited hook and line fishing and prohibited spear fishing and collecting (23%) compared to areas that did not restrict those activities (25%). Coral disease prevalence was higher in sites with more restrictions (5%) than in sites with fewer restrictions (3%). The lowest disease prevalence was in sites that prohibited all fishing (1%). Coral damage was also higher in sites with more restrictions (4%) than in sites with fewer restrictions (2%). In October and November 2012, surveys were conducted at nine fished sites that limited the amount of hook and line fishing and prohibited spear fishing and collecting (27 transects) and 11 fished sites that allowed hook and line fishing, spear fishing, and collecting (33 transects). In addition, 21 sites where all fishing was prohibited (63 transects) were also surveyed. Surveys were conducted along 15 × 2 m belt transects.

A before-and-after study in 1998–2011 at two sites containing deep sea cold-water coral mounds west of Scotland, UK (17) found that after designating a Marine Protected Area that prohibited some fishing and collecting there was no change in coral *Lophelia pertusa* and *Madrepora oculata* cover at one site and a disappearance of most corals at the other site. At one site coral cover was similar eight years after protection was established (47%) compared to 3–5 years before protection (55%). At the other site, cover was 0% eight years after protection compared to 45% in the 3–5 years before protection. Video data after protection found a few cases of coral regrowth but no evidence of coral recolonization from larval settlement. In addition, there was a significant reduction of trawling after protection compared to before, particularly at the site where all corals were lost (data presented graphically). In 2003, an area containing deep sea cold-water coral mounds was closed to bottom trawling and designated as a permanent protected area in 2004. Video

and sonar surveys were conducted in 1998–2000 (3–5 years prior to protection), and follow up surveys were carried out in 2011, eight years after initial fishery closures. The proportion of survey trips that travelled over live coral and presence of trawling scars were recorded.

A replicated, site comparison study [year not specified] at 14 areas of coralligenous habitat off western Italy (18) found that protected areas that prohibited some fishing and collection had similar community assemblages (including corals) compared to unprotected areas but showed different patterns of spatial variability. Average invertebrate cover (including corals) was 6% in protected areas and 4% in unprotected areas (result not tested for statistical significance). Overall, community assemblages were similar in protected areas and unprotected areas, but communities in protected and unprotected areas varied at different spatial scales (data reported as statistical model results). Protected areas had higher variation at the smallest spatial scale (individual survey plots) than unprotected areas, but lower variation at the largest spatial scale (the study areas) than unprotected areas (reported as pseudo-variance). Variation in the abundance of each species was also dependent on the spatial scale considered. Seven Marine Protected Areas and seven unprotected areas were chosen (10s of km apart), with two sites/area and three 10 $m^2$ locations/site selected for sampling. Ten survey plots/location (40 × 50 cm plots) were randomly sampled in June and July using photographs (60 images/protected or unprotected area) to assess cover by different species (including corals).

A replicated, site comparison study in 2009–2013 in 16 coral reef sites along the Belize Barrier Reef, Belize (19) found that protected areas that prohibited some types of fishing had similar coral cover as unprotected areas and areas prohibiting all types of fishing. Coral cover was similar in areas with some fishing restrictions (18%), unprotected areas (21%) and fully protected areas (20%). In addition, cover was similar across different enforcement levels (good: 21%, moderate: 15%, inadequate: 19%, absent: 20%) and did not change due to time since protection started (see paper for details). Four sites were selected with some restrictions (limited fishing licenses and banned use of traps, nets and longlines), four with no protection (although fishing of herbivorous fish and Nassau groupers *Epinephelus striatus* was restricted at all sites), and four that were fully protected (only non-extractive activities allowed).

Each site was monitored in May and June in 2009, 2010, 2012, and 2013 via six 10 m transects, spaced around 10 m apart. Coral cover was recorded, and corals were identified to species level.

A replicated, before-and-after study in 2007–2016 at four sites in Lyme Bay, UK (20) found that after a protected area that prohibited all towed fishing gear was established, abundance of one soft coral species *Alcyonium digitatum* increased over 8–10 years and abundance of another *Eunicella verrucosa* did not change. In 2016, *A. digitatum* abundance was higher than in 2007 (before protection) at a site that had previously been unprotected and dredged (after: 45 individuals/100 $m^2$, before: 1 individual/100 $m^2$) and a site that had previously been unprotected but not dredged (after: 27 individuals/100 $m^2$, before: 6 individual/100 $m^2$). *A. digitatum* abundance remained similar over time in two sites that had been voluntarily protected in 2006 (2016: 24–57 individuals/100 $m^2$; 2007: 7–21 individuals/100 $m^2$). Abundance of *E. verrucosa* remained similar over time at all sites (2016: 13–62 individual/100 $m^2$, 2007: 4–28 individual/100 $m^2$). Authors also report that by 2016, individual *A. digitatum* were larger in both sites that were unprotected in 2007, and that *E. verrucosa* was larger at one of two sites that were unprotected in 2007 (see paper for details). In 2008, a protected area was established that covered four sites and all towed fishing gear was prohibited. Two of the sites had been voluntarily protected with the same restrictions since 2006. In 2007, five locations within each site were surveyed: one site that had previously been dredged and one that had not. Follow up surveys were carried out in 2016. Previous fishing effort was estimated from tracks of five dredging vessels between 2000–2006. Coral cover was assessed using video footage, with cameras towed for around 10 minutes at 0.5 knots.

A replicated, site comparison study over six years [years unknown] from 56 sites spanning six years along the Great Southern Reef off Australia (21) found that in protected areas that prohibited some types of fishing and collection, diversity and cover of sessile invertebrate (reported as "sponges, soft corals, ascidians, etc.") was lower compared to unprotected areas. See original paper for data. In addition, protected areas where all fishing and collecting was prohibited had similar diversity and cover of sessile invertebrates than unprotected areas. Sites either restricted some types of fishing (18 sites in 11 areas, with

a range of different restrictions; see paper for details), fully restricted all fishing and collecting (19 sites in 10 areas) or were outside of a protected area (19 sites). Using data from an online database, a total of 1,971 photo quadrats (33% from partially protected, 46% from fully protected, 21% from unprotected areas) taken along 50 m transects were used to quantify diversity and cover of sessile invertebrates using the Collaborative and Annotation Tools for Analysis of Marine Imagery.

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## 11.6 Designate a Marine Protected Area and prohibit/limit recreational activities (including anchoring)

- **Six studies** evaluated the effects on corals of designating a Marine Protected Area and prohibiting/limiting recreational activities (including anchoring). One study was in each of the US Virgin Islands[1], Israel[2], Spain[3], Egypt[4], Mexico and Cuba[5], and Bonaire[6].

### COMMUNITY RESPONSE (2 STUDIES)

- **Richness/diversity (2 studies):** One site comparison study in Mexico and Cuba[5] found similar coral species richness in a site closed to diving/snorkelling and a site with no diving/snorkelling restrictions. One replicated, site comparison study in Egypt[4] found that in protected areas that limited anchoring by using mooring buoys (and prohibited some fishing), a biodiversity index (which included corals) was higher or similar compared to unprotected areas.

### POPULATION RESPONSE (4 STUDIES)

- **Abundance/Cover (4 studies):** Three site comparison studies (inclunding one replicated, before-and-after study) in Bonaire[6], Mexico and Cuba[5], and the US Virgin Islands[1] found

that prohibiting diving/snorkelling[5,6] or prohibiting anchoring (and some fishing)[1] had mixed effects on coral cover or densities when compared to unprotected areas[1,6] or protected areas with no diving/snorkelling restirctions[5]. One study in Egypt[4] found that in protected areas that limited anchoring by using mooring buoys (and prohibited some fishing) one species group of corals increased in one of three protected areas, but other species showed no change.

- **Condition (2 studies):** One of two site comparison studies (one replicated) in Israel[2] and Spain[3] found that stony corals in protected areas that prohibited diving had higher growth, lower tissue or skeletal loss, and lower predation than corals in areas with some restrictions or fully open[2]. The other study[3] found that in protected areas that prohibited diving (and fishing) fewer coral *Paramuricea clavata* colonies had other organisms growing on them compared to areas where diving and/or fishing was permitted. The study also reported that colonies with organisms growing on them had fewer reproductive cells than those without.

**Background**

A range of recreational and non-extractive activities can directly threaten corals. Prohibiting some or all of these activities within Marine Protected Areas is a widely used conservation action (Kriegl *et al.* 2021), though this is commonly implemented alongside restrictions on fishing activities.

Recreational diving and anchoring (often associated with diving) can cause physical damage to corals (Hasler & Ott 2008, Giglio *et al.* 2017), and damage caused by high levels of tourism has been associated with greater prevalence of coral disease (Lamb *et al.* 2014).

Prohibiting or limiting access for recreational activities in protected areas has the potential to reduce these threats and allow corals and the reefs they form to recover and flourish.

Giglio V.J., Ternes M.L., Mendes T.C., Cordeiro C.A. & Ferreira C.E. (2017) Anchoring damages to benthic organisms in a subtropical scuba dive hotspot. *Journal of Coastal Conservation*, 21, 311–316. https://doi.org/10.1007/s11852-017-0507-7

Hasler H. & Ott J.A. (2008) Diving down the reefs? Intensive diving tourism threatens the reefs of the northern Red Sea. *Marine Pollution Bulletin*, 56, 1788–1794. https://doi.org/10.1016/j.marpolbul.2008.06.002

Kriegl M., Elías Ilosvay X.E., von Dorrien C. & Oesterwind D. (2021) Marine protected areas: At the crossroads of nature conservation and fisheries management. *Frontiers in Marine Science*, 8, Article 676264. https://doi.org/10.3389/fmars.2021.676264

Lamb J.B., True J.D., Piromvaragorn S. & Willis B.L. (2014) Scuba diving damage and intensity of tourist activities increases coral disease prevalence. *Biological Conservation*, 178, 88–96. https://doi.org/10.1016/j.biocon.2014.06.027

A replicated, site comparison study in 2003–2008 at two coral reef sites off St John, US Virgin Islands (1) found that in protected areas where anchoring was prohibited and some fishing and collection was also prohibited, hard coral cover was lower in the protected areas compared to outside, and soft coral cover was higher in one area compared to outside but similar in a second. Hard coral cover was lower in the protected areas (4% and 4%) compared to the unprotected areas (15% and 8%) and soft coral cover was higher in the protected areas in one comparison (inside: 22%, outside: 13%) and similar in a second (inside: 12%, outside: 11%). In one case, coral cover declined in the protected area (2003: 7%, 2008: 3%), but declined more dramatically in the unprotected area (2003: 26%, 2008: 6%), and in a second case, cover in the protected area was 4% in 2003 and 2% in 2008, and in the unprotected area cover was 10% in 2003 and 6% in 2008 (results were not tested for statistical significance). Two protected areas were selected, one on the mid-shelf reef and one in a bay. Anchoring was prohibited, alongside fishing and collection of all species except the blue runner *Caranx crysos*. Sites in the protected areas (18–30 sites/year) and in adjacent unprotected areas (15–25 sites/year) were surveyed annually from 2003–2008. Coral cover was assessed at one location/site within a 15 m diameter area.

A replicated, site comparison study in 2003–2004 at five coral reef sites in Eilat, Israel (2) found that stony corals at a site prohibiting diving had higher growth, lower tissue and skeletal loss, and lower predation compared to corals on sites limiting diving or completely open. After one year, average

growth/colony of *Acropora hemprichii* was higher at the prohibited site (94 g/year) than open sites (28 g/year). The percentage of stony corals with damage (complete or partial tissue loss and skeletal damage) was lower at the prohibited site (21%) than limited (33%) and open (39%) sites. After one year, the percentage of colonies attacked by corallivorous snails *Drupella* spp. had increased significantly in the open site (2003: 4%; 2004: 13%). In July 2003 five sites within the Coral Beach Nature Reserve were identified with different diving restrictions (prohibited: no diving since 1992; limited: <4,000 dives/year; and three open sites: ~6,000–16,000 dives/year). Twenty undamaged *Acropora hemprichii* colonies at each of the prohibited and open sites were measured, photographed next to a ruler and tagged. Separately, all large (>10 cm diameter) undamaged stony coral colonies (mainly *Acropora hemprichii*, *Pocillopora verrucosa* and *Stylophora pistillata*) were selected along a 300 m stretch of the reserve incorporating the prohibited, limited and one open site (338 colonies: 58–168/site). Colonies were photographed, tagged, and presence of corallivorous snails recorded. All colonies were photographed again after twelve months to determine growth of *A. hemprichii*, tissue and skeletal loss, and snail presence for *A. hemprichii*, *P. verucossa*, and *S. pistillata*.

A site comparison study in 2010–2011 at nine coral reef sites in Cap de Creus and Medes Islands, off Spain in the northern Mediterranean (3) found that in a protected area that prohibited diving, and also prohibited all fishing, fewer coral *Paramuricea clavata* colonies had other organisms growing on them (likely due to injury/damage) than in areas where diving and/or fishing was permitted. In the protected area, 4–10% of colonies had other organisms growing on them, compared to 10–33% in unprotected areas. Colonies with organisms growing on them had fewer reproductive cells (5–13 gonads/coral polyp) than those without (10–25 gonads/coral polyp), and authors also reported on differences in concentrations of lipids, carbohydrates and proteins in coral branches (see paper for details). One area of a marine park (established in 1996) where both diving and fishing was prohibited was selected, along with six other sites in the same area (with a mix of diving and recreational fishing) and two sites in a different area (with some diving permitted but no fishing). In June 2010 and January 2011, a total of 15 surveys across the nine locations were carried out (4 in the fully protected area)

by divers along transects (6–20 m long, 16–38 m deep).

A replicated, site comparison study in 2007–2010 in five coral reef sites in the Red Sea off Egypt (4) found that in protected areas that introduced mooring buoys for dive boats and also prohibited some fishing and collection, a biodiversity index (which included corals) was higher in two of the three protected areas compared to one of two unprotected areas. There was no difference between sites in other comparisons (data reported as statistical model results). For fire corals *Millepora sp.*, frequency of sightings increased over time in one of three protected areas (2007: 85% of surveys, 2010: 90%), but did not increase in the two unprotected areas. Other coral species did not show trends over time (data reported as statistical model results). Three protected areas in Sharm el-Sheikh where commercial and sport fishing were prohibited and two unprotected areas were selected. In 2007–2010, over 7,000 volunteer divers carried out surveys at 100 locations across the five sites (17,900 surveys, 14,500 hours of survey time). Divers completed a questionnaire where they recorded species that they had seen (14 named coral species and option to report other corals) and estimated the number of individuals. Volunteer surveys were validated against surveys carried out by experts.

A site comparison study in 2014–2015 at four coral reef sites in Mexico and Cuba (5) found that in protected areas that limited the number of snorkellers or divers, there were not clear differences in coral richness and abundance compared to areas with no restrictions, though there were some differences in community composition. In Mexico, a site with no divers/snorkellers had 23 species and a site with high diver/snorkeller numbers had 20 species, whereas two sites in Cuba with low diver/snorkeller numbers had 35 and 36 species. In Mexico, the no divers/snorkellers site had greater coral density (20–23 colonies/$m^2$) in two of four comparisons than the site with high diver/snorkeller numbers (17–19 colonies/$m^2$), but lower coral cover in three of four comparisons (no divers/snorkellers: 27–44 $cm^2$/$m^2$, high divers/snorkellers: 46–53 $cm^2/m^2$). Coral communities varied between sites in terms of relative abundance, and differences in species density and live coral cover were larger between the sites with no divers/snorkellers and high diver/snorkeller numbers in Mexico

than between the two sites with low diver/snorkeller numbers in Cuba (data reported as graphical analysis). Two sites were selected in protected areas in both Mexico and Cuba. One Mexican site was closed to all divers/snorkellers, and the other received around 100 divers or snorkellers/day. Both Cuban sites received around 15 divers or snorkellers/day. Each site was sampled four times over two years, with eight transects (1 × 10 m) established in each site, and five 1 $m^2$ quadrates sampled/transect (40 samples/site).

A replicated, before-and-after, site comparison study in the early 1980s–2009 in four coral reef sites in Bonaire (6) found that one of two protected areas that prohibited diving/snorkelling showed an increase in hard coral cover, while the other, and two unprotected sites, showed declines. Results were not tested for statistical significance. The sheltered, protected area had higher hard coral cover and lower number of coral patches after protection (cover: 83%, patches: 3) than before (cover: 66%, patches: 7), but the exposed protected area and both unprotected areas (sheltered and exposed) had lower cover (after: 41–54%, before: 79–90%) and more patches (after: 3–9, before: 0–7) after protection. Authors also reported on other metrics including patch size and connectivity. Two marine reserves were established in 1991 that excluded divers and other underwater visitors. One was exposed to storms and the other was sheltered. Adjacent unprotected sites were also selected, one exposed and one sheltered. In the early 1980s, maps of coral cover were created through aerial photographs and scuba diving surveys. In 2008–2009, satellite images were acquired for the same locations, along with 17 underwater video transects. Habitat was classed as coral (live hard coral cover >20%), or sand (>50% sand) or sand-coral mixture (<20% hard coral and <50% sand, where additional cover could include octocorals).

(1) Monaco M.E., Friedlander A.M., Caldow C., Hile S.D., Menza C. & Boulon R.H. (2009) Long-term monitoring of habitats and reef fish found inside and outside the US Virgin Islands Coral Reef National Monument: A comparative assessment. *Caribbean Journal of Science*, 45, 338–347. https://doi.org/10.18475/cjos.v45i2.a18

(2) Gunzer B., Novplansky A., Shalit O. & Chadwick N.E. (2010) Indirect impacts of recreational SCUBA diving: patterns of growth and predation in branching stony corals. *Bulletin of Marine Science*, 86, 727-742. https://www.ingentaconnect.com/content/umrsmas/bullmar/2010/00000086/00000003/art00012

(3) Tsounis G., Martinez L., Bramanti L., Viladrich N., Gili J.M., Martinez Á. & Rossi S. (2012) Anthropogenic effects on reproductive effort and allocation of energy reserves in the Mediterranean octocoral *Paramuricea clavata*. *Marine Ecology Progress Series*, 449, 161–172. https://doi.org/10.3354/meps09521

(4) Branchini S., Pensa F., Neri P., Tonucci B.M., Mattielli L., Collavo A., Sillingardi M.E., Piccinetti C., Zaccanti F. & Goffredo S. (2015) Using a citizen science program to monitor coral reef biodiversity through space and time. *Biodiversity and Conservation*, 24, 319–336. https://doi.org/10.1007/s10531-014-0810-7

(5) Perera-Valderrama S., Hernández-Arana H., Ruiz-Zárate M.Á., Alcolado P.M., Caballero-Aragón H., González-Cano J., Vega-Zepeda A., Victoria-Salazar I., Cobián-Rojas D., González-Méndez J., Hernández-González Z. & de la Guardia-Llansó E. (2017) Temporal dynamic of reef benthic communities in two marine protected areas in the Caribbean. *Journal of Sea Research*, 128, 15–24. https://doi.org/10.1016/j.seares.2017.07.007

(6) Relles N.J., Patterson M.R. & Jones D.O.B. (2019) Change detection in a Marine Protected Area (MPA) over three decades on Bonaire, Dutch Caribbean. *Journal of the Marine Biological Association of the United Kingdom*, 99, 761–770. https://doi.org/10.1017/S0025315418000565

## 11.7 Designate a Marine Protected Area without setting management measures, usage restrictions, or enforcement

https://www.conservationevidence.com/actions/4062

- We found no studies that evaluated the effects of designating a Marine Protected Area without setting management measures, usage restrictions, or enforcement on corals.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Designation of Marine Protected Areas is a widely used conservation action (Kriegl *et al*. 2021). However, there is much variation in governance structures and the extent of restrictions, with many protected areas operating as "paper parks" where management plans are not implemented or enforced (Rife *et al*. 2013, Álvarez-Fernández *et al*. 2020). There is a growing understanding of the features that lead to successful Marine Protected Areas (Edgar *et al*. 2014) Carefully considering management objectives and potential for enforcement is likely to lead to better outcomes than designations alone (Di Minin & Toivonen 2015; Jones & De Santo 2016).

Álvarez-Fernández I., Freire J., Naya I., Fernández N. & Sánchez-Carnero N. (2020) Failures in the design and implementation of management plans of Marine Protected Areas: An empirical analysis for the North-east Atlantic Ocean. *Ocean & Coastal Management*, 192, 105178. https://doi.org/10.1016/j.ocecoaman.2020.105178

Di Minin E. & Toivonen T. (2015) Global protected area expansion: creating more than paper parks. *BioScience*, 65, 637–638. https://doi.org/10.1093/biosci/biv064

Edgar G., Stuart-Smith R., Willis T., Kininmonth S., Baker S.C., Banks S., Barrett N.S., Becerro M.A., Bernard A.T.F., Berkhout J., Buxton C.D., Campbell S.J., Cooper A.T., Davey M., Edgar S.C., Försterra G., Galván D.E., Irigoyen A.J., Kushner D.J., Moura R., Parnell P.E., Shears N.T., Soler G., Strain E.M.A. & Thomson, R.J. (2014) Global conservation outcomes depend on marine protected areas with five key features. *Nature*, 506, 216–220. https://doi.org/10.1038/nature13022

Jones P.J. & De Santo E.M. (2016) Viewpoint–Is the race for remote, very large marine protected areas (VLMPAs) taking us down the wrong track? *Marine Policy*, 73, 231–234. https://doi.org/10.1016/j.marpol.2016.08.015

Kriegl M., Elías Ilosvay X.E., von Dorrien C. & Oesterwind D. (2021) Marine protected areas: At the crossroads of nature conservation and fisheries management. *Frontiers in Marine Science*, 8, Article 676264. https://doi.org/10.3389/fmars.2021.676264

Rife A.N., Erisman B., Sanchez A. & Aburto-Oropeza O. (2013) When good intentions are not enough... Insights on networks of "paper park" marine protected areas. *Conservation Letters*, 6, 200–212. https://doi.org/10.1111/j.1755-263X.2012.00303.x

## 11.8 Establish community-based coral reef management

https://www.conservationevidence.com/actions/4004

- **Two studies** evaluated the effects of establishing community-based coral reef management. One study was in Kenya and Tanzania[1], and one was in Kenya[2].

**COMMUNITY RESPONSE (1 STUDY)**

- **Richness/diversity (1 study)**: One replicated, site comparison in Kenya and Tanzania[1] found that community managed areas had similar diversity of hard coral species compared to protected areas managed by the government.

**POPULATION RESPONSE (2 STUDIES)**

- **Abundance/Cover (2 studies)**: Two replicated, site comparison studies in Kenya and Tanzania[1] and Kenya[2] found that community managed areas had similar coral cover compared to government managed areas.

**OTHER (1 STUDY)**

- **Human behaviour (1 study)**: One study in Kenya and Tanzania[1] found that in a community managed area there was a decrease in the amount of blast fishing over time.

**Background**

There may be a range of social, ecological and economic considerations when protecting corals and other marine resources, which may impact compliance with the range of restrictions that are put in place (Kriegl *et al*. 2021). Community-based management offers an alternative to top-down models of protection that gives local people more control over how they manage their marine resources (Gutiérrez *et al*. 2011). Community-based fisheries management is often based on a partial protection strategy, which uses one or more spatial management measures (for instance measures that restrict some aspect of the fishery; Cinner & Aswani 2007). Given the complex aims of community-based management systems, careful consideration is needed when assessing the effectiveness of these approaches for delivering both ecological and human livelihood and wellbeing outcomes (O'Garra *et al*. 2023).

We have included studies in this action that compare outcomes in areas with community-based management with outcomes in protected areas that are managed through other means, such as those managed by the state.

Studies that report the effect of establishing private coral reef management are described in *Establish private coral reef management*.

Cinner J.E. & Aswani S. (2007) Integrating customary management into marine conservation. *Biological Conservation*, 140, 201–216. https://doi.org/10.1016/j.biocon.2007.08.008

Gutiérrez N.L., Hilborn R. & Defeo O. (2011) Leadership, social capital and incentives promote successful fisheries. *Nature*, 470, 386–389. https://doi.org/10.1038/nature09689

Kriegl M., Elías Ilosvay X.E., von Dorrien C. & Oesterwind D. (2021) Marine protected areas: at the crossroads of nature conservation and fisheries management. *Frontiers in Marine Science*, 8, Article 676264. https://doi.org/10.3389/fmars.2021.676264

O'Garra T., Mangubhai S., Jagadish A., Tabunakawai-Vakalalabure M., Tawake A., Govan H. & Mills M. (2023) National-level evaluation of a community-based marine management initiative. *Nature Sustainability*, 6, 908–918. https://doi.org/10.1038/s41893-023-01123-7

A replicated, site comparison study in 1996 and 2003–2004 in 12 marine areas in coastal Kenya and Tanzania (1) found that areas with community-based coral reef management with a history of destructive fishing had similar coral cover and diversity compared to Marine Protected Areas with no fishing. Coral cover was similar in community-managed sites (1996, before bleaching: 24 cm/m; 2004, post-bleaching: 28 cm/m) compared to protected sites (1996, before bleaching: cover 22 cm/m; 2004, post-bleaching: 32 cm/m). The diversity of hard coral species was also similar between community-managed sites (1996, before bleaching: 12 coral species/90 m transect; 2004, post-bleaching: 13) compared to protected sites (1996, before bleaching: 12; 2004, post-bleaching: 13). Community enforcement reduced dynamite fishing from 180 blasts/month in 1995 to <5/month in 2003. Coral reef sites of similar aspects (four small collaboratively managed reefs in the Mtang'ata Community Managed Area and three separate sites in Marine Protected Areas with no fishing) were compared and included Malindi, Watamu and Mombasa Marine National Parks. The Mtang'ata collaboratively managed area reefs were included in 1994 as recognition of the degradation due to dynamite fishing and illegal mangrove cutting and to enhance the well-being of the coastal communities by improving management. Communities were surveyed using nine 10 m line transects/site with benthic biota >3 cm in length classified into nine gross categories; hard coral were further identified to genus.

A replicated, site comparison study in 2011 at six sites off the coast of Kenya (2) found that areas with community-based management had similar coral cover compared to government-managed no-take zones and fished areas. Coral cover did not vary based on management type and was 26 and 46% in community closures, 20 and 27% in Government closures and 7 and 35% in fished areas. Two community-managed areas were closed to fishing in 2005 and 2010. Two government closures were protected since 1968 and 1991. Two fished areas were fished intensively with a range of gear (including spearguns, nets, traps). Coral cover

was surveyed using randomly placed 10 m transects (nine transects/site).

(1) McClanahan T.R., Verheij E. & Maina J. (2006) Comparing the management effectiveness of a marine park and a multiple-use collaborative fisheries management area in East Africa. *Aquatic conservation: marine and freshwater ecosystems*, 16, 147–165. https://doi.org/10.1002/aqc.715

(2) Humphries A.T., McClanahan T.R. & McQuaid C.D. (2014) Differential impacts of coral reef herbivores on algal succession in Kenya. *Marine Ecology Progress Series*, 504, 119–132. https://doi.org/10.3354/meps10744.

## 11.9 Establish private coral reef management

https://www.conservationevidence.com/actions/4003

- **Two studies** evaluated the effects of establishing private coral reef management on corals. One study was in Malaysia and the Philippines[1], and one was in China[2].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (2 STUDIES)**

- **Abundance/Cover (2 studies)**: One replicated, site comparison study in China[2] found that privately managed areas had similar coral cover to protected areas managed by the government. One site comparison study in Malaysia and the Philippines[1] found that privately managed areas had higher hard coral cover than a collaborative- and government-managed area.

**Background**

Privately managed protected areas may well play an important role (Stolton *et al*. 2014) in meeting ambitious global targets for protected area coverage, such as protecting 30% of land, water and seas by 2030 under the Kunming-Montreal Global Biodiversity Framework (Ainsworth *et al*. 2022).. Such areas may be managed by individuals or groups, non-governmental organisations, corporations, for profit owners or research or religious entities (Stolton *et al*. 2014). While the IUCN definition of a privately protected area requires that areas should qualify as a protected area (as defined by the IUCN), there may be potential for sites that do not meet that definition to confer benefits to corals where management goals and practices have co-benefits for corals and the reefs they form.

We have included studies in this action that compare outcomes in privately managed areas with outcomes in protected areas that are managed through other means, such as those managed by the state.

Studies that report the effect of establishing community-based coral reef management are described in *Establish community-based coral reef management*.

Ainsworth D., Collins T. & d'Amico F. (2022) *Nations adopt four goals, 23 targets for 2030 in landmark UN biodiversity agreement*. Available from: https://www.cbd.int/article/cop15-cbd-press-release-final-19dec2022. Accessed 24 April 2024

Stolton S., Redford K.H. & Dudley N. (2014) *The Futures of Privately Protected Areas*. IUCN: Gland, Switzerland. Available from: https://portals.iucn.org/library/sites/library/files/documents/PATRS-001.pdf

A site comparison study in 2000–2007 at three protected areas that prohibited all fishing and extractive activities in Malaysia and the Philippines (1) reported that privately managed areas had higher

hard coral cover than a collaborative- and government-managed area. Results were not tested for statistical significance. Hard coral cover was 42% in the privately managed area, 26% in the collaboratively managed area and 23% in the government managed area. Cover by other species, including soft coral and algae, was 7% (private), 4% (collaborative) and 35% (government). Collaborative management involved the government appointing an organisation to take partial or complete responsibility for managing the area. Authors reported data from one privately managed area (established: 2001, surveyed: 2007), one collaboratively managed area (established: 1988, surveyed: 2000) and one government managed area (established: 1985, surveyed: 2005). In the privately managed area, coral cover was recorded along 20 × 5 m transects (number of transects not reported). Survey methods for other areas are not reported. **Costs (US$)**: Private management cost $3/ha of protected area. Cost included wages and salary (34%), administration (7%), depreciation (23%), sublease (12%), education and information (7%) and operational costs (17%). The area generated revenue of $5.70/visitor/night, and in 2006 revenues were $1.47/ha.

A replicated, site comparison study in 2014 at nine coral reef sites in Sanya Bay, Hainan, China (2) found that privately managed areas had similar coral cover to protected areas managed by the government but lower cover than unprotected, unmanaged areas. Coral cover was similar in privately managed protected areas (10%), privately managed unprotected areas (8%), and government-managed protected areas (10%), but lower than in unprotected areas without private management (36%). In addition, fishing restrictions were well enforced in privately managed sites, but enforcement was lacking in protected areas without private management (see paper for details). In 2014, nine sites were selected that varied in management (privately managed or not) and protection status (protected vs unprotected). Privately managed areas were managed by three different companies for tourism (including diving, snorkelling, and other water sports) and protected areas were established in 1990. At each site, three 50 m transects were surveyed at each of two depths (2–3 m and 6–8 m), with photographs taken 25 times along each transect using evenly spaced quadrats (50 × 50 cm).

(1) Teh L. C., Teh L. S. & Chung F. C. (2008) A private management approach to coral reef conservation in Sabah, Malaysia. *Biodiversity and Conservation*, 17, 3061–3077. https://doi.org/10.1007/s10531-007-9266-3

(2) Huang H., Wen C.K.C., Li X., Tao Y., Lian J., Yang J. & Cherh K.L. (2017) Can private management compensate the ineffective marine reserves in China? *Ambio*, 46, 73–87. https://doi.org/10.1007/s13280-016-0808-3

## 11.10 Enforce protected area restrictions and regulations

https://www.conservationevidence.com/actions/4002

- **Two studies** evaluated the effects of enforcing protected area restrictions and regulations on corals. One study was in China[1], and one was in Belize[2].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (2 STUDIES)**

- **Abundance/Cover (2 studies)**: Two replicated, site comparison studies in China[1] and Belize[2] found that areas with high levels of enforcement had similar coral cover compared to areas with lower levels of enforcement.

**OTHER (1 STUDY)**

- **Human behaviour (1 study)**: One replicated, site comparison study in China[1] found that areas with high levels of enforcement had fewer fishing boats and fishers and lower response times by authorities compared to areas with low levels of enforcement.

**Background**

Marine Protected Areas may ultimately fail to achieve their stated aims if the restrictions and regulations set out in their management plans are not enforced (so called "paper parks", Rife *et al*. 2013, Álvarez-Fernández *et al*. 2020). Enforcement can be a particular challenge in large protected areas, where costs of surveillance can be very high and resources available for enforcement are often lacking (Wilhelm *et al*. 2014). Developing better understanding of where and when illegal fishing and other incursions take place within specific protected areas may help develop improved plans for enforcement (Arias *et al*. 2016).

We have included studies in this action that compare outcomes in areas with higher levels of enforcement with outcomes in areas with lower levels or no enforcement.

Álvarez-Fernández I., Freire J., Naya I., Fernández N. & Sánchez-Carnero N. (2020) Failures in the design and implementation of management plans of Marine Protected Areas: An empirical analysis for the North-east Atlantic Ocean. *Ocean & Coastal Management*, 192, 105178. https://doi.org/10.1016/j.ocecoaman.2020.105178

Arias A., Pressey R.L., Jones R.E., Álvarez-Romero J.G. & Cinner J.E. (2016) Optimizing enforcement and compliance in offshore marine protected areas: a case study from Cocos Island, Costa Rica. *Oryx*, 50, 18–26. https://doi.org/10.1017/S0030605314000337

Rife A.N., Erisman B., Sanchez A. & Aburto-Oropeza O. (2013) When good intentions are not enough… Insights on networks of "paper park" marine protected areas. *Conservation Letters*, 6, 200–212. https://doi.org/10.1111/j.1755-263X.2012.00303.x

Wilhelm T.A., Sheppard C.R., Sheppard A.L., Gaymer C.F., Parks J., Wagner D. & Lewis N.A. (2014) Large marine protected areas–advantages and challenges of going big. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 24, 24–30. https://doi.org/10.1002/aqc.2499

A replicated, site comparison study in 2014 at nine coral reef sites in Sanya Bay, Hainan, China (1) found that areas with greater enforcement of restrictions had similar coral cover to protected areas

with low enforcement, but lower cover than unprotected areas with no enforcement. Coral cover was similar in protected areas with high enforcement (10%) and low enforcement (10%) and also similar to an unprotected area with high enforcement (8%). The highest cover was in an unprotected area with no enforcement (36%). In 2014, nine sites were selected that varied in management (privately managed or not) and protection status (protected vs unprotected). Privately managed areas were managed by three different companies for tourism (including diving, snorkelling and other water sports) and protected areas were established in 1990. At each site, three 50 m transects were surveyed at each of two depths (2–3 m and 6–8 m), with photographs taken 25 times along each transect using evenly spaced quadrats (50 × 50 cm). Levels of enforcement were determined through recording response time of management authorities following entry into the area and recording the number of fishing boats and fishers in the area. High enforcement areas were managed privately and had fewer fishing boats and fishers (average of 0, response time of enforcement 17–36 minutes) than low enforcement areas managed by local government (fishing boats: 1, fishers: 4, response time of enforcement >2 h).

A replicated, site comparison study in 2009–2013 in 16 coral reef sites along the Belize Barrier Reef, Belize (2) found that sites with greater enforcement of fishing restrictions had similar coral cover compared to sites with lower or no enforcement. Coral cover was similar in areas where enforcement was considered good (21%), moderate (15%), inadequate (19%) or absent (20%). In addition, cover was similar across sites with different protection (fully protected: 20%, some fishing restrictions: 18%, no protection: 21%) and did not change due to time since protection started (see paper for details). Sixteen sites were selected (15−18 m depth) and classified based on the level of enforcement of restrictions. Enforcement was classified as good (regular patrols and satisfactory compliance), moderate (regular patrols but some poaching and insufficient legal outcomes), inadequate (irregular patrols, greater poaching, insufficient legal outcomes, and a high level of concern from the local community) or absent. Each site was monitored in May and June in 2009, 2010, 2012, and 2013 via six 10 m transects, spaced around 10 m apart. Coral cover was recorded, and corals were identified to species level.

(1) Huang H., Wen C.K.C., Li X., Tao Y., Lian J., Yang J. & Cherh K.L. (2017) Can private management compensate the ineffective marine reserves in China? *Ambio*, 46, 73–87. https://doi.org/10.1007/s13280-016-0808-3

(2) Cox C., Valdivia A., McField M., Castillo K. & Bruno J.F. (2017) Establishment of marine protected areas alone does not restore coral reef communities in Belize. *Marine Ecology Progress Series*, 563, 65–79. https://doi.org/10.3354/meps11984

## 11.11 Identify/designate high biodiversity areas

https://www.conservationevidence.com/actions/4001

- **One study** examined the effects of identifying/designating high biodiversity areas on corals. The study was in Australia[1].

**COMMUNITY RESPONSE (1 STUDY)**

- **Richness/diversity (1 study)**: One site comparison study in Australia[1] found that a site given a designation due to its high biodiversity had a distinct community assemblage compared to a site with no designation.

**POPULATION RESPONSE (1 STUDY)**

- **Abundance/Cover (1 study)**: One site comparison study in Australia[1] found that a site with a high biodiversity designation had lower cover of all benthic species (including hard and soft corals) across four depth categories compared to a site with no designation.

**Background**

A range of approaches have been developed for identifying areas that are particularly important for biodiversity and conservation. For example, Key Biodiversity Areas are "sites contributing significantly to the global persistence of biodiversity" that consider whether sites hold threatened or geographically restricted biodiversity, hold intact ecological communities, support key biological processes or display high levels of irreplaceability (IUCN 2016). While identifying and designating sites as having globally important biodiversity may not confer any specific protections per se, it may be a key step towards achieving those protections as governments and other actors seek to expand protected area networks. It may be challenging to understand the impacts of these designations given that sites receiving them will be those with existing high levels of biodiversity.

IUCN (2016) *A Global Standard for the Identification of Key Biodiversity Areas*. Version 1.0. First edition. IUCN: Gland, Switzerland.

A site comparison study in 2013 at two coral reef sites off the Pilbara coast, northwest Australia (1) found that a site that had been designated as an area of high biodiversity had a distinct community assemblage (including corals), but lower cover of benthic taxa (including hard and soft corals) compared to another site with no designation. Percentage cover of coral families and morphologies were both more strongly influenced by site than a range of environmental variables (data reported as statistical model result, see paper for details). The site with a biodiversity designation had lower cover of all benthic taxa (including hard and soft corals) at four depth categories (21, 9, 2 and <1%) compared to the site with no designation (44, 36, 21 and 6%). The highest hard coral cover at the designated site was recorded at <40 m depth (3%), but the undesignated site had cover of 23% at this depth. The highest soft coral cover at the designated site was found at <40 m (4%) but found at 40–≥80 m at the undesignated site (1–2%).

Two isolated reef sites were selected, one that was designated as a Key Ecological Feature (14,700 ha) and one with no designation (1,700 ha). Surveys of community composition and cover (including corals) were conducted in 2013 via towed video surveys (2 km tows), with 53 tows in the designated site and 23 in the undesignated site. A range of methods were used to assess other environmental variables (see paper for details).

(1) Abdul Wahab M.A., Radford B., Cappo M., Colquhoun J., Stowar M., Depczynski M., Miller K. & Heyward A. (2018) Biodiversity and spatial patterns of benthic habitat and associated demersal fish communities at two tropical submerged reef ecosystems. *Coral Reefs*, 37, 327–343. https://doi.org/10.1007/s00338-017-1655-9

# 12. Habitat restoration and creation

**Background**

Habitat destruction is the greatest threat to biodiversity worldwide and habitat protection remains one of the most important and frequently used conservation actions. However, in many parts of the world, restoring damaged habitats, improving habitats through altering management regimes, or creating new habitat may also be possible. The role of restoration ecology in conservation is well established (Dobson *et al*. 1997), and there is a growing movement within the more specific field of coral reef restoration (Vardi *et al*. 2021), with a rapidly developing evidence base (Boström-Einarsson *et al*. 2020).

Habitat restoration for corals includes actions aimed at stabilizing damaged reefs and the use of natural materials or unnatural materials and structures to restore, repair or create habitat for natural coral settlement. This includes the use of settlement tiles and the repurposing and modification of existing and obsolete man-made offshore structures.

For studies describing attempts to restore habitats indirectly through the designation of legal or other protections, see *Habitat protection*, and for those restoring habitats through cultivating or transplanting of corals see *Species management*.


https://doi.org/10.11647/OBP.0453.12

Here, descriptive studies of biodiversity on or around man-made structures already in place, such as oil rigs and wind farms, are not included, unless they were specifically deployed or modified to enhance local coral diversity or left in place following decommissioning, to act as artificial reefs.

Boström-Einarsson L., Babcock R.C., Bayraktarov E., Ceccarelli D., Cook N., Ferse S.C., Hancock B., Harrison P., Hein M., Shaver E., Smith A., Suggett D., Stewart-Sinclair P.J., Vardi T. & Mcleod I.M. (2020) Coral restoration – A systematic review of current methods, successes, failures and future directions. *PloS One*, 15, e0226631. https://doi.org/10.1371/journal.pone.0226631

Dobson A.P., Bradshaw A.D. & Baker A.J. (1997) Hopes for the future: restoration ecology and conservation biology. *Science*, 277, 515–522. https://www.science.org/doi/10.1126/science.277.5325.515

Vardi T., Hoot W.C., Levy J., Shaver E., Winters R.S., Banaszak A.T., Baums I.B., Chamberland V.F., Cook N., Gulko D., Hein M.Y., Kaufman L., Loewe M., Lundgren P., Lustic C., MacGowan P., Matz M.V., McGonigle M., McLeod I., Moore J., Moore T., Pivard S., Joseph Pollock F., Rinkevich B., Suggett D.J., Suleiman S., Shay Viehman T., Villalobos T., Weis V.M., Wolke C. & Montoya-Maya, P.H. (2021) Six priorities to advance the science and practice of coral reef restoration worldwide. *Restoration Ecology*, 29, e13498. https://doi.org/10.1111/rec.13498

# Natural habitat restoration/creation

## 12.1 Use natural materials to restore/repair/create habitat for corals to encourage natural coral settlement

https://www.conservationevidence.com/actions/3987

- **Four studies** evaluated the effects of restoring / repairing / creating habitat for corals using natural material to encourage coral settlement. Two were in Indonesia[2a,b] and one study was in each of Israel[1] and Australia[3].

**COMMUNITY RESPONSE (1 STUDY)**

- **Richness/diversity (1 study)**: One site comparison study in Israel[1] found that large rocks placed in an orderly pattern had a lower diversity of coral species than natural reef patches.

**POPULATION RESPONSE (2 STUDIES)**

- **Abundance/Cover (4 studies)**: Three of four studies (two replicated including one controlled, and one site comparison) in Israel[1], Indonesia[2a,b], and Australia[3] found that using piles of rocks to create reefs led to higher numbers of corals colonizing when rocks were randomly aggregated compared to orderly[1], in different patterns[2b] or bare rubble[2a,b]. The fourth study[3] found that repositioned coral columns ('bommies') retained live coral tissue and were colonized by other coral species.

**Background**

Man-made reefs provide a solution to the pressure of human activity by expanding the available habitat on which corals can naturally settle and colonize (Abelson & Schlesinger 2002). Using natural material to restore or create habitat for corals to settle on can provide a more sustainable option than using unnatural materials. Natural materials can be coral rock/rubble, limestone rock, or calcium carbonate substrate such as giant clam *Hippopus* and *Tridacna* shells (Neo *et al*. 2015). They can also be 'living' materials such as coral outcrops (sometimes known as 'bommies'), comprising habitat-forming species of coral (e.g., *Porites* spp.) that can provide a substrate for other corals to colonize. Using natural material to construct reefs can enable corals to settle, particularly if the material being used is similar to nearby substrate. In addition, natural materials can offer an opportunity to design a reef which closely resembles the natural surroundings.

Here we focus on the creation of reef structures using natural materials to encourage subsequent settlement by wild coral from existing populations in the vicinity. Other similar actions include *Use structures made from unnatural materials to restore/repair/create habitat for corals to encourage natural coral settlement; Stabilize damaged or broken coral reef substrate*. Actions relating to transplanting or cultivating coral species on natural substrates are covered in *Transplant nursery-grown corals onto natural substrate; Transplant wild-grown corals onto natural substrate; Cultivate coral fragments in an artificial nursery located in a natural habitat*; and *Cultivate coral larvae in an artificial nursery located in a natural habitat*.

Abelson A. & Shlesinger Y. (2002) Comparison of the development of coral and fish communities on rock-aggregated artificial reefs in Eilat, Red Sea. *ICES Journal of Marine Science*, 59, S122–S126. https://doi.org/10.1006/jmsc.2002.1210

Neo M.L., Eckman W., Vicentuan K., Teo S.L.-M. & Todd P.A. (2015) The ecological significance of Giant Clams in coral reef ecosystems. *Biological Conservation*, 181, 111–123. https://doi.org/10.1016/j.biocon.2014.11.004

A site comparison study in 1989–1998 of two man-made reefs in the Gulf of Aqaba, near Eilat, Israel (1), found that a reef comprising randomly aggregated piles of smaller rocks had a greater number of coral species than one comprising orderly aggregated piles of larger rocks, and the orderly aggregated reef had a lower number of coral species than the nearby natural reef. Species richness was higher on a reef with randomly aggregated piles of small rocks (33 species) than one with orderly aggregated piles of larger rocks (25 species) after 8.3 years. The average number of coral species and number of individuals were significantly lower on the orderly aggregated reef (8 species, 17 individuals) compared to a natural reef located 100 m away (18 species, 58 individuals) after seven years. Two artificial reefs, constructed using limestone rocks to imitate the substrate on the nearby natural reef, were deployed in December 1989, one hundred meters south of a Coral Reserve. One reef comprised

randomly aggregated piles of rocks (area: 4.9 $m^2$, average rock diameter 18.9 cm) and the other orderly aggregated piles of rocks (area: 12 $m^2$, average rock diameter 49.5 cm). Coral species were visually recorded on the two artificial reefs every 4–6 months for four years and eight months, then with a single survey eight years and four months after deployment. Comparison between the orderly aggregated and natural reef was made during a single transect survey in 1996.

A replicated, controlled study in 2000–2003 at nine coral rubble sites in the Komodo National Park, Indonesia (2a) found that using rock piles to create reefs led to higher numbers of stony coral recruits and greater area covered by coral than sites left as bare rubble. The average number of stony corals increased during the study period from 1–21/$m^2$ (six months after rock pile installation) to 1–42/$m^2$ (three years after installation) (data not statistically tested). The average area covered by corals increased from 0–19 $cm^2/m^2$ (six months after installation) to 14–1262 $cm^2/m^2$ (three years after installation). There was no detectable increase in coral numbers or coverage on the bare rubble control site. In spring 2000, piles of limestone and lithic sandstone rocks (0.5–2.0 $m^3$) were placed inside three or four 10 $m^2$ areas of coral rubble substrate at each of nine sites. Rock piles were 70–90 cm high and placed 2–4 m apart. Surveys were carried out every 6 months until May 2002 then a final survey in March 2003. Coral recruits were counted, and area covered by coral was measured using 1 $m^2$ quadrats.

A study in 2002–2003 at four coral rubble sites in the Komodo National Park, Indonesia (2b), reported that stony corals settled on rocks piled in different patterns whereas none settled on areas of bare rubble. Six–twelve months after rock piles were installed, average coral numbers were 7/$m^2$ (4–14/$m^2$) and the average size of corals was 8 $cm^2$ (3–11 $cm^2$). Data were not statistically tested. In March–September 2002, rock piles each ~140 $m^3$ and comprising limestone and lithic sandstone were installed in different patterns at four sites with >1000 $m^2$ of coral rubble substrate. Site 1: rocks completely covered the site ~75 cm high; site 2: rock piles 1–2 $m^3$ were placed every 2–3 m; site 3: spurs ~75 cm high, 2 m wide were placed every 2–3 m parallel to the prevailing current; site 4: spurs ~75 cm high, 2 m wide were placed every 2–3 m perpendicular to the prevailing

current. Sites were surveyed once in March 2003 (6–12 months after rocks were installed). Coral recruits were counted and measured using 1 $m^2$ quadrats. An area of bare rubble adjacent to each site was surveyed for comparison.

A replicated study in 2017–2018 off Whitsunday Island, Great Barrier Reef, Australia (3) found that following the repositioning of displaced column-shaped coral outcrops ('bommies') of stony coral *Porites* spp. colonies, some live tissue was retained, and other coral species colonized them. Sixteen months after bommies were repositioned, coverage of original live tissue ranged from 0–20% (average 6%) with 16 of the 22 bommies surveyed still retaining some live tissue. Thirteen of the 22 bommies were colonized by other corals including species of *Pocillopora, Cyphastrea, Favia, Favites, Goniastrea, Psammocora* and *Hydnophora*). Eight bommies had at least one coral recruit, four had at least two, and one had six. Recruits ranged from 3–15 cm in diameter. In March 2017, a cyclone dislodged bommies of *Porites* spp. colonies (1–3 m diameter) and deposited them on the intertidal zone. In June 2017 heavy machinery was used to roll the bommies back into the subtidal region along with 100 $m^3$ of dead coral rubble. Divers surveyed coral bommies in October 2018, recording live tissue coverage (%) and identifying coral species recruited onto the bommie. **Costs (AUS$)**: The costs (reported in 2019) to reposition dead coral rubble were ~AUS$30,000 (it is not reported whether this included the bommie repositioning).

(1) Abelson A. & Shlesinger Y. (2002) Comparison of the development of coral and fish communities on rock-aggregated artificial reefs in Eilat, Red Sea. *ICES Journal of Marine Science*, 59, S122–S126. https://doi.org/10.1006/jmsc.2002.1210

(2) Fox H.E., Mous P.J., Pet J.S., Muljadi A.H. & Caldwell R.L. (2005) Experimental assessment of coral reef rehabilitation following blast fishing. *Conservation Biology*, 19, 98–107. https://doi.org/10.1111/j.1523-1739.2005.00261.x

(3) McLeod I.M., Williamson D.H., Taylor S., Srinivasan M., Read M., Boxer C., Mattocks N. & Ceccarelli D.M. (2019) Bommies away! Logistics and early effects of repositioning 400 tonnes of displaced coral colonies following cyclone impacts on the Great Barrier Reef. *Ecological Management and Restoration*, 20, 262–265. https://doi.org/10.1111/emr.12381

## 12.2 Stabilize damaged or broken coral reef substrate or remove unconsolidated rubble

https://www.conservationevidence.com/actions/3988

- **Six studies** examined the effects of stabilizing damaged or broken coral reef substrate or removing unconsolidated rubble on coral colonies. Three studies were in Indonesia[2,3,6], and one was in each of the Maldives[1], the Phillipines[4,] and Puerto Rico[5].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (4 STUDIES)**

- **Abundance/Cover (5 studies)**: Five studies (three replicated, including two controlled) in the Maldives[1], Indonesia[2,3,6], and the Philippines[4] reported that in areas where degraded coral reefs were stabilized, coral numbers[1,2,3] and coverage[1,3,4,6] increased compared to those with unstablized coral rubble. One of the studies[3] found that coral numbers and coverage varied between reefs stabilized with rock piles compared to other materials, another study[6] found density varied with different configurations of rock piles and one study[1] found more corals on structures designed to provide a high level of stability.
- **Survival (1 studies)**: One controlled study in the Philippines[4] found that on areas where coral reef was stabilized stony coral survived and survival was higher than in unstabilized areas.
- **Condition (1 study)**: A study in Puerto Rico[5] reported that stabilizing a patch of damaged coral reef, as well as transplanting wild-grown and nursery-grown fragments of staghorn coral, led to the patch of restored reef more than doubling in size, whereas no growth was reported on an unstabilized patch.

**Background**

Historically, coral 'rock' has been extracted from reefs for use in construction. The practice involves removing the top 0.5 m of the coral structure (Clark & Edwards, 1999). The remaining reef comprises broken/loose coral and coral 'rubble' (Clark & Edwards, 1999). Other actions, such as 'blast fishing' (the use of dynamite to bring fish to the surface) have a similarly devastating effect on coral reefs (Raymundo *et al*. 2007). Stabilizing damaged or degraded coral reefs using natural or unnatural materials can provide a stable substrate enabling coral colonies to re-establish. Stabilized reefs are likely to be more resilient to the impact of storms (Raymundo *et al*. 2007).

This action is specifically related to the effectiveness of 'stabilizing' an existing coral reef/rubble substrate. Actions relating to the restoration or creation of reefs using natural or unnatural materials are summarized in sections *Use natural materials to restore/repair/create habitat to encourage coral settlement; Use structures made from unnatural materials to restore/repair/create habitat to encourage coral settlement*. Coral settlement happens by natural colonization from existing wild colonies in the vicinity. Actions relating to cultivating or transplanting corals onto stabilized reefs are covered in *Cultivate coral fragments in an artificial nursery located in a natural habitat; Cultivate coral larvae in an artificial nursery located in a natural habitat; Transplant nursery-grown corals onto natural substrate; Transplant nursery-grown corals onto artificial substrate; Transplant wild-grown corals onto natural substrate; Transplant wild-grown corals onto artificial substrate; Change transplant attachment method*.

Clark S. & Edwards A.J. (1999) An evaluation of artificial reef structures as tools for marine habitat rehabilitation in the Maldives. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 9, 5–21. https://doi.org/10.1002/(SICI)1099-0755(199901/02)9:1<5::AID-AQC330>3.0.CO;2-U

Raymundo L.J., Maypa A.P., Gomez E.D., Cadiz P. (2007) Can dynamite-blasted reefs recover? A novel, low-tech approach to stimulating natural recovery in fish and coral populations. *Marine Pollution Bulletin*, 54, 1009–1019. https://doi.org/10.1016/j.marpolbul.2007.02.006

A study in 1990–1993 at an area of degraded coral reef in Galu Falhu, Maldives (1) reported that using artificial structures to provide greater stability to coral rubble substrate led to an increase in the number of coral colonies. After 3.5 years, approximately 500 coral colonies (average density 13/$m^2$) were recorded on structurally complex concrete/PVC blocks that provided high substrate stability. After 3.5 years, average density on concrete mats that provided medium stability was 3 recruits/$m^2$ but 18/$m^2$ on the edges. After 3.5 years, some corals were observed attached to chain link fencing designed to provide low stability (numbers not reported). After 2.5 years, coral coverage on the unstabilized rubble had declined from 0.8% to 0.19%. In 1990–1991, four 10 × 5 m areas of previously mined coral rubble substrate at four sites each received one of three artificial substrate-stabilizing structures or were left unstabilized. Structures comprised complex concrete/PVC blocks (providing high stability), concrete mats (medium stability), or chain-link fencing (low stability) (see paper for design). Structures were deployed 0.5–1.8 m deep and were either sufficiently heavy to prevent movement by wave action or, for the concrete mats and chain-link fencing, weighted down using paving slabs. Monitoring took place at 8–12 month intervals for 2.5–3.5 years. **Costs (UK£)** (presented in 1999): concrete/PVC blocks £210/$m^2$; concrete mats £66/$m^2$; chain-link fencing £26/$m^2$.

A replicated study in 2000 at a degraded coral reef in Komodo National Park, eastern Indonesia (2) reported that stabilizing damaged coral substrate using piles of quarried rocks led to an increase in stony coral numbers compared to unstabilized coral rubble. Results were not tested for statistical significance. After six months, stony coral numbers on the stabilized reef ranged from 1–20/$m^2$ and after 12 months 1–36/$m^2$ compared to no observed increase in coral numbers on the unstabilized areas (data not reported). In April 2000, three or more 0.5–2.0 $m^3$ rock piles were installed at each of nine sites with coral-rubble substrate (comprising dead coral fragments) across Komodo National Park. Sites were surveyed for stony coral recruits in October 2000 and April 2001 using six 1 $m^2$ quadrats/site. **Costs (US$)**: US$ 5–10/$m^2$ (reported in 2001).

A replicated, controlled study in 1998–2001 at nine coral rubble sites in the Komodo National Park, Indonesia (3) found that stabilizing coral rubble using piles of rocks led to a higher number and coverage of coral recruits compared to rubble stabilized using cement blocks, or netting, or unstabilized rubble. After three years, the average number of corals was highest on rock piles (13/plot) followed by cement blocks (11/plot) and netting (7/plot) and lowest on unstabilized rubble (5/plot). Average area ($cm^2$/plot) covered by coral recruits was highest on rock piles (476 $cm^2$), followed by cement blocks (270 $cm^2$), and netting (253 $cm^2$), and lowest on unstabilized rubble (188 $cm^2$). In October and November 1998, two–four 1 $m^2$ plots were placed at each site with either rock piles (20–40 cm high, rocks 20–30 cm diameter), cement blocks, or netting (~5 cm mesh) pinned to the substrate. An additional four plots/sites were left as unstabilized rubble. The number of coral recruits and area covered was recorded every six months for three years. Plots began to degrade after 2.5 years due to strong currents.

A controlled study in 2003–2006 on a platform/patch coral reef in Negros Oriental, Philippines (4) found that in plots where rubble was stabilized with plastic mesh carpets and stone piles, new stony corals settled and had greater survival and cover than corals on unstabilized rubble. On stabilized plots established in the spawning season, corals settled within three months and reached 1–8 individuals/$m^2$ after 36 months. On plots established after spawning, they settled within a year and reached 4–7 individuals/$m^2$ after 32 months. Over a 10-month period after settlement, coral survival and colony size was greater on stabilized plots (survival: 63%, diameter: 6 cm) than unstabilized rubble (survival: 6%, diameter: 2–4 cm). Two years after establishment, stabilized plots had a higher average coverage of corals (19%) than unstabilized rubble (8%), but lower than adjacent healthy reef (44%). Five 17.5 $m^2$ plots were established, three in June 2003 (coral spawning season) and two in October 2003 (before storm season). Plots were at the edge of a 2,400 $m^2$ rubble field created by dynamite fishing, within a platform/patch reef in the Calagcalag Marine Protected Area. In the plots and the areas in between, plastic mesh carpets (2 cm mesh) were anchored to the rubble with metal stakes (with holes cut to accommodate existing coral), and rock piles (1 pile/0.5 $m^2$, 1 m high) were placed

on top of the mesh. Corals in plots and in transects through untreated rubble and adjacent healthy reef were counted 1–4 times/year for three years. In May 2004, ten to twelve coral recruits from each plot established in June 2003 (total 30 recruits) and 25 recruits from the rubble field were tagged and monitored for growth and survival for 10 months.

A study in 2006–2014 at a damaged coral reef site in Tallaboa, Puerto Rico (5) reported that stabilizing the substrate along with transplanting wild-grown and nursery-grown fragments of staghorn coral *Acropora cervicornis*, led to the area of restored reef increasing. After eight years, the area of restored reef had grown from 70 $m^2$ to 180 $m^2$. Coral colonies in unrestored areas in the vicinity, with loose rubble and damaged substrate, showed no signs of recovery during the same period. It was not possible to determine from the study how much of the recovery was attributable to stabilizing the substrate, transplanting loose fragments, or transplanting nursery-grown fragments. In 2006, following the destruction of a coral reef by a ship grounding, wire cages and metal stakes were used to stabilize a 70 $m^2$ area of damaged reef. Approximately 227 (10–20 cm) loose fragments of staghorn coral were collected from nearby reefs and attached to the substrate using cement puddles. In 2009–2011, approximately 400 (20–40 cm) fragments of staghorn coral were collected from a nursery and attached to the substrate using masonry nails, cable ties and/or epoxy. Coral recovery was measured using aerial imagery in 2014. No other methods are reported.

A replicated, controlled, before-and-after study in 2002–2016 at four sites in Komodo National Park, eastern Indonesia (6) found that using piles of quarried rocks to stabilize coral rubble substrate resulted in an increase in coral density compared to unstabilized rubble, and coral cover varied on different rock configurations. Average stony coral cover on the rock piles increased over time and reached 45% after 14 years compared to 3% on the adjacent unstabilized coral rubble site. Coral cover varied between rock configurations (range: single rock: 3–68%; small piles: 20–61%; parallel: 24–83%; perpendicular: 39–68%). In 2002, over 6,000 $m^2$ of quarried rock (20–30 cm diameter) was placed 6–10 m deep at four sites within the Komodo National Park (Gillawadarat, Karang Makassar, Padar, and Papagarang). Rocks were placed in different configurations: single rock pile; small piles 1–2 $m^3$; parallel to the prevailing current;

and perpendicular to the prevailing current. Rock piles were surveyed in 2004, 2008 and 2016 using five–eight 1 $m^2$ quadrats that the authors selectively placed to capture the range and type of cover.

(1) Clark S. & Edwards A.J. (1999) An evaluation of artificial reef structures as tools for marine habitat rehabilitation in the Maldives. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 9, 5–21. https://doi.org/10.1002/(SICI)1099-0755(199901/02)9:1<5::AID-AQC330>3.0.CO;2-U

(2) Fox H.E. & Pet J.S. (2001) Pilot study suggests viable options for reef restoration in Komodo National Park. *Coral Reefs*, 20, 219–220. https://doi.org/10.1007/s003380100175

(3) Fox H.E., Mous P.J., Pet J.S., Muljadi A.H. & Caldwell R.L. (2005) Experimental assessment of coral reef rehabilitation following blast fishing. *Conservation Biology*, 19, 98–107. https://doi.org/10.1111/j.1523-1739.2005.00261.x

(4) Raymundo L.J., Maypa A.P., Gomez E.D., Cadiz P. (2007) Can dynamite-blasted reefs recover? A novel, low-tech approach to stimulating natural recovery in fish and coral populations. *Marine Pollution Bulletin*, 54(7), 1009–1019. https://doi.org/10.1016/j.marpolbul.2007.02.006

(5) Griffin S.P., Nemeth M.I. Moore T.D. & Gintert B. (2015). Restoration using *Acropora cervicornis* at the T/V MARGARA grounding site. *Coral Reefs*, 34, 885–885. https://doi.org/10.1007/s00338-015-1310-2

(6) Fox H.E., Harris J.L., Darling E.S., Ahmadia G.N., Estradivari & Razak T.B. (2019) Rebuilding coral reefs: success (and failure) 16 years after low-cost, low-tech restoration. *Restoration Ecology*, 27, 862–869. https://doi.org/10.1111/rec.12935

## Artificial habitat creation

### 12.3 Use structures made from unnatural materials to restore/repair /create habitat for corals to encourage natural coral settlement

https://www.conservationevidence.com/actions/3989

- **Ten studies** examined the effects of using unnatural materials to create habitat to encourage coral settlement. Five studies were in the USA[1,3,5,6,10], two in Singapore[4,9] and one in each of Hong Kong[2], Indonesia[7], and Japan[8].

**COMMUNITY RESPONSE (2 STUDIES)**

- **Richness/diversity (2 studies)**: One site comparison study in the USA[3] found that diversity of corals settled on concrete or limerock was similar to a natural reef. Another site comparison study in Japan[8] found that diversity of corals settled on ropes was higher than on some natural reefs.

**POPULATION RESPONSE (10 STUDIES)**

- **Abundance/Cover (10 studies)**: Ten studies (five replicated, including one controlled, and one randomized, controlled) in the USA[1,3,5,6,10], Hong Kong[2], Singapore[4,9], Indonesia[7], and Japan[8] found that coral settled on unnatural materials. Two of the studies[1,2] found that the number of corals settling depended on settlement substrate material. Two studies[4,10,] found that coral settlement was higher on fibreglass/sand/calcium carbonate[4], and concrete[10] substrate than on the surrounding natural reef. Three studies[3,5,6,] found that coral cover[5,6,] and density[3,5,] on concrete and/or limerock[3,5,] and concrete/limestone[6] substrate became similar to the natural reef. One study[7] found that the number of coral recruits was similar whether concrete structures were next to or away from transplanted adult colonies.
- **Survival (1 study)**: One replicated, controlled study in the USA[10] found that soft coral settled on concrete slabs had lower survival than on a natural reef.
- **Condition (3 studies)**: Two of three studies (one replicated, one site comparison) in the USA[3], Singapore[4] and Japan[8] found that coral that settled on concrete or limerock[3], or fibreglass/sand/calcium carbonate[4] structures were smaller than coral on the surrounding natural reef. The third, replicated, study[8] found that corals settled on ropes experienced less bleaching but higher levels of disease than on a natural reef.

**Background**

The use of unnatural materials, (materials not typically encountered by corals such as concrete, or PVC), to create reefs specifically designed to encourage settlement by coral is a widely-used method that aims to rapidly expand available habitat and encourage corals to settle. However, there is also the potential for negative consequences within the coral ecosystem through pollution or contamination caused by the degradation of unnatural reef materials (such as concrete or PVC) (McManus *et al*. 2018).

Here we focus on the creation of artificial reef structures using unnatural materials to encourage subsequent settlement by wild coral. Other similar actions include *Use natural materials to restore/repair/create habitat for corals to encourage natural coral settlement; Stabilize damaged or broken coral reef substrate; Repurpose obsolete offshore structures to act as structures for restoring coral reefs* (where a man-made structure is no longer being used for its original purpose and has been repurposed as an artificial reef); *Modify existing man-made structures to create artificial reefs* (where a structure was created for another purpose but has been modified to allow colonization by coral or has been colonized in its original state); and *Use settlement tiles to encourage natural coral settlement* (where tiles made from various materials are placed on the substrate). Actions relating to cultivating or transplanting corals onto artificial substrates are covered in *Cultivate coral fragments in an artificial nursery located in a natural habitat; Cultivate coral larvae in an artificial nursery located in a natural habitat; Transplant nursery-grown corals onto artificial substrate*; and *Transplant wild-grown corals onto artificial substrate*.

McManus R.S., Archibald N., Comber S., Knights A.M., Thompson R.C. & Firth L.B. (2018) Partial replacement of cement for waste aggregates in concrete coastal and marine infrastructure: a foundation for ecological enhancement? *Ecological Engineering*, 120, 655–667. https://doi.org/10.1016/j.ecoleng.2017.06.062

A study in 1995–1998 in two artificial reefs in Florida, USA (1) found that three years after concrete blocks embedded with limerock were used to create habitat, stony corals, hydrocorals and octocorals had established on the unnatural substrates. At one site, three years after a ship grounding crater was filled with concrete blocks embedded with limerocks, seven types (species or genera) of coral were found at a density of 3 corals/$m^2$. *Porites astreoides* was the most abundant (>15% of corals) at the site. Sixty percent of corals had settled on the embedded limerocks (25% of the structure), rather than the surrounding concrete (75%). At the other site, three years after a grounding crater was filled with limerock boulders, 11 types of coral were found, at a density of 4 corals/$m^2$. *Porites astreoides, Favia fragum* and *Agaricia* sp. Were the most abundant, each constituting >15% of corals at the site. In October and November 1989, two ships grounded on reefs 6.5 km apart in the northern Florida Keys National Marine Sanctuary, leaving craters. In June–August 1995, at the 2.5 m-deep site, 40 concrete blocks embedded with limerocks were used to fill the crater and sealed with cement. At the other 10 m-deep site large limerock boulders were used to fill the crater. In summer 1998, three years after installation, juvenile coral recruits were mapped and measured on 17 concrete blocks and 17 limerock boulders. The proportion of corals on the embedded limerocks compared to surrounding concrete was measured on nine of the concrete blocks.

A replicated study in 1993–1995 at an artificial reef in Hoi Ha Wan, Hong Kong (2) found that after pulverised fly-ash/cement blocks were used to create habitats, the number of stony coral recruits settling onto the blocks varied according to time immersed, block orientation, composition and species. A total of 387 *Oulastrea crispata* were recorded during the 24-month monitoring period (0–65/$m^2$). More recruits settled on the top and reef-facing sides of the block compared to the sea-facing or bottom sides (data not reported). There was no difference in *Oulastrea crispata* recruitment on blocks comprising different pulverised fly-ash:cement mixes. Thirty *Culicia japonica* recruits were recorded during the monitoring period, with the density fluctuating (range 0–6/$m^2$) and peaking after 24 months. More recruits were recorded on the reef-facing, top and bottom sides compared to the sea-facing (data not reported). More *Culicia japonica* settled on blocks

comprising 3:1 pulverised fly-ash:cement mix (numbers not reported). In December 1993, a total of 176 smooth-sided cube blocks (0.15 $m^3$) were randomly placed on top of an existing artificial reef 7 m deep. Blocks comprised different ratios of pulverised fly-ash:cement (0:1, 1:3, 1:1, 3:1). Coral recruits were counted approximately every three months for 24 months.

A site comparison study in 1995, and 1998–2001 at two damaged coral reefs in the Florida Keys National Marine Sanctuary, USA (3) found that using concrete armor or limerock boulders to repair the reefs led to natural settlement by corals with 70–80% of species the same as on nearby natural reefs, but the diameter of stony coral *Porites asteroides* colonies was lower, and density did not differ between restored and natural reefs. Six years after the artificial structures were installed, 80% of species recorded on concrete armor and 70% of species on limerock boulders were also found on the adjacent natural reefs. Average colony diameter of *P. asteroides* increased from 14 mm (concrete armor) and 18 mm (limerock boulder) in 1998 to 22 mm (concrete) and 23 mm (limerock) in 2001, but was smaller in 2001 than colonies on the adjacent natural reefs (adjacent to concrete 85 mm; adjacent to limerock: 34 mm). Average density of *P. asteroids* increased on the concrete armor reef from 2.1 colonies/$m^2$ in 1998 to 4.5/$m^2$ in 2001 whereas average density was unchanged on limerock boulders (1.4/$m^2$ both years). Average density was not significantly different between either concrete armor or limerock boulders and their adjacent natural reefs (concrete armor: 4.5, adjacent reef: 5.4 colonies/$m^2$; limerock boulders: 1.4; adjacent reef 0.9 colonies/$m^2$). In 1995, six years after two ships (M/V Maitland and M/V Elpis) ran aground, artificial structures comprising 12 concrete armor blocks (Maitland site) and 16 limerock boulders (Elpis site) were installed to repair the damaged reef. The artificial reefs were monitored to record natural settlement by coral species. Density and diameter of *P. asteroides* were recorded in 1998 and 2001 and compared, in 2001, to *P. asteroides* colonies on natural reefs approximately 25 m away.

A replicated study in 2001–2004 at three artificial reefs in Singapore (4) found that after fibreglass/sand/calcium carbonate structures were used to create habitat, stony coral recruits settled, and at one site at a higher density compared to natural coral rubble substrate, although

recruits were smaller. After 24–26 months, the average density of coral recruits across all sites ranged from 0.1 recruits/$m^2$ to 4.8/$m^2$. At one site after 23–31 months, coral density was higher (range: 6–11 recruits/$m^2$) than the adjacent natural coral rubble (range: 4–10 recruits/$m^2$). Although at that site the average size of recruits on the artificial structures grew between month 26 (1.0–1.5 cm) and 31 (2.0–2.5 cm), these were smaller than recruits on the natural substrate (2.5–3.0 cm for both months). *Pocillopora damicornis* was the dominant species at each site (50%, 79%, 100%) with species from six other families also recorded (see paper for list). In October 2001, ninety-six 70 cm diameter 50 cm tall structures, comprising fibreglass mixed with sand and calcium carbonate, were installed at three sites. Structures were fixed to the seabed using 40 cm or 70 cm stakes. A random sample of 10 structures were monitored every 2–3 months for 24–26 months. In addition, from 23–31 months after installation, coral density and growth on five structures at one of the sites were compared to five 1 $m^2$ plots on adjacent natural coral rubble. **Costs (US$)**: Each substrate structure cost US$130 (in 2006) and US$23 for six 40 cm stakes.

A site comparison study in 1999–2004 at an artificial and natural coral reef site in Bal Harbour, Florida, USA (5) found that corals settled on an artificial reef made from concrete and limerock and, over time, the coral community more closely resembled the adjacent natural reef and stony coral coverage and density increased. The coral community on the artificial reef became more similar to the natural reefs during the first 3.5 years after the artificial reef was installed and then stabilized to a similarity of 45–58% (data presented as a Bray Curtis Index). Average cover of stony coral increased on the artificial reef to 1.35% after five years and was reported as similar to one of the natural reefs (0.70%). Density of stony corals increased from 0.21/$m^2$ in year one to 25.29/$m^2$ after five years. In May 1999, an artificial reef comprising a 46 × 23 m rectangle of 8,000 t of 0.9–1.5 m diameter limerock boulders surrounded by 179 prefabricated concrete and limerock modules (see paper for details). These modules were installed between two natural reefs, 3.1 km offshore, 20 m deep. Reefs were monitored every six months for five years from October 1999 using quadrats to record coral diversity and density.

A study in 2007 on artificial and natural reefs in Florida Keys National Marine Sanctuary, Florida, USA (6) reported that hard coral cover was similar on two older concrete and limestone artificial reefs compared to natural reefs but lower on two newer reefs. Percentage of hard coral cover on 12-year-old artificial reefs was similar to adjacent natural reference reefs (Maitland artificial: 5%, natural: 3%; Elpis artificial: 5%, natural: 4%) but newer reefs had lower hard coral cover than natural reefs (Iselin eight-year-old artificial: 2%, natural: 5%; Wellwood five-year-old artificial: 2%, natural: 8%). Results presented as a similarity index including all species recorded. The hard coral community on the 12-year-old artificial reefs was dominated by *Porites asteroides*. In 2007, four 10 metre long line transect surveys were carried out on four concrete and limestone artificial reefs (two 12-, one eight-, and one five-years-old) and adjacent natural reefs. The percentage of hard coral cover was recorded.

A replicated, randomized, controlled study in 2005–2007 at three degraded coral reefs in northern Sulawesi, Indonesia (7) found that concrete structures placed close to transplanted stony coral fragments had similar numbers of stony coral recruits to structures placed further away. The number of coral recruits was similar on concrete structures placed next to transplanted corals compared to structures placed away from corals in eight of nine comparisons (next to transplants: 0.02–0.28 corals/100 $cm^2$, away from transplants: 0.03–0.26 corals/100 $cm^2$), and higher in the ninth comparison (next to transplants: 0.58 corals/100 $cm^2$, away from transplants: 0.36 corals/100 $cm^2$). For limestone plates placed next to, or distant from, transplanted corals there were a similar number of recruits in 15 of 18 comparisons, more recruits in two comparisons, and fewer in one comparison (see paper for data). In July 2005–March 2006, six-thousand-one-hundred-and-sixty-four stony coral fragments (*Acropora yongei, Pocillopora verrucosa, Acropora muricata, Isopora brueggemanni*) were collected from donor colonies near three transplant sites. Two plots (10 × 10 m) at each of three sites, with each plot randomly assigned to either: concrete structures (25/plot) alternating in a 'chessboard' design with transplanted stony coral fragments attached to bamboo frames; or concrete structures only (25/plot). At all plots, six groups of three limestone settlement plates were also installed on metal frames. Coral recruits that settled on concrete structures were

counted after 14–24 months. Recruits on limestone plates were counted every three months for 14–24 months. Plates were replaced every three months.

A site comparison study in 1997 and 2009–2010 at a fish farm and adjacent coral reefs in Setouchi Channel, Japan (8), found that corals that settled and began growing on suspended ropes had lower rates of bleaching but higher instances of infection than corals on natural reefs, and the community differed between the ropes and natural reefs. Three months after monitoring began, the percentage partial bleaching on rope-growing corals was lower (12%) than on corals growing on one of the disturbed reefs (46%), but similar to corals growing on the other disturbed (18%) and protected (12%) reefs. Rates of infection by flatworm *Waminoa* spp. were higher after nine months in rope-growing corals (4%) compared to corals growing on disturbed (0%, 1%) and protected (0%) reefs. Diversity of coral communities on the ropes was significantly higher than communities on the two disturbed sites, and either equaled or was higher than on the protected site (results presented as multivariate analyses, see paper for full species list). Coral responses to other threats (e.g. algae and sponge overgrowth) were not significantly different between rope-growing or naturally growing corals. In 1997, a tuna fish farm was established using floating cages suspended by rope 3 m deep, ~50 m above the seabed. In May and August 2009 and February 2010, surveys were carried out on the ropes and three adjacent coral reefs (two disturbed by outbreaks of crown-of-thorns starfish; one protected through management of crown-of-thorns starfish). Photographs were used to monitor diversity, bleaching, infection, and other threats.

A replicated, site comparison study in 2004 and 2014 at seven artificial reefs off Singapore (9) found that corals settled on fibreglass reefs, and the percentage of organisms that were stony corals increased over 10 years. Stony corals represented on average <1% of organisms on artificial reefs in 2004 and 2–42% (11% average) 10 years later. In 2014, stony coral colonies on average covered <1–32% of artificial reef surfaces and at three of seven sites 25–58% of corals were recorded with eggs (no eggs were recorded at the remaining sites). In the early 2000s, fibreglass artificial reefs were fixed with iron stakes to areas of sand and rubble at seven sites off Singapore's southern offshore islands. The communities on the outer surfaces of all 84 artificial reefs were

surveyed in 2004 and the 44 that remained in 2014. Thirty-five were surveyed in both years. In 2014, three fragments were taken from every adult coral colony ≥12 cm to look for eggs (to determine if the corals were reproductive).

A replicated, controlled study in 2004–2009 at a reef in the South Atlantic Bight, Georgia, USA (10) found that using concrete paving slabs led to higher recruitment of temperate stony coral *Oculina arbuscula* but a higher mortality rate than the natural reef substrate. After almost five years, the average number of coral recruits was higher on concrete paving slabs (17/plot) than on the natural reef (2/plot). The maximum number recorded during one survey was 85 (concrete) and 3 (natural)/plot. Mortality (deaths/plot) was higher at the end of the study for recruits on the concrete paving slabs (5) than on the natural reef (0.25). In June 2004, twenty 30 × 30 cm plots were marked on a hard-bottom reef comprising relict scallop shells on rocky substrate, 20 m deep. Concrete paving slabs (30 × 30 × 5 cm) were placed, unsecured, into 10 plot areas. The remaining plots were left as natural substrate. Twenty surveys were carried out periodically from June 2004–June 2009 to record coral recruitment and mortality using photographs.

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## 12.4 Use settlement tiles made from unnatural materials to encourage natural coral settlement

https://www.conservationevidence.com/actions/3990

- **Sixteen studies** examined the use of settlement tiles to encourage natural coral settlement. Three studies were in Australia[1a,b,11], two in each of the Philippines[2,8c], Israel[6a,b], and the United Arab Emirates[7,12], and one in each of Japan[3], Italy[4], Italy and Spain[5], the US Virgin Islands[8a], Taiwan[8b], Belize[9], and Palau[10].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (16 STUDIES)**

- **Abundance/Cover (16 studies)**: Sixteen replicated studies (including two randomized, one controlled, one site comparison and one paired) in Australia[1a,b,11], the Phillipines[2,8c], Japan[3], Italy[4], Italy and Spain[5], Israel[6a,b], the United Arab Emirates[7,12], the US Virgin Islands[8a], Taiwan[8b], Belize[9], and Palau[10], found that coral naturally settled on settlement tiles. Four of the studies[2,6a,7,11] found that the number of corals settling depended on settlement tile material. Two studies[2,3] found that coral settlement numbers were higher on tiles within a coral reef[2] or near existing adult colonies[3]. Two studies[11,12] found that coral settlement tended to be higher on the underside of settlement tiles, whereas three studies[8a-c] found that more corals settled on the upper tile surface with refuge holes than without.
- **Survival (2 studies)**: One replicated study[4] found that average survival was similar on tiles at different depths. One replicated, site-comparison study[5] found that survival one year after settlement varied on the site.
- **Condition (1 study)**: A replicated study in Italy[4] found settled coral growth and the number of new polyps increased with age.

**Background**

The use of settlement tiles comprising materials not typically encountered by corals such as marble, concrete, terracotta, acrylic, or PVC are used to encourage natural settlement by coral larvae. Tiles with settled corals can be removed to be cultivated in ex-situ or in-situ nurseries or left on site to create additional habitat. However, there is also the potential for negative consequences within the coral ecosystem through pollution or contamination caused by the degradation of unnatural reef materials (such as concrete or PVC) (McManus *et al*. 2018) or the material itself being less suitable than a natural reef thus reducing settlement by larvae (Natanzi *et al*. 2021).

Here we focus on the use of settlement tiles to encourage natural settlement by wild coral. Other similar actions include *Use natural materials to restore/repair/create habitat for corals to encourage natural coral settlement; Use structures made from unnatural materials to restore/repair/create habitat for corals to encourage natural coral settlement; Repurpose obsolete offshore structures to act as structures for restoring coral reefs* (where a man-made structure is no longer being used for its original purpose and has been repurposed as an artificial reef); and *Modify existing man-made structures to create artificial reefs* (where a structure was created for another purpose but has been modified to allow colonization by coral or has been colonized in its original state). Actions relating to cultivating or transplanting corals onto artificial substrates are covered in *Cultivate coral fragments in an artificial nursery located in a natural habitat; Cultivate coral larvae in an artificial nursery located in a natural habitat; Transplant nursery-grown corals onto artificial substrate*; and *Transplant wild-grown corals onto artificial substrate*.

McManus R.S., Archibald N., Comber S., Knights A.M., Thompson R.C. & Firth L.B. (2018) Partial replacement of cement for waste aggregates in concrete coastal and marine infrastructure: a foundation for ecological enhancement? *Ecological Engineering*, 120, 655–667. https://doi.org/10.1016/j.ecoleng.2017.06.062

Natanzi A.S., Thompson B.J., Brooks P.R., Crowe T.P. & McNally C. (2021) Influence of concrete properties on the initial biological colonization of marine artificial structures. *Ecological Engineering*, 159,106104. https://doi.org/10.1016/j.ecoleng.2020.106104

A replicated study in 1994–1995 at two reef sites at Heron Reef, Great Barrier Reef, Australia (1a) found that attaching artificial settlement tiles directly to the substrate did not result in higher natural coral recruitment than tiles attached in pairs or singly to wire mesh racks. Five months after tiles were installed, there was no significant difference between the number of coral recruits on tiles attached to the substrate (Site 1: 1.4, Site 2: 1.2/100 $cm^2$), pairs of tiles on wire racks (Site 1: 1.3, Site 2: 1.4/100 $cm^2$) or single tiles on racks (Site 1: 1.3, Site 2: 1.5/100 $cm^2$). In September 1994, forty unglazed terracotta settlement tiles (110 × 110 × 10 mm) with numerous pits and grooves (<1 × 1 mm) were taken to each of two reef sites 500 m apart. Tiles (10/site) were screwed to a stainless-steel baseplate (100 × 50 × 0.6 mm) and attached to the substrate 1–2 m apart using two screws. Wire mesh, A-frame racks (five/site) were anchored to the substrate 2–3 m apart, 9 m deep using steel pegs. One pair of tiles (one on top of the other) and one single tile were screwed to each side of the A-frame (30 tiles/site). Plates and racks were retrieved in February 1995 and the number of coral recruits was counted using a microscope.

A replicated study in 1994–1995 at two reef sites at Heron Reef, Great Barrier Reef, Australia (1b) reported that attaching artificial settlement tiles to the substrate on different natural substrate features, depths, and at different angles led to natural coral settlement. Five months after tiles were installed, the number of acroporid coral recruits ranged from 0–4/tile at both sites, and pocilloporid coral recruits ranged from 0–16/tile at Site 1 and 0–11/tile at Site 2 (data reported as statistical model results). In September 1994, unglazed terracotta settlement tiles (110 × 110 × 10 mm) with numerous pits and grooves (<1 × 1 mm) were attached to stainless steel base plates and screwed to the substrate at two reef sites (site 1: 228 tiles; site 2: 206 tiles). Tiles were attached on different topographic features categorized as 'level' (flat substrate);

'protected' (located in a depression >5 cm below surrounding substrate); 'raised' (on mound >10 cm above substrate); 'stepped' (located on a series of ledges). Tiles were placed at different angles (0–90° and depths (2.5–8.7 m). Tiles were collected after five months, and the number of coral recruits was counted and identified.

A randomized, replicated study in 1998 on sandy substrate at a coral reef at Danjugan Island, Sulu Sea, central Philippines (2), found that a higher number of stony coral larvae settled on tiles made from consolidated coral rubble or concrete than rubber, and more coral larvae settled on tiles placed within compared to outside an existing reef. After 4.5 months, the average number of stony coral larvae/tile was higher on coral rubble (within reef: 7.7; outside reef: 2.9) and concrete (within reef: 6.9; outside reef: 2.3) than rubber (within reef: 0.45; outside reef: 0.35) tiles and higher on tiles within the existing reef than outside. Almost all settled larvae were from two families (Pocilloporids: 87% within, 88% outside; Acroporids: 11% within, 12% outside). In February 1998, forty-eight 10 × 10 cm tiles comprising 16 each of coral-rubble-cement, concrete, and rubber were randomly arranged on 16 frames (one of each type/frame) and attached using wire ties. Eight frames were placed within an existing coral reef <0.25 m from live coral, and eight placed outside the reef area >5 m from live coral. Frames were placed 12 m deep, 30 cm above the sandy seabed. Frames were retrieved after 4.5 months and larvae were counted and identified under a microscope.

A replicated, site comparison study in 1997–1999 at two coral reef sites in Amakusa, Japan (3) found that placing artificial settlement tiles adjacent to adult stony coral *Pocillopora damicornis* colonies led to higher recruitment than tiles placed 8–10 m away. Three months after larvae were released by the adult colonies, 70 recruits had settled on tiles in July–October 1997 and 65 in July–October 1998 but no recruits settled NovemberJune in 1998 or 1999. The study reports that there were significantly more recruits on tiles placed adjacent to adult *P. damicornis* colonies than on tiles placed 8–10 m away but numbers are not reported. In July 1997, fifteen concrete blocks (40 × 20 × 10 cm) were placed on the substrate, adjacent (5–10 cm) to existing *Pocillopora damicornis* colonies, and a further 15 blocks were placed 8–10 m away from the nearest colony. Six ceramic settlement tiles (10 × 10 × 2 cm) were bolted to each concrete block. Tiles were retrieved after three months and new plates were attached and retrieved in June 1998. The process was

repeated from July 1998–June 1999. *P. damicornis* recruits were identified and counted under a microscope.

A replicated study in 1998–2002 on rocky substrate in Leghorn, Italy (4) found that marble settlement tiles were settled on by Mediterranean red coral *Corallium rubrum* larvae and some survived and grew, with survival similar between depths. Overall, 388 new coral colonies settled on tiles during the four-year study (244 on tiles 25 m deep and 144 at 35 m). After four years, coral density was 19 (at 25 m) and 10 (at 35 m) settlers/10 $cm^2$. Average annual survival of cohorts (survival rate between two consecutive years) was similar across the study period and between depths (76% at 25 m; 75% at 35 m). After four years, 34% (25 m) and 31% (35 m) of the first cohorts (settled in 1998) had survived. Average diameter increased with coral age (1 year old: 0.6; 4 years old: 2.5 mm), height also increased with age (2 years old: 2 mm; 4 years old: 7 mm). The average number of polyps was significantly higher for four-year-old corals (38) than two (9) and one (5) year old. In June 1998 (approximately three weeks before red coral spawning), 20 white marble tiles (90 × 120 mm) were fixed with a steel screw into crevices at 25 m and 35 m depth (10/depth). Tiles were monitored every three months from October 1998–October 2002 when they were removed and red coral settlers counted and measured.

A replicated, site comparison study in 2003–2004 at two sites in Italy and one in Spain (5) found that using marble settlement tiles resulted in recruitment of red coral *Corallium rubrum* with settlement rates, recruitment density and mortality rates varying depending on site. Four months after tiles were installed, there was no significant difference in overall settlement rate between sites (Calafuria: 67%; Elba: 50%; Medes: 50%). Average settler recruitment density varied between sites (Calafuria: 2.8; Elba: 1.1; Medes 1.6 recruits/$cm^2$). One year after installation, average mortality rates varied between sites with 72% (21/29) mortality at Mendes, 14% (7/50) mortality at Calafuria, and 10% (2/20) mortality at Elba. In June 2003, fifty-four marble tiles (9 × 12 cm) were secured using a single central screw to rocky crevices on vertical cliffs 25–35 m deep. Nine tiles were placed at each of two locations in three sites in the Mediterranean (Calafuria and Elba, Italy; Mendes, Spain). Settlement by red coral recruits was photographed and analysed after four months (October 2003) and mortality rate measured

after a year (June 2004).

A replicated study in 1999–2001 at a shallow reef in Eilat, Israel (6a) found that using unglazed ceramic settlement tiles resulted in a higher number of naturally settled hard coral spat (settled larvae) compared to brick tiles but only during the third survey period and no difference in the number of naturally settled soft coral spat. Four months after the third deployment of tiles, there were 255 hard coral and 153 soft coral spat on 66 tiles. Numbers of naturally settled hard coral spat were higher on ceramic tiles (4–10/100 $cm^2$) compared to brick tiles (3–4/100 $cm^2$). There was no difference for soft coral spat (ceramic: 1–2/100 $cm^2$; brick: 1–2/100 $cm^2$). There were 34 hard and 81 soft coral spat recorded four months after the second deployment of tiles but no difference between ceramic or brick tiles. No coral spat was recorded during the first survey period. In November 1999, June 2000, and March 2001, nine unglazed ceramic (100 × 100 × 5 mm) and nine fired brick (115 × 115 × 25 mm) settlement tiles were fixed to the substrate using masonry plugs, and nine of each type attached to one of three wire racks. Tiles were placed 10–20 mm (masonry plug) or 200–400 mm (wire rack), above the substrate, 5 m deep. Tiles were recovered and replaced four months after each deployment. Coral spat were counted and species groups recorded using a dissecting microscope.

A replicated study in 1999–2001 at a shallow reef in Eilat, Israel (6b) found that settlement tiles attached to wire racks had a higher number of naturally settled hard coral spat (settled larvae) compared to tiles attached to the substrate but only during the third survey period and no difference in the number of naturally settled soft coral spat. Four months after the third deployment of tiles, there were 255 hard coral and 153 soft coral spat on 66 tiles. Numbers of naturally settled coral spat were higher on tiles attached to a wire rack (4–10/100 $cm^2$) compared to tiles attached directly to the substrate (3–4/100 $cm^2$). There was no difference for soft coral spat (wire rack: 1–2; substrate: 1–2/100 $cm^2$). There were 34 hard and 81 soft coral spat recorded four months after the second deployment of tiles but no difference between tiles on the rack or the substrate. No coral spat was recorded during the first survey period. In November 1999, June 2000, and March 2001, eighteen settlement tiles (nine 100 × 100 × 5 mm unglazed ceramic; nine 115 × 115 × 25 mm fired brick) were attached using cable ties to one of three wire racks fixed 200–400 mm above the substrate at a

45° angle, 5 m deep. Eighteen tiles were attached to the substrate 5 m deep using masonry plugs leaving a gap of 10–20 mm. Tiles were recovered and replaced after four months. Coral spat were counted and species groups recorded using a dissecting microscope.

A replicated, randomized, controlled study in 2007–2008 on two artificial reefs and two rocky reefs off Dubai, United Arab Emirates (7) found that sandstone, terracotta, granite, gabbro, and concrete settlement tiles had similar densities of settled corals at three of four sites. At one of four sites, juvenile corals were more abundant on gabbro (8 corals/100 $cm^2$) than sandstone (3 corals/100 $cm^2$) and concrete (3 corals/100 $cm^2$) tiles, and more abundant on terracotta (7 corals/100 $cm^2$) than sandstone, with other comparisons showing no differences (granite: 5 corals/100 $cm^2$). At the other sites, few corals were recorded with no significant differences between materials (<1 coral/100 $cm^2$ at all). Settlement tiles (100 × 100 × 15 mm) were made from sandstone, terracotta, granite, gabbro and concrete. Twenty-five of each were randomly arranged horizontally 10–15 mm above the substrate at 4 m depth on each of two breakwaters and two rocky reefs in April 2007 (before May–October spawning season). After 12 months, tiles were brought to the laboratory, immersed in bleach for 24 h to remove organic matter, and juvenile corals on the bottom of each tile were counted. Twenty-five tiles went missing during the experiment.

A replicated study in 2010–2012 on five fringing reefs off St John, US Virgin Islands (8a) found that the upper surfaces of unglazed terracotta or acrylic settlement tiles were colonized by stony corals when they had refuge holes, but not when they were smooth. No corals settled on upper surfaces of tiles without refuge holes during the study. On tiles deployed August 2010–June 2011 coral density did not differ between upper surfaces with refuge holes (0.97 corals/100 $cm^2$) and lower surfaces (1.08 corals/100 $cm^2$), but on tiles deployed June 2011–August 2012 there was lower density on upper surfaces with refuge holes (0.14 corals/100 $cm^2$) than lower surfaces (1.31 corals/100 $cm^2$). See paper for preferences of different coral species. At five sites off St John (<500 m apart), a cluster of 15 unglazed terracotta or acrylic settlement tiles was attached at 45° to horizontal at 5 m depth, 1 cm above the substrate, using stainless steel studs and a spacer which were attached to rocks with epoxy putty. Tiles were deployed August 2010–June 2011, then replaced and left until August 2012. When retrieved, tiles were

cleaned, dried and inspected with a microscope for corals. For each sampling period, authors inspected the lower surface of seventy-five terracotta tiles (15 × 15 × 1 cm) and the upper surface of 20 terracotta tiles topped with acrylic tiles (15 × 15 × 0.6 cm) which had been drilled with holes on the top surface only, and 20 undrilled acrylic-only tiles.

A replicated study in 2010–2012 on three reefs off Lyudao, Lanyu and Kenting islands, Taiwan (8b) found that the upper surfaces of unglazed terracotta or acrylic settlement tiles with refuge holes were colonized by a higher density of stony corals than upper tile surfaces with no holes. Four weeks after deployment, upper surfaces of tiles with refuge holes had a higher density of settled corals (1.6–7.9 corals/100 $cm^2$) than upper surfaces without holes (0.3–1.9 corals/100 $cm^2$) and lower surfaces (0.3–4.7 corals/100 $cm^2$, data is not separated for lower surfaces with or without refuge holes). See paper for preferences of different coral species. Pairs of unglazed terracotta or acrylic tiles (10 × 10 × 1 cm) with a smooth and a grooved surface were stuck together, either with both grooved surfaces facing outwards (refuges) or both smooth surfaces facing outwards (smooth). Off three islands (70–105 km apart), 15–18 pairs of refuge and smooth tiles were fixed a few cm above the substrate at 45° to horizontal using stainless steel bolts at a depth of 5 m. Tile pairs were deployed in March–April (2–3 weeks before coral spawning), off Lyudao in 2010 and off Lyudao, Lanyu and Kenting in 2012, retrieved four weeks later, cleaned, dried and inspected with a microscope for corals.

A replicated study in 2010–2012 on three reefs off Caniogan, Cangaluyan and Lucero islands, Philippines (8c) found that upper surfaces of fibre-cement settlement tiles with refuge holes were colonized by a higher density of stony corals than upper surfaces without holes. Five months after deployment, upper tile surfaces with refuge holes had a higher density of settled corals (1.9–11.4 corals/100 $cm^2$) than smooth upper surfaces (0–1.7 corals/100 $cm^2$) and lower surfaces (0.4–2.8 corals/100 $cm^2$, data is not separated for lower surfaces with or without refuge holes). See paper for settlement surfaces of different coral species. Fifteen fibre-cement tiles (10 × 10 × 1.2 cm) with refuges (drilled with sixty-four 0.5 cm radius holes on each side) and 15 without refuges (smooth) were fixed 1 cm above the substrate at 45° to horizontal using concrete nails at a depth of 5 m on fore-reefs at Caniogan, Cangaluyan and Lucero (11–24 km apart). Refuge and smooth tiles were installed 30 cm apart in February 2012 and

retrieved in July 2012. Peak coral spawning was March–May. Retrieved tiles were cleaned, dried and inspected with a microscope for corals.

A replicated, controlled study in 2007–2008 at two coral reefs at Glovers Reef and Carrie Bow Cay, Belize (9), found that using exclusion devices on settlement tiles to deter herbivorous parrot fish led to a reduction in settlement by coral spat (settled larvae) and an increase in nuisance algae compared to tiles without devices. One year after exclusion devices were installed, the number of coral spat was lower on tiles with exclusion devices (0.3–0.6/tile) compared to tiles with just frames (1.3–1.5/tile) and bare tiles (0.9–1.7/tile). Coverage by nuisance macroalgae was also higher on tiles inside exclusion devices (38–68%) compared to tiles with wire (22–33%) and bare tiles (24–30%). Coral species were mainly *Agaricia* spp. and *Porites* spp. although there were no *Porites* spp. settled on any of the exclusion tiles. In March 2007, parrot-fish exclusion devices were placed around 24 terracotta settlement tiles (10 × 10 × 1 cm). Devices comprised a 20 cm diameter wire star-shaped frame with 15.2 cm vertical stainless-steel bolts attached at 4 cm intervals to resemble a 'cage'. Frames only were attached to 24 tiles and a further 24 were left bare. Twenty-four groups of three tiles (one/treatment) were screwed to the substrate at each of Glovers Reef and Carrie Bow Cay. Coral settlement and algal growth were recorded after one year.

A replicated, paired study in 2008 at Iou Lukes reef, Palau (10), found that settlement tiles allowed to 'biologically condition' for three months had a higher density of artificially enhanced or naturally settled stony coral spat (settled larvae) compared to tiles conditioned for one week, and density was higher on tiles with artificially enhanced coral larvae supply. One week or five weeks after nearby wild-growing stony coral spawned or larvae were artificially introduced to the tiles, density of coral spat was higher on tiles conditioned for three months (natural: 50; artificial: 205/0.1 $m^2$) compared to tiles conditioned for one week (natural: 4; artificial 29/0.1 $m^2$). Density was significantly higher on one-week conditioned and three-month conditioned tiles where larvae supply had been enhanced compared to the natural tiles. In January 2008 and April 2008, four fibre-cement settlement tiles (10 × 10 × 0.6 cm) were attached to each of 28 concrete/limestone 'pallet-balls' (1.2 × 0.9 m) placed 3–5 m apart, 5–8 m deep on the seafloor adjacent to a natural reef. Tiles were allowed to 'condition' (develop biofilm) for three months (January 2008) or one week (April 2008) before coral spawning. In April 2008, seven randomly selected pallet-balls were 'seeded' with nursery-

cultivated stony coral *Acropora digitata* larvae (see paper for methods), and corals on the natural reef spawned. Tiles were retrieved either one or five weeks after wild-growing coral colonies had spawned and the number of coral spat was counted.

A replicated study in 2012–2015 at coral reef patches ('microatolls') off One Tree Island, Great Barrier Reef, Australia (11) found that PVC pipes and the top of ceramic settlement tiles were colonized by a lower number of small stony coral recruits than the underside of ceramic tiles but there was no difference for larger coral colonies or overall coral cover. After 34 months, no coral recruits (<1 cm) were attached to PVC pipes or the top of ceramic tiles, compared to an average of 0.2 (range 0–2) on the underside of ceramic tiles. There was no difference in the average number of coral colonies (>1 cm) attached to PVC pipes (0.7, range 0–8) or the underside (0.6, range 0–7) or topside (0.2, range 0–3) of ceramic tiles. There was no difference in total coral cover (recruits and colonies) between settlement materials (data presented as a figure). In May 2012, thirty PVC pipes and 61 unglazed ceramic tiles were each fixed, horizontally, to a PVC frame attached to the substrate using cable ties. Ceramic tiles were placed in pairs with one tile facing upwards (30 tiles) and one facing down (31 tiles). PVC frames were placed randomly within three microtolls at 1–2 m deep. Corals were counted and measured in March 2015.

A replicated study in 2019 at a reef at Sir Abu Nu'Ayr Island off the United Arab Emirates (12) found that terracotta settlement tiles were naturally settled by stony corals (including *Acropora* spp. and *Porites* spp.). An average of three corals settled/tile, and all but two recruits settled on the grooved underside of the tiles. *Acropora* spp. made up 30% of settled corals, and *Porites* spp. made up 10%. In April 2019, thirty-one terracotta tiles (10 × 10 × 1 cm) were attached to the reef substrate (5 m deep, 2 m apart) using a screw and epoxy, with the grooved surface facing down. In September 2019, tiles were collected, and the number of recruits were counted, and species were identified.

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## 12.5. Repurpose obsolete offshore structures to act as structures for restoring coral reefs

https://www.conservationevidence.com/actions/3991

- **Two studies** evaluated the effects of repurposing obsolete offshore structures to restore coral reefs. One study was in Japan[1] and one in the Gulf of Mexico[2].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (2 STUDIES)**

- **Abundance/Cover (2 studies)**: One study in Japan[1] found that concrete aquaculture boxes had higher coral cover than the surrounding reef. One replicated, site comparison study in the Gulf of Mexico[2] found that toppled oil rig platforms had similar overall stony coral density to rigs left standing, but density of species varied between rigs.

**Background**

Man-made offshore structures, such as oil rigs and aquaculture boxes, provide hard surfaces that may allow coral larvae to settle in areas where there is otherwise a lack of suitable substrate. Once these structures are no longer used for their intended commercial purpose they can be removed from the marine environment or be made into artificial reefs. If being repurposed for biodiversity, they can either be left in the same location (standing or toppled) or can be moved to a new location to increase the likelihood of natural colonization by corals and fish communities. Programmes that encourage the repurposing of obsolete structures as artificial reefs, such as 'Rigs-to-Reefs' in the Gulf of Mexico, operate under the premise that the structures will provide benefits for nature by providing new habitats and benefits for businesses by reducing the costs for decommissioning and removing obsolete equipment (Macreadie *et al*. 2011).

Other similar actions include *Use structures made from unnatural materials to create habitat to encourage coral settlement* (where a structure has specifically been made as a reef) and *Modify existing man-made structures to create habitat to encourage coral settlement* (where a structure was created for another purpose but has been modified to allow colonization by corals or has been colonized in its original state).

Macreadie P.I., Fowler A.M. & Booth D.J. (2011) Rigs-to-reefs: Will the deep sea benefit from artificial habitat? *Frontiers in Ecology and the Environment*, 9, 455–461. https://doi.org/10.1890/100112

A replicated study in 1996–2003 at an aquaculture site at Miako Island, Okinawa, Japan (1) reported that coral cover was higher inside empty aquaculture boxes compared to the surrounding reef. After seven years, coral cover inside five boxes originally designed to be used for rearing top-shell snails *Trochus niloticus* was 90% compared to 20% on the surrounding reef (data not statistically tested). By 2003, twenty-six species had colonized the base of the boxes; the dominant species being *Acropora* spp. which had grown to 40–65 cm in diameter. In 1996, five concrete aquaculture boxes (2.1 × 2.1 × 0.6 m) in shallow water (depth not specified) were left empty to enable coral to grow on the base. The box bases were made from plastic lattice reinforced with quartz sand-coated fibreglass to which the corals could attach. Monitoring frequency and other methods are not reported.

A replicated, site comparison study (years not given) on seven decommissioned oil rig platforms in the Gulf of Mexico (2) found that toppled platforms did not have greater overall density of stony corals than standing platforms, but densities of three of four stony coral species varied between toppled and standing platforms. There was no significant difference between the average density of all corals on toppled oil platforms (90 corals/10 $m^2$) and standing platforms (20 corals/10 $m^2$). However, on average, *Madracis decactis* and *Tubastraea coccinea* densities were higher on toppled (*Madracis decactis*: 0.4 corals/10 $m^2$; *Tubastraea coccinea*: 28 corals/10 $m^2$) than standing platforms (*Madracis decactis*: 0.3 corals/10 $m^2$; *Tubastraea coccinea*: 19 corals/10 $m^2$). In contrast, *Phyllangia*

*americana* density was lower on toppled (1 coral/10 $m^2$) than standing platforms (4 corals/10 $m^2$). There was no difference in *Oculina diffusa* density between toppled (2 corals/10 $m^2$) and standing platforms (2 corals/10 $m^2$). Surveys for stony corals were carried out on two standing oil platforms deployed 15–30 years prior (sea level to maximum depth of 101 m and 113 m) and five obsolete oil platforms cut at the base and toppled 13–20 years prior (minimum depth: 23–30 m; maximum: 48–195 m). Monitoring was carried out using photos and videos taken by remotely operated vehicles along two to four vertical and two horizontal struts/platform (20 m to a maximum of 110 m deep).

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(2) Sammarco P.W., Lirette A., Tung Y.F., Boland G.S., Genazzio M. & Sinclair J. (2014) Coral communities on artificial reefs in the Gulf of Mexico: Standing vs. toppled oil platforms. *ICES Journal of Marine Science*, 71, 417–426. https://doi.org/10.1093/icesjms/fst140

## 12.6 Modify existing man-made structures to encourage natural coral settlement

https://www.conservationevidence.com/actions/3992

- We found no studies that evaluated the effects on corals of modifying existing man-made structures to encourage natural coral settlement.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Modifying existing man-made structures, such as sea walls, breakwaters, and renewable energy structures (e.g. wind turbines), to create substrate that encourages natural coral settlement can offer an option for establishing new coral reefs. Coral larvae can settle on many man-made substrates, including concrete (Burt *et al*. 2009), so existing structures can provide an ideal surface on which to settle and grow. These structures can be modified to encourage settlement and growth by, for example, creating crevices or drilling holes in the surface of the structure.

This action covers structures that continue to be used for their original or intended purpose but are modified in some way to encourage coral larvae to settle. Other similar actions include *Use structures made from unnatural materials to restore/repair/create habitat for corals to encourage natural coral settlement* (where a structure has specifically been made as a reef), *Repurpose obsolete offshore structures to act as structures for restoring coral reefs* (where a man-made structure is no longer being used for its original purpose and has been repurposed as an artificial reef). This action covers natural settlement by coral onto existing structures that are modified. Other studies investigating cultivating and transplanting coral onto existing structures are covered *Cultivate coral fragments in an artificial nursery located in a natural habitat; Cultivate coral larvae in an artificial nursery located in a natural habitat; Transplant nursery-grown coral fragments onto artificial substrate*; and *Transplant wild-grown coral onto artificial substrate*.

Burt J., Bartholomew A., Usseglio P., Bauman A. & Sale P.F. (2009) Are artificial reefs surrogates of natural habitats for corals and fish in Dubai, United Arab Emirates? *Coral Reefs*, 28, 663–675. https://doi.org/10.1007/s00338-009-0500-1

# 13. Species management

**Background**

Most of the chapters in this book are aimed at minimizing threats, but there are also some actions which aim specifically to increase the diversity of coral communities, the size and extent of populations, or to increase certain demographic rates or condition of individuals. These actions may be used in response to a wide range of threats. This chapter includes the cultivation and transplant of corals, as well as some actions designed to increase rates of coral settlement and growth. Many of these actions – particularly coral transplanting and larval enhancement – are also considered as options for coral restoration (Boström-Einarsson *et al*. 2020), though are included in this chapter as they involve directly managing coral species. For studies describing attempts to restore corals through restoring, repairing or creating habitats for natural coral settlement, see *Habitat restoration and protection*, and for those describing attempts to restore habitats indirectly through the designation of legal or other protections, see *Habitat protection*.

Boström-Einarsson L., Babcock R.C., Bayraktarov E., Ceccarelli D., Cook N., Ferse S.C., Hancock B., Harrison P., Hein M., Shaver E., Smith A., Suggett D., Stewart-Sinclair P.J., Vardi T. & Mcleod I.M. (2020) Coral restoration – A systematic review of current methods, successes, failures and future directions. *PloS One, 15*, e0226631. https://doi.org/10.1371/journal.pone.0226631

 https://doi.org/10.11647/OBP.0453.13

# Cultivate coral

## 13.1. Cultivate coral fragments in an artificial nursery located in a natural habitat

https://www.conservationevidence.com/actions/3993

- **Twenty-seven studies** evaluated the effects of cultivating coral fragments in an artificial nursery located in a natural habitat. Eight studies were in Taiwan[5a-h], six in Puerto Rico[2a-d,3,12], five in each of the USA[8,10,11,13,15], and the Philippines[1,4,6,9a,b], and one in each of Israel[7], the Dominican Republic[14], and the British Virgin Islands[16].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (27 STUDIES)**

- **Reproductive success (1 study)**: One study in the Dominican Republic[14] found that most colonies in an artificial nursery in a natural habitat spawned.
- **Survival (14 studies)**: Fourteen studies (eleven replicated including one randomized), in the Philippines[1,4,6,9a], Puerto Rico[2b,c,d,3], Taiwan[5h], Israel[7], the USA[11,13,15] and the British Virgin Islands[16], found that some of every coral species cultivated in an artificial nursery located in a natural habitat survived. Two of the studies[2b,c], found medium and large fragments had lower mortality than small[2b], and younger fragments had lower mortality than older[2c]. One study[5h] found that cultivated small fragments were more likely to survive if algae were cleared from their racks and another study[15] found higher survival and lower partial mortality for fragments cultivated horizontally rather than vertically. One study[1] found higher survival for fragments cultivated at 10 m, than 1 m depth[1]. One study found that survival of fragments attached to giant clam shells varied by species and fragment size[9a]. One study found that survival was higher on wire frames or coral rubble than sand[2d], whereas another found no difference in survival of fragments cultivated on frames painted with anti-fouling

paint or unpainted[7]. One study found no difference in survival for suspended or fixed fragments[6], but another found lower survival for suspended fragments than fragments attached to concrete blocks[11].

- **Condition (22 studies)** Twenty of twenty-two studies (twenty replicated including, one randomized, controlled, one randomized, one paired, and one controlled) in the Phillipines[1,4,6,9a,b], Puerto Rico[2a,d], Taiwan[5a-h], the USA[8,10,11,13,15], the USA and Puerto Rico[12], and the British Virgin Islands[16] found that on average all coral fragments cultivated in an artificial nursery located in a natural habitat grew[1,2a,d,4,5a-h,6,8,10,11,12,13,15,16]. Three of the studies[1,5b,8] found larger cultivated fragments had greater overall growth than smaller fragments, but one study[5c] found that fragments cut in half had greater growth than intact. Fragments had greater growth[2a,d,5d,e,13,15] and grew new branches[2a,5d,e], when cultivated on wire frames[2a], above the substrate[2d], at 5 m rather than 10 m deep[5d], when pointing upwards rather than downwards[5e], on arrays rather than concrete blocks[13], and horizontally rather than vertically[15] Two studies[11,13] found that suspended fragments had greater growth than on blocks[,11,13], had later onset of bleaching and fewer breakages[11], but no difference in weight[11], whereas one study[6], found that suspended and fixed fragments both grew. One study[5h] found that clearing algae from nursery racks made no difference to fragment growth. Two replicated studies in the Philippines[9a,b] found that attachment time for fragments cultivated on giant clam shells varied by species[9a,b] and fragment size[9a].

**Background**

Coral fragments can be cultivated in artificial nurseries that are located within the natural reef habitat (a process known as 'coral gardening') (Meesters *et al*. 2013). Cultivation methods involve taking coral fragments from an existing nursery or wild colony. Fragments are attached to or suspended from temporary structures, such as PVC frames, that act as artificial nurseries (Meesters *et al*. 2013). The aim is to grow the coral colonies and allow them to develop. Once grown, they can be removed from the temporary nursery and transplanted into the wild (also known as outplanting), onto natural substrate, such as degraded coral reefs or the sea floor, or onto more permanent artificial structures. Cultivating corals using these methods allows more direct action and manipulation to encourage the corals to grow.

This action specifically refers to cultivating coral fragments in an artificial nursery in a natural habitat. Studies that report the effect of cultivating coral larvae or spat (settled larvae) in an in-situ nursery are described in *Cultivate coral larvae in an artificial nursery located in a natural habitat*. Studies that report the effect of cultivating corals in an ex-situ nursery are described in *Cultivate corals in an ex-situ nursery*. Studies that report the effect of transplanting corals are described in *Transplant nursery-grown corals onto natural substrate; Transplant nursery-grown corals onto artificial substrate, Transplant wild-grown corals onto natural substrate*; and *Transplant wild-grown corals onto artificial substrate*.

Meesters E.H.W.G., Smith S.R. & Becking L.E. (2013) *A review of coral reef restoration techniques*. Report number C028/14. IMARES: Wageningen UR. Available from: https://library.wur.nl/WebQuery/wurpubs/fulltext/333153

A controlled, before-and-after study in 1994–1995 in an artificial nursery on a coral reef in Pangasinan, Philippines (1) found that cultivating stony coral *Porites cylindrica* and *Porites rus* fragments on grids at 1 m depth led to lower survival but higher monthly mass increase than those cultivated at 10 m depth, and small fragments had a greater

proportional mass increase than large fragments but similar survival. Average mass/month gained over 16 months was higher for fragments cultivated at 1 m depth (average g/30 days: small *Porites cylindrica* 0–5, small *Porites rus* 4–8, large *Porites cylindrica* 0–16, large *Porites rus* 17–24) than fragments cultivated at 10 m depth (average g/30 days: small *Porites cylindrica* 2–3, small *Porites rus* 3–6, large *Porites cylindrica* 0–7, large *Porites rus* 0–9). Small coral fragments gained less mass than large ones (average g/30 days: small *Porites cylindrica* 0–5, large *Porites cylindrica* 0–16; small *Porites rus* 3–8, large *Porites rus* 0–24), but had a greater percentage increase in mass (average %/30 days: small *Porites cylindrica* 0–21%, large *Porites cylindrica* 0–13%; small *Porites rus* 6–19%, large *Porites rus* 0–12%). Survival was lower for fragments transplanted at 1 m (6 months 67–93%, 12 months 47–86%, 16 months 0–40%) than 10 m (6 months 93–100%, 12 months 94–100%, 16 months 0–34%). There was no effect of size on survival. In July 1994, small (average weight: *Porites cylindrica* 19–22 g; *Porites rus* 37–44 g) and large (average weight: *Porites cylindrica* 118–168 g; *Porites rus* 185–203 g) wild-grown fragments of *Porites cylindrica* and *Porites rus* were stuck with superglue to Plexiglass plates. Plates were attached with plastic-coated copper wire to steel grids (5 cm mesh) raised 20 cm above the substrate at 1 m or 10 m depth (fifteen fragments/species/size/depth). From August 1994–November 1995, survival of fragments was surveyed every month, and fragments were weighed in the laboratory every two months.

A replicated study (years not given) at five artificial coral nurseries in sheltered back reefs in Puerto Rico (2a) found that fragments of staghorn corals *Acropora cervicornis* and *Acropora prolifera* cultivated on wire frames increased in size and grew new branches, and growth of *Acropora cervicornis* fragments over the frames was greater for those collected from the reef front than from the sheltered back reef. After one year, fragments of staghorn corals cultivated on wire frames increased in size (relative growth: *Acropora cervicornis* = 17–25 × original length; *Acropora prolifera* = 29–38 × original length), grew new branches (total length of new branches: *Acropora cervicornis* = 221–282 cm; *Acropora prolifera* = 349–494 cm), and grew over the frames (total of all overgrowths: *Acropora cervicornis* = 11–31 cm; *Acropora prolifera* = 13–34 cm). For *Acropora cervicornis*, fragments collected at the reef front had greater average overgrowth (13 cm) than those collected at the

sheltered back reef (8 cm). At each of five sites, for each of four staghorn coral varieties (*Acropora cervicornis* and *Acropora prolifera* collected from reef front and back reef sites), 10–12 fragments (8–12 cm long) were attached to weighted 'A-shaped' wire mesh frames (1 m long, 25 cm high, 2.5 × 5 cm PVC-coated mesh), 5–10 cm above sandy substrate. Surviving fragments, new branches and overgrowth were measured after one year.

A replicated study (years not given) at four artificial coral nurseries in sheltered back reefs in Puerto Rico (2b) reported that cultivating staghorn coral *Acropora cervicornis* and *Acropora prolifera* fragments of 15–22 cm or 8–12 cm led to lower mortality compared to fragments of 3–5 cm. Results were not tested for statistical significance. After six months, mortality rates of 15–22 cm and 8–12 cm fragments were 25% and 4–22% for *Acropora cervicornis*, respectively, and 25% and 2–4% for *Acropora prolifera*, whereas mortality rates of 3–5 cm fragments were 32–44% for *Acropora cervicornis* and 32% for *Acropora prolifera*. At each of four sites, and for each of three coral varieties (*Acropora cervicornis* collected from back reef and reef front sites, *Acropora prolifera* collected from back reef), 10 fragments of each of three sizes (3–5, 8–12 and 15–22 cm) were attached to 5–6 m fishing lines using cable ties, spaced 10–30 cm apart. Lines were secured at one end to metal stakes randomly placed in a sheltered area of coral rubble substrate. Mortality was recorded for each fragment after six months.

A replicated study (years not given) at four artificial coral nurseries in sheltered back reefs in Puerto Rico (2c) found that cultivating younger fragments of staghorn corals *Acropora cervicornis* and *Acropora prolifera* led to lower mortality compared to when older fragments were cultivated. After six months, overall mortality rates were lower for younger coral fragments collected from outer branches of colonies (17%) than older fragments collected from >10–15 cm below the apex (0–10%). The same was true when older fragments collected from >20–25 cm below the apex (30–50%) were compared to younger fragments collected from outer branches (0–10%). At each of four reef sites, for each of three coral varieties (*Acropora cervicornis* collected from back reef and reef front sites, *Acropora prolifera* collected from back reef), 10 × 8–12 cm fragments of each of two relative ages (younger fragments collected from outer branches, older fragments collected from >10–15 cm below

the apex) were attached to 5–6 m fishing lines using cable ties. Three additional lines had 10 *Acropora cervicornis* fragments collected from >20–25 cm below the apex and 10 younger fragments collected from outer branches. Lines were secured at one end to metal stakes randomly placed in a sheltered area of coral rubble substrate. Mortality was recorded for each fragment after six months.

A replicated study (years not given) at four artificial coral nurseries in sheltered back reefs in Puerto Rico (2d) found that cultivating fragments of staghorn corals *Acropora cervicornis* and *Acropora prolifera* on wire frames or coral rubble led to greater survival than fragments placed directly on sand, and fragments on frames suspended above the sand had greater growth than those in contact with sand. After six months, no coral fragments placed directly on sand survived, whereas 15–30% of fragments placed on coral rubble survived, and 1–49% of fragments on wire frames survived. After one year, relative growth (length gained as a proportion of original length) was greater for fragments suspended on frames 5–10 cm above the sand (12–22 × original length) than for those with their bases in contact with sand (7–17 × original length; difference not statistically tested). At each of four sites, for each of 3–4 staghorn coral varieties (*Acropora cervicornis* and *Acropora prolifera* collected from back reef and reef front sites), 10–12 fragments (8–12 cm long) were either scattered directly on sand, supported on weighted 'A-shaped' wire mesh frames (1 m long, 25 cm high, 2.5 × 5 cm PVC-coated mesh) with 0.5–1 cm of fragment bases covered with sand, supported on wire mesh frames 5–10 cm above the sand, or scattered directly on coral rubble. Mortality of each fragment was recorded after six months. Relative growth was recorded for the three largest fragments/treatment/frame after one year.

A replicated study in 1998–1999 at three reef sites in La Paraguera, Puerto Rico (3) found no difference in survival between fragments of stony coral transplanted onto artificial reef structures than fragments attached to dead coral. After 12 months, overall survival was 90%. There was no difference in survival between structures with 42/45 fragments (93%) surviving on the artificial structure and 20/23 (85%) on the dead coral. In addition, after 12 months, corals on the artificial structure had grown <0.5 cm (data not analysed). In March 1998, three Bay Ball artificial reef structures were constructed with holes (9 cm diameter)

over the surface of the ball (see paper for details). Small fragments (< 20 cm diameter) of several species of stony coral (*Diplora* spp., *Montastraea* spp., *Colpophyllia* spp., and *Siderastrea siderea*) were collected from shallow (<0.5 m) reefs near the survey site. Most fragments were found unattached on the seabed. Fragments were attached to the Bay Balls™ and dead coral heads using underwater cement. Bay Balls™ were deployed 3–5 m deep on sandy substrate at Enrique Reef (east and west) and Mario Reef. Dead coral heads were located in Enrique and Mario Reefs. Survival was recorded 12 months after Bay Balls™ were installed.

A randomized, replicated study in 1996–1998 at a reef and an ex-situ marine laboratory in Bolinao, Philippines (4) found that cultivating stony coral *Porites cylindrica* and *Porites rus* fragments in an artificial nursery in a natural habitat led to a higher survival rate and, for *Porites cylindrica*, a higher rate of growth than fragments cultivated in an ex-situ nursery. After nineteen months, survival rate was higher for fragments in the artificial nursery in natural habitat (*P. cylindrica*: 44%; *P. rus*: 42%) than ex-situ (*P. cylindrica*: 12%; *P. rus*: 0.5%). Average growth rate was higher for artificial, natural habitat nursery (0.8–3.75 g/30 days) than ex-situ *P. cylindrica* fragments (0.17–0.76 g/30 days). There was no statistical difference in growth rate for *P. rus* (natural habitat nursery: 3.5–8.5 g/30 days; ex-situ: 1.6–5.2). Actual growth was higher for natural habitat nursery than ex-situ fragments of both species (*P. cylindrica*: natural habitat nursery 78 g, ex-situ 18 g; *P. rus*: natural habitat nursery 165 g, ex-situ 80 g). In November 1996, sixty fragments each of *P. cylindrica* and *P. rus* were collected from wild colonies and transported to an ex-situ marine laboratory, trimmed, and attached to acrylic plates using cyanoacrylate glue (superglue). Thirty fragments from each species were combined and taken to a sandy lagoon. Fragments were distributed evenly across six 1 $m^2$ steel grids, 20 cm above the seabed. The remaining sixty fragments were similarly mixed and placed on plastic grids, but in one of three seawater-filled plastic tanks in the laboratory. Fragments were cleared of all fouling organisms and mortality was recorded every two weeks. Growth was measured every two months then extrapolated to determine growth/30 days.

A replicated, paired study in 1996–1998 at an artificial coral nursery on a sandy substrate near Henchun, southern Taiwan (5a) found that

cultivating fragments of wild-grown branching stony coral *Acropora pulchra* cut from below the final branching point led to fragments developing new branches sooner and having increased skeletal growth compared to fragments cut from above the final branching point. Three months after being attached to the nursery frame, there were more below-final-branch cut fragments showing new branch growth (16/19) than above-final-branch cut fragments (6/19), however there was no difference after four months (below: 18/18, above: 17/18). Average skeletal growth was greater for below-final-branch cut fragments after two months (0.75 cm) and three months (1.03 cm) compared to above-final-branch cut fragments (two months: 0.46 cm, three months: 0.82 cm). In 1996–1998, twenty branches of healthy branching stony coral were collected from the wild. Two 4 cm length fragments were cut from each branch—one from below and one from above the final branching point. Fragments were tied to a rack and suspended vertically 30 cm above the sea floor 6 m deep. New branch growth was counted after three and four months. Skeletal growth (cm/30 days) was measured one, two and three months after fragments were attached to the rack (months and years not provided).

A replicated study in 1996–1998 at an artificial coral nursery on sandy substrate near Henchun, Taiwan (5b) found that cultivating wild-grown stony coral *Acropora pulchra* fragments of 7 cm or 4 cm led to greater new branch growth and skeletal growth compared to fragments of 1 cm. Although there was no difference after three months in the number of fragments with new branch growth between 7 cm (8/20), 4 cm (6/19) and 1 cm (3/17) fragments, after four months the number of 7 cm (18/20) and 4 cm (17/19) fragments with new branch growth was higher than for 1 cm fragments (8/17). Skeletal growth was higher for 7 cm (0.99 cm) compared to 4 cm (0.76 cm) and 1 cm fragments (0.23 cm) and greater for 4 cm compared to 1 cm fragments. In 1996–1998, sixty fragments (20 each of 7 cm, 4 cm, and 1 cm length) taken from branches of healthy stony coral were tied to a rack and suspended vertically 30 cm above the sea floor at 6 m deep. New branch growth was counted after three and four months. Skeletal growth (cm/30 days and cm/30 days/cm fragment) was measured three months after fragments were attached to the rack (months and years not given).

A replicated study in 1996–1998 at an artificial coral nursery on sandy

substrate near Henchun, Taiwan (5c) found that cultivating fragments of wild-grown stony coral *Acropora pulchra* when 6 cm fragments were divided into two 3 cm lengths led to greater skeletal growth compared to when they were left intact. Average skeletal growth was greater for the combined growth of each pair of 3 cm fragments after two (1.6 cm) and three months (2.2 cm) compared to single 6 cm fragments (two months: 1.3 cm, three months: 1.4 cm). For the 3 cm fragments and 6 cm fragments, there was no difference after one month (combined 3 cm fragments: 0.8 cm; 6 cm fragments: 0.9 cm). In 1996–1998, twenty-nine 6 cm long fragments were taken from branches of healthy stony coral. Fifteen were kept intact and 14 were divided into two 3 cm lengths. All fragments were tied to a rack and suspended vertically 30 cm above the sea floor at 6 m deep. Skeletal growth (cm/30 days) was measured one, two and three months after fragments were attached to the rack (months and years not given).

A replicated study in 1996–1998 at an artificial coral nursery on sandy substrate near Henchun, southern Taiwan (5d) found that cultivating fragments of wild-grown stony coral *Acropora pulchra* at 5 m deep led to more new branch development and greater skeletal growth compared to fragments cultivated at 10 m. After four months, new branch growth was recorded on more fragments at 5 m (18/20) compared to fragments at 10 m (8/17), although there was no difference after three months (5 m: 9/20, 10 m: 3/17). After two and three months, average skeletal growth was higher for fragments at 5 m (two months: 0.6 cm, three months: 0.9 cm) compared to fragments at 10 m (two months: 0.3 cm, three months: 0.5 cm), although there was no difference after one month (5 m: 0.02 cm, 10 m: 0.04 cm). In 1996–1998, thirty-seven 4 cm long fragments cut from branches of healthy stony coral were tied to racks and suspended vertically 30 cm above the sea floor. One rack (20 fragments) was placed at 5 m deep and one rack (17 fragments) at 10 m. New branch growth was recorded after three and four months. Skeletal growth rate (cm/30 days) was measured one, two and three months after fragments were attached to the rack (months and years not given).

A replicated study in 1996–1998 at an artificial coral nursery on sandy substrate near Henchun, Taiwan (5e) found that the upward-pointing cut-end of fragments of wild-grown stony coral *Acropora pulchra* cultivated in an artificial nursery developed more new branches

compared to downward-pointing cut ends regardless of their original orientation on the donor colony, but there was no difference in skeletal growth. New branch growth was recorded on 16/29 upward pointing ends compared to 3/29 downward pointing ends. There was no difference in skeletal growth between upward and downward pointing ends (data not reported). In 1996–1998, twenty-nine 4 cm long fragments were cut from branches of healthy stony coral and with the end tip cut off to create two cut ends on each fragment. Fragments were tied to a rack (15 with original end tip pointing up and 14 with original end tip pointing down) and suspended vertically 30 cm above the sea floor 6 m deep. Skeletal growth (cm/30 days) was measured, and new branch growth was recorded four months after fragments were attached to the rack (months and years not given).

A replicated study in 1996–1998 at an artificial coral nursery on sandy substrate near Henchun, Taiwan (5f) found that suspending fragments of wild-grown stony coral *Acropora pulchra* horizontally resulted in skeletal growth and new branch growth. After four months, average skeletal growth from the two cut ends was 0.77 cm/30 days (range 0.48–1.03 cm/30 days) and 0.42 cm/ 30 days (range 0–1.08 cm/30 days). New branch growth was also recorded (data not provided). In 1996–1998, twenty 6 cm fragments of stony coral were cut from wild colonies. Fragments were suspended 30 cm above the seabed from iron and plastic racks 6 m deep. Skeletal growth (cm/30 days) was measured, and new branch growth was recorded four months after fragments were attached to the rack (months and years not given).

A replicated study in 1996–1998 at an artificial coral nursery on sandy substrate near Henchun, southern Taiwan (5g) found that cultivating damaged fragments of wild-grown stony coral *Acropora pulchra* led to more new branch growth from the middle of the fragment, but slower new branch growth from the ends and less skeletal growth compared to undamaged fragments. After three months, more damaged fragments showed new branch growth from the middle of the fragment (7/13) compared to undamaged fragments (1/15). There was no difference after two months (damaged: 5/13, undamaged: 1/15). None of the 13 damaged fragments showed new branch growth from the cut end compared to 9/15 undamaged. There was no difference after three months (damaged: 4/13, undamaged: 9/15). After one, two and three

months, there was less skeletal growth for damaged fragments (one month: 0.3 cm, two months: 0.7 cm, three months: 0.8 cm), compared to undamaged (one month: 0.8 cm, two months: 1.3 cm, three months: 1.4 cm). In 1996–1998, twenty-eight 6 cm fragments were cut from branches of healthy stony coral. A 1 cm wide band of tissue was removed from the central section of 15 fragments while 13 fragments were left undamaged. All fragments were attached to a rack and suspended vertically 30 cm above the sea floor 6 m deep. New branch growth was recorded after two and three months. Skeletal growth (cm/30 days) was measured one, two and three months after the fragments were attached to the rack (months and years not given).

A replicated study in 1996–1998 at an artificial coral nursery on sandy substrate near Henchun, southern Taiwan (5h) found that when cultivating fragments of wild-grown stony coral *Acropora pulchra*, clearing problematic algae from the nursery rack led to a higher survival rate for 1 cm fragments, but not for 6 cm fragments. There was no difference in skeletal growth for 1 cm or 6 cm fragments, compared to fragments on racks where algae was not cleared. After four months, all seventeen 1 cm fragments survived when algal growth was regularly cleared from the nursery rack compared to 7/17 fragments where algae was not cleared. There was no difference in survival for 6 cm fragments (algae cleared: 18/18, algae not cleared: 19/20 survived). There was no difference in skeletal growth for 1 cm or 6 cm fragments where algal growth was cleared compared to where it was not cleared (data not reported). In 1996–1998, thirty-four 1 cm and thirty-eight 6 cm fragments were taken from branches of healthy stony coral. Seventeen 1 cm and eighteen 6 cm fragments were tied to a rack and suspended vertically on fishing line 30 cm above the sea floor 6 m deep. Each month, algae was removed from the line. The number of surviving coral fragments was recorded after four months. Skeletal growth (cm/30 days) was measured four months after fragments were attached to the rack (months and years not given).

A controlled study in 2005–2006 on two artificial coral nurseries in a lagoon in Pangasinan, the Philippines (6) found that cultivated fragments of stony coral on suspended or fixed nursery structures all grew, and there was no difference in survival or detachment between fragments on different structures. After one year, there was no difference

in survivorship or detachment for any species between suspended (average: survivorship: 91%; detachment: 5%) and fixed (average: survivorship: 85%; detachment: 5%) coral nurseries. On average, all species increased in size (branching species: height: 0.57–1%/day, width: 0.92–3.15%/day; non-branching species: surface area growth: 0.08–0.87%/day). See paper for details of individual species. In June–September 2005, two adjacent nurseries were constructed: a suspended nursery (seventy 60 × 80 cm plastic mesh trays on a PVC frame buoyed by floats and tethered at 0.5–1 m above the lagoon floor), and a fixed nursery (fifty 60 × 80 cm plastic mesh trays on a PVC frame attached to the floor on 1 m legs). Seventy wild-grown fragments of *Merulina scabricula, Montipora digitata, Echinopora lamellosa* and *Pocillopora damicornis* were transplanted in both nurseries, and 70 *Acropora formosa, Porites rus and Montipora aequituberculata* were transplanted only in the suspended nursery. Fragments were glued with cyanoacrylate glue (superglue) to plastic tubing inserted into the mesh trays or directly onto the mesh. Monitoring was carried out for one year with fragments monitored fortnightly for survival and 10 fragments/tray photographed monthly to monitor growth. **Costs (US$)**: Construction materials cost $1,645 (2005 value) and the project used 2,610 person-hours (including time spent constructing the nurseries and preparing and attaching fragments).

A replicated study in 2006 at an artificial coral nursery in a natural habitat near a fish farm in Eilat, Israel (7) found that stony coral *Stylophora pistillata* fragments cultivated on frames painted with anti-fouling paint had similar survival rates to those on unpainted frames, but survival was lower when pins or pinheads that corals were attached to were also painted. After four months, average survival rates were similar for coral fragments cultivated on frames painted with anti-fouling paint (83% survived, 13% detached, 4% died) and unpainted frames (85% survived, 14% detached, 1% died). Survival rates were lower when paint was also applied to pins (62% survived, 8% detached, 29% died) or pinheads (11% survived, 58% detached, 31% died) which the fragments were attached to. Cleaning time was reduced by 90% for corals on painted frames (2 min/10 coral tips) compared to unpainted frames (5 min/10 coral tips plus 15 min to clean nets). Corals on painted pins/pinheads did not require cleaning. In April 2006, four treatments

were each applied to 12 PVC frames (30 × 50 cm) containing plastic nets (0.25 $cm^2$ mesh) and pins (9 cm long, 2 cm diameter head): paint applied to entire frames and pins; paint applied to frames and pins but scraped off pinheads; paint applied to frames only; no paint applied. Two coats of anti-fouling paint (Aqua-guard M250) were used. Coral fragments were glued onto pins (60 fragments/frame). Frames were suspended at a depth of 8 m, approximately 10 m from a fish cage. After 126 days, surviving, detached and dead coral fragments were counted. Surviving corals were cleaned following protocols regularly used during coral nursery maintenance or before transplantation.

A replicated study in 2015 at an artificial offshore coral nursery in a natural habitat near Looe Key, Florida, USA (8) found that cultivating nursery-grown fragments of staghorn coral *Acropora cervicornis* by suspending them on lines resulted in greater linear growth but lower skeletal density compared to fragments cultivated on blocks attached to the seabed. There was no difference in buoyant weight. After six months, average length was greater for line-suspended (15 cm) compared to block-attached fragments (10 cm). Skeletal density was lower for line-suspended (0.05 g $cm^3$) compared to block-attached fragments (0.10 g $cm^3$). There was no difference in buoyant weight (line-suspended: 15.2 mg/day, block-attached: 16.3 mg/day). Six months after the fragments were attached (6th October), fragments were completely bleached (but still living) during a bleaching event in the summer. Three weeks later (28th October) all fragments were dead. In April 2015, twenty-one stony coral branch tips (average length 6.8 cm) were collected from nursery-grown colonies on site. Nine fragments were suspended using fishing line from a PVC 'tree' attached to the seabed 6.4 m deep. Twelve fragments were attached to PVC discs using epoxy putty and bolted onto a PVC pipe attached to a cement block placed on the seabed 7.9 m deep. Average linear growth (cm/day), skeletal density (mg/day) and buoyant weight (mg/day) were calculated on 30 October 2015.

A replicated study in 2005–2006 at an artificial nursery off Silaqui Island, Phillipines, (9a) found that cultivating fragments of wild-grown stony and blue coral on giant clam shells attached to a pvc frame resulted in variations in attachment time and survival rates depending on species and fragment size. *Acropora muricata* fragments were quickest to attach

to the substrate (average time: large fragments 31 days, small fragments 39 days) and *Echinopora lamellosa* fragments were slowest (average time: large >250 days, small 167 days). Survival rate after seven months was lowest for *Acropora muricata* (80–88%) compared to 100% for *Heliopora coerulea, Montipora digitata, Hydnophora rigida, Porites cylindrica, Porites rus, Pocillopora damicornis*. In December 2005–January 2006, fifty small (average diameter 27 mm) and 50 large (average diameter 60 mm) fragments of 10 stony (see paper for full list) and one blue *Heliopora coerulea* coral were collected from reefs near the study site. Fragments (one/species) were attached to giant clam *Tridacna gigas* shells (small: 11 fragments/shell, large: 11 fragments/two shells) using epoxy clay. Shells were fixed to a pvc frame 0.5 m above the seabed 2.9–3.4 m deep at five sites 50–220 m apart. Attachment (measured as the percentage of coral tissue attached to the substrate or the number of secondary attachment points) was recorded after one month and then every two weeks for seven months.

A replicated study in 2007 at an artificial nursery off Silaqui Island, Phillipines, (9b) found that cultivating fragments of wild-grown stony corals *Acropora hyacinthus* and *Acropora digitifera* on giant clam shells attached to a pvc frame led to a faster attachment time than cultivated fragments of *Acropora muricata*. After seven days, fragments of *Acropora hyacinthus* and *Acropora digitifera* had started attaching to the substrate whereas *Acropora muricata* took 10 days. More than 50% of *Acropora hyacinthus* and *Acropora digitifera* fragments had fully attached to the substrate after 16 days compared to 24 days for *Acropora muricata* fragments. After 34 days, all *Acropora hyacinthus* and *Acropora digitifera* fragments had fully attached to the substrate compared to 35/50 (70%) *Acropora muricata* fragments. In April 2007, 50 fragments (average diameter 34 mm) were taken from 25 colonies of *Acropora hyancinthus, Acropora digitifera*, and *Acropora muricata* (two fragments/colony). Fragments were attached to 50 empty giant clam *Tridacna gigas* shells (one fragment from each species/shell) using epoxy clay. Shells were fixed to a pvc frame 0.5 m above the seabed 2.9–3.4 m deep at five sites 50–220 m apart (10 shells/site). Attachment (% of fragment attached and the time taken for fragments to fully attach) was recorded every 3 or 4 days for 34 days.

A replicated study (year not given) at an in-situ coral nursery in a

natural habitat in Florida, USA (10), found that transplanting nursery-grown fragments of staghorn coral *Acropora cervicornis* of different genotypes (genetic makeup) led to differences in buoyant weight, linear growth, and number of branches between some fragments. Thirteen months after transplanting, net buoyant weight of fragments was higher for genotypes U41 (74 g), K2 (72 g) and U73 (69 g) than for U25 (33 g). Growth (total linear extension) was greater for U41 (133 cm), K2 (124 cm) and U73 (117 cm) compared to U25 (43 cm). There were no other significant differences in buoyant weight or growth between fragments. The number of branches recorded after 291 days ranged from 8–30/fragment and average number of branches/fragment varied between genotypes (numbers not reported). Ten known genotypes (K1, K2, K3, U25, U41, U44, U47, U73, U77, U78) of staghorn coral were selected. Four non-branched tips (~5 cm) were clipped from each of three colonies/genotype (12 fragments/genotype). Fragments were each weighed and randomly suspended from one of four PVC tree structures using monofilament and aluminium crimps. Tree structures were placed on the sea floor. Buoyant weight was recorded at the start and then at days 122 and 390 (the end of the experiment). Linear growth and number of branches were recorded at the start and every 45 days.

A replicated study in 2014–2015 at an in-situ nursery in a natural habitat in Florida, USA (11) found that cultivating nursery-grown fragments of staghorn *Acropora cervicornis* coral by suspending them from tree structures led to fragments growing longer and experiencing a later onset of bleaching and fewer breakages but having a lower survival rate than fragments attached to concrete blocks. After one year, linear growth of tree-cultivated fragments was higher (238 mm) than block-cultivated (110 mm). Bleaching was first observed on tree-cultivated fragments later (278 days) than block-cultivated (246 days). None of the tree-cultivated fragments broke compared to 19% of block-cultivated fragments. However, tree-cultivated fragments did not survive as long (297 days) as block-cultivated (305 days). In December 2014, two hundred and forty staghorn fragments from four genetically different colonies were collected (wild- or nursery-grown not specified). One hundred and twenty fragments were attached to individual cement disks using epoxy. Disks were attached to PVC pipes and placed vertically into concrete bases in groups of 10. Bases were placed on the seabed

8 m deep. The remaining 120 fragments were suspended from branches of a tree-structure (material not specified), 4–6 m deep. Monitoring was carried out monthly for one year. Growth (total linear extension) was measured for each fragment (colony) and evidence of disease or bleaching, and any breakages were recorded.

A review of six restoration projects established in 2007–2010 at locations in natural habitats in Florida and Puerto Rico, USA, (12), reported that most staghorn coral *Acropora cervicornis* fragments cultivated in an artificial nursery in a natural habitat survived and grew, and fragments attached to floating arrays had a higher growth rate than those attached to concrete blocks. After 1–2 years, the average survival for nursery-cultivated fragments was 91% (85–96%). Growth was higher for fragments on floating arrays (53 cm/year) compared to fragments on concrete blocks (average 18 cm/year, range: 11–30 cm/year). This paper presents survival and growth results from six projects cultivating staghorn coral fragments. Wild-growing staghorn coral colonies (>25 cm diameter) were selected and 3–4 branches or ≤10% of the colony were taken. Branches were broken into smaller fragments and attached to concrete blocks on the substrate (5/6 projects) or floating underwater arrays (1/6 projects). Survival (including fragments with partial tissue loss) was determined by counting the number of fragments with some live tissue. Growth (total linear extension) was measured using a flexible ruler. Fragments were monitored for 1–2 years.

A replicated study in 2015 at an artificial offshore coral nursery in a natural habitat near Looe Key, Florida, USA (13) found that cultivating nursery-grown staghorn coral *Acropora cervicornis* fragments by suspending them on lines resulted in greater linear growth but lower skeletal density compared to fragments cultivated on blocks attached to the seabed, but there was no difference in buoyant weight. After six months, average length was greater for line-suspended (15 cm) compared to block-attached fragments (10 cm). Skeletal density was lower for line-suspended (0.05 g $cm^3$) compared to block-attached fragments (0.10 g $cm^3$). There was no difference in buoyant weight (line-suspended: 15.2 mg/day, block-attached: 16.3 mg/day). By 6$^{th}$ October (six months after the fragments were attached), fragments were completely bleached (but still living) following a bleaching event in summer. By 28 October all fragments were dead. In April 2015, twenty-

one stony coral branch tips (average length 6.8 cm) were collected from nursery-grown colonies on site. Nine fragments were suspended using fishing line from a PVC 'tree' attached to the seabed 6.4 m deep. Twelve fragments were attached to PVC discs using epoxy putty and bolted onto a PVC pipe attached to a cement block placed on the seabed 7.9 m deep. Six months later, average linear growth (cm/day), skeletal density (mg/day) and buoyant weight (mg/day) were calculated.

A study in 2015 and 2016 at an artificial coral nursery in a natural habitat in the Dominican Republic (14) reported that most nursery-grown colonies of staghorn coral *Acropora cervicornis* spawned. In both years, 6–7 days after the full moon, approximately 80% of 500 staghorn coral colonies released egg/sperm bundles. The nursery (a 150 $m^2$ area, 12.5 m deep) was established in 2011 and consisted of 25 structures, including ropes, frames, domes and tables. Spawning was observed on two field trips in September 2015 and August 2016.

A replicated study in 2011–2012 using an artificial line-nursery on sandy substrate off Fort Lauderdale, Florida, USA (15) found that cultivating fragments of nursery-grown staghorn coral *Acropora cervicornis* by suspending them from a horizontal line led to higher survival, lower partial mortality, higher rates of self-attachment (fragment growing onto the substrate), greater linear growth, and greater new branch growth (>5 cm), compared to fragments fixed directly to vertical lines. After three months, self-attachment was higher for suspended (97%) compared to vertical fragments (79%). After 12 months, survival was higher for suspended (100%) compared to vertical fragments (43%). Average partial mortality/month was lower for suspended (1%–3%) than vertical fragments (3%–38%). Also, average linear growth was greater for suspended (61 cm) than vertical (10 cm) fragments and the average number of new branches >5 cm was higher for suspended (7.1) than vertical (0.3) fragments. In 2011, six H-shaped line-nurseries comprising one 2 m horizontal line 1 m above the seabed and two 1 m fixed vertical lines that were anchored to the seabed 7 m deep. Each frame contained 24 nursery-grown staghorn coral fragments >5 cm, twelve suspended from the horizontal line using rope, and six fixed to each vertical line. Live fragments, partial mortality (% live tissue), self-attachment (fragment grown over attachment wire), linear growth (cm/month) and the number of new branches >5 cm

were recorded monthly from January 2011–January 2012.

A replicated, randomized, controlled study in 2013–2014 at an artificial nursery in a natural habitat and natural reef site near Guana Island, British Virgin Islands (16), found no difference in survival or growth between wild-grown staghorn coral *Acropora cervicornis* fragments cultivated in an in-situ nursery in a natural habitat before transplanting or transplanted directly onto the reef. Both had higher survival and growth than fragments placed unattached on the reef. After 15 months, there was no significant difference in survival between nursery-cultivated-then-attached fragments (49%) and directly-attached (58%) but both had higher survival than unattached fragments (7%). Growth was not significantly different between nursery-cultivated fragments (42 cm) and directly-attached (78 cm) but both were higher than unattached (28 cm). In August 2013, loose fragments of staghorn coral (780) were collected from two reefs, measured and randomly assigned to one of three treatments: nursery-cultivated-then-attached (291); directly-attached (306); unattached (183). Nursery-cultivated fragments were tied 25 cm apart to one of seven PVC-frame line-nurseries 5–7 m deep. Direct-transplanted fragments were attached to the substrate using cable ties, and unattached fragments were placed loose on the reef substrate. After three months, nursery-cultivated fragments were transplanted and attached to the reef using cable ties. The length of all live branches was recorded for each fragment using scaled photographs. Survival and growth were monitored after 12, 24, and 64 weeks. **Costs (US$)**: number of survivors and growth/US$ (including e.g. cable ties, and nursery-frame materials, excluding e.g. SCUBA and snorkel equipment). Nursery-cultivated 1.0 survivors, 1.8 cm growth/US$; direct transplant 3.3 survivors, 9.2 cm growth/US$.

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## 13.2 Cultivate coral larvae in an artificial nursery located in a natural habitat

https://www.conservationevidence.com/actions/3994

- **Fifteen studies** evaluated the effects of cultivating coral larvae in an artificial nursery located in a natural habitat. Six studies were in Japan[2,3a,b,8,9,12], two in the Phillipines[4,10], and one in each of Australia[1], Belize[5], Israel[6], French Polynesia[7], Curaçao[11], the Dominican Republic[13], and Palau[14].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (15 STUDIES)**

- **Survival (15 studies)**: Fifteen studies (twelve replicated including three controlled, one randomized, one randomized, before-and-after study) in Australia[1], Japan[2,3a,b,8,9,12], the Philippines[4,10], Belize[5], Israel[6], French Polynesia[7], Curaçao[11], the Dominican Republic[13], and Palau[14] found that some larvae or spat (settled larvae) cultivated in an artificial nursery located in a natural habitat survived. Coral spat survived on settlement tiles[14], at different depths[1]; when settled on crustose coralline algae[5]; when settled on plastic trays[6,12], or cement/

plastic plugs[10]. Survival was higher for spat in crevices on tiles[3a], on vertical tiles[3b], initially at 3.5 m than 2 m deep[8], over time in narrower grids on fibreglass plates[8], in smaller refuges on cement plates[9], and on clay tripods in the wild compared to in a land-based nursery[11]. Two studies found that survival was higher for coral spat on tiles under cages[4] or in cages with topshell snails[2], whereas two studies[7,14] found no difference in survival between caged and uncaged spat. One study found low survival for spat on tiles hung 1 m above the sea floor[13].

- **Condition (4 studies)** Three out of four studies (two replicated), in Japan[3b,12], the Phillipines[10], and Curacao[11] found that coral spat (settled larvae) cultivated in an artificial nursery located in a natural habitat grew[3b,10,12]. The three studies found growth was higher for one species of coral spat on vertical tiles than horizontal[3b], and coral spat on cement/plastic plugs[10] and on plastic trays[12] grew. One study[11] found no difference in growth between spat cultivated on clay tripods in the wild compared to in a land-based nursery.

**Background**

Coral larvae can be cultivated in artificial nurseries that are located within the natural reef habitat (a process known as 'coral gardening') (Meesters *et al*. 2013). Cultivation methods involve taking coral spat (settled larvae) or single individual animals (polyps) attached to settlement 'tiles', from an existing nursery or wild colony. Settlement tiles are attached to or suspended from temporary structures, such as PVC frames, that act as artificial nurseries (Meesters *et al*. 2013). The aim is to grow the coral colonies and allow them to develop. Once grown, they can be removed from the temporary nursery and transplanted into the wild (also known as outplanting) onto natural substrate, such as degraded coral reefs or the sea floor, or onto more permanent artificial structures. Cultivating corals using these methods allows more direct action and manipulation to encourage the corals to grow.

This action specifically refers to cultivating nursery-grown coral larvae and spat (settled larvae) in an artificial nursery in a natural habitat. Studies that report the effect of cultivating coral fragments in an in-situ nursery are described in *Cultivate coral fragments in an artificial nursery located in a natural habitat*. Studies that report the effect of cultivating corals in an ex-situ nursery are described in *Cultivate corals in an ex-situ nursery*. Studies reporting the effect of transplanting corals are described in *Transplant nursery-grown corals onto artificial substrate*. Studies that report the effect of trasnplanting corals are described in *Transplant nursery-grown corals onto natural substrate; Transplant nursery-grown corals onto artificial substrate, Transplant wild-grown corals onto natural substrate* and *Transplant wild-grown corals onto artificial substrate*.

Meesters E.H.W.G., Smith S.R. & Becking L.E. (2013) *A review of coral reef restoration techniques*. Report number C028/14. IMARES: Wageningen, UR. Available from: https://library.wur.nl/WebQuery/wurpubs/fulltext/333153

A replicated study in 1994–1995 at an artificial nursery on a coral reef at Orpheus Island, Great Barrier Reef, Australia (1) found that some transplanted nursery-grown stony coral *Goniastrea aspera, Oxypora lacera* and *Platygyra daedalea* spat (settled larvae) on settlement tiles at different depths survived and survival rates were broadly similar between species and depths. Three months after transplanting, less than 15% of spat had survived. Twelve months after transplanting, there was no significant difference in average survival between species transplanted at different depths (*Goniastrea aspera*: shallow 1.4%, mid 0.3%, lower 3.1%; *Oxypora lacera*: shallow 1.1%, mid 2.4%, lower 1.6%; *Platygyra daedalea*: shallow 2.1%, mid 1.4%, lower 1.6%). In December 1994, egg/sperm bundles were collected from 8–10 mature colonies of three species of stony coral and transferred to settlement jars for four days to develop. Larvae were transferred to PVC trays for five days to settle onto terracotta tiles (11 × 11 × 1 cm). Ten replicate settlement tiles/species were transferred to the reef and bolted to the substrate at each of three depths (shallow: 0 m; mid: 5 m; lower: 10 m). Survival and number of juvenile corals were counted after three and six months when tiles were retrieved for

examination and returned to the reef within 24 hours. Tiles were then collected and examined at the end of the 12 month experiment. Tiles at the second site were retrieved and examined after three months but not returned.

A replicated, controlled study in 2004–2005 at an artificial coral nursery in a natural habitat at Akajima Island, Okinawa, Japan (2) reported that stony coral *Acropora tenuis* polyps cultivated on settlement tiles in cages containing juvenile top shell snails *Trochus niloticus* survived longer than polyps cultivated in cages without top shells. Nine months after settlement, 10–39 polyps on each settlement tile in cages with juvenile top shells had grown to ~40 mm diameter whereas all polyps in cages without top shells died within four months of settlement (no statistical analysis carried out). Settlement plates in the cages without top shells were observed covered in algae, sponges, hydroids and sea squirts. In June 2004, stony coral *Acropora tenuis* polyps were attached to concrete 10 $cm^2$ settlement tiles. Tiles were placed into plastic cages (number not reported) that also contained juvenile top shell snails. A control cage was prepared containing stony coral polyps on settlement tiles without top shells. Cages were suspended 2 m above the seafloor at a depth of 3.4 m. No information on sample sizes, sampling method and frequency was reported.

A replicated, randomized study in 2006–2007 at an artificial nursery in a natural habitat at Nichidomari, southwestern Japan (3a) found that when cultivating wild-grown stony coral *Echinophyllia aspera, Favites pentagona* and *Platygyra contorta* spat (settled larvae), survival was higher for spat growing in artificial crevices on slate tiles compared to spat growing on the tile surface. After four months, all larvae that settled on the surface of the tile between crevices had died, whereas after one year, survival rate for coral spat in the crevices ranged from 1.5% (*Favites pentagona*) to 12% (*Platygyra contorta*) (data presented as a figure). In August 2006, larvae collected from wild-growing colonies of *Echinophyllia aspera, Favites pentagona* and *Platygyra contorta* corals were taken to a laboratory and placed into settlement containers (five/species) each containing four 10 × 9 × 0.5 cm slate tiles with 70 artificial crevices drilled into the surface. Larvae settled on the tiles for two to three weeks before tiles were moved to a nearby bay and randomly fixed horizontally (8 or 10/species) or vertically (10/species) to rocky

substrate 2 m above the sea floor, 5 m deep. Tiles were retrieved every month for 12 or 13 months and survival and position of each spat was recorded using a dissecting microscope before tiles were returned to the site. Growth was measured using photographs.

A replicated study in 2006–2007 at an artificial nursery in a natural habitat at Nichidomari, southwestern Japan (3b) found cultivating stony coral *Echinophyllia aspera, Favites pentagona* and *Platygyra contorta* spat (settled larvae) led to mixed results for survival and growth on slate tiles fixed vertically compared to horizontally. After approximately one year, coral spat survival rate was higher on tiles fixed vertically compared to horizontally for *Echinophyllia aspera* (10% vs 1.5%) and *Favites pentagona* (7.3% vs 1.5%) but lower for *Platygyra contorta* (5% vs 12%). Growth of *Echinophyllia aspera* spat on vertical tiles was higher (3.3 $mm^2$) compared to horizontal (1.1 $mm^2$) but there was no difference for *Favites pentagona* (vertical: 3.7 $mm^2$, horizontal: 2.9 $mm^2$) or *Platygyra contorta* (vertical: 4.4 $mm^2$, horizontal: 3.3 $mm^2$). In August 2006, larvae collected from wild-growing colonies of *Echinophyllia aspera, Favites pentagona* and *Platygyra contorta* corals were taken to a laboratory and placed into settlement containers (five / species) each containing four 10 × 9 × 0.5 cm slate tiles each with 70 artificial crevices drilled into the surface. Larvae settled on the tiles for two to three weeks before tiles were removed to a nearby bay and randomly fixed horizontally (8 or 10/ species) or vertically (10/species) to rocky substrate 2 m above the sea floor, 5 m deep. Tiles were retrieved every month for 12 or 13 months and survival and position of each spat was recorded using a dissecting microscope before tiles were returned to the site. Growth was measured using photographs.

A replicated, controlled study in 2007 at two depths at an artificial nursery on a coral reef in Malilnep channel, Philippines, (4) found that using full or partial caging to exclude predators from transplanted stony coral *Acropora tenuis* spat (settled larvae) resulted in higher survival compared to uncaged spat. After three months, average survival rate at two depths of caged spat (4 m: 17%, 9 m: 33%) and partially caged spat (4 m: 19%, 9 m: 23%) was higher than uncaged spat (4 m: 5%, 9 m: 11%). There was no difference in survival between caged and partially caged spat. In June 2007, sixty pairs of fibre-cement settlement tiles each with 20 coral spat (>1 cm apart) were attached 20 cm apart on a metal

rod suspended 10 cm above the substrate on two metal posts. Forty structures were fully covered by 1 $cm^2$ PVC mesh cage, 40 were partially covered (open-sided cage) and 40 were left uncaged. Structures were placed at 4 m and 9 m (60 structures/depth). Survival of spat was recorded after three months in September 2007.

A replicated study in 2008 at an artificial nursery on a coral reef in South Water Cay, Belize (5) found that some elkhorn coral *Acropora palmata* and staghorn coral *Acropora cervicornis* spat (settled larvae) transplanted onto crustose coralline algae fragments attached to tiles survived, but survival of staghorn coral depended on the crustose coralline algae species used. After six weeks, on average, similar percentages of elkhorn coral spat survived on fragments of *Hydrolithon boergesenii* (17%) and *Titanoderma prototypum* (19%). For staghorn coral, 13% of spat survived on *Titanoderma prototypum* fragments and no spat survived on *Hydrolithon boergesenii* fragments. In August 2008, wild-collected elkhorn and staghorn coral egg/sperm bundles were cross-fertilized in a laboratory. Larvae were settled on fragments (1 × 1 cm) of two crustose coralline algae species. For each coral species, 1–2 fragments of *Hydrolithon boergesenii* and *Titanoderma prototypum*, each with 1–5 spat, were attached to 14–15 terracotta tiles (10 × 10 cm) using underwater epoxy. Tiles were bolted face down on a reef (3 m deep), with a 2 cm gap between the tile and reef. After six weeks, all tiles were retrieved, and proportions of surviving spat recorded using a dissecting microscope.

A study (years not given) at an artificial coral nursery in a natural habitat in Eilat, Israel (6) reported that the majority of stony coral *Stylophora pistillata* spat (settled larvae) cultivated in artificial nurseries in a natural habitat survived. Four months after being placed in artificial nurseries, 428 of 480 spat (89%) survived and began to form 3D structures. Larvae collected from wild-growing colonies of *Stylophora pistillata* were taken to a laboratory and placed in settlement containers (petri dishes containing paper discs). At the age of 1–2 months, 480 spat were individually glued onto plastic pins placed in trays (45 × 32 cm) made of PVC pipes and plastic netting (0.5 $cm^2$ mesh). Trays were attached to the coral nursery, located on the seabed 9 m deep, with cable ties and covered with white plastic nets (1 $cm^2$ mesh). Algae and encrusting invertebrates were removed from trays monthly. Each tray was photographed monthly (dates not given) to monitor spat mortality and growth.

A replicated study in 2004 at an artificial nursery in a natural habitat off the coast of Moorea, French Polynesia (7) found that transplanting nursery-grown stony coral *Acropora striata* recruits on tiles covered in cages led to similar survival to those on uncaged tiles. After one week, the average proportion of surviving *Acropora striata* recruits did not differ significantly between caged (48%) and uncaged tiles (44%). On average, caged tiles had a higher proportion of intact dead recruits (47%), a lower proportion of removed or damaged recruits (5%) and greater algae cover (82%) than uncaged tiles (intact dead recruits: 28%; removed/damaged recruits: 28%; algae cover: 24%). In 2004, *Acropora striata* larvae reared in the laboratory were placed in tanks and settled on 18 terracotta tiles (11 × 11 × 1 cm). After one week, 1 mm coral recruits on six tiles (average 35 recruits/tile) were placed at each of three depths in the sea (6, 12 and 18 m). Half of the tiles at each depth were covered with 1.2 cm wire cages, and the other half were left uncovered. Coral survival and algae cover were assessed after one week using photographs.

A replicated, randomized, before-and-after study in 2010 at an artificial nursery on two coral reefs off Ishigaki Island, Japan (8) found that 72 h after stony coral *Acropora spp*. larvae were deposited on artificial reefs there was higher settlement on reefs at 3.5 m than 2 m depth, but no difference related to grid size or plate arrangement, whereas after seven months there was higher survival on plates with narrower grids, but no difference related to plate arrangement or depth. After 72 h, more juvenile corals settled on settlement plates at 3.5 m (0.5–0.9 corals/cm$^2$) than 2 m depth (0.1–0.2 corals/cm$^2$). After seven months, survival was higher on plates with narrower grids than wider grids (average corals/plate: 2.5 cm grid = 52; two stacked 4 cm grids = 8; a single 4 cm grid = 1; 8 cm grid = 0). See original paper for non-statistically significant results. In February 2010, experimental plates (comprising fibreglass-reinforced plastic/polycarbonate) with different grid sizes were set up at two sites (one 2 m deep and one 3.5 m deep), and plates arranged to create different structures (see paper for designs) were also set up at the 3.5 m depth site. Plates were 50 cm$^2$ with a grid size of 2.5 cm, 4 cm, two 4 cm plates stacked, or 8 cm (three replicates at each site). Different grid sizes and arrangements were placed randomly, >2 m apart. Mature stony coral colonies were collected from the reef and taken to an artificial nursery to spawn. Egg/sperm bundles were

collected and placed into tanks to enable larvae to develop. Larvae were introduced to the plates four days later (see paper for methods). Coral spat were surveyed 72 h after larvae were introduced and survival was surveyed one, three, six and seven months later.

A replicated study in 2007–2009 at an artificial nursery in a bay in Otsuki, Japan (9) found that transplanting stony coral *Echinophyllia aspera* and *Favites abdita* spat (settled larvae) on plates with smaller refuge structures led to greater survival compared to those on plates with larger structures. Two years after transplantation, 16% of *Echinophyllia aspera* spat survived on plates with 5 mm refuge structures, whereas all spat on plates with 15 and 25 mm structures died within 3 and 12 months respectively. For *Favites abdita*, 12% of spat survived on plates with 5 mm refuge structures, 2.5% survived on plates with 15 mm structures, and all spat on plates with 25 mm structures died within two years. In July–August 2007, eggs and sperm were collected from wild-growing *Echinophyllia aspera* and *Favites abdita* colonies and cross-fertilized. Larvae were placed in tanks containing cement slate plates (10 × 9 × 0.5 cm) with each of three sizes of refuge structure (5, 15 and 25 mm wide) created from 4 mm projections of underwater epoxy. After three weeks, plates with spat of each species (175–325 plates/refuge size) were randomly selected and bolted vertically to rocky substrate (10–30 cm apart, 5 m deep) with refuge structures facing outwards. Plates were retrieved at one, two, three, six, 12 and 24 months after transplantation and spat survival assessed using a dissecting microscope.

A study in 2008 at a laboratory and artificial nursery on a natural coral reef in Luzon, Philippines (10) reported that nursery-reared stony coral *Acropora millepora* spat (settled larvae) settled onto cement and plastic 'plug-ins' and then cultivated in an artificial nursery on a reef survived and grew. One month after ~102,500 larvae were placed in settlement tanks, there were 1,390 (1.4%) surviving coral spat (settled larvae) on 531/840 artificial settlement plug-ins (range: 1–41 corals/plug). Plug-ins supporting at least one coral were transferred to an in-situ nursery for a further six months, with 200 (14.4%) corals surviving on 153 plugs (1–4 corals/plug). Approximately seven months after fertilization, the average diameter of surviving corals was 4.4 mm. In March 2008, three wild-growing colonies of *Acropora millepora* were collected from the reef and taken to a laboratory where they spawned. Egg/sperm bundles were

collected and allowed to fertilize. Four days after spawning, ~102,500 larvae had developed from ~120,000 eggs (85.4%). Larvae were placed in tanks for seven days to settle onto plug-ins (comprising a cylindrical 20 × 15 mm cement head with a plastic screw plug attached) on racks made from sections of PVC pipe. Racks were transferred to rearing tanks in the laboratory for one month then live coral was counted and any plugs supporting live coral were transferred to an artificial nursery on a nearby reef. Coral survival and growth were measured after six months in the in-situ nursery (seven months after fertilization).

A replicated study in 2012–2014 at an artificial nursery in a natural habitat in Curaçao, Southern Caribbean (11) found a higher survival rate for elkhorn coral *Acropora palmata* larvae outplanted on clay tripods in the wild compared to larvae reared in a land-based nursery, but no difference in size. Overall survival rate after 31 months was 6.8 times higher for outplanted (3.4%) compared to nursery-grown corals (0.5%). After 31 months, 32% of substrate structures in the wild supported at least one settler compared to 3% of structures in the nursery. There was no difference in average size of corals between outplanted (17 $cm^2$) and nursery-grown coral (13 $cm^2$) after 31 months. In August 2012, approximately 4,000 egg-sperm bundles were collected from four colonies of elkhorn coral at 1–5 m deep. These were transferred to an ex-situ nursery and allowed to settle onto 320 clay pottery tripods (see paper for methods). After two weeks, 30 tripods each hosting an average of 11.1 settled larvae were transferred back to the reef and fixed to the substrate. Larvae on the other 30 tripods (average 11.1/tripod) were grown in the nursery. Survival (number of live settlers) was recorded after one, six, 11, 17 and 31 months. Size was measured after 17 and 31 months by photographing colonies against a ruler for scale. **Costs (US$)** reported in 2015: Nursery maintenance (including labour and utilities) = US$12,875/year. Larval rearing costs (including labour and materials) = US$8,814. Outplanting and monitoring (including labour and materials) = US$6,284. See paper for full cost breakdown.

A replicated study in 2014–2016 at artificial nurseries at five natural sites of mixed substrate in Sekisei Lagoon, Japan (12), found that some nursery-grown stony coral *Acropora tenuis* and *Acropora selago* larvae cultivated in plastic trays in a natural habitat survived and grew. After 72 h, there were approximately 25–30 settled larvae/100 $cm^2$ at four of

five sites (fifth site: ~150/100 $cm^2$). After one month, survival ranged from ~20–80% falling to 1–15% after 15 months. The average number of juvenile corals/plate decreased from 4–89/plate 15 months after settlement to 4–48/plate after 26 months. Average juvenile colony size grew from 8–14 mm 15 months after settlement to 13–28 mm after 26 months. In May 2014, ten colonies each of *Acropora tenuis* and *Acropora selago* were collected from the wild and taken to a laboratory where spawning was induced. After four days, 1,000 larvae from each species were placed into 40 bags each containing a fiber-reinforced plastic settlement plate with thirty-six 4 × 4 × 4 cm cells in a 6 × 6 grid. Eight plates were attached to the substrates using anchoring bolts and cable ties or, on sandy substrate, using an iron rod. Survival and settlement numbers were recorded after 72 h, and one, three, six, and 15 months by removing and examining the plates under a microscope. Juvenile corals were counted in-situ 15 and 26 months after settlement.

A study in 2019–2020 at an artificial nursery in a natural habitat off southeastern Dominican Republic (13) reported that transplanting nursery-grown stony coral *Dendrogyra cylindrus* spat (settled larvae) on tiles hung 1 m above the sea floor resulted in very low survival. A year after settled larvae were transplanted only one had survived (of an estimated 380 larvae). In August 2019 (after sunset, three nights after full moon), sperm and eggs were collected in-situ from two male and two female coral colonies. Sperm and eggs were mixed in the laboratory (83% fertilization rate) and larvae were fully developed within 24 hours. Twenty star-shaped ceramic tiles were added and left for 10 days for larvae to settle and the number of settlers on two tiles were counted. A month later, tiles were transferred to an in-situ nursery (12 m deep) and hung 1 m above the seafloor. Survival was assessed in May 2020 and September 2020.

A replicated, controlled study in 2017–2018 at an artificial nursery in a natural coral reef off Palau (14) found that cultivating coral *Acropora digitifera* larvae in a natural habitat with a protective cage resulted in higher survival than when a partial cage or no cage was used in one of two experiments. One experiment using 'clean' settlement tiles found higher survival for corals on caged tiles (24–34%) than for corals on partially caged or open tiles (1–12%). The other experiment using settlement tiles 'conditioned' on the reef found similar low survival for

corals on caged (1 of 121 larvae) and uncaged tiles (3 of 215 larvae). In 2018, settlement tiles (10 × 10 cm) comprising a checkerboard arrangement of 1 cm × 1 cm raised squares were placed on the reef in 12 plots of three tiles each (one caged tile, one partially caged and one open). After four weeks, the tiles were retrieved and any algae removed. A second set of tiles of the same design was left (caged or uncaged) to 'condition' on the reef for six months before being returned to the aquarium. In April 2018, coral larvae were collected from eight adult colonies, transferred to ex-situ aquaria and settled on to the 'clean' or 'conditioned' tiles (10 × 10 cm) In the first experiment, clean tiles with coral spat (settled larvae) were returned to the reef. Tiles were arranged in 12 plots with one each of caged, partially caged and open tile. Survival was assessed after 28 days, then every 14 days until 70 days had passed. In the second experiment, the tiles with coral spat were returned to the reef and placed, caged or uncaged, in 12 plots on the reef. Survival for corals in the second experiment was assessed after 70 days.

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## 13.3. Cultivate corals in an ex-situ nursery

https://www.conservationevidence.com/actions/4016

- **Thirty-six studies** evaluated the effects of cultivating corals in an ex-situ nursery. Ten studies were in the USA[13a-c,15a-c,16,18,22,25], five in Israel[3a-c,4,12], and Australia[17,20a,b,26,27], three in the Philippines[5,6a,b], two in Singapore[10a,b], and one in each of the Bahamas[1], Germany[2], Colombia[7], Spain[8], Puerto Rico[9], Belize[11], Japan[14], Dominican Republic[19], Fiji[21], Mexico[23], and Taiwan[24].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (36 STUDIES)**

- **Abundance (9 studies)**: Seven of nine studies (seven replicated, including one randomized) in the Philippines[6a], Singapore[10a,b], Belize[11], Israel[12], the USA[13a-c], and Japan[14], reported that larvae cultivated in ex-situ nurseries settled on different substrate materials including ceramic[14], cement mixed with coral rubble[10a,b], and crustose coralline algae[11], or material colours[13a-c]. One study[12] found settlement was higher in petri dishes with lids rather than uncovered. One study [6a] found that larvae cultivated in an ex-situ nursery had a higher settlement rate than on the natural reef.
- **Reproductive success (2 studies)**: One of two replicated studies (one controlled, one randomized controlled) in Mexico[23] and Taiwan[24] found that the percentage of eggs that yielded larvae was lower when frozen sperm was used compared to fresh[23]. The other study[24] found that fragments released fewer larvae in warmer water temperatures than in average temperatures.
- **Survival (20 studies)**: Eight of 20 studies (sixteen replicated including six randomized controlled, and five randomized) in Germany[2], Israel [3a,c], the Philippines[5,6a,b], Colombia[7], Israel[12], Japan[14], the USA[15a,c,18,25], Australia[17,26,27], the Dominican Republic[19], Fiji[21], Mexico[23], and Taiwan[24], found survival of cultivated coral larvae or spat (settled larvae) varied by orientation[2], settlement substrate[14,18], settlement container[7,12], presence of adult coral[17], the number of settlers and water source[26], or presence of beneficial algae[27], or when cultivated from adults in a protected area[21]. One study[6b] found survival was higher for groups or pairs than single spat. One study[23] found that coral spat (settled larvae) survival was higher when frozen sperm had been used. Two studies[24,25] found larvae survival was temperature dependent. Two studies[3c,6a], found survival was higher for ex-situ cultivated nubbins (small fragments)[3c], and larvae[6a], than on a natural reef whereas another study[5] found that ex-situ cultivated fragments had

lower survival than in-situ. One study[15a] found that most nubbins survived under different levels of shade and water flow, whereas another study[15c] found no difference in survival between fed and unfed nubbins. One study[3a] found no difference in survival of nubbins taken from the donor branch tip or mid-branch. One study[19] found most coral spat died when settled on coral rocks or cement substrate.

- **Condition (23 studies)**: Five of 23 studies (seventeen replicated including six randomized, controlled, five randomized, and three controlled) in the Bahamas[1], Germany[2], Israel[3a-c,4], the Philippines[5,6b], Spain[8], Puerto Rico[9], Belize[11], Japan[14], the USA[15a-c,16,18,22,25], Australia[17, 20a,b] and Taiwan[24] found that coral spat (settled larvae) grew[2,3c,9,14] and developed polyps[2,8] when cultivated in ex-situ nurseries. Two studies[3b,15a] found that nubbins (small fragments) developed more polyps under a combination of lights[3b] and that the growth of nubbins under different levels of shade and water flow varied by species[15a]. One study[24] found larvae growth rate was not temperature dependent. One study[6b] found a higher number of polyps developed on juveniles cultivated in groups. Three studies[15b,c,20b] reported mixed effects of supplementary feeding on growth and weight of fragments. Two studies[17,22] found higher levels of zooxanthellae uptake by cultivated coral spat in settlement tanks containing sediment and an adult coral, whereas another study[25] found no difference in zooxanthellae uptake by coral spat cultivated under different temperatures. One study[3a] found no difference in the number of polyps that developed on nubbins taken from the donor branch tip than mid-branch. Two studies[1,8] found ex-situ cultivated fragments had higher growth than wild-growing colonies, whereas one study[5] found that weight gain was lower for ex-situ cultivated fragments than in-situ. One study[4] found lower bleaching of fragments exposed to ultraviolet radiation. Three studies found that growth[16,18] and self-attachment[20a] were higher for corals cultivated on tiles cleared of algae[16], on rhyolite breccia and coral skeletons[18], and upside down rather than the right way up[20a].

**Background**

Cultivating corals in an ex-situ (also known as a land-based) nursery can provide a stable environment for larvae to settle and grow before being transplanted (outplanted) to restore degraded reefs or enhance existing reefs (Petersen & Tollrian 2001). Such facilities also provide environments where corals can be subjected to controlled environmental manipulation to test methods that aim to improve the survival rate and growth of cultivated corals or increase resilience to threats such as climate change (Yap & Molina 2003). However, these facilities are expensive to operate and require trained staff.

This action describes the cultivation of corals at various life-stages including larval, spat (settled larvae), nubbins (small fragments), and larger fragments in ex-situ nurseries away from the natural reef. Outcomes are generally survival, growth and reproductive success. Studies describing cultivating corals in an in-situ nursery are covered in *Cultivate coral fragments in an artificial nursery located in a natural habitat*; and *Cultivate coral larvae in an artificial nursery located in a natural habitat*. Studies describing transplanting nursery-grown coral onto natural or artificial substrates or wild-grown corals onto natural or artificial substrates are covered in *Transplant nursery-grown corals onto natural substrate; Transplant nursery-grown coral fragments onto artificial substrate; Transplant wild-grown corals onto natural substrate*; and *Transplant wild-grown coral onto artificial substrate*.

Petersen D. & Tollrian R. (2001) Methods to enhance sexual recruitment for restoration of damaged reefs. *Bulletin of Marine Science*, 69, 989–1000. Available from: https://www.ingentaconnect.com/content/umrsmas/bullmar/2001/00000069/00000002/art00057?crawler=true

Yap H.T. & Molina R.A. (2003) Comparison of coral growth and survival under enclosed, semi-natural conditions and in the field. *Marine Pollution Bulletin*, 46, 858–864. https://doi.org/10.1016/S0025-326X(03)00064-X

A replicated, randomized study in 1997–1998 at an ex-situ aquarium and natural reef in the Bahamas (1), found that fragments of staghorn *Acropora cervicornis* and elkhorn *Acropora palmata* corals cultivated in seawater tanks had higher basal growth than fragments cultivated on a natural reef, but similar or lower vertical and calcium-carbonate growth. After 10 months, basal growth was higher for fragments cultivated in an aquarium tank (staghorn: 56; elkhorn: 84 mm) compared to fragments cultivated on a natural reef (staghorn: 28; elkhorn 45 mm). Vertical growth of elkhorn fragments was lower for tank (17 mm) than natural reef (41 mm) cultivated fragments but there was no significant difference for staghorn fragments (tank: 40; reef: 31 mm). Calcium carbonate growth of elkhorn fragments was lower for tank (65 mg/day) than natural reef (112 mg/day) but no difference for staghorn fragments (tank: 39; reef 24 mg/day). In July 1997, twelve fragments (~7 cm long) were taken from each of two colonies of staghorn and elkhorn corals (48 fragments). Each fragment was attached upright to a PVC plate using epoxy. Fragments were weighed before-and-after attachment. Fragments were randomly chosen to be transplanted into an open water tank 1 m deep or transplanted onto a steel array 3 m deep at a nearby patch reef site. After 10 months, corals were collected and weighed to determine calcium carbonate growth. Vertical and basal growth were assessed using photographs.

A study in 1998–1999 at an aquarium in Munich, Germany (2), reported that some wild-grown stony coral *Acropora florida* larvae cultivated in an ex-situ nursery settled (vertically and horizontally), and that surviving spat (settled larvae) grew and produced polyps. Two weeks after 400 stony coral larvae were placed in settlement tanks, 99/400 (25%) had settled onto ceramic tiles (46 on vertical surfaces and 53 on horizontal). After seven weeks, overall survival rate of spat was 83% decreasing to 72% after 19 weeks. Survival rate remained at 27% from week 3240. Coral spat settled on vertical surfaces had a higher survival rate (19/46, 41%) after 40 weeks compared to horizontal (8/53, 15%). The average number of polyps increased from one/colony after two weeks to 13/colony after 32 weeks. In June 1998, four hundred larvae from wild-growing stony coral colonies in Okinawa, Japan, were flown to an ex-situ nursery at an aquarium in Munich, Germany. These were placed into tanks filled with artificial seawater and allowed to settle onto conditioned ceramic settlement tiles. After 32 weeks, tiles with juvenile

colonies on their surface were transferred to an aquarium and attached to the rocky substrate using a screw. Survival and growth (number of new polyps) of coral spat and juveniles were monitored after two, seven, 19, 32 and 40 weeks.

A study (year not given) at an aquarium in Eilat, Israel (3a), found that cultivated stony coral *Stylophora pistillata* nubbins (small fragments) survived and grew and there was no difference in survival or growth rate for nubbins taken from the donor colony branch tip compared to mid-branch. After 90 days, all 20 nubbins taken from branch tip or mid-branch survived. The average growth rate was 0.42 polyps/nubbin/day and the average number of polyps had increased from 5–44/nubbin (branch-tip) and 5–47/nubbin (mid-branch). There was no significant difference between average polyp numbers on tip or mid-branch nubbins. One *S. pistillata* colony was collected from 4–6 m deep and taken to an ex-situ aquarium. Twenty nubbins (~five polyps each) were taken from the colony (eight from branch tips; 12 from mid-branch ~3 cm below the tip), super-glued to glass slides, and placed in a 16 L running seawater aquarium. Survival and growth (number of new polyps) were measured weekly for 90 days using a binocular microscope.

A replicated study (year not given) at an aquarium in Eilat, Israel (3b) found that cultivated stony coral *Stylophora pistillata* nubbins (small fragments) in an ex-situ nursery under a combination of Floura, Cool-White, and Blue-Blue lighting had a greater increase in new polyp development than nubbins under individual lights. After 107 days the average number of polyps/nubbin was higher for nubbins cultivated under the combination lighting (34 polyps/nubbin; 195% increase) than those under single Floura (31 polyps/nubbin; 160% increase), Cool-White (24 polyps/nubbin; 128% increase), or Blue-Blue (20 polyps/nubbin; 116% increase) lights. Three wild-growing *S. pistillata* colonies were collected from 4–6 m deep and taken to an ex-situ aquarium where 192 nubbins (64/colony) were collected from branch tips and mid-branch. Nubbins were super-glued to glass slides and distributed evenly between four 16 L aquarium tanks. Each tank was subjected to a different light regime with three fluorescent tubes/aquarium comprising either Blue-Blue, Cool-White, Fluora, or a combination of all three lights. The number of new polyps was counted each week using a binocular microscope.

A replicated study (year not given) at an aquarium and natural reef in Eilat, Israel (3c) found that stony coral *Stylophora pistillata* nubbins (small fragments) had a higher survival rate in an ex-situ nursery than an in-situ nursery, and that ex-situ nubbins grew. After 40 days, 95% of ex-situ cultivated nubbins had survived compared to none of the in-situ cultivated. After 103 days, 60% of the ex-situ nubbins survived. After 40 days, 153/300 (51%) ex-situ nubbins had grown across the substrate and after 103 days, 117/300 (39%) had grown. Three wild-growing *S. pistillata* colonies were collected from 4–6 m deep and taken to an ex-situ aquarium where 200 nubbins (5–15 polyps/nubbin) were removed from each colony. Nubbins were super-glued onto 10 × 10 cm plastic squares. Three hundred were placed into ex-situ aquaria and 300 were placed 5 m deep, 1 m above the natural reef substrate (method not reported). Survival and growth were measured after seven, 40, and 103 days (method not reported).

A replicated, controlled study in 2000 at a laboratory near Sdot-Yam, Israel (4) found that stony coral *Oculina patagonica* fragments exposed to ultraviolet radiation (UVR) did not show signs of bleaching, and levels of the bacteria *Vibrio shiloi*, known to cause bleaching, were undetectable compared to fragments shielded from UVR. After 25 days, none of the 20 fragments infected with *Vibrio shiloi* and exposed to direct sunlight with UVR showed any signs of bleaching. All 20 fragments infected and then exposed to direct sunlight but shielded from UVR showed total bleaching (100% loss of pigmentation) after 10 days. After eight hours, the number of *V. shiloi* detected in infected fragments exposed to UVR decreased by 97% and was no longer detectable after 10 hours. The number of *V. shiloi* in infected fragments shielded from UVR increased from 42,000 to 25 million/cm$^2$ after six hours and remained constant for seven days. Ten uninfected control fragments did not show any sign of bleaching whether they were exposed to or shielded from UVR. Forty stony coral *Oculina patagonica* fragments were each infected with the bacteria *Vibrio shiloi* (42,000/cm$^2$). Twenty fragments were placed in a tank and exposed to direct sunlight, the other 20 were exposed to direct sunlight but were shielded from UVR by a 5 mm Plexiglass cover that blocks 100% UVR but allows 95% visible light through. Ten control fragments not infected with *V. shiloi* were placed in direct sunlight (five with and five without UVR shielding).

A replicated, randomized study in 1996–1998 at a laboratory and reef in Bolinao, Philippines (5) found that ex-situ-cultivated fragments of stony coral *Porites cylindrica* and *Porites rus* survived and grew but less so than fragments in an in-situ (reef) nursery. After 19 months, survival rate was lower for laboratory-cultivated (*P. cylindrica*: 12%; *P. rus*: 0.5%) than reef-cultivated (*P. cylindrica*: 44%; *P. rus*: 42%) fragments. Weight gain was lower for ex-situ fragments of both species than transplanted (*P. cylindrica*: ex-situ 18 g, transplanted 78 g; *P. rus*: ex-situ 80 g, transplanted 165 g). In November 1996, sixty fragments each of *P. cylindrica* and *P. rus* were collected and transported to a marine laboratory, trimmed, and super-glued to acrylic plates. Thirty fragments from each species were randomly selected and transplanted to a sandy lagoon. Fragments from both species were distributed evenly across six 1 $m^2$ steel grids, 20 cm above the seabed. The remaining 60 fragments were similarly distributed on plastic grids in one of three seawater-filled plastic tanks in the laboratory. Mortality was recorded every two weeks. Growth was measured every two months for 19 months.

A replicated, randomized study in 1997–1998 at a laboratory and natural coral reef in central Philippines (6a), found that cultivating *Pocillopora damicornis* larvae in an ex-situ nursery led to a higher settlement rate and survival compared to larvae settled in-situ on a natural reef. Average settlement success was higher for ex-situ cultivated larvae (59%) compared to in-situ cultivated (12%). One week after settlement, average survival was higher for ex-situ cultivated larvae (76%) than for in-situ (42%). Ex-situ cultivated juveniles grew an average of 2.3 mm/week over six months (data not reported for in-situ cultivated). On each of nine sampling months from February 1997–August 1998, five wild-growing *P. damicornis* colonies were collected from a reef and kept in ex-situ aquarium tanks to spawn. Larvae were collected and either placed into tanks with limestone settlement tiles (48 $cm^2$) or taken to the reef and injected into settlement traps (6–12 traps with 4050 larvae/trap) attached to the reef substrate. Tiles with settled larvae (360–1500 larvae) were transferred to cultivation tanks with flowing unfiltered seawater. Settlement traps on the reef were removed after 24 h and settlement success was measured. Survival was measured each day for a week and then weekly for one month. Growth was measured weekly for six months on a randomly selected group of 20 ex-situ cultivated juveniles settled in February 1997.

A replicated, randomized study in 1998–1999 at a laboratory in central Philippines (6b), found that cultivated stony coral *Pocillopora damicornis* larvae survived and developed polyps. Coral spat (settled larvae) that naturally joined in pairs or groups had higher survival rates and developed more polyps than single spat. Average weekly survival over six months was higher for groups (99.9%) and pairs (99.8%) compared to single juveniles (98.2%) and there was no significant difference in survival rate between groups and pairs. After six months, the average number of polyps was higher in groups (48 polyps/colony) than in pairs (30 polyps/colony) and single (15 polyps/colony), there was no difference between pairs and single colonies. In July 1998, three wild-growing *P. damicornis* colonies were collected from the reef and kept in ex-situ aquarium tanks to spawn. Larvae were collected and placed into one of three settlement tanks (60 larvae/500 ml water) to settle onto limestone settlement tiles (48 $cm^2$). Settlement and joining were measured after 24 h. Tiles containing settled juveniles were transferred to cultivation tanks with flowing unfiltered seawater. Survival was recorded weekly or every two weeks for six months, and the number of polyps/colony was recorded weekly for six months for all joined colonies and 40 randomly selected single colonies.

A replicated study (year not given) at an aquarium in Colombia (7) found that embryos of mountainous star coral *Montastraea faveolata* cultivated in vials with aerated seawater died within two hours, whereas all embryos in vials without aeration survived. Two hours after the embryos were placed in the vials, all embryos in the aerated water had turned white (indicating they had died). Embryos in the non-aerated water remained orange (indicating they were still alive). Egg-sperm bundles from mountainous star coral were placed into sealed vials with seawater and submerged in a seawater lagoon for 12 h to allow natural fertilization. Once fertilized, 70–100 embryos were transferred to each of 50 vials. Seawater was aerated in 25 of the vials. All vials were placed in aquarium water baths at 28°C.

A study in 2006–2007 at an ex-situ nursery in Spain (8) reported that fragments of cold-water coral *Lophelia pertusa* and *Madrepora oculata* cultivated in aquaria grew and developed new polyps and growth rate and polyp development were similar, and sometimes higher, than wild-growing coral. Vertical linear extension of ex-situ cultivated *Lophelia*

*pertusa* was 15–17 mm/year and *Madrepora oculata* was 3–18 mm/year. Polyps developed on *Lophelia pertusa* (4/year) and on *Madrepora oculata* (5/year). Growth rate and polyp development is reported as similar and sometimes higher than in-situ measurement rates (data from other studies). In August 2006, fragments of cold-water coral *Lophelia pertusa* and *Madrepora oculata* (number and size not reported) were collected from approximately 200 m deep in the Cap de Creus Canyon, NW Mediterranean. Fragments were maintained in dark aquaria and fed daily with *Artemia* sp. and Mysidacea. Water temperature was 11.5–12.5°C. Growth rate and number of polyps were measured in December 2006, and in March, May and November of 2007.

A study in 2006–2007 at ex-situ aquaria in Puerto Rico and 10 other locations (9) reported that elkhorn coral *Acropora palmata* larvae reared from field-collected eggs developed into juvenile corals. In August 2006, eggs from the threatened elkhorn coral were collected and fertilized in a laboratory aquarium. Half the 900,000 fertilized eggs were distributed among 10 aquaria located around the world. Approximately 20% of these eggs settled onto ceramic settlement tiles. Eighteen months after settlement, 800 juvenile corals were established in the aquaria, with some colonies 9 cm in diameter. In August 2006, eggs were collected from elkhorn coral at a reef at the Trés Palmas Reserve, Rincon. No other methodological details are reported.

A replicated study in 2006–2007 at an ex-situ nursery in Singapore (10a) found that a cultivated stony coral *Pocillopora damicornis* larvae settled in higher numbers on tiles made from cement mixed with coral rubble than on any of five other artificial substrate materials. After ten days, the average number of coral spat (settled larvae)/tile was highest on cement tiles mixed with 10% coral rubble (13 spat/tile) than any of the other materials (acrylic plates: one spat; cement tiles: three spat; ceramic tiles: four spat; glass plates: five spat, and PVC plates: seven spat/tile). Significantly more larvae settled on PVC plates than acrylic or cement tiles. Five wild-growing colonies (10–25 cm diameter) of *Pocillopora damicornis* were collected from reefs at Kusu Island and Raffles Lighthouse and maintained in tanks. From May 2006–March 2007, larvae were collected from the *Pocillopora damicornis* colonies and maintained for 1–2 days. Fifty larvae were randomly selected and placed into one of 24 two-litre polythene tanks containing one of the

settlement materials (200 larvae/material and four tanks/material). Larvae settlement was counted daily for 10 days.

A replicated study in 2006–2007 at an ex-situ nursery in Singapore (10b) found that a higher number of stony coral *Pocillopora damicornis* spat (settled larvae) on cement tiles mixed with 10% coral rubble than tiles with other cement and rubble mixes. After ten days, the average number of spat was higher on tiles comprising cement with 10% coral rubble (13/tile) than on tiles comprising cement with 0% rubble (6/tile), 1% rubble (7/tile), 5% rubble (6/tile), and 20% rubble (8/tile). Five colonies (10–25 cm diameter) of *Pocillopora damicornis* were collected from reefs at Kusu Island and Raffles Lighthouse and maintained in tanks. From May 2006–March 2007, larvae were collected from the *Pocillopora damicornis* colonies and maintained in darkened plastic containers for 1–2 days. Fifty larvae were randomly selected and placed into a two-litre polythene tank containing one cement tile mixed with 0%, 1%, 5%, 10%, or 20% coral rubble. There were four tanks for each tile type. Larvae settlement was counted daily for 10 days.

A replicated, randomized study in 2008 at a laboratory at Carrie Bow Cay, Belize (11) found that some five- and seven-day old elkhorn coral *Acropora palmata* and staghorn coral *Acropora cervicornis* larvae settled in containers, and settlement rates of seven-day old elkhorn coral larvae were higher in containers containing crustose coralline algae *Hydrolithon boergesenii* than in containers with three other crustose coralline algae species or coral skeletons. Seven-day old elkhorn coral larvae had higher average settlement rates in containers with *Hydrolithon boergesenii* fragments (81%) than in containers with fragments of *Paragoniolithon solubile* (46%), *Porolithon pachydermum* (39%), *Titanoderma prototypum* (34%) or staghorn coral skeleton (52%). Settlement rates of five-day old elkhorn coral larvae (17–23%) and five-day old staghorn coral larvae (17–21%) did not differ significantly between the five treatments. In August 2008, wild-collected elkhorn and staghorn coral egg/sperm bundles were cross-fertilized in a laboratory. Five and seven-day old elkhorn coral larvae and five-day old staghorn coral larvae were placed in wells in culture plates or petri dishes (10–20 larvae/container) with fragments of one of four crustose coralline algae species or a fragment of elkhorn coral skeleton (15 containers/treatment). Larave settlement was recorded after 24 h.

A replicated study (years not given) in a laboratory in Eilat, Israel (12) found that cultivating stony coral *Stylophora pistillata* larvae in lidded petri dishes containing two paper discs submerged in seawater led to higher settlement rates than cultivating larvae in open petri dishes containing one paper disc placed in a humidity chamber, and mortality rates did not differ between treatments. On average, a greater percentage of larvae settled in lidded petri dishes containing two paper discs submerged in seawater (60–63%) than in open petri dishes with one paper disc placed in a humidity chamber (45%). Larval mortality rates did not differ significantly between treatments (lidded dishes: 19–23%; open dishes: 27%). Adding silicone plugs to lidded dishes did not have a significant effect on larval settlement or mortality rates (see original paper). Larvae collected from wild-growing colonies of *Stylophora pistillata* were placed in 24–30 petri dishes (9 ×1.5 cm; 1–69 larvae/dish) for each of three treatments. Treatments consisted of open dishes fitted with one paper disc (in the bottom) placed in a humidity chamber; and lidded dishes, with and without silicone plugs, fitted with two paper discs (in the top and bottom) submerged in a flow-through seawater table. Discs (made from polyester, double-sided matte paper) were submerged in seawater for at least two months prior to the experiments. After each of two 48-h periods, paper discs were removed and spat (settled larvae) counted. Mortality rates were calculated at the end of the experiment.

A replicated study in 2006 in a laboratory in Miami, USA (13a) reported that mustard hill *Porites astreoides* and elkhorn *Acropora palmata* coral larvae settled on red and/or orange but not blue, green or white plastic cable ties, and only settled when illuminated. Mustard hill coral larvae settled on red cable ties (total 32 larvae, 6 of 6 dishes), whereas none settled on green or white cable ties. Elkhorn coral larvae settled on red and orange cable ties (both: total 8 larvae, 4 of 5 dishes), whereas none settled on blue cable ties. For both species, no larvae settled on cable ties in dark conditions. In June 2006, mustard hill coral larvae were collected from eight wild-grown colonies maintained in the laboratory. Ten larvae (<24 h old) were placed in each of twelve 100 ml dishes containing seawater and one each of red, green and white cable ties secured around a white fiberglass rod. In August 2006, egg/sperm bundles were collected from wild elkhorn coral colonies and cross-

fertilized to generate larvae. Ten larvae (5 days old) were placed in each of ten 100 ml dishes with seawater and one each of red, orange and blue cable ties secured around a white fiberglass rod. Half of the dishes in each experiment were placed in darkness and half under fluorescent lights for 12 h/day. After 48 h, settled larvae were counted using a dissecting microscope.

A replicated study in 2009 in a laboratory in Miami, USA (13b) found that mustard hill coral *Porites astreoides* larvae settled on nylon buttons, but in greater numbers on red nylon buttons than on buttons of six other colours. Overall, a greater number of larvae settled on red buttons (total 24 of 100 larvae, 6 of 10 dishes) than on pink (0 larvae), orange (3 larvae, 2/10 dishes), green (0 larvae), blue (1 larva, 1/10 dishes), purple (10 larvae, 1/10 dishes) or white buttons (1 larva, 1/10 dishes). In June 2009, mustard hill coral larvae were collected from eight wild-grown colonies maintained in the laboratory. Ten larvae (<24 h old) were placed in each of 10 petri dishes containing seawater and one each of red, pink, orange, green, blue, purple and white nylon buttons (1.6 cm diameter). Dishes were placed under fluorescent lights for 12 h/day. After 48 h, settled larvae were counted using a dissecting microscope.

A replicated study in 2009 in a laboratory in Miami, USA (13c), found that elkhorn coral *Acropora palmata* larvae settled on tiles, but in greater numbers on red ceramic tiles than on orange, yellow, green or blue ceramic tiles or limestone tiles. Overall, a greater number of larvae settled on red tiles (total 16 of 100 larvae, 4 of10 tiles) than on orange (6 larvae, 3/10 tiles), yellow (6 larvae, 4/10 tiles), green (4 larvae, 3/10 tiles), blue (1 larvae, 1/10 tiles) or limestone tiles (6 larvae, 2/10 tiles). In August 2009, egg/sperm bundles were collected from wild elkhorn coral colonies and cross-fertilized to generate larvae. Thirty larvae were added to each of ten 20 L tanks containing seawater and one one each of red, orange, yellow, green and blue acrylic tiles, and one limestone tile. All tiles (5 × 5 cm) had grooves carved into them. Tanks were placed under fluorescent lights for 12 h/day. After one week, settled larvae were counted using a dissecting microscope.

A study in 2007–2008 at an ex-situ nursery in Okinawa, Japan (14) reported that more than half of stony coral *Acropora tenuis* larvae cultivated in tanks settled on ceramic tiles, and more than half the spat (settled larvae) survived and grew. In total, 111,000 of 205,000 larvae

(54%) settled on ceramic tiles within tanks (average 173 larvae/tile). After 10 months, 66,000 of 111,000 spat (59%) survived and grew into juvenile corals. In June 2007, eggs and sperm were collected from eight wild-grown captive *Acropora tenuis* colonies and cross-fertilized. Five-day-old larvae were placed in four rectangular tanks (1.7 × 0.8 × 0.4 m), each containing seawater and 160 ceramic tiles (each 12 × 12 × 2.5 cm with five rows of 1.5 $cm^2$ holes) arranged in two layers. After 4–5 days, numbers of settled larvae were estimated and tiles transferred to aerated outdoor tanks (5.2 × 0.8 × 0.4 m) with flow-through seawater, snails, and young fish, and covered with shade nets or transparent vinyl tents. Numbers of surviving juvenile corals were estimated in April 2008. **Costs (¥)**: Total cultivation cost ¥7,963,000 (2011 value), plus ¥12,600,000 for collection and transport of coral colonies.

A replicated, randomized, controlled study in 2006 at a laboratory in Hawaii, USA (15a) reported that almost all stony coral *Porites compressa* and *Montipora capitata* nubbins (small fragments) cultivated under different levels of shade and water flow rates survived and grew. After 41 days, 478/480 (99.4%) of fragments survived. Under low water flow, growth rate of *P. compressa* fragments was highest (0.65 mm increase) under direct sunlight (0× shade) and lowest (0.36 mm increase) under the most shade (3×). Conversely, growth rate of *M. capitata* fragments was lowest in 0× shade (0.03 mm) and highest in 1× and 2× shade (both 0.16 mm) and 3× shade (0.08 mm). Under high water flow, growth rate was slightly lower than for low water flow and varied between different shading although results were not significantly different for either species (*Porites compressa*: 0.44–0.52 mm; *Montipora capitata*: 0.02–0.11 mm). In 2006, eighty nubbins (1 $cm^2$) were collected from each of three colonies of *P. compressa* and *M. capitata* and placed in alternating rows on 1 $cm^2$ plastic mesh (15 nubbins/species/mesh). Mesh sheets were placed into one of 16 buckets and randomly assigned to one of four shade treatments under either high (~11 cm/s) or low (~4 cm/s) water flow. Shade was provided by using layers of 50% shade cloth (1×, 2×, and 3×, and a 0× control). Buckets were cleaned and all nubbins were photographed weekly. Growth (area) was measured after 19 and 41 days.

A replicated, randomized, controlled study in 2006–2007 at a laboratory in Hawaii, USA (15b) found that providing additional food

to ex-situ cultivated stony coral *Montipora capitata* and *Porites damicornis* nubbins (small fragments) led to an increase in weight compared to unfed nubbins but there was no difference for *Porites compressa*. After three months, average overall weight increase (all species) was significantly higher for nubbins in tanks with unfiltered seawater fed with Reef Chili® (6.5%) and Reef-Roids® (7.5%) compared to unfed nubbins (2.1%). Weight increase for nubbins in tanks fed with Oyster Eggs® (2.7%) and Roti-Feast® (3.1%) were not significantly different from unfed nubbins. Nubbins of *M. capitata* fed additional food showed the highest increase in weight (fed: 0.06–0.14; unfed: 0.02), and *P. damicornis* (fed: –0.03–0.04; unfed: –0.02) whereas weight change was similar for fed and unfed *P. compressa* (fed: 0.02–0.07; unfed: 0.06). Ten tanks were set up, each with 18 nubbins (6 fragments/species) on plastic mesh. Fragments were collected from wild-growing stony coral *M. capitata, P. damicornis* and *P. compressa* colonies. Tanks were randomly assigned to one of four feeding treatments (Oyster Eggs®, Roti-Feast®, Reef Chili®, and Reef-Roids®) or the unfed control (two tanks/treatment). Corals were fed four times/week according to manufacturers' recommendations for 12 weeks. Measurements of wet weight (g) and displacement (ml) for each nubbin were taken at the start of the experiment and again three months later.

A replicated, randomized, controlled study in 2006 at a laboratory in Hawaii, USA (15c) found that stony coral *Porites compressa* and *Montipora capitata* nubbins (small fragments) cultivated in an ex-situ nursery with additional food supplements survived and grew and there was no difference in growth between fed and unfed nubbins, but growth varied with higher doses of supplements. After 45 days, 99% of *P. compressa* and 51% of *M. capitata* fragments had survived. Tissue growth had increased by 65% (*P. compressa*) and 35% (*M. capitata*). There was no difference in average net growth after 45 days for fragments fed with the recommended dose of supplements (*P. compressa*: 0.73 $cm^2$; *M. capitata*: 0.35 $cm^2$) compared to unfed fragments (*P. compressa* 0.70 $cm^2$; *M. capitata*: 0.37 $cm^2$) but net growth of *M. capitata* decreased with higher doses of supplements (3 × dose: 0.32 $cm^2$, 10 × dose: 0.26 $cm^2$). In October 2006, nubbins from *P. compressa* and *M. capitata* colonies (240 nubbins/species) were attached to 6 × 6 inch ceramic tiles (15 nubbins/species/tile) using marine epoxy. Tiles were

placed into one of 16 buckets (18 L) and randomly assigned to one of three feeding treatments (1 × manufacturers' recommended dosage, 3 × recommended, 10 × recommended) or the unfed control. Food supplements (comprising MicroVert®, MarineSnow Plankton Diet®, Phytoplan®, and Salifert Coral Food®) were provided with seawater filtered through a 500 μm filter. Buckets were cleaned and nubbins photographed each week. Coral tissue area was measured using scaled photographs. Area measurements were taken at the start of the experiment and 45 days later.

A replicated, randomized, controlled study in 2006 at an ex-situ nursery in Florida, USA (16) found that staghorn coral *Acropora cervicornis* cultivated on tiles with algae manually removed or grazed by variegated sea urchins *Lytechinus variegatus* had greater growth than those on undisturbed tiles. Average growth rates were greater for staghorn coral on tiles with algae manually removed (3.1 mm/day) or grazed by sea urchins (1.9 mm/day) than on those left undisturbed (–0.8 mm/day). In April 2006, circular pieces of staghorn coral (10 mm diameter) were attached to ceramic tiles (5 × 5 cm). Three tiles were added to each of nine containers (24 × 24 × 20 cm) within a fiberglass trough. Three containers were randomly assigned to each of three treatments: tile surfaces scraped every seven days using a razor blade; four variegated sea urchins (1 cm diameter) added; or tiles left undisturbed. Containers were replaced every seven days and corals randomly re-assigned to each treatment. Larval fish, shrimp and zooplankton were added to all containers weekly. Corals were measured on nine occasions over 210 days using photographs. **Costs (US$)**: Production costs for 100–2,000 × 50 $cm^2$ coral ramets were $9,620 (2013 value) using sea urchins and $6,302–16,790 using manual scraping. All cost estimates included labour and facility rental and operating costs.

A replicated, randomized, controlled study in 2011–2013 at an aquarium on Heron Island, Australia (17) found that cultivating stony coral *Acropora millepora, Acropora selago* and *Isopora palifera* spat (settled larvae) in tanks containing sterilized sediment plus an adult coral fragment led to increased uptake of zooxanthellae (beneficial algae), but mixed results for survival. After 9–12 days, 61–73% of spat in the sediment+adult coral tanks had acquired zooxanthellae compared to 20–52% (sediment only); 14–47% (adult coral only); and 15–19%

(seawater control). Survival rates for *A. selago* were highest in the adult-coral-only (55%) than the sediment+adult coral (27%) and sediment-only tanks (20%). There were no significant differences in survival between treatments for *A. millepora* (57%–78%) or *I. palifera* (data reported as statistical model results). For three consecutive years, wild-growing colonies of *A. millepora* (2011), *A. selago* (2012), and *I. palifera* (2013) were taken to an aquarium to spawn or release larvae. Egg/sperm bundles and larvae developed and settled onto pre-conditioned terracotta tiles. Tiles with spat were randomly allocated to one of four treatment tanks filled with sterilized seawater (sediment+adult coral; sediment only; adult coral only; seawater only). One tile was suspended 1 cm above the bottom of the tank in each of five tanks/treatment. Adult coral fragments (5 × 1 cm) were taken from the recently spawned colonies. Sediment was collected from the reef flat and sterilized at 134°C for 20 minutes. Zooxanthellae cells were counted periodically for 12 days (*A. millepora* and *A. selago*) and eight days (*I. palifera*) using a microscope.

A replicated study (year not provided) at an ex-situ coral nursery in Honolulu, Hawai'i, USA (18) found some variation in tissue growth and survival of stony coral *Montipora capita* and *Porites lobata* fragments transplanted onto different natural and synthetic substrate tiles. Average live tissue coverage at the start of the experiment was 1.61 $cm^2$. After 78 days, there was no significant difference between percentage increase in tissue growth on fragments on different substrate types (range: 60% amygdaloidal basalt to 33% porcelain tiles). After 184 days, increase in tissue surface area was higher for fragments on rhyolite breccia (99%) and amorphous coral skeletons (94%) compared to black 'A'ā lava (53%) but no other significant differences in tissue growth between the other 53 comparisons, or between species (see paper for results). After 365 days, survival was higher for fragments on marble tiles (100%) than on glass tiles (50%). There were no other significant differences in survival between substrate types and no difference between species (see paper for results). Fragments (2–3 cm long) were collected from colonies of *Montopora capita* and *Porites lobata* (132 fragments/species). Three fragments from each species were fixed, using marine expoxy, onto a 100–324 $cm^2$ tile. There were four tiles for each of 11 materials (total 264 fragments, 44 tiles). Percentage increase in tissue surface area was

measured after 78 and 184 days, survival was measured after 365 days.

A study in 2015–2016 at a laboratory in the Dominican Republic (19) reported that some staghorn coral *Acropora cervicornis* larvae settled in plastic buckets containing coral rocks or cement substrates, but most cultivated spat (settled larvae) and polyps died within 30 days. In 2015, during 4–9 days after fertilization, 50% of larvae settled on coral rocks and the walls of plastic buckets containing them. In 2016, during 8–30 days after fertilization, 35% of larvae settled in buckets with cement substrates. Most spat and polyps (90%) died within 30 days of settling. In September 2015 and August 2016, egg/sperm bundles were collected from staghorn coral colonies at an in-situ nursery and cross-fertilized. Fertilized eggs were cleaned with filtered seawater and placed in twelve 1.5 L buckets in a laboratory. Once the larval stage was reached, coral rocks (2015) or red and white cement substrates (2016) were added to the buckets. After 20 days, polyps on half of the settlement substrates were moved to a 45 L aquarium with coral fragments and sediment from the in-situ nursery. In 2015 and 2016, numbers of settled larvae were counted daily for 33 days after fertilization. Spat, and the polyps that grew from spat, were monitored for 332 days.

A replicated study (year not stated) at an *ex-situ* coral nursery in New South Wales, Australia (20a) found that cultivating wild-grown stony coral *Hydnophora rigida* fragments upside-down rather than the right-way-up led to higher rate of self-attachment, shorter time to self-attachment and greater attachment-surface growth but similar height gain and weight. After 100 days, self-attachment to the glass substrate was greater for upside-down (20 of 23, 87%) than right-way-up fragments (14 of 24, 58%). Average time to self-attachment was shorter for upside-down (71 days) than right-way-up fragments (81 days). Average monthly attachment-surface growth was greater for upside-down (75 $mm^2$) than right-way-up fragments (31 $mm^2$). However, there was no significant difference in average height gain after 100 days (upside-down: 0.6 mm; right-way-up: 0.9 mm) or average weight (upside-down: 400 mg; right-way-up: 400 mg). Forty-seven 3 cm long fragments were collected from six stony coral *Hydnophora rigida* colonies at the Great Barrier Reef. Fragments were fixed to individual glass plates using cyanoacrylate glue (superglue), 23 upside-down and 24 the right-way-up. Self-attachment (growth of fragment over the attachment plate)

was recorded every 20 days for 100 days. Average monthly attachment surface growth ($mm^2$), height increment (mm) and weight (mg) were calculated from measurements taken after 100 days.

A replicated, controlled study (year not stated) at an *ex-situ* coral nursery in New South Wales, Australia (20b) found that providing supplementary food (*Artemia* sp. or lipid-enriched *Artemia* sp.) to cultivated fragments of wild-grown stony coral *Hydnophora rigida* led to increased attachment growth and height, but not weight, compared to unfed fragments. After 100 days, attachment growth was greater for fragments fed with normal *Artemia* sp. (68 $mm^2$/month) or enriched *Artemia* sp. (68 $mm^2$/month) compared to unfed (19 $mm^2$/month); there was no significant difference between normal and enriched *Artemia* sp. Average height growth was greater for normal (1.0 mm/month) and enriched *Artemia* sp. (0.8 mm/month) than unfed (0.4 mm/month); there was no significant difference between normal and enriched *Artemia* sp. There was no significant difference in average monthly weight (normal: 500 mg, enriched: 400 mg, unfed: 300 mg). Forty-seven 3 cm long fragments were collected from six colonies of stony coral *Hydnophora rigida* at the Great Barrier Reef near Cairns. Fragments were super-glued to individual glass plates. They were fed with normal *Artemia* sp. (16 fragments) or lipid-enriched *Artemia* sp. (16), every two days, or were unfed (15). Self-attachment (fragment growth over the attachment plate) was recorded every 20 days for 100 days. Average monthly attachment-surface growth ($mm^2$), height increment (mm) and weight (mg) were calculated from measurements taken after 100 days.

A replicated study in 2014 in Fiji (21) found that cultivating stony *Pocillopora damicornis* corals in an ex-situ setting resulted in higher short-term survival for larvae originating from a protected area compared to those from a fished reef. Over a six-day period, larvae from the protected area had higher survival (94%) than larvae from the fished area (26–66%). This was true for larvae reared in protected area water (protected area larvae: 94%, fished area larvae: 66%) and fished area water (protected area larvae: 94%, fished area larvae: 26%). In addition, no differences were found in the microbiomes of larvae due to their origin (protected or fished reef) or the water they were held in (data reported as graphical analysis). In 2014, fragments of *P. damicornis*

colonies were collected from a protected area and an adjacent fished reef (12 colonies/area, 100–500 m between protected and fished areas) and held in separate containers. Four colonies from each area released larvae, and 10 larvae/colony were used to assess the microbiome. Additional larvae were gathered, and four treatments were established based on the origin of the larvae (protected or fished reef) and the origin of water (protected or fished reef), with 10 replicates/treatment, each with 10 larvae. Survival of larvae in containers was assessed after six days.

A replicated, randomized, controlled study in 2012 at a laboratory in Austin, Texas, USA (22) found that cultivating stony coral *Pseudodiploria strigosa* larvae in tanks containing sediment from their collection site and an adult stony coral *Orbicella faveolata* led to a higher uptake of zooxanthellae (beneficial algae) than larvae cultivated without sediment or adult fragment or in natural seawater. Fifty-six days after settled larvae were placed into tanks, uptake of zooxanthellae was significantly higher for larvae in the sediment+coral tanks (100%) compared to the sediment only (67%), coral only (11%) and seawater control (0%). There were no other significant differences between treatments. In August 2012, egg/sperm bundles collected from eight wild-growing *Pseudodiploria strigosa* colonies at Flower Garden Banks reef, Gulf of Mexico, USA, were left in plastic tubs to cross-fertilize before being transferred to a laboratory and to settle onto plastic settlement tiles. In addition, six one-gallon bags of sediment collected from 23 m deep immediately below the coral colonies and a large fragment from an adult *Orbicella faveolata*, collected at the same time, were also taken to the laboratory. Twelve tanks, filled with artificial seawater, were assigned one of four treatments (3 cm layer of sediment + *O. faveolata* fragment; sediment only; coral fragment only; or seawater control). Settled larvae were randomly assigned to one of the 12 tanks. Recruits were monitored daily for a week then every three days for a further 55 days. Uptake of zooxanthellae was assessed using a fluorescent stereomicroscope.

A replicated, controlled study in 2017–2020 in a laboratory near Puerto Morelos Reef National Park, Mexico (23) found that corals *Diploria labyrinthiformis* could be cultivated in an ex-situ setting and 68–88% survived for at least two weeks, with some variation due to whether previously frozen or fresh sperm was used for fertilization. On average, post-settlement polyp survival varied from 76–88% when

fertilized with frozen sperm compared to 68% when fresh sperm was used. The percentage of eggs that yielded swimming larvae was lower for sperm frozen for 30 minutes or 12–13 months (22–26%) compared to fresh sperm (53%), but similar for sperm frozen for one month (40%). Authors present additional results on sperm motility, fertilization and settlement (see paper for details). In 2017 and 2018, egg/sperm bundles were collected from the reef and transported to the laboratory. Eggs and sperm were separated, and sperm were frozen at –80°C for 30 minutes, one month, 12 months or 13 months before being thawed out. Fresh eggs were fertilized with either sperm that had been frozen, or fresh sperm. Resulting larvae were placed in 2 L containers with a settlement substrate (1:2 mixture of white cement and sea sand). The number of eggs that developed into larvae was monitored. Post-settlement survival was assessed after two weeks.

A replicated, randomized, controlled study in 2017 in a laboratory in southern Taiwan (24) found that cultivated stony coral *Pocillopora acuta* larvae survived for at least 7–9 weeks at two different temperatures, but growth rate was not affected by temperature. For larvae sourced from adults held at 26°C, survival was similar at both settlement temperatures (49–56% after 7 weeks). For larvae sourced from adults held at 29.5 °C, survival was similar for all temperature treatments in April (33–50% after 9 weeks), but in May, survival was higher when they settled at 26°C (57% after 7 weeks) than when they settled at 29.5 °C (31% after 7 weeks). Larval growth was not affected by settlement temperature, but larvae from adults held at 26°C were larger in six out of seven comparisons than those from adults held at 29.5°C. In addition, colonies held at 26°C released more larvae than those held at 29.5°C in March and April (26°C: 571–1,160 larvae; 29.5°C: 516–671 larvae,) but released fewer larvae in May (26°C: 693 larvae; 29.5°C: 908 larvae). In February 2017, twenty-four coral colonies (diameter 14 cm) were collected from a reef and transported to an ex-situ laboratory (flow-through tanks). Colonies were randomly assigned to one of two temperature treatments (26°C, average spring temperature; or 29.5°C, above average spring temperature) with 12 colonies/treatment. In March, April and May 2017, coral larvae released from adult colonies were collected on peak release days and split across 24 containers (12 containers/temperature treatment). Containers were randomly assigned to tanks at 26°C or

29.5°C. Each tank contained a ceramic tile, and settlement onto the tile was recorded daily for a week. Survival and growth were then monitored one, three, seven and nine weeks after settlement.

A replicated, controlled study in 2016–2019 in laboratory conditions in Florida and New York, USA (25) found that soft coral *Antillogorgia bipinnata* polyps cultivated at 26°C had higher survival than those cultivated at 30°C. Polyp survival after 52 days was higher at 26°C (48–74%) than at 30°C (15–52%). An average of 40–100% of polyps took up symbionts *Breviolum antillogorgium*, with statistically similar uptake for different temperatures and symbiont genotypes. The number of symbionts/polyp taken up after 69 days varied with temperature and genotype (see paper for details). In 2018, branches of coral were collected from a reef and brought into the laboratory. Coral larvae were collected and settled in polypropylene containers. Coral symbionts of one of five genotypes were added to containers (see paper for details of schedule), and 9–10 containers/genotype were kept at 26°C and 6–8 containers/genotype were kept at 30°C. Symbiont cells had been collected from a reef two years previously and maintained in the lab for two years at 26°C (three genotypes) or 30°C (two genotypes). Survival was recorded every 3–4 days, and polyps were visually inspected to assess uptake of the symbionts.

A replicated, randomized, controlled study in 2017 in a laboratory setting in Queensland, Australia (26) reported that 1–30% of coral larvae settled (depending on the number of species mixed together) and 44–92% of settled larvae survived (depending on the number of settlers and source of water). When a single coral larva settled, survival was similar for corals held in water sourced from tanks containing adult corals (68%) compared to tanks with reef water (70%). When 60 larvae settled, survival was lower with water from coral tanks (44%) than with reef water (92%). Settlement was lower when larvae from two or three species were mixed than when just a single species was used (*Acropora millepora*: 8–10% vs 38%, *Acropora valida*: 5–7% vs 30%, *Leptoria phrygia*: 1% vs 7%). In 2017, colonies of six coral species (*Acropora hyacinthus*, *Acropora millepora*, *Acropora valida*, *Astrea curta*, *Leptoria phrygia* and *Porites cylindrica*) were collected and maintained in flow-through aquaria until spawning. One-litre cylindrical containers with a settlement tile were placed into outdoor raceways and randomly assigned to different water

treatments (water from tanks containing adult corals, or reef water), densities of coral larvae (10, 50, 100), or species (six species), with five replicates for each combination. Survival was assessed 14 days after settlement. In addition, seven containers received 50 larvae of two species (25/species), seven received 51 larvae of three species (17/species), and these were compared to five containers with 50 larvae of a single species. Settlement was assessed after six days.

A replicated, randomized, controlled study in a laboratory in Australia (27) reported 69–96% survival of cultivated corals *Acropora tenuis* over 28–72 days, depending on the temperature and type of symbiont added. At 27°C, survival was lower for corals with heat tolerant symbionts (89%) than for those with natural symbionts (95%) after 28 days, but similar after 72 days (heat tolerant: 80%, natural: 87%). At 31°C or 32.5°C, survival was similar for both types of symbiont at 28 days (93–96%) and 72 days (69–82%). For two measures of growth, corals with heat tolerant symbionts had lower growth than those with natural symbionts in five of 10 comparisons across three temperatures. The study also reports results on symbiont density and performance. Symbiont cells were extracted from wild corals and a heat tolerant strain was developed over 21 generations in laboratory conditions. A mixture of natural strains was also obtained. Coral larvae were settled on aragonite plugs (1,628 plugs), and following uptake of symbionts, transferred to tanks (9 with heat tolerant symbionts, 9 with natural symbionts). Tanks were split evenly between three temperature treatments (27, 31, 32.5°C). Survival and growth were assessed after 28 and 72 days.

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# Transplant coral

## 13.4. Transplant nursery-grown coral onto natural substrate

https://www.conservationevidence.com/actions/4059

- **Twenty-two studies** evaluated the effects of transplanting nursery-grown coral onto natural substrate. Eight studies were in the USA[10,13,14,16,18,20–22], four in the Philippines[1,3,17, 19], two in each of Japan[2,9], and Puerto Rico[5,6], and one in each of Jamaica[4], the US Virgin Islands[7], the Seychelles[8], the Cayman Islands[11], the USA and Puerto Rico[12], and Fiji[15].

**COMMUNITY RESPONSE (1 STUDY)**

- **Richness/diversity (1 study)**: One replicated, site comparison study in the USA[16] found that transplanting nursery-grown colonies of staghorn coral *Acropora cervicornis* onto natural substrate did not increase overall coral diversity.

**POPULATION RESPONSE (22 STUDIES)**

- **Abundance/Cover (2 studies)**: Two studies (one replicated) in Puerto Rico[5], and the USA[16], found that transplanting nursery-grown staghorn coral onto natural substrate led to an increase in coral cover[5] and a higher number of juvenile stony coral species[16].
- **Reproductive success (3 studies)**: Three studies (including one replicated, site comparison) in the Seychelles[8], Japan[9],

and the USA[16] found that after nursery-grown corals were transplanted onto natural substrate they successfully reproduced.

- **Survival (18 studies)**: Sixteen of eighteen studies (twelve replicated, including one randomized, controlled, and one before-and-after) in the Philippines[1,17,19], Japan[2], USA[10,13,14,18,20–22], Jamaica[4], Puerto Rico[5,6], the Cayman Islands[11], USA and Puerto Rico[12], and Fiji[15], found that most nursery-grown corals transplanted onto natural substrate survived[2,4–6,10–15,18,20–22]. The other two studies[1,19] found that most fragments transplanted onto natural substrate died. Two of the studies found that the proportion surviving depended on transplant depth[4,11], and another that survival was higher at lower transplant density[10]. Four studies found that medium[21] and large[6,17,19,21] fragments had higher survival than small, but another found survival did not depend on size[6]. One study found that transplants on substrate taken from a protected area had a higher survival than those on substrate from a fished reef[15].
- **Condition (15 studies)**: Fifteen studies (ten replicated including three randomized) in Japan[2], the Philippines[3,17,19], Jamaica[4], Puerto Rico[6], the US Virgin Islands[7], the USA[10,13,14,18,21,22], the Cayman Islands[11], and the USA and Puerto Rico[12] found that after nursery-grown corals were transplanted onto natural substrate on average they grew. In one study, the amount of growth depended on transplanting density[7], although there were mixed results in another study[10]. Four studies found that medium[21] and large[6,17,19,21] fragments tended to grow more than small fragments, but another study[14] found that small fragments had less bleaching and more live tissue growth than large. One study found that fragments transplanted deeper decreased in size more than those transplanted at a shallower depth[4], and another study found that corals transplanted at 10 m and 16 m depths grew, but those transplanted at 1 m depth decreased in size[11].

**Background**

Corals can be transplanted from a nursery (either ex-situ or in-situ) with the aim of restoring coral populations in the wild. Corals are grown from larvae or fragments and, once colonies are large enough to survive without additional action, they are removed from the nursery and transplanted into the wild (also known as outplanting) onto natural substrate, such as degraded coral reefs or the sea floor. They may be fixed directly onto the substrate using adhesives or tethered with nails, wire or cable ties (Meesters *et al*. 2013). Transplanting corals onto natural, rather than artificial substrates, may be seen as advantageous because corals may be more likely to attach to natural substrates already present in the environment. Transplanting corals onto natural substrates also reduces the risk of pollution caused by man-made materials in the environment.

This action specifically refers to transplanting nursery-grown corals onto natural substrates. Studies that report the effect of transplanting corals onto artificial substrate are described in *Transplanting nursery-grown corals onto artificial substrate*. Studies that report the effect of cultivating corals in nurseries are described in *Cultivate coral fragments in an artificial nursery located in a natural habitat*; and *Cultivate coral larvae in an artificial nursery located in a natural habitat*.

Meesters E.H.W.G., Smith S.R.& Becking L.E. (2013) *A review of coral reef restoration techniques*. Report number C028/14. IMARES: Wageningen, UR. Available from: https://library.wur.nl/WebQuery/wurpubs/fulltext/333153

A replicated study in 1998 in a lagoon in Pangasinan, Philippines (1) found that stony coral *Porites rus* fragments transplanted onto live or dead *Porites cylindrica* colonies had lower survival after 14 weeks than fragments transplanted onto suspended metal grids, and that no *Porites cylindrica* fragments survived transplantation. After 14 weeks, *Porites rus* fragments removed from their nursery grids and transplanted onto live *Porites cylindrica* colonies had higher survival (44%) than those attached to dead

colonies (11%), but both had lower survival than fragments suspended above the reef on grids (86%). No *Porites cylindrica* fragments using any of the three transplantation methods survived. In 1996, twenty-eight *Porites cylindrica* and 25 *Porites rus* fragments were obtained from a reef 1 km from the experiment site, attached to 1 $m^2$ metal grids coated in white epoxy paint, and allowed to grow at 2–3 m depth. In June 1998, fragments were divided with some remaining on the grids (which were then suspended 40 cm above the sandy substrate on metal stakes). The other fragments were removed from the grids and tied onto existing live or dead *Porites cylindrica* colonies using plastic-coated copper wire. The three treatments were replicated at three sites (number of fragments/site not provided). Fragments were monitored every two weeks for 14 weeks.

A study in 2006–2007 at a degraded coral reef site at Akajima Island, Okinawa, Japan (2) reported that the majority of nursery-grown juvenile stony coral *Acropora tenuis* colonies transplanted onto the coral substrate survived and increased in size. After six months, 89% of colonies were still alive, attached to the substrate and had grown from an average diameter of 5.8 cm to 9.1 cm. In December 2006, approximately 2,000 colonies of stony coral *Acropora tenuis* (average diameter 5.8 cm) were collected from an in-situ nursery located at Akajima Island. Colonies were transplanted onto nearby outcrops (2 m high, 6 m deep) and fixed to the degraded coral substrate using pegs and underwater glue. No other methods are reported.

A study in 2008–2011 in Bolinao, northwestern Philippines (3) reported that three out of 12 colonies of stony coral *Acropora millepora* transplanted to a natural reef reached sexual maturity after three years compared to 17 of 19 colonies that remained in a nursery, and sexually mature colonies were larger than non-mature colonies. After three years, sexually mature (with eggs present) colonies had average diameters ranging from 12.3–13.7 cm (transplanted) and 14.4–28.3 cm (nursery-reared) compared to non-mature colonies (transplanted: 7.8–11.0 cm; nursery-reared 11.8–13.5 cm). Data were not statistically tested. In 2008, stony coral spat (settled larvae) on artificial settlement tiles in an outdoor nursery were transferred to an in-situ nursery. After six months, 12 colonies were transplanted onto a natural reef and 19 colonies remained in the nursery. After three years, the number of sexually mature colonies (colonies with eggs) was counted, and all colonies were measured.

A randomized, replicated study in 2007–2008 at two reef sites in Montego Bay, Jamaica (4) found that transplanting nursery-grown staghorn *Acropora cervicornis* fragments at a shallower depth led to higher survival, and less decrease in length than fragments transplanted deeper. After 11 months, 28 of 36 (78%) transplanted at the shallow site had survived (8 were lost), and 7 of 29 (24%) transplanted at the deeper site survived (11 died, 11 were lost). Partial mortality (measured as a % live tissue loss and decrease in total length of live tissue) was lower for shallow fragments (47%, 2.5–13 m) than deeper (91% from 17–1.5 m). The average number of polyps/fragment was lower after 11 months (shallow: 7/fragment, deep: 6/fragment) compared to the start (shallow: 9/fragment, deep: 6/fragment) (data not statistically analysed). In September 2007, sixty-five staghorn fragments (5 cm) were selected at random from an established in-situ line nursery and each cable-tied to a nail driven into the rock or dead coral substrate 3–5 m or 12.5–14 m deep. Survival was recorded, growth (length of live tissue) measured and the number of polyps/fragment counted three times in the first four months then again after 11 months using scaled photographs.

A before-and-after, site comparison study in 2006–2014 at a damaged coral reef site in Tallaboa, Puerto Rico (5) reported that following transplanting nursery-grown, along with wild-grown, fragments of staghorn *Acropora cervicornis* coral onto stabilized natural substrate, the area of restored reef increased. After eight years, the area of restored reef had increased from 70 $m^2$ to 180 $m^2$. Coral colonies in unrestored areas in the vicinity, with loose rubble and damaged substrate, showed no signs of recovery during the same period. It was not possible to determine from the study how much of the recovery was attributable to transplanting nursery-grown fragments, transplanting wild-grown fragments, or stabilizing the substrate. In 2006, following the destruction of a coral reef by a ship grounding, wire cages and metal stakes were used to stabilize a 70 $m^2$ area of damaged reef. In the same year, approximately 227 (10–20 cm) fragments of staghorn coral were collected from nearby reefs and attached to the substrate using cement puddles. In 2009–2011, approximately 400 (20–40 cm) fragments of staghorn coral were collected from a nursery and attached to the substrate using masonry nails, cable ties and/or epoxy. Coral recovery was measured using photos taken in 2006 and 2014 as well as aerial imagery in 2014. No other methods are reported.

A replicated study in 2011–2012 on two coral reefs off Culebra Island, Puerto Rico (6) found that a year after transplanting nursery-grown staghorn coral *Acropora cervicornis* fragments onto natural substrate, there was no difference in survival between large and small fragments, but large fragments grew faster. One year after transplanting, there was no difference in average survival between large (62%) and small (57–68%) coral fragments. However, large fragments on average grew more quickly (0.3–0.4 cm/day) and produced more branches (9–14 branches/year) than small fragments (0.1–0.2 cm/day; 6–11 branches/year). In May 2011, large (>25 cm) and small (<25 cm) *Acropora cervicornis* branches were clipped from colonies grown in two in-situ nurseries and transplanted to two nearby reefs. The branches were attached directly to the natural substrate (3–4 m deep) with concrete nails and plastic cable ties. Survival was monitored one month later and every three months thereafter for one year in total. Growth was calculated using photographs taken at transplantation and at the final survey.

A study in 2012 at a coral nursery at Cane Bay, US Virgin Islands (7) found that transplanting fragments of nursery-grown staghorn coral *Acropora cervicornis* onto natural substrate at lower densities resulted in higher total linear growth, secondary branch growth and numbers of new branches than those transplanted at higher densities. After three months, average linear growth ranged from 15 cm (1 fragment/plot) to 5 cm (16 fragments/plot). Secondary branch growth decreased as the number of fragments/plot increased ranging from 13 cm (1/plot) to 4 cm (16/plot), but there was no significant difference in primary branch growth (1.2 cm for 7/plot to 3.5 cm for 18/plot). The number of new branches recorded declined with increasing density from five (1 fragment/plot) to one (18/plot). In April 2012, one hundred and forty-six fragments of staghorn coral (average length 6.5 cm) were collected from a nursery and attached vertically at their base onto bare reef using marine epoxy. Fragments were arranged in clusters from 1–12, 14, 16, 18, 20 individuals/plot (16 plots, 146 fragments) each fragment 5 cm apart from its neighbour. Primary and secondary branch growth, and new branch development were measured after three months.

A before-and-after, site comparison study in 2012–2014 at a coral reef off Cousin Island, Seychelles (8) found that transplanting fragments of nursery-grown stony coral *Acropora* and *Pocillopora* onto natural substrate

led to a higher density of settled larvae (coral spat) than at healthy or degraded sites without transplants, and a higher number of juvenile corals than the degraded site. Twenty months after transplantation, density of all coral spat was higher at the transplantation site (124 spat/$m^2$) than at nearby healthy (68 spat/$m^2$) and degraded (78 spat/$m^2$) sites without transplants. However, 24 months after transplanting, the transplantation site had lower density of all juveniles (3 juveniles/$m^2$) than the healthy site (5 juveniles/$m^2$) but higher than the degraded site (2 juveniles/$m^2$). In November 2012–June 2014, a total of 24,431 nursery-grown coral colonies of seven *Acropora* and four *Pocillopora* species were transplanted at a 0.52 ha area of degraded reef (<3% coverage of corals in 2012, 16% in 2014). Transplantation methods not provided. A 0.12 ha healthy site (14% coverage in 2012, 35% in 2014) and a 0.13 ha degraded site (<3% coverage in both years), both without transplantations, were chosen for comparison. Sites were adjacent, 50 m apart from their neighbouring site. In January 2014, forty ceramic tiles were placed 8–10 m deep at each site, retrieved in July 2014 and inspected for coral spat of any species and returned to the sites. Juvenile corals were surveyed along six 10 m transects and in three randomly placed 1 $m^2$ quadrats at each site before transplantation started, and at 12, 18 and 24 months after.

A study in 2015 at a coral reef restoration site in Maeganeku, Japan (9) reported that nursery-grown colonies of stony coral *Acropora tenuis* outplanted onto natural substrate spawned. In June 2015, twenty-five nights after the full moon, almost all 2,800 transplanted *Acropora tenuis* colonies released egg/sperm bundles. Since 1998, more than 40 stony coral species have been outplanted on the periphery of existing coral reefs. Methods not reported.

A replicated, randomized, controlled study in 2013–2014 at a coral reef site off Plantation Key, Florida, USA (10) found that transplanting staghorn coral *Acropora cervicornis* fragments at lower density led to a higher survival rate compared to fragments transplanted at higher densities, but results for growth were mixed. Thirteen months after transplanting, fragment survival was higher in the 3 fragments/plot treatment (100%) compared to the 12-clumped/plot and 24/plot (both 58%) and higher in the 6/plot (84%), 12/plot (88%) and 12-clumped/plot compared to 24/plot. Daily growth rate (skeletal extension) did

not vary between treatments until the final survey period when growth was higher for 12/plot fragments (0.82 cm/day) compared to 24/plot (0.44 cm/day). There was no difference in daily growth rate between 3 (0.67 cm/day), 6 (0.68 cm/day), and 12-clumped (0.82 cm/day) plots compared to 24/plot (0.44 cm/day). In May 2013, twenty-four 4 $m^2$ plots were marked on a reef 5–7 m deep. Staghorn coral fragments (~85 cm long) were transplanted from a nearby nursery and attached to the substrate using marine epoxy in densities: 3/plot (0.75 corals/$m^2$); 6/plot (1.5/$m^2$); 12/plot (3/$m^2$); 12-clumped/plot (12/$m^2$); and 24/plot (6/$m^2$). Fragments were evenly distributed within each plot except the 12-clumped which were placed within 1 $m^2$ inside the plot. Each plot had four replicates and an additional four plots were left without transplants as controls. Plots were surveyed in August and December 2013 and June 2014. Growth was measured as total skeletal extension (length, width, and height) of all fragments. Survival (% fragments alive) was recorded at each survey.

A controlled study in 2015 on a reef in Little Cayman, Cayman Islands (11) found that transplanting nursery-grown staghorn coral *Acropora cervicornis* fragments onto natural substrate at 10 m deep led to higher survival and similar growth to those at 16 m but similar survival and greater growth than those at 1 m. Survival 85 days after transplantation was higher amongst staghorn coral fragments at 1 m (100%) and 10 m (95%) deep, than those at 16 m deep (60%). However, those at 1 m on average decreased in height (–4 cm), whereas those at 10 m and 16 m had similar average increases (12 cm and 7 cm, respectively). Sixty 11–33 cm staghorn coral fragments grown in an in-situ nursery since 2012 were transplanted in May 2015 in plots containing 10 fragments each at depths of 1 m, 10 m and 16 m (two plots/depth). Fragments were attached with cable ties to nails embedded in the sea floor in a 1 × 4 m grid, and then to the floor with epoxy putty. Plots were photographed approximately every 30 days for 85 days.

A review of six restoration projects established in 2007–2010 at locations in Florida and Puerto Rico, USA, (12) reported that most nursery-grown staghorn coral *Acropora cervicornis* fragments transplanted to a natural substrate survived and grew. After one year, 85% (range 75–93%) of fragments survived and after two years (at the 3 sites monitored over that period), survival rate was 67–80%. Growth

rate varied between sites from 26–81 cm/year (average 46 cm/year). The paper presents survival and growth results from six projects that transplanted nursery-grown staghorn coral fragments. Fragments of staghorn coral were cultivated and transplanted to nearby reefs, attached to the substrate using nails, cable ties, and/or epoxy (see paper for details). Survival (including fragments with partial tissue loss) was determined by counting the number of fragments with some live tissue. Growth (total linear extension) was measured using a flexible ruler. Fragments were monitored for 1 year (3 of 6 projects) and 2 years (3 of 6 projects).

A replicated study in 2016–2017 at two coral reef sites in Florida, USA (13) found that most nursery-grown staghorn coral *Acropora cervicornis* colonies transplanted onto the natural reef survived and increased in size. One year after transplanting, the average survival rate of fragments did not differ between the two sites (86 and 92%). Although the average size of colonies at transplanting did not differ between sites (site 1: 171 mm; site 2: 189 mm), size was higher at site 1 (751 mm) than site 2 (598 mm) one year later. In May 2016, one hundred nursery-grown staghorn colonies (100–200 mm with at least two branches) were transplanted 4–5 m deep on two coral reefs. At each site, colonies were arranged in two arrays (~3–5 m apart) each with five clusters (~0.5–1 m apart) of five colonies (~10–15 cm apart). Colonies were attached to the substrate using masonry nails and cable ties. Survival was recorded after one year. Growth (measured as total linear extension including all branches) was recorded approximately three, six, and 12 months after transplanting.

A replicated study in 2014–2016 at three coral reef sites off the Florida coast, USA (14), found that transplanted smaller fragments of elkhorn *Acropora palmata* experienced less bleaching and produced more live tissue than larger fragments, but both had similar survival. Three months after transplanting, all fragments showed some signs of bleaching. However, the percentage of fragments partially or >90% bleached white was lower for small fragments (part-bleached: 30–76%; >90% bleached: 8–18%) than large fragments (part-bleached: 65–83%; >90% bleached: 16%). After 30 months, increases in live-area-index (a proxy for size) were greater for smaller fragments (214 $cm^2$) than larger fragments (103 $cm^2$). There was no difference in survival between fragments after 30 months (small:

27%; large 26%). In May 2014, small (average 51 $cm^2$ live-area-index – see paper for detail) and large (average 108 $cm^2$ live-area-index) fragments of elkhorn coral (126 of each size) were transplanted in pairs 1 m apart across three fore-reef sites. Size was measured as live-area-index (see paper for details). Survival and increase in live-area-index were measured after 1, 6, 13, and 30 months. Two additional surveys were carried out during an extreme thermal stress event in August and September 2014 to assess condition and record bleaching between 0 (no bleaching) – 4 (>90% of the fragment completely bleached).

A replicated study in 2014 in one protected coral reef and one fished reef off Viti Levu, Fiji (15) found that transplanting nursery-grown corals *Pocillopora damicornis* onto natural substrate resulted in higher survival for juveniles when they were transplanted on substrate taken from and transplanted to the protected area compared to on substrate taken from and transplanted to the fished reef. Survival was higher for settled larvae on substrate taken from and transplanted to the protected area (day 4: protected area larvae: 49%, fished reef larvae: 64%, day 26: protected area larvae: 22%, fished reef larvae: 39%) than on substrate taken from and transplanted to the fished reef (day 4: protected area larvae: 12%, fished reef larvae: 29%, day 26: protected area larvae: 5, fished reef larvae: 8%). In addition, survival was lower for juveniles transplanted to the fished area on substrate fouled with macroalgae (day 4: 15%, day 26: 9%) compared to on unfouled substrate in the fished area or unfouled substrate in the protected area (day 4: 43−51%, day 26: 22−28%). In 2014, fragments of coral colonies were collected from 12 colonies from a protected area and 12 from an adjacent fished reef (100–500 m between protected and fished areas). Larvae from adults from the protected and fished areas were added to separate plastic dishes (10 larvae/dish) and settled on substrate gathered either from the protected area or fished reef (20 dishes/treatment). Settled larvae on either substrate were transplanted after four days either to the protected area or fished reef (13−18 pieces of substrate/treatment) and attached using nails and cable ties. Additionally, larvae on 14–15 pieces of substrate were transplanted in each of the three treatments: transplanted to fished area with or without macroalgae on the substrate or transplanted to the protected area with no macroalgae. Survival was assessed after four and 26 days.

A replicated, site comparison in 2014 in four coral reefs in Florida, USA (16) found that transplanting nursery-grown colonies of staghorn coral *Acropora cervicornis* onto natural substrate led to a higher abundance of *Acropora* coral species juveniles and, at one of four reefs, a higher abundance of non-*Acropora* coral juveniles than plots without transplants but did not increase overall coral diversity. There was higher coral cover on plots with transplants (5–15%) than plots without (1–3%), but this was mostly due to increases in *Acropora* species, which made up 78–89% of corals in plots with transplants and 0–7% in plots without. At one of four reefs there was a higher abundance of non-*Acropora* juvenile corals on sites with transplants than those without (Pickles Reef: with transplants: 4 corals/50 $m^2$, without: 1 coral/50 $m^2$) but at the other three there was no difference (with transplants: 2–18 corals/50 $m^2$, without: 2–15 corals/50 $m^2$). There was no difference in coral diversity between plots with transplanted *Acropora cervicornis* and those without (presented as Shannon-Weiner index). The four reefs in the Florida Keys National Marine Sanctuary had undergone limited *Acropora cervicornis* transplanting from 2–11 years prior (<100 corals transplanted), but more extensive transplanting since 2011 (Molasses Reef: 2,300 corals), 2012 (Pickles Reef: 1,150 corals, Snapper Reef: 500 corals) or 2013 (Conch Reef: 500 corals). Nursery-grown coral colonies were transplanted onto reefs using epoxy putty. In July–August 2014, at each reef, five 25 m transects were swum in an area with transplants and five in an area without (≥5 m away) to record coral abundance and species.

A replicated study in 2015–2016 in two coral reef sites in Lingayen Gulf, Philippines (17, same experimental set-up as 19) found that a year after transplanting nursery-grown stony coral *Acropora verweyi* onto natural substrate, large transplants had higher survival rates than small transplants at one of two sites. After a year, large transplants had higher survival rates than small transplants at one of two sites (site 1: 32% large vs 14% small, site 2: 36% for both). Large transplants also grew more than small transplants (large: 13 mm/year, small: 9 mm/year). In 2015, eleven *Acropora verweyi* colonies were collected and transplanted to an ex-situ setting. All colonies were placed in a plastic tank for spawning, and egg/sperm bundles were collected and settled on dead coral rubble. Four months after fertilization, 240 pieces of coral rubble with a single coral

colony (120 large: 10–15 mm at time of transplant; 120 small: 3–5 mm at time of transplant) were transplanted to one of two sites, distributed evenly between four bommies (coral outcrops) at each site and inserted into drilled holes with putty. Survival and growth were monitored 62, 93, 185, 278, and 376 days post-transplant. **Costs (US$)**: Transplanting nursery-grown coral in 2015 cost $2.67 for each transplanted juvenile, and $11.49 for each transplanted coral that survived for one-year post-transplant. Costs included surveys and collection of donor corals, ex-situ cultivation and rearing, transplanting and monitoring over one year.

A replicated, randomized study in 2016–2017 at a coral reef in Florida, USA (18) found that the majority of nursery-grown colonies of three stony coral species transplanted onto natural substrate survived, and surviving colonies of one of three coral species increased in size, while the other two decreased. After 17–18 months, 46 of 60 (77%) staghorn coral *Acropora cervicornis* colonies, 43 of 60 (72%) great star coral *Montastraea cavernosa* colonies, and 55 of 60 (92%) mountainous star coral *Orbicella faveolata* colonies survived. On average, surviving staghorn coral colonies increased in volume by 1,015%, whereas great star coral and mountainous star coral colonies decreased in surface area by 23% and 11%, respectively. Staghorn coral colonies (66–575 $cm^3$) were collected from an ex-situ nursery, and great star (45–120 $cm^2$) and mountainous star (38–130 $cm^2$) coral colonies from an in-situ nursery. In March 2016, sixty colonies of each species were transplanted onto the hard substrate of a coral reef (≥2 m apart, 8 m deep) using nails and zipties or cement and Plaster of Paris. Areas around half of the transplant sites were cleared of algae and zoanthids *Palythoa caribaeorum*. After 17–18 months, surviving colonies were counted and measured in the field or from photographs.

A replicated study in 2015–2019 in two coral reef sites in Lingayen Gulf, Philippines (19, same experimental set-up as 17) found that transplanting nursery-grown stony coral *Acropora verweyi* onto natural substrate resulted in 18% of corals surviving for four years, with higher survival for larger transplants at one of two sites. Survival was higher for larger transplanted corals than smaller corals at one site (22% of large vs 15% of small) but survival was similar at the other (15% of large vs 12% of small). Average diameter after four years was 16 cm and did not differ for larger or smaller transplanted corals. In 2015, eleven *Acropora*

*verweyi* colonies were collected and transplanted to an ex-situ setting. All colonies were placed in a plastic tank for spawning, and egg/sperm bundles were collected and settled on dead coral rubble. Four months after fertilization, 240 pieces of coral rubble with a single coral colony (120 large: 1–2 cm at time of transplant; 120 small: 0.3–0.5 cm at time of transplant) were transplanted to one of two sites, distributed evenly between four bommies (coral outcrops) at each site and inserted into drilled holes with putty. Survival and size were assessed in June 2019, four years after transplant.

A study in 2019 at a coral reef restoration site off Florida, USA (20) found that transplanting nursery-grown staghorn coral *Acropora cervicornis* onto natural substrate resulted in most surviving for at least four months, with no difference between methods of attachment. Tissue mortality was similar for corals transplanted using cement, or nails and cable ties (0–27% partial mortality, 0–13% full mortality). Transplants using a range of cement mixes or epoxy found average tissue mortality of 2% (cement) or 0% (epoxy) after eight days, with no additional mortality after one month and recovery after five months. Divers were able to transplant around 11 corals/dive using cement compared to six/dive using nails and cable ties (not tested for statistical significance). A total of 225 coral fragments were used to compare a range of cement mixes and epoxy. Five bases (8–10 cm diameter) were deployed for each mix, and three fragments were placed on each base. Survival was assessed after eight days and then again at one and five months. A further 50 fragments were used to compare the best performing cement with the nail and cable tie method (25 fragments/method) and coral survival was assessed after one and four months. **Costs (US$)**: Transplanting nursery-grown coral in 2019 cost $0.05/coral when using cement, $0.47 when using epoxy and $0.50 using the nail and cable tie method. Costs included materials only and did not include any shipping costs for materials.

A replicated study in 2012–2018 at around 68 coral reef sites across the Florida reef tract, USA (21) found that after transplanting nursery-grown staghorn corals *Acropora cervicornis* fragments onto natural substrates, medium and large fragments had higher survival than small fragments, and survival increased with latitude. Survival was higher for medium and larger coral fragments (65–67 % after 800 days)

than smaller fragments (51% after 800 days). Survival increased with latitude of transplant site (48% at 24.5°N, 85% at 26.5°N after 800 days). In addition, authors reported differences in survival due to the specific reef habitat but no differences in survival due to attachment method or genetic diversity of coral transplants. Authors collated data from six coral transplanting programs on survival for a total of 22,634 corals transplanted in 2012–2018 (405–15,917 corals/program). Corals were raised in nurseries along the Florida reef tract and transplanted out to six natural reef habitats using nails and cable ties or epoxy. Survival was monitored one month and one-year post-transplant, and at some sites annually for four years. Corals were grouped by size (small: 1–15 cm, medium: 16–50 cm, large: 51–160 cm) for analysis.

A replicated study in 2017–2018 at two reef sites and an in-situ nursery site on the Florida Reef Tract, USA (22) reported that transplanting nursery-grown staghorn coral *Acropora cervicornis* onto natural substrates resulted in survival and growth over at least 480 days. Overall, 83 of 120 transplanted colonies (69%) survived for at least 480 days. Survival differed across sites, with highest survival in the nursery (98% of 40 survived), followed by Tennessee reef (83% of 40), then Cheeca rocks (28% of 40). Colonies grew at all sites, and average size after 480 days ranged from 99–156 $cm^3$ at reef sites to 12,720 $cm^3$ at the nursery site. Egg/sperm bundles were gathered from an in-situ nursery, settled on tiles, and moved to an ex-situ aquaculture facility where they were allowed to grow for 20 months. Three transplant sites were selected: two reefs, and one in-situ nursery. In 2017, corals were fragmented, and 40 fragments/site were attached directly to the reef using a masonry nail, epoxy and cable tie. Colonies were monitored approximately two weeks, one month, three, six, and sixteen months after transplanting.

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## 13.5 Transplant nursery-grown coral fragments onto artificial substrate

https://www.conservationevidence.com/actions/4060

- **Fifteen studies** evaluated the effects of transplanting nursery-grown corals onto artificial substrate. Five studies were in the USA[2a,b,7,13,14], four in the Philippines[3,4,6,10], three in Israel[1,5,8] and one in each of Japan[9], Singapore[11] and Curaçao [12].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (15 STUDIES)**

- **Reproductive success (3 studies)**: Three studies (including one replicated and one controlled) in Israel[1,8] and Curaçao[12] found that after transplanting/outplanting nursery-grown corals onto artificial substrate some produced eggs[1,12] and sperm[12], and released a higher number of larvae than wild growing corals[8].
- **Abundance/cover (2 studies)**: One replicated, controlled study in the Philippines[6] found that transplanting coral fragments onto concrete structures at higher density did not result in an increase in natural coral settlement compared to transplanting at lower density or on structures without transplants. One replicated study in Japan[9] found that coverage of transplanted corals on ceramic tiles was highest when tiles were shaded, caged and facing up.
- **Survival (11 studies)**: Eleven replicated studies (including three randomized and one controlled study) in Israel[1,5] the USA[2b,7,13,14], the Philippines[3,4,10], Singapore[11], and Curaçao[12] found that nursery-grown corals transplanted onto artificial substrate survived. Three of the studies found that survival varied by coral species[4,10] or the substrate that corals were transplanted/outplanted onto[1,4]. Five of the studies found that survival was higher for larger fragments[3,7], fragments cultivated in the nursery for longer before transplantation[10] , for soft and stony corals transplanted onto metal racks rather than onto a natural reef[5], or for corals transplanted onto an

artificial reef rather than in an aquarium[2b] and, for one species, than fragments transplanted onto a natural reef[2b].

- **Condition (5 studies)**: Five studies (three replicated including two randomized) in the USA[2a], Philippines[3], Singapore[11], Curaçao[12] and the USA[14], found that most nursery-grown corals transplanted/outplanted onto artificial substrate that survived, grew. One of the studies[2a] found that growth was higher in the second year after transplanting compared to the first.

**Background**

Nursery-grown corals are cultivated from larvae or fragments collected from wild colonies or taken from existing nursery colonies. Once colonies are large enough to survive without additional intervention, they are removed from the nursery and transplanted onto substrates in the wild (also known as outplanting). Transplanting/outplanting corals from the nursery onto artificial rather than natural substrates can provide additional stability to the corals as substrates can be designed specifically to accommodate coral fragments. Transplanted/outplanted corals can be fixed to the artificial substrate using a variety of attachment methods such as cable-ties, epoxy, cement, nails or wire. Substrates are designed to be permanent or semi-permanent structures made from materials such as metal, PVC, bamboo, or concrete, which can be free-standing, attached to the seabed or used to stabilize degraded reefs. Although artificial substrates provide stability for transplanted coral, there is a danger that the materials used could degrade and pollute the habitat.

This action specifically refers to transplanting nursery-grown corals onto artificial substrates. Studies that report the effect of transplanting corals onto natural substrates are described in *Transplant nursery-grown corals onto natural substrate; Transplant wild-grown corals onto artificial substrate; Transplant wild-grown corals onto natural substrate*; and *Change transplant attachment method*. Studies that report the effect of cultivating corals in nurseries are described in *Cultivate coral in an artificial nursery located in a natural habitat*.

A controlled study in 1997–1998 on a coral reef in Eilat, Israel (1) found that nursery-grown stony coral *Stylophora pistillata* branches transplanted onto plastic crates had higher survival than those transplanted onto cement tiles, and fragments on plastic crates produced eggs but there was no difference in survival between branches transplanted onto plastic crates placed at 5 m or 10 m depth. After six months, there was higher survival of coral branches transplanted onto plastic crates (83%) than those transplanted onto cement tiles (25%). After 18 months, survival of branches on crates was 60% (survival on tiles not recorded). Of branches on crates, there was no difference in survival between those at 5 m and 10 m (data not provided). All branches removed from their colonies during larval release had eggs 18 months after removal (0.8–1.5 eggs/polyp), and 67% removed during gonad development had eggs 12 months after removal (0–1.8 eggs/polyp). In 1997, *Stylophora pistillata* coral branches were cut from 5–10 m depth in-situ donor colonies at the Marine Biology Laboratory (MBL) in the Red Sea, either during larval release (April) or during gonad development (October). A total of 310 branches (~60 branches/crate) were placed on plastic crates (1 × 0.5 × 0.4 m, with 1 $cm^2$ mesh), which were fastened to the reef at MBL at 5 m (two crates) or 10 m depth (three crates). Sixty branches were attached to clips glued onto cement tiles (10 branches/tile), which were attached either 0.5 m above or directly onto the reef at a depth of 10–12 m. One to three branches from the April and October removals placed on crates were removed in November 1998 (18 and 12 months after removal, respectively), and inspected for female gonads to

determine reproductive status. Survival of corals on tiles was recorded at one, three and six months, whereas survival of corals on crates was recorded at six, 12 and 18 months.

A randomized, replicated study in 1996–1999 at a coral reef site in Florida, USA (2a), found overall skeletal growth of nursery-grown colonies of *Montastraea faveolata* transplanted onto ceramic pedestals on the reef was higher in the second year after transplanting compared to the first year, but there was no difference in skeletal growth between fragments attached horizontally, vertically or in between. Skeletal growth was higher in the second year after transplanting (65.1 mg/day) compared to the first (32.2 mg/day). There was no difference in skeletal growth between transplanted colonies attached to the reef horizontally, vertically, or in between (data not reported). In May 1996, twelve cores (5.1 cm diameter) were taken from each of three wild-growing colonies of *Montastraea faveolata* and transported back to a land-based aquarium. Cores were fixed to mushroom-shaped ceramic pedestals using epoxy before being placed into aquarium tanks. After one year, pedestals were taken to a nearby degraded natural reef and transplanted into holes drilled into the substrate and secured using epoxy. Pedestals were randomly assigned to be transplanted horizontally, vertically, or in between. In May 1998 and May 1999 pedestals were removed from the site and taken to the lab to be weighed.

A randomized, replicated study in 1997–1999 at a natural reef and an aquarium in Florida, USA (2b), found that transplanting fragments of stony coral *Montastraea faveolata* onto ceramic pedestals and staghorn coral *Acropora cervicornis* onto limestone blocks then onto an array on a natural reef led to longer survival than fragments of either species transplanted directly onto the natural reef substrate or in an aquarium. After 21 months, survival of *Montastraea faveolata* was higher for fragments on ceramic pedestals on the array (8/12 survived) compared to those on pedestals placed directly onto the reef substrate (1/12) or in an aquarium (3/12). There was no statistical difference in survival between direct natural reef and aquarium fragments of *Montastraea faveolata*. After 18 months, survival of staghorn coral was higher for those on limestone blocks on the array (2/12) compared to on limestone block in the aquarium (0/12), but no difference compared to fragments on limestone blocks placed directly onto the reef substrate (7/12). In

August 1997, twelve 2.5 cm cores were taken from each of three colonies of *Montastraea faveolata* and attached to mushroom-shaped ceramic pedestals using epoxy. In November 1997, twelve fragments (~7 cm long) were taken from each of three colonies of staghorn coral and fixed to limestone blocks. Corals attached to ceramic pedestals or limestone blocks were randomly selected to be placed on a 5.5 m deep array (no detail provided) or placed directly onto the natural reef substrate or placed in the aquarium. Survival was recorded every three months for 21 months (*Montastraea faveolata*) and 18 months (staghorn coral).

A randomized, replicated study in 1997–1998 at a laboratory and natural coral reef in central Philippines (3), found that transplanting larger nursery-grown juvenile stony coral *Pocillopora damicornis* onto limestone settlement tiles led to higher survival and growth rate than smaller juveniles. One year after transplanting, juvenile survival rate was highest (40/80) for the largest size class, compared to the next largest (16/80), the third largest (2/80), and smallest (0/80). Average size after one year also varied depending on the size at transplant (largest: 51 mm; next largest: 26 mm; third largest 8 mm; smallest: 0 mm). Each four months between February and July 1997, five *Pocillopora damicornis* colonies were collected from the wild and kept in laboratory aquarium tanks to spawn. Larvae were collected and placed into tanks to settle onto limestone settlement tiles (48 $cm^2$). Eighty settled larvae/month were placed into cultivation tanks with flowing unfiltered seawater. In August 1997, three hundred and twenty juvenile corals attached to individual tiles were measured and sorted into one of four size classes ($\leq$3 mm; 3.1–6.0 mm; 6.1–10 mm; >10 mm). Tiles were taken to the reef and attached to the substrate 4 m deep using marine epoxy. Survival and growth of juveniles were monitored for one year after transplanting.

A replicated study in 1998 in a lagoon in Pangasinan, Philippines (4) found that nursery-grown stony coral *Porites rus* fragments transplanted onto a reef on metal grids had higher survival than fragments removed from their grids and transplanted directly onto live or dead *Porites cylindrica* colonies, and no *Porites cylindrica* fragments survived transplantation. After 14 weeks, *Porites rus* fragments attached to the reef on metal grids had the highest survival (86%), and of those removed from their grids, those transplanted onto live *Porites cylindrica* colonies had higher survival (44%) than those transplanted

onto dead colonies (11%). No *Porites cylindrica* fragments using any of the three transplantation methods survived. In 1996, twenty-eight *Porites cylindrica* and 25 *Porites rus* fragments were obtained from a reef 1 km from the experiment site, attached to 1 $m^2$ metal grids coated in white epoxy paint, and allowed to grow at 2–3 m depth. In June 1998, fragments were divided with some remaining on the grids (which were then suspended 40 cm above the sandy substrate on metal stakes). The other fragments were removed from the grids and tied onto existing live or dead *Porites cylindrica* colonies using plastic-coated copper wire. The three treatments were replicated at three sites (number of fragments/site not provided). Fragments were monitored every two weeks for 14 weeks.

A replicated study in 2004 at an artificial and natural coral reef site in Eilat, Israel (5) found that survival was higher for soft coral *Dendronephthya hemprichi* and stony coral *Pocillopora damicornis* fragments transplanted onto PVC plates on an artificial reef compared to on an adjacent natural reef, but there was no difference for fragments transplanted in different orientations or directions. After 20 days, fragments on the artificial reef had a higher survival rate (soft coral: 73%; stony coral 87%) than fragments on the natural reef (soft coral: 44%; stony coral: 24%). There was no difference in survival between soft coral fragments on horizontal plates compared to vertical, or inward rather than outward facing plate surfaces (horizontal: inner 81%, outer 66%; vertical: inner 66%, outer 71%). In February 2004, small fragments (0.5 cm) of nursery-grown soft coral *Dendronephthya hemprichi* (6–10 colonies/plate) and stony coral (4/plate) *Pocillopora damicornis* were fixed to four (soft coral) or five (stony coral) PVC plates using superglue. Plates were fixed to 400 × 20 × 10 cm metal racks either horizontally or vertically and attached to the outward (away from the reef) or inward (towards the reef) facing side of the rack. Racks were attached, 14 m deep, to the artificial reef (comprising a PVC fence anchored to the sea floor) or adjacent natural reef. Survival was checked on each plate after one, two, four, six, 13 and 20 days.

A replicated, controlled study in 2007 near a coral reef in Bolinao, northwestern Philippines (6) found that transplanting stony coral fragments on concrete reef structures at higher densities (with or without topshell snails *Trochus niloticus* added) did not lead to higher natural coral settlement. Five months after transplanting, there was

no significant difference in the average density of coral spat (settled larvae) on structures with fragments transplanted at high density (with topshells: 8/m$^2$; without topshells: 22/m$^2$), low density (with and without topshells: 19/m$^2$), or with no transplanted fragments (with topshells: 30 spat/m$^2$; without topshells 16/m$^2$). Overall, a total of 2,189 coral spat were recorded with 85% being pocilloporids, 8% poritids, 4% acroporids and 6% unidentifiable. In January 2007, nursery-reared fragments (~5 cm diameter) from five stony coral species (*Pocillopora damicornis, Acropora muricata, Porites cylindrica, Montipora digitata*, and *Echinopora lamellosa*) were transplanted onto 42 concrete pallet balls (1.2 m diameter, 0.9 m high) (see paper for full design). Fragments were transplanted onto 14 pallet balls at low density (five fragments/species/ball: 9.5 fragments/m$^2$) and 14 at high density (10 fragments/species/ball: 19 fragments/m$^2$) fragments. The final 14 balls had no fragments attached (as a control). Half the pallet balls also had topshell snails added (10/ball). Pallet balls were placed 4–8 m deep on sandy substrate 3–5 m from a natural coral reef. Coral spat was counted on each pallet ball after approximately five months.

A replicated study in 2009 at coral nursery in Biscayne National Park, Florida, USA (7) reported that larger nursery-grown fragments of staghorn *Acropora cervicornis* coral transplanted onto ceramic disks had higher survival than smaller fragments. After 24–39 days, 25 of 27 (93%) larger (3.5 cm) fragments survived compared to 13 of 27 (48%) smaller (2.5 cm) fragments. In June 2009, thirty branch-tip fragments (15 × 2.5 cm and 15 × 3.5 cm length) were collected from 15 nursery-grown colonies of staghorn coral. These were taken to a boat and kept in buckets of water whilst being attached individually to ceramic disks using epoxy. In July 2009, twenty-four branch-tip fragments (12 × 2.5 cm and 12 × 3.5 cm length) were collected from the same nursery. These were attached to ceramic disks whilst underwater. Ceramic disks were attached to a PVC frame using cable ties and placed on the substrate 5.5 m deep within the nursery. Survival was recorded after 39 days (June fragments) and 24 days (July fragments).

A replicated, controlled study in 2005–2010 at five knolls in Eilat, Israel (8) found that nursery-grown stony coral *Stylophora pistillata* colonies transplanted onto natural substrate released a greater number of larvae in most cases compared to wild-grown resident colonies. During three

reproductive seasons in each of two trials, greater numbers of larvae were released on average by transplanted nursery-grown colonies (4–23 larvae/colony) than wild-grown resident colonies (0–2 larvae/colony). In a third trial, transplanted nursery-grown colonies released more larvae than resident colonies during the first reproductive season (2 vs 12 larvae/colony), but the difference was not significant in the second season (2 vs 6 larvae/colony). In November 2005, May 2007 and September 2008, *Stylophora pistillata* colonies reared for 8–24 months in a floating nursery were transplanted onto five bare knolls, along with six other branching or stony coral species (total 1,400 colonies). Colonies were attached using pegs and masonry anchors inserted into drilled holes and secured with epoxy glue. During three reproductive seasons in 2007, 2009 and 2010, collection devices were placed over 20–54 transplanted and 10–40 nearby resident *Stylophora pistillata* colonies for several nights from sunset to sunrise. Collected larvae were counted using a dissecting microscope.

A replicated study in 2008–2010 at a coral reef in Okinotorishima, Japan (9) found that transplanting nursery-grown stony coral *Acropora tenuis* on unshaded, upward-facing ceramic tiles covered in cages led to greater coral coverage compared to shaded, downward-facing or uncaged tiles. After 22 months, average coral cover was greater on unshaded, upward-facing tiles with cages (26%) than on shaded upward-facing tiles with cages (8%) or on unshaded, shaded, upward or downward-facing tiles without cages (3–6%). In June 2007, eggs and sperm were cross-fertilized from eight wild *Acropora tenuis* colonies taken from the transplantation site to a land-based nursery. Larvae were settled on ceramic tiles (each 12 × 12 × 2.5 cm with five rows of 1.5 $cm^2$ holes). In April 2008, pairs of tiles with 10 month old corals were attached to steel rods and secured to reef knolls using epoxy cement. In each pair, tiles were placed one above the other with the upper tile shading the lower tile. Tile pairs were arranged in three configurations: tiles fixed 1 cm apart with corals facing upwards and covered with a vinyl-coated wire cage (5 cm mesh; 43 pairs) or not covered with a cage (33 pairs), or tiles fixed 3 cm apart with corals facing each other (one upward, one downward) without a cage (32 pairs). Live coral coverage on each tile was measured using a 10 $cm^2$ quadrat at eight, 10 and 22 months after transplantation. **Costs (¥)**: Cultivation and transplantation cost

¥50,563,000 (2011 value), including materials, equipment, personnel, electricity, water, fuel and transport (see paper for detailed breakdown).

A replicated, randomized study in 2008–2010 at an in-situ nursery and natural reef in Malinep, Philippines (10) found that growing fragments of stony coral *Acropora millepora* on cement 'plug-ins' for longer in a nursery before transplanting led to a higher survival rate compared to fragments transplanted after less time in the nursery. After 31 months, survival rate was higher for fragments transplanted after 19 months in the nursery (47%) compared to fragments transplanted after 14 (12%) or seven months in the nursery (8%). In April 2008, PVC-pipe racks with 200 coral 'plug-ins' (comprising a cylindrical 20 × 15 mm cement head with a plastic screw plug attached) each supporting at least one juvenile stony coral, were taken from an ex-situ rearing tank to an artificial nursery on a nearby reef. After seven months, sixty plugs were randomly selected and transplanted to a natural reef and attached using holes drilled into the substrate. After 14 months, a further 60 plug-ins were transplanted, with the final 30 plug-ins transplanted after 19 months. Survival was monitored approximately every month from October 2008–October 2010, and size (average diameter) every six months. **Costs (US$)**: Cost/surviving 2.5 year old coral was estimated by dividing the total project cost by the number of plug-ins supporting one juvenile coral transplanted at each stage. Cost for corals transplanted at seven months (US$284), 14 months (US$217), 19 months (US$61). Full details in the original paper.

A study in 2010–2012 on an intertidal seawall off Changi, Singapore (11) found that some nursery-grown stony and soft coral fragments outplanted onto a seawall survived, depending on the species, and most survivors had grown. Thirteen months after outplanting stony corals, survival was higher for *Goniastrea minuta* (90%) than *Diploastrea heliopora* (10%). *Diploastrea heliopora* fragments had negative growth rates (–1.2 $cm^2$/month), while the other five surviving species had positive growth rates (1.2–17.7 $cm^2$/month). Twenty-four months after outplanting stony corals, survival was higher for *Porites lobata* (47%) than *Pocillopora damicornis* and *Hydnophora rigida* (both 0%). Soft coral survival was higher for *Lobophytum* sp. (88%) than *Cladiella* sp. (37%) and *Sinularia* sp. (13%). Coral fragments were collected from an ex-situ nursery and outplanted onto a granite boulder seawall. Fragments

(≥3 cm diameter) of three stony coral species (18–38 fragments/species) and three soft coral species (30–40/species) were outplanted in May 2010 (see paper for full species list). Fragments ≥3 cm diameter of two additional stony coral species (30 fragments/species) were outplanted in April 2011. Soft corals were grown on concrete plates (5 cm diameter, 0.5 cm thick) in the nursery, which were then attached to the seawall, whereas stony corals were outplanted directly onto the seawall (both using epoxy putty). Coral survival was monitored monthly during low tide for 24 months for the first transplants and for 13 months for the second batch. Growth was measured using photographs taken during the final survey visit in May 2012.

A study in 2011–2015 at a breakwater in Curaçao, Caribbean (12) reported that some nursery-grown elkhorn *Acropora palmata* coral colonies settled on clay tiles then outplanted onto artificial substrate, survived, grew and spawned. After four years, seven out of nine outplanted colonies survived and grew to 30–40 cm diameter and 20–30 cm height. Four years after outplanting, two colonies were observed releasing egg/sperm bundles. In 2011, egg/sperm bundles were collected from eight elkhorn coral colonies in the wild and cross-fertilized to generate larvae. Viable larvae were settled onto clay tiles and reared at a land-based nursery for one year. After one year, nine colonies were outplanted to a breakwater 2–5 m deep off Curaçao. Monitoring was carried out using photographs.

A replicated study in 2019–2020 at six sites in the Florida Keys Reef Tract, USA (13) found that three species of nursery-grown corals *Montastraea cavernosa, Orbicella faveolata*, and *Psuedodiploria clivosa* transplanted on cement or ceramic discs had high survival after 12 weeks. After 12 weeks, 347 of 360 (96%) transplanted colonies still had live tissue, nine completely died (3%) and four were missing (1%). A lower percentage of fragments from in-situ nurseries on cement discs were predated (6–80%) than those from ex-situ nurseries on ceramic discs (23–99%). The percentage of live tissue/colony initially decreased after transplanting for in-situ corals (99% on transplant day and 95% after 1 week) and ex-situ corals (100% on transplant day and 88% after 1 week), but began to increase from six weeks after transplanting for in-situ nursery corals (reaching 96% after 12 weeks) and one week after transplanting for ex-situ nursery corals (reaching 92% after 12 weeks).

At each of three locations, one offshore continuous reef site (5–6 m depth, 6–9 km from shore) and one inshore patch reef site (3–5 m depth, 3–5 km from shore) were selected. At each of the six sites, 60 coral colonies (20 colonies/species) were transplanted, half sourced from an in-situ nursery and half from an ex-situ nursery. Colonies had all been fragmented at their nurseries. Colonies were attached using epoxy to a cement disc (in-situ colonies) or ceramic disc (ex-situ colonies) and attached to the natural substrate via a drilled hole and epoxy. Sites were monitored one, two, six, and 12 weeks after transplanting.

A replicated study in 2018–2019 at two reef sites and an in-situ nursery site on the Florida Reef Tract, USA (14) reported that transplanting nursery-grown staghorn coral *Acropora cervicornis* onto artificial substrates resulted in survival and growth over at least 480 days. Overall, 107 transplanted colonies (89%) survived for at least 480 days, and survival was similar at all sites (85–95% of 40 survived). Colonies grew at all sites, and average size after 480 days ranged from 156–229 $cm^3$ at reef sites to 2,330 $cm^3$ at the nursery site. Egg/sperm bundles were gathered from an in-situ nursery, settled on tiles, and moved to an ex-situ aquaculture facility where they were allowed to grow for eight months. Three transplant sites were selected: two reefs, and one in-situ nursery. In 2018, recruits were grown on 3 × 3 cm ceramic tiles, and 40 fragments/site were transplanted. Tiles were mounted to an argonite and concrete pyramid with epoxy, which was then affixed to the reef. Colonies were monitored approximately two weeks, one month, three, six, and sixteen months after transplanting.

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## 13.6. Transplant wild-grown coral onto natural substrate

https://www.conservationevidence.com/actions/4061

- **Thirty-seven studies** evaluated the effects of transplanting wild-grown coral onto natural substrate. Eight studies were in the British Virgin Islands[14a-c, 15a,b, 16–18], five studies were in the Philippines[9,12,19,20,22], four in Japan[3, 8a-c], Puerto Rico[1a-c,21], and the USA[5,11,13,28], three in the US Virgin Islands[2a,b,10], two in Spain[25,27],and one in each of Israel[4], Indonesia[6], Kenya[7], Mexico[23], north-western Mediterranean[24], Australia[26] and Mauritius[29].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (37 STUDIES)**

- **Abundance/Cover (2 studies)**: Two studies (including one replicated, controlled study) in the Philippines[12] and Puerto Rico[21] found that after transplanting wild-grown coral onto natural substrate, numbers of new coral colonies increased[21] or were similar to areas without transplants[12].
- **Reproductive success (7 studies)**: Seven studies (including six replicated) in the US Virgin Islands[2a,b], Japan[3,8a-c], and north-western Mediterranean[24], found that transplanted wild-grown coral on natural substrate spawned[8a-c], released larvae[2a,b,3], or showed potential to reproduce[24]. Transplanting to different depths affected larvae production[2a], but cutting fragments in half did not[2b]. Large and/or vertically attached fragments had a higher spawning rate than small or medium[8a,b], and/or horizontal fragments[8a,c].
- **Survival (30 studies)**: Thirty studies (twenty-four replicated, including five controlled, and one controlled, before-and-after) in Puerto Rico[1a-c,21], the US Virgin Islands[2a,b,10], the USA[5,11,28],

Indonesia[6], Kenya[7], Japan[8a-c], the Philippines[9,12,19,20,22], the British Virgin Islands[14a,b,16,18], Mexico[23], NW Mediterranean[24], Spain[25,27], Australia[26] and Mauritius[29], found that some fragments of at least one wild-grown coral species transplanted onto natural substrate survived. Six of the studies found that survival depended on substrate[1a,6], fragment size[8a], or orientation[2b,8a,c], or whether fragments were cut in half[2b], or broken[16] before transplanting. Seven of the studies found that survival varied depending on whether fragments were transplanted inside or outside a protected area[7], at their collection site or a different site[5,14b,25], at high or low density[19,20], or in single- or mixed-species groups[22]. One of the studies found that survival was similar for fragments transplanted onto reefs with or without existing coral[11].

- **Condition (29 studies)**: Twenty-five of twenty-nine studies (twenty-three replicated including three controlled, two randomized, one controlled, before-and-after, and two paired) in Puerto Rico[1a-c], the US Virgin Islands[2a,b], Israel[4], the USA[5,11,13], Kenya[7], Japan[8a-c], the Philippines[9,19,20,22], the British Virgin Islands[14a,b,15a,b,16,17a,b,18], Mexico[23], north-western Mediterranean[24], Spain[25], and Mauritius[29] found that some wild-grown corals transplanted onto natural substrate increased in size or percentage live tissue coverage or growth[1a-c,2a,b,7,8a-c,9,11,13,14a,b,15a,16,17a,b,18–20,22–25]. One study[5] found that growth was reduced for all transplanted corals. Eight of the studies found that growth was higher for fragments in a protected area[7], for smaller fragments[8b], fragments transplanted horizontally rather than v ertically[8a,20], at high rather than low density[19,20], when they were recently broken rather than healed[23], or was site-dependent[2a,8a,b], but one study found that growth was lower for broken fragments than intact[16]. Five of the studies found that growth of transplanted corals was similar when fragments were transplanted at their collection site or a different site[5,14b,25], whether they were cut in half or not[2b], or transplanted with or without existing corals[11]. One study found transplanting fragments in single- or mixed-species groups produced mixed results[22]. Four of the studies found the percentage of live tissue

cover or growth on transplanted fragments was affected by orientation[1b], depth[1c], stabilization[13], and transplant site[15a]. The other three of twenty-nine studies found levels of bleaching or surface damage were affected by depth[4], or location inside or outside damselfish territory[15b,29].

**Background**

A widely used method for restoring degraded coral reefs is to transplant corals onto them using fragments from live corals on nearby reefs, which either broke off during natural disasters or human activities or were intentionally removed for the purpose of transplantation. Corals used for transplanting procedures can be in colony, fragment, branch, nubbin (a small number of polyps) or settled larvae (coral spat) form (Forrester *et al*. 2011). Corals can be attached directly to the substrate or to a natural material placed on or fixed to the substate, such as limestone, using epoxy resin, cable ties, cement, or metal grids.

This action specifically refers to transplanting wild-grown corals onto natural substrates. Studies that report the effect of transplanting wild-grown coral onto artificial substrates or nursery-grown corals onto natural or artificial substrates are described in *Transplant wild-grown corals onto artificial substrate; Transplant nursery-grown coral onto natural substrate; Transplant nursery-grown corals onto artificial substrate*. Studies that report the effect of cultivating corals are described in *Cultivate coral in an artificial nursery located in a natural habitat*.

Forrester G.E., O'Connell-Rodwell C., Baily P., Forrester L.M., Giovannini S., Harmon L., Karis R., Krumholz J., Rodwell T. & Jarecki L. (2011), Evaluating methods for transplanting endangered Elkhorn Corals in the Virgin Islands. *Restoration Ecology*, 19, 299–306. https://doi.org/10.1111/j.1526-100X.2010.00664.x

A study in 1997–1999 at a damaged coral reef site off Mona Island, Puerto Rico (1a) found that more than 50% of broken fragments of elkhorn

*Acropora palmata* coral survived reattachment but fewer survived when reattached to dead coral skeletons compared to coral reef substrate, some developed new upward growth, and some had fused to the attachment surface. Two years after reattachment, 405/705 of fragments were still alive (retaining some live tissue cover), 182/705 had died and 118/705 were missing (removed from analysis). There was a lower proportion of live fragments attached to coral skeletons (173/269) compared to the reef substrate (232/318). Only 58/705 fragments had fused to the dead coral or reef substrate; 128/705 fragments had either fully or partially overgrown the attachment wire. New upward growth (2–10 cm) was recorded on 108/705 of the fragments. In September–October 1997, following a ship grounding, 1,857 broken elkhorn coral fragments (15 cm–3.4 m) were reattached to dead standing elkhorn skeletons using stainless steel wire and cable ties, or to the reef substrate using wire and nails. Monitoring was carried out in August 1999 on a representative sample of the fragments (38%, 705/1,857). New upward growth was measured, and an estimate was made of the level of surface attachment (fused, fully or partially overgrown wire).

A study in 1997–1999 at a damaged coral reef site off Mona Island, Puerto Rico, (1b) found a greater percentage of live tissue cover on broken fragments of elkhorn *Acropora palmata* coral reattached the right way up on dead coral skeletons or reef substrate compared to fragments attached upside down, but no difference in the survival rate or length of fragments. Two years after reattachment, average live tissue cover was higher on fragments attached the right way up (54%; relative to their original orientation before breakage) compared to the average cover on fragments attached upside down (47%). There was no difference in survival for fragments reattached the right way up (71% 267/376) compared to upside down (63% 120/190). Average length of fragments did not vary between those attached the right way up (66 cm) and upside down (59 cm). In September–October 1997, following a ship grounding, 1,857 broken elkhorn coral fragments (15 cm–3.4 m) were reattached to dead standing elkhorn skeletons using stainless steel wire and cable ties, or to the reef substrate using wire and nails. Fragments were attached either the right way up or upside down. Monitoring was carried out in August 1999 on a representative sample of the fragments (38%, 705/1,857). Proportion of live tissue cover remaining on the upper

(visible) surface of the fragment was recorded by two divers directly above the fragments.

A study in 1997–1999 at a damaged coral reef site off Mona Island, Puerto Rico, (1c) reported a higher overall survival rate of broken fragments of elkhorn *Acropora palmata* coral reattached to dead coral skeletons or coral reef substrate in shallow water compared to deeper water, and there was a higher percentage of live tissue cover on fragments reattached to dead coral skeletons in shallower water compared to deeper. After two years, survival was higher for fragments reattached at shallow (<3 m) and intermediate (3–4 m) depths (shallow: 71%, 122/172; intermediate 74%, 188/253) compared to deeper (>4 m; 59%, 95/162). Survival rate was higher for fragments attached to dead coral skeletons at shallow (68%, 64/94) and intermediate (70%, 71/101) depths compared to deeper (49%, 36/73). Live tissue cover was higher on fragments reattached to dead coral at shallow (49%) and intermediate (60%) depths than deeper (38%). There was no difference in live tissue cover between depths for fragments attached to the reef substrate. In September–October 1997, following a ship grounding, 1,857 broken elkhorn coral fragments (0.15–3.40 m) were reattached to dead-standing elkhorn skeletons using stainless steel wire and cable ties, or to the reef substrate using wire and nails. Fragments were attached 2–7 m deep and categorized as shallow: 2–3 m, intermediate: 3–4 m, and deep: 4–7 m. Monitoring was carried out in August 1999 on a representative sample of the fragments (38%, 705/1,857). The number of live fragments and proportion of live tissue cover remaining on the upper (visible) surface of the fragment were recorded by two divers directly above the fragments.

A replicated study (years not given) at a coral reef in Saint Croix, US Virgin Islands (2a) found that after transplanting wild-grown mustard hill coral *Porites astreoides* onto natural substrate most colonies survived, and that transplanting colonies to different depths to those they originated from had significant effects on growth and numbers of larvae produced. After 21 months, 87–100% of transplanted colonies survived. Average growth rates were higher for colonies transplanted to shallower sites (3.5 mm/year) and lower for colonies transplanted to deeper sites (1.7 mm/year) compared to those transplanted to their depth of origin (2.6 mm/year). Average larval production rates were

higher for colonies transplanted to their depth of origin at shallow sites (11 larva/40 polyps) than for those transplanted to their depth of origin at deep sites (2.6 larva/40 polyps) or transplanted to deeper (4.4 larva/40 polyps) or shallower sites than their origin (4.5 larva/40 polyps). Thirty-two mustard hill coral colonies (each 7–15 cm diameter) were collected at each of two depths (9 and 24 m) and taken to a laboratory. Each colony was cut in half and stained with red dye for 36–48 h, before being transported back to the reef and transplanted onto natural substrate using underwater epoxy. One half of each colony was transplanted to the depth it was collected from, and the other to a new depth (9 or 24 m). After 21 months, survival was recorded. Growth and larvae numbers were assessed in the laboratory for 16–19 transplanted colonies/depth.

A replicated study (years not given) at a coral reef in Saint Croix, US Virgin Islands (2b) found that transplanting wild-grown mustard hill coral *Porites astreoides* colonies cut in half onto natural substrate led to similar survival, growth and larval production rates compared to when colonies were left intact when transplanted. After 21 months, the average percentage of surviving colonies was similar for cut (88–100%) and intact colonies (87–100%). Average growth and larval production rates were reported to be similar for cut (1.7–3.5 mm/year; 2.6–11 larva/40 polyps) and intact colonies (1.5–3.3 mm/year; 3.5–13 larva/40 polyps), although the results were not tested for statistical significance. Thirty-two mustard hill coral colonies (each 7–15 cm diameter) were collected at each of two depths (9 and 24 m), cut in half, and stained with red dye for 36–48 h in a laboratory. Intact colonies of a similar size were collected from the same depths (50–52 colonies/depth) and stained for 36 h in plastic bags anchored in a sand channel. Cut and intact colonies were transplanted onto natural substrate on the reef using underwater epoxy. Half were transplanted to the depth they were collected from, and half to a new depth (9 or 24 m). After 21 months, survival was recorded. Growth and larvae numbers were assessed in the laboratory for 58 cut and 69 intact colonies.

A study in 1997–1998 at two coral reef sites in Amakusa, Japan (3) reported that transplanted stony coral *Pocillopora damicornis* fragments released larvae. Two weeks after larvae were released, average numbers of recruits ranged from 0.3–4.8/625 $cm^2$ in 1997 and 0.1–5.6/625 $cm^2$ in

1998. In February 1997, fifty *Pocillopora damicornis* colonies were collected from Ōshima Island and transplanted to an area of Satsuki where they had not previously been recorded. Colonies were attached to the rocky substrate using epoxy over a 10 m diameter area. Coral recruits were measured using quadrats two weeks after larval release in 1997 and new recruits (i.e. <2 cm) were recorded two weeks after larval release in 1998.

A replicated study in 2000 at a coral reef near Sdot-Yam, Israel (4) found that exposing stony coral *Oculina patagonica* fragments to higher levels of ultraviolet radiation by transplanting to shallower depths led to a reduction in bleaching and the quantity of *Vibrio shiloi* bacteria that cause bleaching compared to fragments transplanted deeper. Three months after transplanting, no bleaching was recorded on the fragments that remained at 0.8 m deep or fragments transplanted from 4 m deep to 0.8 m deep. Bleaching (approx. 5% of each colony) was recorded on 8% of intact colonies growing at 0.8 m. By contrast, >90% of fragments remaining at 4 m, intact colonies, and 100% of fragments transplanted from 0.8 m to 4 m showed bleaching (32–35% of surface area bleached). *Vibrio shiloi* was not detected in eight non-bleached fragments transplanted from 4 m to 0.8 m but was detected in the eight bleached fragments transplanted from 0.8 m to 4 m. In May 2000, two fragments (7 $cm^3$) were taken from each of 24 *Oculina patagonica* colonies at 0.8 m and 4 m deep. Twenty-four fragments were each glued to the substrate at their original depth, the other 24 were swapped so fragments from 0.8 m were transplanted to 4 m and vice versa. Bleaching was monitored monthly for seven months. In August 2000, eight transplanted fragments from each depth were collected and examined for the presence of *Vibrio shiloi*.

A replicated, controlled study in 1997–2000 in five reefs in Hawaii, USA (5) found that 19–25 months after transplanting onto natural substrate, black corals *Antipathes ulex* and *Antipathes dichotoma* had 0–70% survival and had reduced in height, and survival and growth were not affected by transplantation next to or far from their parent colony. Results were not tested for statistical significance. One transplanted *Antipathes ulex* fragment showed no growth during the survey period and died within 25 months. Nineteen or 24 months after transplantation, *Antipathes dichotoma* fragments transplanted next to their parent colony had

0–70% survival and an average height reduction of 64%, and fragments transplanted far from their parent colony had 56–70% survival and an average height reduction of 22–67%. In July 1997, one *Antipathes ulex* fragment was cut from its parent colony in Oahu, transplanted 2 m away at 46 m deep, and surveyed in October 1997, May 1998, August 1998 and June 1999. In June 1998, nineteen *Antipathes dichotoma* fragments were cut from their colony in Hawaii, 10 were transplanted 2 m away at 27 m deep and nine were transplanted 83 km away at 25 m deep. Both transplanted colonies were surveyed in June 2000 (24 months later). In July 1998, twenty *Antipathes dichotoma* fragments were cut from a colony in Maui, 10 were transplanted 2 m away at 34 m deep, 10 were transplanted elsewhere in Maui at 26 m deep (distance between sites not given), and both transplanted colonies were surveyed in April 2000 (19 months later). Transplanted fragments were attached to new substrate with cable ties and epoxy.

A replicated study in 1999–2000 at a coral rubble site in Bunaken National Park, North Sulawesi, Indonesia (6), found that attaching transplanted stony coral *Acropora yongei* fragments to pieces of coral rubble led to lower survival than fragments fixed to the substrate. Twelve months after attachment, the survival rate of fragments attached to pieces of coral rubble was lower (40%) than fragments fixed on the substrate (65%). In April 1999, one hundred and forty fragments (~10 cm long with 2–4 branches) were collected from a single wild *Acropora. yongei* colony. Eighty-two were attached to pieces of coral rubble using wire and were able to be moved by the current. Fifty-eight fragments were attached to PVC pipes with wire and cable ties and secured by being driven down to the level of the coral rubble. Survival of fragments was recorded six and 12 months after transplantation.

A replicated, before-and-after, site comparison study in 1999–2001 in six marine sites in coastal Kenya (7), found mixed levels of short-term survival and growth of transplanted coral fragments in protected, unfished areas compared to unprotected, fished areas, with or without cages. Live cover of transplanted corals varied between fragment size and coral species, with massive *Porites* species experiencing the greatest losses among the four taxa (see paper for details). Live coral cover was higher at one out of three protected, no fishing sites (9%; 19% and 59%) compared to unprotected, fished reefs (5%; 12% and 19%). Growth

of transplanted corals was generally higher at protected, no fishing sites (average 2.6 cm) than at fished reefs (0.7 cm). Cage presence for transplanted corals had no impact on their condition but turf algae cover was negatively associated with mortality of the transplanted coral fragments (see paper for details). Coral fragments (192 fragments 5–6 cm and 550 fragments 10–15 cm long) of four stony coral species (*Porites* spp., *Pocillopora damicornis*, *Pavona decussata* and *Pavona frondifera*) were removed with hammer and chisel from one site, immersed in aerated seawater in separate buckets, transported to site and immediately placed into a large underwater cage for 1–3 days before attachment with epoxy putty and later only masonry cement to the substratum. Transplanted corals were monitored for 22–35 days. Sites included three protected unfished sites (Malindi, Watamu, and Mombasa Marine National Parks). Some transplanted coral fragments (64 each in Marine National Park and fished reefs) were kept inside cages for approximately 14 days prior to removing the cages and exposing them to predators.

A replicated study in 1999–2001 at a coral reef at Akajima Island, Japan (8a) found that transplanting large fragments of stony coral *Acropora formosa* led to higher survival rates, growth, and spawning than medium and small fragments, and depended on vertical or horizontal orientation. After six months, large fragments had a higher survival rate (vertical: 100%; horizontal 94%) than medium (vertical: 91%; horizontal: 68%) and small fragments (vertical: 76%; horizontal: 30%) with most vertical fragments having a higher survival rate than horizontal (see paper for data). Survival after 18 months was higher for large fragments (vertical: 98%, horizontal: 92%) than medium (vertical 84%, horizontal: 0%) or small fragments (vertical: 29%, horizontal: 7%). Monthly growth rate was higher for large (5%) and medium (7%) vertical fragments compared to horizontal fragments (large: 4%; medium 4%), but there was no difference for small fragments (vertical: 5%; horizontal: 7%). A greater number of large vertical fragments spawned (81%) compared to medium (4%) and small vertical fragments (0%), and large vertical fragments had a higher spawning rate (81%) than large horizontal fragments (20%). In November 1999, six wild-growing *Acropora formosa* colonies each had 6–10 fragments taken in three different size classes (small: 5 cm; medium: 10 cm; large 20 cm). Fragments were transplanted 2–3 m deep, 15–20 m from the donor colonies, and attached either

vertically or horizontally to the substrate using nails and cable ties. Survival and growth were measured every two months for 18 months. Spawning was recorded during May and June 2000, 2001, and 2002.

A replicated study in 2000–2003 at a coral reef at Akajima Island, Japan (8b), found that transplanted stony coral *Acropora formosa* and *Acropora hyacinthus* fragments survived and spawned and that small fragments of *A. formosa* had a higher growth rate than medium or large fragments. After 14–18 months, 87–100% of *A. formosa* and 100% of *A. hyacinthus* fragments had survived. Spawning rate ranged from 0–20% (small and medium fragments) and 46–100% (large) fragments of *A. formosa*, and 18% for *A. hyacinthus* fragments. Growth of small *A. formosa* fragments was higher (13–14%) than medium (6–7%) or large fragments (3–5%). In March and August 2000, six wild-growing *A. formosa* colonies each had 6–10 fragments taken in three different size classes (small: 5 cm; medium: 10 cm; large 20 cm). In February 2002, fragments were taken from 11 colonies of *A. hyacinthus*. All fragments were transplanted 2–3 m deep, 15–20 m from the donor colonies, and attached vertically to the substrate using nails and cable ties. Survival was monitored every two months for 14–18 months, spawning was recorded when it took place. Growth of *A. formosa* was measured after one year. Growth of *A. hyacinthus* could not be measured as many fragments died during a bleaching event in August 2002.

A replicated study in 2001–2002 at a coral reef at Akajima Island, Japan (8c) found that stony coral *Acropora hyacinthus* fragments transplanted vertically had higher survival, greater attachment to the substrate, and higher numbers spawned, compared to fragments transplanted horizontally, but all fragments had new bud growth. After four months, survival of vertically attached fragments (45%) was higher than horizontally attached (20%), and after 14 months survival was higher for vertically attached fragments (32%) than for horizontally attached (2%). After two months, 80% of vertically attached fragments had fused to the substrate whereas none of the horizontal fragments had fused after one year. A small number of vertical fragments (0.4%) spawned but none of the horizontal fragments. New bud growth was observed on all surfaces of both vertical and horizontal fragments (data not reported). In July 2001, fragments (average width 5.1 × length 10.3 cm) were taken from 11 different *A. hyacinthus* colonies (see paper

for methods). Fragments were transplanted 2–3 m deep, 15–20 m from the donor colonies. Half the fragments were attached vertically and half horizontally to the substrate using nails and cable ties. Survival and attachment to the substrate were monitored every two months for 14 months. Growth could not be measured as many fragments died during a bleaching event in August 2002.

A replicated study in 2005 at three sites on a coral reef near Bolinao, north-western Philippines (9) found that transplanting 11 stony coral nubbins (small fragments) and and one non-stony coral nubbin onto natural substrate led to mixed results for survival and self-attachment (tissue growth onto the substrate) depending on species. After five months, a total of 310/540 (57%) nubbins were alive (>40% live tissue). There was a significant difference for survival rate after five months between species ranging from 98% (*Pavona frondifera*) to 18% (*Pocillopora verrucosa*). A total of 225/540 nubbins exhibited self-attachment (tissue growth over the adhesive and directly onto the substrate) and there was a significant difference in self-attachment rate between species ranging from 92% (*Pavona frondifera*) to 41% (*Montipora digitata*). In June and August 2005, a total of 540 coral nubbins (fragments 2–3 cm in length) were collected from 11 wild-grown stony coral colonies and one non-stony coral colony at three sites (see paper for full species list). Substrate was created using 15 giant clam half shells deployed at each of three sites (site 1 and 2 in June 2005, site 3 in August 2005) at a depth of 2–4 m. Twelve nubbins (one/ species – see original paper) were attached to each shell using marine epoxy, epoxy putty or cyanoacrylate glue (superglue). Fifteen shells were deployed at each of three sites. Survival and self-attachment were recorded every two weeks for five months (dates not given).

A replicated, controlled study in 1999–2004 at four reefs around St John's Island, US Virgin Islands (10), found that transplanting storm-generated fragments of elkhorn *Acropora palmata*, staghorn *Acropora cervicornis* and stony *Porites porites* corals onto degraded coral substrate led to mixed results for survival compared to existing coral colonies. After five years, survival of transplanted elkhorn coral (20%) was lower than existing (53%) corals, but staghorn (transplanted: 0%, existing: 6%) and *P. porites* (transplanted: 27%, existing: 13%) corals did not differ significantly from existing coral. In July 1999, storm-generated

fragments of elkhorn (15), staghorn (30) and *P. porites* (15) corals were collected from 1–3 m deep at two sites. These were transported 1–5 km to recipient reefs and attached to dead, upright coral skeletons (mostly elkhorn) using nylon cable ties. Fragments were placed near existing colonies of the same species for comparison (15 elkhorn, 45 staghorn and 15 *P. porites* corals). Each colony was photographed, sketched, and live tissue on each branch and base was measured every six months from 1999–2001 then annually until 2004. Dead or detached colonies were removed from the analysis. **Costs (US$)**: Transplantation cost $1,250 (2008 value), including materials, boat and scuba, and salary ($21/transplant with and $5/transplant without salary costs).

A replicated study in 2004–2005 in two sites Viosca Knoll, Gulf of Mexico, USA (11) found that transplanting fragments of deep-water coral *Lophelia pertusa* on to reefs with existing coral coverage did not result in higher survival, growth rate or number of new polyps than corals transplanted onto bare rock without existing coral. Thirteen and a half months after transplanting, survival rate was the same for fragments transplanted into areas with existing coral or onto bare rock (both 91% survival). Similarly, there was no significant difference in total linear growth (existing: 20 mm, bare rock: 11 mm), average growth/polyp (existing: 4 mm, bare rock: 3 mm) or the average number of new polyps/fragment (existing: 3.3, bare rock: 3.5). In July 2004, fragments of deep-water coral were collected from Viosca Knoll. Thirty-two fragments each with 10–20 polyps were stained using red dye and photographed before being fixed into a 2 cm PVC pipe using cement. Fragments were attached to frames (4/frame) before being transferred to the transplant sites. Four frames were placed 460 m deep in an area with existing coral coverage and four frames were placed 507 m deep on bare rock ~0.25 km from the other site. Fragments were removed after 13.5 months. Survival was recorded and the number of new polyps counted. Growth was measured using photographs.

A replicated, controlled study in 2000–2002 in four areas of a coral reef in Luzon, Philippines (12) found that 9–27 months after *Acropora* and *Porites* stony corals were transplanted onto natural substrate, 83–94% survived, but plots with and without transplanted corals had similar numbers of new coral colonies. Nine to 27 months after transplantation, 94% of *Acropora palifera*, 85% of *Porites lobata* and 83% of

*Porites cylindrica* colonies survived. Plots where corals were transplanted had similar average numbers of new coral colonies (0.62) to interspersed plots without transplantation (0.88) and plots 100 m away without transplantation (0.51). All three had lower numbers of new coral colonies than plots at a nearby healthy reef from which the transplanted corals had been sourced (10.39). Each of four sites of rocky seabed had eighteen 1 $m^2$ plots: six with transplanted corals and six without corals (interspersed amongst each other), six 100 m away without corals, and six at the transplanted coral source site (a healthy reef). Between April 2000 and November 2001, colonies of three coral species were chiselled from a nearby healthy reef and at each transplantation plot either attached directly to the rock with cement or tied to plastic screens covering the plots: *Acropora palifera* (5–19 cm diameter, 2/plot), *Porites cylindrica* (11–30 cm diameter, 2/plot), and *Porites lobata* (7–19 cm diameter, 3/plot). From January 2001–July 2002 (9–27 months after transplanting), survival and new coral colonies were recorded during visual census by divers in all experimental plots every 3–4 months.

A randomized, replicated study in 2006–2007 at a coral reef in Florida, USA (13) reported that stabilizing transplanted wild-grown elkhorn coral *Acropora palmata* fragments led to better 'performance' than unstabilized fragments and there was no difference between attachment methods for stabilized fragments. Forty-four weeks after transplanting, more stabilized fragments were classed as 'best-performing' (8/18 attached with cable-ties, 10/18 epoxied) compared to unstabilized fragments (0/18). Fewer stabilized fragments were classed as 'worst-performing' (4/18 for cable-tied and epoxied) compared to unstabilized fragments (10/18) and fewer stabilized were classed as 'intermediate performing' (cable-tied: 6/18; epoxied: 4/10) than unstabilized (8/18). For stabilized fragments, there was no difference in performance between cable-tied and epoxied fragments. In August 2006, fifty-four naturally occurring elkhorn coral fragments (<40 cm) were collected from 3 m deep and taken to a reef ~350 m away. Fragments were placed in groups of three according to size and % live tissue coverage (18 groups) and randomly assigned to be stabilized (attached with cable-tie or epoxy) or unstabilized (tethered to substrate using a 1 m line). The substrate was cleared of sediment and micro-algae before attachment. Fragments were monitored after seven, 24, and 44 weeks, and any

unstable fragments were reattached at seven weeks. After 44 weeks, fragments were categorized as performing worst (including lost or dead fragments); intermediate; or best, according to the % live tissue and % natural attachment to the substrate (category parameters not reported).

A controlled study in 2005–2009 at four coral patch-reef sites near Guana Island, British Virgin Islands (14a), found that transplanting storm-generated fragments of elkhorn coral *Acropora palmata* onto natural substrate led to a greater survival and increased surface area of live tissue compared to fragments left unattached. After one year, 28 of 35 (80%) transplanted fragments survived compared with one of seven (14%) unattached fragments. All unattached fragments were dead after two years. After four years, 14 of 35 (40%) transplanted fragments were alive and the average surface area of live tissue had increased by 1,160% to reach 1,453 $cm^2$. In July–November 2005, thirty-five fragments of elkhorn coral that had become naturally detached were transplanted onto four patch-reef restoration sites 0.4–3.6 km away. Fragments were attached onto limestone and coral rubble substrate 0.4–1.6 m deep using epoxy resin, ensuring live tissue contacted the attachment surface. Seven fragments were left unattached at the original site. Survival was recorded and surface area of live tissue measured using photographs annually for four years.

A replicated, controlled study in 2007–2009 at coral patch-reef sites near Guana Island, British Virgin Islands (14b), found transplanting storm-generated fragments of elkhorn *Acropora palmata* coral onto natural substrate at a new site did not lead to higher survival or increased live tissue growth compared to fragments transplanted at their original site. There was no significant difference in survival rate, one year after transplanting, between new site (2007: 30%; 2008: 54%) and original site fragments (2007: 56%; 2008: 62%). There was also no significant difference in increase of live tissue between new site (2007: 413%; 2008: 80%) versus original site fragments (2007: 322%; 2008: 111%). In July–August 2007 and 2008, storm-generated fragments of elkhorn coral were collected from patch-reefs. Thirty fragments (collected in 2007) and 167 (collected in 2008) were transplanted at reef restoration sites 0.4–3.6 km away, and 27 (collected in 2007) and 70 (collected in 2008) were transplanted at the original collection site. Fragments were either attached to bare reef or dead coral skeletons, using cable ties, marine

epoxy or cement, ensuring live tissue was in contact with the attachment surface. Survival was recorded after one year and growth measured after two and 12 months using scaled photographs.

A replicated, randomized study in 2010–2011 in the British Virgin Islands (15a), found that storm-generated fragments of elkhorn coral *Acropora palmata* transplanted onto natural substrate at a new site had higher levels of bleaching and tissue loss than fragments transplanted within the original site or established fragments transplanted two years before, but growth was lower for same-site than new-site or established fragments. After 10–16 days, a higher number of new-site fragments (57/84) showed some bleaching (>0%) compared to original-site ([figure shows] 9/45) and established fragments (9/45). The number of new-site fragments showing some tissue loss (>0%) was higher (72/84) than original-site (26/48) and established (24/45) fragments. After three months, long-term tissue loss as a percentage of the original fragment size was greater in original-site (68% tissue loss) compared to new-site (37%) and established (28%) fragments. In July–August 2010, one-hundred-and-thirty-two storm-generated elkhorn fragments were collected from two sites in the British Virgin Islands (Harris Ghut and Little Camanhoe). Eighty-four fragments were randomly selected and transplanted to an existing restoration site at White Bay and attached to the substrate using cable-ties. Remaining fragments were re-attached to the substrate at their original site (22 at Harris Ghut, 26 at Little Camanhoe) using cable-ties. As a comparison, 45 fragments that had been transplanted in July–August 2008 at White Bay were surveyed. Percentage tissue loss and bleaching was recorded after 2–4 days for new-site fragments and 10–16 days after transplanting for all fragments. Growth was measured using photographs after three months. Hurricane Earl affected the area in September 2010 — two weeks after transplanting.

A replicated study in 2010–2011 in the British Virgin Islands (15b), found that transplanting storm-generated fragments of elkhorn coral *Acropora palmata* outside known damselfish *Stegastes planifrons* territory did not result in less tissue loss or bleaching than fragments transplanted inside territories. After 10–16 days, there was no significant difference in the number of fragments showing some tissue loss between newly or established transplanted fragments inside (new transplants: 21/25, 84%; established transplants: 8/13, 62%) or outside damselfish territory

(new: 51/59, 86%; established: 16/32, 50%). There was also no significant difference in the number of fragments showing some bleaching inside (new: 13/25, 52%; established: 3/13, 23%,) or outside damselfish territory (new: 44/59, 75%; established: 16/32, 19%). In July-August 2010, eighty-four storm-generated elkhorn fragments were collected from two sites in the British Virgin Islands (Harris Ghut and Little Camanhoe) and transplanted to an existing restoration site at White Bay. Fragments were attached to the substrate using cable ties either inside (25) or outside (59) known damselfish territories. As a comparison, 45 fragments that had been transplanted in July–August 2008 at White Bay were surveyed. Of those, 13 were inside of damsefish territories and 32 outside. Percentage tissue loss and bleaching were visually assessed after 10–16 days and recorded on a 0–5 scale. Hurricane Earl affected the area in September 2010 — two weeks after transplanting.

A replicated, paired study in 2011–2012 at a coral reef near Guana Island, British Virgin Islands (16), found that transplanting elkhorn coral *Acropora palmata* fragments broken into smaller pieces led to lower growth and survival compared to fragments left intact. One year after transplanting, the % change in colony size ($cm^2$) was lower for broken (49%) compared to intact fragments (97%). After one year, survival rate for broken fragments was lower (90/138, 65%) than intact fragments (45/55, 82%). In August 2011, one hundred and ten elkhorn fragments were collected from two sites (Harris Ghut and Great Camanoe). Average fragment size was 233 $cm^2$. Fragments were paired by size and one fragment/pair was broken into 2–5 pieces and the other was left intact. Each pair was placed close together (0.3–4.5 m apart) and the individual fragments were attached to the reef, 0.4–1.6 m deep, using nylon cable ties. Colony growth and survival were measured after three and 12 months using photographs and image analysis software.

A study in 2007–2008 and 2010–2011 at four coral reef sites near Guana Island, British Virgin Islands (17a) found that transplanting storm-generated fragments of elkhorn coral *Acropora palmata* onto natural substrate led to live tissue growth for fragments from two of three collection sites. One year after transplanting, live tissue surface area of fragments collected from two sites increased by an average of 67–304%, whilst fragments collected from a third site decreased by an average of 38%. In July–August 2007 and 2010, storm-generated elkhorn

coral fragments were collected on 1–2 occasions from three reefs (7–45 fragments/reef) and transplanted at a reef restoration site. Fragments were attached to bare reef or dead coral skeletons using cable ties or marine epoxy. Attachment sites were scraped with a wire brush prior to transplanting to remove macroalgae. Growth (surface area of live tissue) was measured using photographs immediately after transplanting and 12 months later.

A replicated, paired study in 2007–2011 at four coral reef sites near Guana Island, British Virgin Islands (17b) found that transplanting storm-generated fragments of elkhorn coral *Acropora palmata* onto natural substrate at a new site led to slower live tissue growth or a reduction in live tissue compared to fragments transplanted at their original site. One year after transplanting, average increases in live tissue surface area for three fragment groups were lower at the new site (40–283%) than at original sites (218–349%). For one fragment group, live tissue surface area decreased by an average of 34% at the new site and increased by 135% at the original site. In July–August 2007, 2008, and 2010, four groups of storm-generated elkhorn coral fragments were collected from three reefs (14–32 fragments/reef). Each fragment was split into two sub-fragments using a hammer and chisel. One sub-fragment was reattached at the original collection site, and the other transplanted at a reef restoration site. Fragments were attached to bare reef or dead coral skeletons using cable ties or marine epoxy. Attachment sites were scraped with a wire brush prior to transplanting to remove macroalgae. Growth (surface area of live tissue) was measured using photographs immediately after transplanting and 12 months later.

A study in 2005–2012 at a coral reef site off Guana Island, British Virgin Islands (18) found that transplanted storm-generated fragments of elkhorn coral *Acropora palmata* survived and grew. Twelve months after transplanting, survival rate for each group of transplants was at least 50% (range: 50–85%) and remained relatively constant until 2012 (results presented as log survival). Survival was lower for the groups of fragments transplanted in 2007 (21%) and 2010 (30%) due to severe storms. After three months, average fragment size across all groups had decreased from 108 $cm^2$ to 92 $cm^2$ then increased to 156 $cm^2$ after 12 months, reaching 2064 $cm^2$ after 72 months. Data were not statistically tested. In July-August 2005–2011, a total of 832 storm-

generated fragments of elkhorn coral were collected from reefs within 4 km of the transplant site. Fragments ranged from 2–1016 $cm^2$ (average 108 $cm^2$). Groups (ranging from 19–257 fragments) were transplanted to a nearby restoration site on the leeward side of Guana Island. Fragments were fixed to the reef 0.4–1.6 m deep using nylon cable ties, or marine epoxy, or hydrostatic cement. Survival and growth (surface area live tissue) were recorded three and 12 months after transplanting and then annually until 2012.

A replicated, controlled, before-and-after study in 2010–2012 in three coral-reef sites in Pangasinan, Philippines (19) found that stony coral *Acropora pulchra* and *Acropora intermedia* fragments transplanted onto natural substrate grew more quickly, but had similar survival, when attached at high rather than low density. Twelve months after transplantation, corals attached at high density had grown faster (*Acropora pulchra*: 1,646 $cm^3$/month, *Acropora intermedia*: 1,115 $cm^3$/month) than corals at low density (*Acropora pulchra*: 1,125 $cm^3$/month, *Acropora intermedia*: 824 $cm^3$/month). Survival after 19 months did not differ between high- and low-density plots (68–89%), but *Acropora pulchra* had higher average survival (85%) than *Acropora intermedia* (72%), regardless of density. Three clusters (>50 m apart) of three 4 $m^2$ experimental plots were demarcated on a degraded 2–3 m deep back-reef. Within each cluster in July 2010, fifty fragments each of *Acropora pulchra* and *Acropora intermedia* were transplanted in one plot (high density), 25 fragments of each were transplanted in another plot (low density), and no fragments were transplanted in a third control plot. Fragments (>25 cm) had been cut from a reef 21 km away a few days prior. They were then inserted into the sand and tethered with wire to a 40 cm bamboo stake driven halfway into the sand. Every two months for a year, and then once after 19 months, survival was monitored, and 10 fragments of each species/plot were measured. **Costs (US$)**: Transplantation cost $0.90 USD/$m^2$ (2010 value), including collection and transplantation tools and boat fuel, but not boat rental, labour or snorkelling gear (provided for free by volunteers).

A replicated study in 2006–2007 at two coral reef sites in a lagoon in northwestern Philippines (20) found that survival and growth of stony coral *Porites cylindrica* fragments transplanted onto natural substrate varied depending on density, and attachment orientation. Survival rate

after 20 months ranged from 80–89% at Malilnep and 94–98% at Binlab. Survival differed between sites only for transplants placed horizontally at low density (Malilnep: 89%, Binlab: 96% survival). Overall vertical growth did not vary significantly between sites (Malilnep: 1.5–3.5; Binlab: 2.2–4.5 mm/30 days; (data reported from figures, which do not match data in text). Radial growth was higher for horizontally placed transplants (1.2–2.3 mm/30 days) compared to vertically placed (1.0–2.1 mm/30 days) irrespective of density or site. In March 2006, nine hundred and sixty, mostly loose, fragments of *Porites cylindrica* (4–6 cm) were collected from a single bommie (coral outcrop) within the lagoon. Fragments were transplanted onto three other bommies selected at each of two sites with substrate comprising either dead massive *Porites* corals or solid substrate (Malilnep) or dead branching *Porites* corals or perforated substrate (Binlab) (no other substrate information provided). Fragments were placed into a depression created in the substrate of the bommies and secured using marine epoxy clay. Forty fragments were each placed either horizontally or vertically and at low density (30 cm apart) or high density (15 cm apart; total 160 fragments) on each of the three bommies at each site (480 fragments/site). From March 2006–October 2007, survival was monitored bi-monthly, height was measured every three months and radial growth was determined every six months.

A study in 2006–2014 at a damaged coral reef site in Tallaboa, Puerto Rico (21) reported that following transplanting of loose fragments of staghorn *Acropora cervicornis* coral, along with nursery-grown fragments, onto stabilized natural substrate, fragment survived, attached to the substrate and the area of restored reef increased. After eight years, the area of restored reef had grown from 70 $m^2$ to 180 $m^2$. Coral colonies in unrestored areas in the vicinity, with loose rubble and damaged substrate, showed no signs of recovery during the same period. It was not possible to determine from the study how much of the recovery was attributable to transplanting loose fragments, transplanting nursery-grown fragments, or stabilizing the substrate. In 2006, following the destruction of a coral reef by a ship grounding, wire cages and metal stakes were used to stabilize a 70 $m^2$ area of damaged reef. Approximately 227 (10–20 cm) fragments of staghorn coral were collected from nearby reefs and attached to the substrate using cement puddles. In 2009–

2011, approximately 400 (20–40 cm) fragments of staghorn coral were collected from a nursery and attached to the substrate using masonry nails, cable ties and/or epoxy. Coral recovery was measured using aerial imagery in 2014. No other methods are reported.

A replicated study in 2008–2009 at two sites of degraded coral reef at Bolinao, northwestern Philippines (22) found that transplanting wild-grown fragments of stony coral *Pavona frondifera* and *Porites cylindrica* onto natural substrate in mixed compared to single-species groups resulted in mixed results for survival and growth depending on species and site. After 12 months, *Pavona frondifera* survival was higher in mixed-species (9%) compared to single-species groups (0%) at Malilnep, but similar at Binlab (mixed: 51%, single: 63%). *Porites cylindrica* survival was similar for mixed and single species groups at both sites (mixed: 87 and 91%, single: 97 and 82%). For *Pavona frondifera*, average monthly linear growth did not differ significantly after 12 months between mixed or single-species groups at Binlab (range mixed: 7.3–9.1, single: 7.9–10.2 mm/30 days), but at Malilnep all *Pavona frondifera* fragments had died by August 2008. After 12 months, average monthly linear growth of *Porites cylindrica*, was similar for mixed and single species groups at both sites (range Binlab mixed: 9.4–17.8, single: 8.5–12.3; Malilnep mixed: 8.8–24.4, single 11.3–18.1 mm/30 days). Over 12 months at Binlab average monthly radial growth of *Pavona frondifera* was similar for mixed and single species groups (range mixed: 0.8–1.3, single 0.9–1.3 mm/30 days) but higher for *Porites cylindrica* in mixed-species (range 2.4–2.5 mm) compared to single-species groups (range 1.6–1.7 mm), but similar at Malilnep (range mixed: 1.9–4.2, single: 2.0–2.7 mm/30 days). In 2008, fragments of *Pavona frondifera* and *Porites cylindrica* were collected from areas of degraded reef at Malilnep and Binlab or taken from live coral colonies on site. Three areas (1–3 m deep) were selected at each site, each with three plots. Each plot contained either 40 fragments of *Porites cylindrica*, 40 of *Pavona frondifera* or 40 each of both species (mixed group) attached to the dead coral substrate using epoxy clay putty. Over 12 months, survival was monitored monthly for three months then every three months, linear growth (mm/30 days) every three months, and radial growth (mm/30 days) every six months.

A replicated study in 2013–2015 at a coral reef in Playa Las Gatas, Mexico (23) found recently broken fragments of stony coral transplanted

onto natural substrate had greater survival, growth, and attachment compared to healed fragments. Twelve or 13 months after transplanting, survival rate was higher in both rainy and dry seasons for recently broken (dry: 91%, rainy: 63%) compared to healed fragments (dry: 63%, rainy: 46%). Vertical growth was higher for recently broken (dry: 161%, rainy: 210%) compared to healed fragments (dry: 88%, rainy: 124%). Horizontal growth was greater for recently broken (dry and rainy: 107%) compared to healed fragments (dry: 73%, rainy: 100%). Substrate attachment in the dry season was higher and happened faster for recently broken fragments (98% in nine months) compared to healed (86% in 12 months). There was no difference in attachment rate after 12 months for fragments transplanted in the rainy season (broken: 89%, healed: 84%). In November 2013 (dry) and August 2014 (rainy), 250 randomly selected naturally broken fragments of *Pocillopora verrucosa*, *Pocillopora capitata* and *Pocillopora damicornis* were collected from around Playa Las Gatas. Fragments were assessed as 'recently broken' (no signs of healing) or 'completely healed' (healed at their breaking point). Twenty-five fragments were attached to one of ten 15 × 15 cm steel grids (one fragment type/grid) using plastic strips. Grids were fixed to one of ten 1 $m^3$ boulders ensuring fragments touched the boulder. Survival, vertical and horizontal growth (% increase) and attachment (% fused to the substrate), were monitored every 2–3 months for 12 or 13 months.

A replicated study in 2003–2015 on a rocky substrate in the north-western Mediterranean (24) found that transplanted red coral *Corallium rubrum* showed similar survival, growth, and reproductive potential to natural colonies. Four years after transplanting, 99% of transplanted red coral colonies survived and the average annual survival rate, of 100% was similar to natural populations (100%). Most transplanted red coral colonies were <35 mm in height and there was no significant difference between transplanted and natural red coral growth rates (data reported on log scale). There was no significant difference in the proportion of fertile colonies (transplanted: 28%; natural 33%) or the average number of larvae/polyp (transplanted: 0.37; natural 0.28). In 2011, authorities seized 14.5 kg of illegally harvested red coral. Approximately 300 colonies from this seizure were transplanted onto a rocky wall, 15–17 m deep, and attached using epoxy putty. Four transect surveys were carried out immediately after transplanting, in May 2011, then again

four years later. Survival rates were recorded using photographs. In June 2015, reproductive potential was measured by counting red coral larvae inside polyps of fertile females from a sample of 35 transplanted colonies and 35 adjacent natural colonies. Survival rates of natural red coral colonies were calculated using long-term data (2003–2011) on eight natural populations.

A replicated, controlled study in 2012–2013 at two coral reef sites on the Granada coast, southern Spain (25) found transplanting fragments of orange coral *Asteroides calycularis* onto natural substrate within the same site resulted in a higher survival rate, but not area growth or the number of polyps that developed, than fragments transplanted to a different site and, at one site higher than colonies left intact. After 12 months, survival rate of same-site transplants was higher (88% and 86%) compared to different-site transplants (81% and 64%), and higher at one site than intact coral (78% and 90%). There was a significant difference in average overall growth between sites (Punta del Vapor: 3.3 $cm^2$; Punta de la Mona: 5.2 $cm^2$) but not in average area growth between same-site, different-site and intact coral (range after six months: same 3.0–4.0; different 2.4–4.0 $cm^2$; intact 0.8–5.0 $cm^2$, after 12 months: same 0.2–1.3; different 0.2–0.6; intact 0.8–1.3 $cm^2$), or the average number of polyps that developed (range after six months: same 7.0–7.7; different 7.7–17.0; intact 2.6–18.0, after 12 months same 2.2–7.0; different 2.4–7.7; intact 0.6–2.7. In July 2012, three areas (8 m deep) were selected at each of two sites. Thirty-six fragments of orange coral were collected from colonies at each site, 18 of these were transplanted in their original site (same site), 18 were swapped between the two sites (different site) and an additional 18 remained in place (intact). Fragments were secured to the substrate using marine epoxy resin. Survival, area growth and the number of polyps that developed was measured after six and 12 months.

A replicated study in 2017–2018 off Whitsunday Island, Great Barrier Reef, Australia (26) found that after transplanting displaced column-shaped coral outcrops ('bommies') of stony coral *Porites* species colonies onto natural substrate, some live tissue was retained. Sixteen months after bommies were transplanted, coverage of original live tissue ranged from 0–20% (average 6%) with 16 of the 22 bommies surveyed still retaining some live tissue. In March 2017, a cyclone dislodged bommies of *Porites* species colonies (1–3 m diameter) and deposited them on the

intertidal zone. In June 2017 heavy machinery was used to transplant 22 bommies back into the subtidal region. Divers surveyed coral bommies in October 2018, and recorded live tissue coverage (%).

A replicated study in 2018–2019 in three sites in the north-western Mediterranean off Cap de Creus, Spain (27) found that after transplanting soft *Eunicella cavolini* corals onto natural substrate by dropping them from a boat, 84–90% of the corals at one site landed upright, whereas at the other two sites corals were obscured by seagrass *Posidonia oceanica* or fine sediments. Surveys at one site in 2019 detected 460 of the 526 corals (88%) that were transplanted in 2018–2019, across an area of 0.23 hectares. For corals transplanted on natural cobbles, 89% landed upright, and for those transplanted on artificial cobbles it was 73%. At the other two sites all corals were obscured by either seagrass or fine sediments. In 2018–2019, a total of 805 coral colonies were recovered from trammel nets (468 in 2018, 337 in 2019). Corals were held in aquaria for a few weeks to three months, fragmented into nubbins and attached to either natural cobbles (693) or artificial concrete cobbles (133) via a drilled hole and epoxy putty. In 2018, corals were released into the water at one of three locations (150–151/location; 80–120 m depth), and in 2019, all corals were released at one location (375; all on natural cobbles). In November 2018 and September 2019, surveys were conducted using an Autonomous Underwater Vehicle (AUV) with onboard cameras. **Costs (€)**: Full costs of the transplants and monitoring (including all staff costs) was €106,783 (see original paper for cost breakdown).

A replicated study in 2018–2019 at three reefs in the southwestern Gulf of California (28) reported that when *Pocillopora* corals were transplanted in areas with high numbers of crown-of-thorns starfish *Acanthaster* cf. *solaris*, mortality varied from 39–88%. Fragment mortality was 39–88% (out of 192–200 fragments), and on average, fragments survived for 134–197 days. The site with highest mortality (88%) had a higher abundance of starfish (0.3 individuals/$m^2$) than at the other two sites (0.08–0.09 individuals/$m^2$). At each study site, 5 cm fragments were transplanted to plots (50 × 25 m) and fixed to the substrate using plastic straps and epoxy at depths of 2–9 m (192–200 fragments/site). Each site was visited five times over 12–15 months to assess survival of coral fragments. The number of starfish was also recorded during these visits.

A replicated study in 2018–2019 at six coral reef sites off Mauritius (29) found that some *Porites lutea* transplanted onto natural substrates survived and had similar predator bite density and surface damage as corals left in-situ. Twenty-eight of 55 (51%) transplanted colonies were lost or died. Transplanted and in-situ corals had similar predator bite density (transplanted: 0.3–0.7 bites/cm$^2$, in-situ: 0.2–0.6 bites/cm$^2$) and surface area damage (transplanted: 11–33%, in-situ: 8–32%). In addition, bite density and surface area damage were lower for corals transplanted to damselfish *Stegastes* spp. territories (bite density: 0.3–0.4 bites/cm$^2$, surface damage: 11–20%) compared to outside territories (bite density: 0.7 cm$^2$, surface damage: 33%). A total of 55 colonies were transplanted. Ten colonies were transplanted to each of three sites containing damselfish territories, and a further 25 colonies were transplanted to adjacent degraded areas with no damselfish territories (5–10 colonies/area). Colonies were transplanted whole (80 cm$^2$ average surface area) and placed directly among the branches of *Acropora muricata* colonies. Transplanted colonies were monitored in February–March, April, and June 2019 to assess survival and predation. A total of 651 in-situ corals were also monitored across six sites every two months from September 2018 to June 2019.

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## 13.7. Transplant wild-grown coral onto artificial substrate

https://www.conservationevidence.com/actions/4008

- **Twenty-two studies** evaluated the effects of transplanting wild-grown coral onto artificial substrate. Five studies were in Indonesia[5,8,10,13,16], three in Egypt[3a,b,9], two in each of the Philippines[4,7], and the USA[18,19], and one was in each of the Maldives[1], the Netherlands Antilles[2], Tanzania[6a,b], Spain[11], Antigua[12], Singapore[14], Kenya[15], Belize[17], and Mauritus[20].

**COMMUNITY RESPONSE (1 STUDY)**

- **Richness/diversity (1 study)**: One randomized, before-and-after study in Singapore[14] found that transplanting wild-grown corals on a subtidal seawall led to an increase in coral species richness.

**POPULATION RESPONSE (22 STUDIES)**

- **Abundance/Cover (5 studies)** Four of five studies (including two replicated, before-and-after, and one randomized, before-and-after) in the Maldives[1], the Philippines[7], Indonesia[10,16], and Singapore[14] found some increases in coral coverage for wild-grown corals transplanted onto artificial substrates[7,10,14,16]. One of the studies[7] found greater coverage of transplanted branching wild corals on concrete blocks whereas another

study[1] reported similar coral cover on concrete mats with and without transplants. Two of the studies[10,16] found greater coverage when corals were transplanted onto older artificial reefs[10], and coverage increased over time[16].

- **Survival (17 studies)**: Seventeen studies (ten replicated including three controlled and two before-and-after) in the Maldives[1], the Netherlands Antilles[2], Egypt[3a,b,9], Indonesia[5,8,13,16], Tanzania[6a], the Phillipines[7], Singapore[14], Belize[17], the USA[19a-c], and Mauritius[20], found that wild-grown corals transplanted onto artificial substrates (including concrete/cement[1,7,10,13,19a-c], plastic[3a,b,17], polythene[6a], PVC[2,5,9,15], and a subtidal seawall[14]) survived, but some results were species-[8,17], site[2]- or location[3b,17]-dependent. One of the studies[6a] found that coral fragments transplanted onto polythene-string grids had lower partial mortality than unattached fragments, another study[2] found no difference in survival for fragments transplanted onto PVC grids. Two of the studies[5,11] found higher survival for transplanted fragments placed above soft coral[5] or turf algae[11]. Four of the studies found fewer fish bites[15] and lower predation mortality[19a-c] for transplanted fragments protected from predators[15,19b], transplanted in clusters[19c], or near existing coral colonies[19a], whereas another study[20] found similar predation mortality on corals next to or away from existing colonies.
- **Condition (11 studies)**: Nine of eleven studies (ten replicated including three controlled) in the Netherlands Antilles[2], Egypt[3a,b], the Philippines[4], Tanzania[6a,b], Spain[11], Antigua[12] Kenya[15], Belize[17], and the USA[18], found that growth/ weight increased for fragments transplanted onto artificial substrates (including plastic mesh[3a,b], PVC[2,4,6b,11], polythene string[6a], cement[12], and plastic[17]), but some results were site-dependent[2,3b,17]. Two studies[4,11] found that growth was higher for fragments transplanted next to live colonies[4], and above turf algae[11], whereas another study18 found similar growth between fragments transplanted with or without beneficial invertebrates. One study[15] found growth was lower for fragments transplanted under cages.

**Background**

Using artificial materials to provide a structure to support coral fragments is a widely used method for coral reef restoration, particularly where the reef substrate is degraded. Fragments can be taken from live corals on nearby reefs, which either broke off during natural disasters or human activities or were intentionally removed for the purpose of transplantation. Corals used for transplanting procedures can be in colony, fragment, branch, nubbin (a small number of polyps) or settled larvae (coral spat) form (Meesters *et al*. 2013). Corals can be attached either directly or, in the case of coral spat, attached to a settlement tile and then to the substrate using epoxy resin, cable ties, rope, cement, or metal grids. Substrates are permanent or semi-permanent structures constructed from artificial material such as metal, concrete, or PVC.

This action specifically refers to transplanting wild-grown corals onto artificial substrate. Studies describing transplanting wild-grown coral onto natural substrates or nursery-grown corals onto natural or artificial substrates are covered in *Transplant wild-grown coral onto natural substrate, Transplant nursery-grown coral onto natural substrate, Transplant nursery-grown coral onto artificial substrate*. Studies describing cultivating corals are covered in *Cultivate coral in an artificial nursery located in a natural habitat*.

Meesters E.H.W.G., Smith S.R. & Becking L.E. (2013) *A review of coral reef restoration techniques*. Report number C028/14. IMARES: Wageningen, UR. Available from: https://library.wur.nl/WebQuery/wurpubs/fulltext/333153

A study in 1990–1993 at an area of degraded coral reef in Galu Falhu, Maldives (1) reported that transplanting coral fragments onto flexible concrete mats (Armorflex) did not lead to an increase in coral cover compared to mats naturally colonized by coral recruits, but approximately half the transplanted fragments survived. After 10 months, coral recruits were observed on the edges of the paving slabs anchoring the Armorflex mats and, after 16 months, recruits were observed on the mats both with

and without transplants (data not reported). There was no difference in density of coral recruits after 2.5 years with an average of 4/m$^2$ recorded on mats and 18/m$^2$ on the vertical edges of the paving slabs with and without transplants. After 2.5 years, 41–59% of coral transplants were still alive on the mats. In 1990–1991, Armorflex mats, weighted down using paving slabs, were installed on two 10 × 5 m areas of previously mined coral rubble substrate 0.5–1.8 m deep at four sites. Fragments of coral (number and species not reported) were taken from colonies near the study site and attached to one of the Armorflex mats at each site using marine cement; the other mat was left bare. Monitoring took place every 8–12 months for 2.5 years. **Costs (UK£)**: Armorflex mats with transplants cost £97/m$^2$ and bare Armorflex mats cost £66/m$^2$ (1999 value).

A replicated, controlled study in 1997 at three reefs at Curaçao, Netherlands Antilles (2) found transplanted fragments of stony coral *Madracis mirabilis* had a lower growth rate than unfragmented colonies, growth and survival rates of transplants varied between sites and there was no difference in growth or survival for fragments transplanted between or within sites or prepared using different methods. Sixteen weeks after transplanting, average growth of fragments across the four sites ranged from 8–12 mm/year (transplanted) to 16 mm/year (unfragmented). Average growth of fragments from Carmabi Buoy (13 mm/year) and Rif St. Marie (12 mm/year) were lower than unfragmented colonies at the same sites (both 16 mm/year), there was no difference in growth rate of fragments from Janthiel Bay (transplanted: 8 mm/year, unfragmented: 9 mm/year). Overall survival after 16 weeks ranged from 20–49% (average 38%) and there was no difference for fragments transplanted between or within sites. There was no difference in growth rate or survival between fragments prepared on the surface (growth 13 mm/year, survival 48%) compared to fragments prepared underwater (growth 12 mm/year, survival 34%). In April 1997, colonies of *Madracis mirabilis* 5–8 m deep were collected and cut into 10 cm fragments. Fragmentation took place either on the surface (in buckets of seawater) or underwater (see paper for full methods). Six PVC grids (1 × 1 m), each supporting 100 fragments, were secured to permanent quadrats 5–6 m deep at three sites (Carmabi Buoy, Rif St. Marie and Janthiel Bay). Approximately 21–30 colonies were left unfragmented at

each site. Photographs were used to measure growth and survival of fragments after two, four, eight, 12 and 16 weeks.

A replicated study (year not given) at a coral reef at Hurghada, Egypt (3a) reported that transplanted wild-grown stony coral fragments attached to plastic mesh substrate survived, and some grew. Those fragments attached using epoxy had higher survival than fragments attached without epoxy but mixed results for growth. One year after transplanting, 64% of *Favia stelligera* and 11% of *Stylophora pistillata* fragments attached without epoxy survived. Fragments of *Acropora humilis* and *Pocillopora damicornis* attached with epoxy had a higher survival rate (*A. humilis*: 21%; *P. damicornis*: 11%) than fragments attached without epoxy (*A. humilis*: 14%; *P. damicornis*: 8%). Growth after one year was 0.3 cm for *Favia stelligera* and growth was higher for epoxy-attached *Acropora humilis* (0.7 cm) than non-epoxy attached (0.3 cm). *Stylophora pistillata* and *Pocillopora damicornis* fragments did not grow. Stony coral fragments (78 *A. humilis*, 93 *P. damicornis*, 54 *S. pistillata*, and 11 *F. stelligerea*) were collected from colonies near the island of El-Fanadir and transported to Hurghada. Artificial substrate (comprising plastic mesh) was secured to the reef, 5–7 m deep, using nylon thread tied to a rock or fixed iron bar. Fifty-four *S. pistillata*, 11 *F. stelligera*, 36 *A. humilis* and 37 *P. damicornis* fragments were attached to the mesh by pushing fragments through holes. The remaining 42 *A. humilis* and 56 *P. damicornis* fragments were pushed through holes in the mesh then secured to the mesh using epoxy. Survival and growth of fragments were measured after six and 12 months.

A replicated study (year not given) at a coral reef at El-Fanadir, Egypt (3b) reported that two species of wild-grown stony coral fragments transplanted onto artificial substrate on the sea-facing (windward) side of the reef had higher survival and growth than fragments on the reef-facing (leeward) side, another three species fragments transplanted on the leeward side survived and two of those grew. One year after transplanting, survival was higher for windward fragments (*Acropora humilis*: 75%; *Pocillopora damicornis*: 79%) compared to leeward fragments (*A. humilis*: 69%; *P. damicornis*: 71%). Growth was higher for windward fragments (*A. humilis*: 0.6 cm; *P. damicornis*: 0.7 cm) compared to leeward (*A. humilis*: 0.5 cm; *P. damicornis*: 0.4 cm). Data were not statistically tested. Most fragments of the other three

stony coral species transplanted on the leeward side survived (*Acropora verweyi* 66%; *Acropora hemprichii 56%; Stylophora pistillata* 60%), and two of three species grew (*A. verweyi* 0.3 cm; *S. pistillata* 0.5 cm). Stony coral fragments (32 *A. humilis*, 28 *P. damicornis*, 15 *A. verweyi*, 18 *A. hemprichii* 15 *S. pistillata*) were collected from wild-growing colonies near the island of El-Fanadir and transported to the study site. Artificial substrate (comprising plastic mesh) was secured to the reef, 4–5 m deep on the leeward side, using nylon thread tied to a rock or fixed iron bar, and fragments were pushed into the plastic mesh. In addition, 42 of the 78 *A. humilis* and 56 of the 93 *P. damicornis* fragments were attached to the mesh secured on the windward side of the reef. Survival and growth of fragments were measured after six and 12 months.

A replicated study in 1996–1997 at two coral reefs in central Philippines (4) found that at one of the two sites transplanted fragments of stony coral *Porites attenuata* placed on artificial substrate next to live colonies of the same species had higher linear and surface area growth but produced fewer branches than fragments next to dead colonies. After 13 months, at Apo, linear growth of fragments was significantly higher next to live colonies (78–95 mm) than fragments next to dead colonies (70–77 mm), but there was no difference at Bais (live: 49–50 mm; dead: 58–66 mm). Average weekly surface area growth was also higher at Apo for fragments next to live colonies (28–51 $mm^2$) compared to dead (26–32 $mm^2$), but there was no difference at Bais (live 7–9 $mm^2$; dead: 13–15 $mm^2$). Fragments next to live colonies at Apo produced fewer branches (average 5/fragment), compared to those next to dead colonies (average 10–12/fragment), but there was no difference at Bais (average 1–2/fragment). In June 1996, twenty colonies of *Porites attenuata* were selected at each reef and four unbranched fragments (~4 cm long) were taken from each colony and fixed into 1-inch PVC pipe using marine epoxy. At each reef, eight cement platforms (1 × 0.3 m) were anchored 30 cm above the substrate, 10–11 m deep. Half the fragments at each site were swapped with fragments from the other site and all were fixed (9/platform) alternating with living or dead colonies. Growth was measured weekly for 13 months, and the number of branches was counted after 13 months.

A replicated study in 1999–2000 at a coral rubble site in Bunaken National Park, North Sulawesi (5) found that transplanting stony coral

*Acropora yongei* fragments on a PVC frame elevated above the soft coral canopy led to higher survival than fragments transplanted within the soft coral. Twelve months after transplanting, the survival rate for fragments on the frame was significantly higher (75%) than those within the soft coral (30%). In April 1999, ninety-nine fragments (~10 cm long with 2–4 branches) were collected from a single wild *Acropora yongei* colony. Forty-nine were attached to a PVC pipe and elevated 5–10 cm above the soft coral canopy. The remaining fragments were attached directly to the coral rubble substrate within the soft coral. Survival was recorded six and twelve months after transplanting.

A replicated, controlled study in 1998–1999 at Titia Reef, Tanzania (6a) found transplanted fragments of wild-grown stony corals *Acropora muricata* and *Acropora vaughani* on polythene string-grids had a higher survival rate, lower partial mortality, and a greater relative increase in weight of live tissue than unattached fragments, but no difference in total relative weight gain. After one year, 97% of attached fragments survived compared to 87% of unattached fragments. Although most fragments showed some partial mortality (dead tissue), 13% of attached fragments had no tissue loss compared to 7% of unattached fragments. Relative weight gain (weight gained as a proportion of original weight) of living tissue was higher for attached (1.6 × original weight) compared to unattached (1.1 × original weight) fragments. There was no significant difference in total relative weight gain (including live and dead tissue) between attached (1.9 × original weight) and unattached (1.6 × original weight) fragments. In November 1998, branches (average 34 cm long) from seven colonies of *Acropora muricata* and six of *Acropora vaughani* were collected within 2 km of the study site. Twenty-eight fragments were taken from each branch and weighed. Twenty fragments were tied to 2 × 1 m lengths of polythene string at 10 cm intervals, eight fragments were left unattached on the substrate (total 260 attached, 104 unattached fragments). Several (number not specified) of the 1 m lengths of string were tied together to form a grid which was placed (unsecured) on the substrate, 3 m deep. Survival and weight were measured after one year.

A replicated, controlled, before-and-after study in 1998–1999 at Titia Reef, Tanzania (6b) found transplanting damaged fragments of stony coral *Acropora muricata* onto a PVC rack did not result in any difference in relative weight of live tissue compared to undamaged fragments. After

eight months, there was no difference in relative weight gain (weight gained as a proportion of original weight) of living tissue between damaged (1.5–3.4) and undamaged (1.5–3.7) fragments. In November 1998, eighteen fragments of stony coral *Acropora muricata* were collected from each of 12 different coral colonies. Nine fragments from each colony were randomly selected and damaged (to simulate handling damage) by scraping a knife along the main branch and trimming all branch tips by 2 cm to remove soft tissue creating a scar 3–5 mm wide and 1 mm deep. Nine fragments were left undamaged. Fragments (108 damaged and 108 intact) were attached vertically to a PVC rack 3 m deep using cable ties. Fragments were weighed immediately before the treatment and again eight months later.

A replicated, before-and-after study in 2004–2005 on five coral patch-reefs in Pangasinan, Philippines (7) reported that transplanting branching *Acropora* and *Pocillopora* stony coral fragments onto concrete blocks led to a >72% survival of transplanted corals and an increase in percentage cover of branching corals (including growth of existing colonies, transplants and wild recruits) but a similar percentage cover of wild non-branching coral (including growth of existing colonies and recruits) in the surrounding plot compared to before transplantation. Statistical results were not reported. In the plots with transplanted branching *Acropora* and *Pocillopora* corals, the average percentage cover of branching corals increased from 2% the month before transplantation, to 11% the month after, and 16% one year after, whereas the average cover of non-branching corals in the plots before-and-after was similar (7% before, 6% after one month, 8% after one year). There was >72% survival of transplanted corals each month. In December 2005, fifty wild-grown *Acropora* and 50 *Pocillopora* (~15 cm width) fragments were collected from nearby reefs and transplanted in five 5 $m^2$ plots on degraded reefs (>20 m apart). Fragments were cemented individually onto 20 × 20 × 5 cm concrete blocks. Corals were surveyed using digital photographs taken monthly from September 2004–November 2005. Corals that were found dead during surveys were replaced (numbers not given).

A replicated study in 2005–2007 at three degraded coral reefs in North Sulawesi, Indonesia (8) found transplanting wild-grown stony coral species onto pre-cast concrete blocks attached to bamboo frames

led to up to 81% of fragments surviving but results were species-dependent. *Isopora brueggemanni* had the highest overall survival (81% after 11 months) followed by *Acropora yongei* (33% after 20 months) and *Acropora muricata* (21% after 15 months) fragments. *Pocillopora verrucosa* had the lowest survival (5–11% after 15–20 months) (results not tested statistically). Between September 2005 and June 2006, a 100 $m^2$ quadrat was set up at each of three sites of predominantly coral rubble. Quadrats were sub-divided into 100 squares (1 $m^2$) each containing a bamboo frame. Coral fragments (5–10 cm), collected from locally abundant colonies near each transplant site, were fixed onto precast concrete bases using epoxy then attached to the bamboo frames (approximately 50 fragments/frame) using cable ties. The following fragments were attached at each site: Gangga: *Acropora yongei* (1,855) and *Pocillopora verrucosa* (475), Meras: *Acropora muricata* (1,677) and *Pocillopora verrucosa* (378), Benaken: *Isopora brueggemanni* (1,749). Monitoring began several months after transplanting and was approximately monthly (Gangga: September 2005–May 2007, Meras: March 2006–June 2007, Bunaken: June 2006–May 2007). Survival was recorded for fragments attached to the concrete base with living tissue on at least one branch. **Costs (IDR)**: Materials cost 4,395,000 IDR (2010 value). The cost included the bamboo frames, cable ties, epoxy glue and concrete bases but did not include work time for setting up the experiment.

A study in 2009–2011 on a coral reef in the Red Sea, Egypt (9) found that all wild-grown stony corals *Acropora digitifera* and *Acropora selago* transplanted onto an artificial substrate survived for at least nine to 15 months. Nine or 15 months after transplanting, all transplanted *Acropora digitifera* and *Acropora selago* colonies survived. Wild colonies of *Acropora digitifera* and *Acropora selago* (approximately 25 cm diameter) were cut from their natural 1–2 m depth reef with a saw, through the coral rock close to their base, rather than live tissue. Four colonies of each species were transplanted in October 2009 and 12 of each species in April 2010. Colonies were attached with epoxy resin to 10 $cm^2$ or 12 $cm^2$ PVC plates, each of which was attached to a 28 × 35 × 5 cm concrete block with two 1.5 × 5 × 0.1 cm steel plates fixed with steel screws and plastic dowels, which could be removed to allow the PVC plate and colony to be weighed. Colonies were orientated towards the same cardinal direction as in their original location. Both colonies were surveyed in January 2011.

A site comparison study in 2009 on 15 artificial reefs and one natural reef in Aceh, Indonesia (10) found that transplanting stony coral *Acropora subglabra* and *Acropora formosa* fragments onto artificial reefs led to increased coral coverage two and three years after transplantation compared to one year after transplantation. There was no difference in the number of coral recruits with age of reef. There was no difference in average coral cover on two- (65%) and three-year-old (64%) artificial reefs and a nearby natural reef (50%), but these were all higher than on a one-year-old artificial reef (24%). However, the average number of coral recruits on one- (53 corals/$m^2$), two- (39 corals/$m^2$) and three-year-old (57 corals/$m^2$) artificial reefs was similar, with one-year-old and three-year-old artificial reefs having more coral recruits than the natural reef (31 corals/$m^2$). There were more types of coral on three-year-old (16) than one- (12) and two-year-old (11) artificial reefs (not tested statistically). In June 2006, August 2007 and December 2008–January 2009, artificial reefs consisting of nine 1.25 $m^2$ concrete cylinders enclosed within four concrete oblong blocks were placed on the substrate. These were topped with plastic pipes (one/cylinder and four/block). After 4–6 weeks, 25 cm stony coral fragments were attached to the pipes with cable ties. Fragments were obtained in 2006 from nearby healthy reefs and in 2007 and 2008 from colonies that had established on the artificial reefs. In November 2009, five artificial reefs from each deployment were randomly chosen and surveyed using digital photos and transects. As a comparison, corals were also surveyed on a nearby natural reef which was denuded in 2005.

A replicated study in 2008–2009 at a site in Menorca, Spain (11) found that transplanting fragments of juvenile temperate soft coral *Eunicella singularis* on PVC and rubber plates above patches of nuisance turf algae led to higher survival and growth compared to corals transplanted on plates attached onto the rocky substrate and exposed to turf algae overgrowth. Fifteen months after transplanting, survival rate of fragments not exposed to turf algae growth was 90% compared to 30% for fragments exposed to turf algae. Growth was higher for fragments not exposed (2.8 mm) compared to exposed turf algae (2.1 mm). In April 2008, small end-tips (<5 cm) were collected from 80 individual colonies of *Eunicella singularis*. Tips were attached to ten PVC plates overlayed with rubber (eight tips/plate) using holes and slits cut into

the rubber. Five plates were raised above the substrate on a frame, so they were not exposed to turf algae growth. Five plates were attached directly onto the rocky substrate, so they were exposed to turf algae growth. Survival was recorded in-situ after fifteen months. Growth was measured in a laboratory using 20 randomly selected tips (10 each from exposed and non-exposed plates).

A study in 2004 and 2010 at a coral reef restoration site on Maiden Island, Antigua (12) found that transplanting stony coral fragments on artificial structures (Reef Balls) led to lower growth rates for staghorn *Acropora cervicornis* coral but similar growth for elkhorn *Acropora palmata* and *Porites porites* compared to naturally-growing corals (data from other studies). Six years after transplanting, average growth of staghorn fragments was lower (4.9 cm/year, range: 1.67–7.93 cm/year) than reference values for naturally-growing colonies (10.8 cm/year, range 2.52–26.4 cm/year). Average growth of *Porites porites* fragments was similar (0.96 cm/year, range: 0.21–2.21 cm/year) to reference values (1.31 cm/year). The estimated average growth of two elkhorn fragments was similar (9.6 cm/year) to reference values (7.59 cm/year, range: 5.2–10 cm/year). In 2004 loose fragments of stony coral were collected and broken into 1–3 cm nubbins (small fragments). Nubbins were fixed to cement plugs and attached to approximately 3,500 artificial structures (Reef Balls) (data and methods from other studies). In 2010, six years after Reef Balls were installed, growth rate (linear extension) of fragments from three of the transplanted stony coral species (staghorn *Acropora cervicornis*, elkhorn *Acropora palmata*, and *Porites porites*), was measured using scaled photographs and based on an estimated average length of 1.9 cm/fragment at transplant. For comparison, reference data for naturally growing colonies was taken from previously published papers (see original paper for details).

A replicated study in 2013 at a coral reef site in Pulau, Indonesia (13) found that transplanting juvenile wild-grown stony corals *Porites lobata* and *Pocillopora damicornis* into crevices on artificial settlement tiles led to a higher survival rate compared to transplants in partial crevices or fully exposed on the tile surface, but no difference between small or large crevices. Twenty-nine days after settlement tiles were installed, survival rate for *Porites lobata* was higher in the full crevices (93%) compared to partial crevices (68%) and fully exposed (28%) and higher in partial

crevices compared to fully exposed. All juvenile *Pocillopora damicornis* except one had died by day eight although those in the full crevice survived longer (8 days) than partial crevice or fully exposed (both 6 days). There was no significant difference between survival rates for either species in different size crevices (data not reported). In June 2013, four hundred and eighty micro-nubbins (juveniles) were taken from each of five wild-grown colonies of *Porites lobata* and *Pocillopora damicornis*. Crevices (either 1.2 × 1.2 × 1.0 cm or 2.0 × 2.0 × 2.0 cm) were cut into 40 sand and cement settlement tiles (10 × 10 cm) creating a 'chequerboard' pattern. Twenty-four juveniles from the same species were glued to each tile in the full crevice (4/tile), partial crevice (open on one side 8/tile), or tile surface (fully exposed 12/tile). Tiles were placed on the sea floor, 7 m deep. Juveniles were monitored nine times during the 29-day experiment.

A randomized, before-and-after study in 2015–2016 on a seawall in the Singapore Strait, Singapore (14) reported that 58–100% of corals transplanted onto a subtidal seawall survived, depending on the species, that individuals of five of six species grew, and coral species richness and cover increased. After six months, average transplant survival was lower for *Pocillopora damicornis* (58%) than for all other stony coral species (*Echinopora lamellosa*: 100%; *Hydnophora rigida*: 100%; *Merulina ampliata*: 91%; *Platygyra sinensis*: 97%; *Podabacia crustacea*: 92%). All surviving fragments had positive growth rates (*Echinopora lamellosa*: 11 $cm^2$/month; *Hydnophora rigida*: 14 $cm^2$/month; *Platygyra sinensis*: 4 $cm^2$/month; *Pocillopora damicornis*: 26 $cm^2$/month; *Podabacia crustacea*: 4 $cm^2$/month) except *Merulina ampliata* (–1 $cm^2$/month). Coral species richness and cover on the seawall was higher (8 species, 21% cover) than before corals were transplanted (2 species, 3% cover). In August–December 2014, forty-two coral colonies (approximately 60 cm diameter) were collected from natural reefs and fragmented into 7–10 cm diameter colonies. Fragments were cultivated on nursery tables adjacent to a granite boulder seawall, elevated 0.5 m above the seabed 4 m deep for nine months. In April–August 2015, surviving colonies (213) were transplanted and fixed onto the seawall using epoxy putty. Fragments (diameter: 9–16 cm; area: 48–160 $cm^2$) of six stony coral species (36 fragments/species) were randomly arranged in four patches on the seawall at 3 m deep. Corals were counted on a 20 × 3 m section

before and six months after transplants were attached. Transplants were monitored from photographs over six months. **Costs (US$)**: Cultivation and transplantation cost $21,634 (2017 value).

A replicated, controlled study in 2016 at a coral reef near Wasini Island, Kenya (15) found that transplanting stony coral *Acropora verweyi* fragments under cages to exclude fishes led to fewer bites by coral-eating fishes, but lower growth and survival, and higher levels of biofouling, than uncaged or partially caged fragments. Bite rates by coral-eating fishes were lower for caged fragments (0 g/min) compared to uncaged (0.32 g/min) and partially-caged (0.09 g/min), but there was no difference between uncaged and partially-caged. Specific growth rate/day (see original paper for equation) of caged fragments was lower (0.0047) than uncaged (0.0078) and partially-caged (0.0099). After 100 days, survival was lower for caged (89%) than uncaged (98%) and partially-caged (99%) fragments. There was no difference in growth or survival between uncaged and partially-caged fragments. Total fouling (including molluscs, algae, and crustose coralline algae) was higher in caged (484 g/m$^2$) compared to uncaged (61 g/m$^2$) and partially-caged (78 g/m$^2$) structures, and there was no difference between uncaged and partially-caged. In April 2016, forty-five frames, comprising four 26 cm PVC pipes forming a cross, were installed 3 m deep at each of 15 locations along a 100 m stretch of reef. Four hundred and fifty naturally broken fragments of stony coral were collected from a reef, cut into 4 cm lengths, and suspended from the frames by fishing line (10 fragments/frame). A wire cage (0.5 × 0.25 × 0.25 m, 1.3 × 1.3 cm mesh size) was attached to 15 frames, a wire cage with two open sides was placed on 15 frames, and the remaining 15 frames were left uncovered. Bite rate (reported as fish-size-related mass in g/min – see original paper), growth, and survival were estimated each month using photographs. The experiment lasted 100 days.

A replicated, before-and-after study in 2013–2017 at a degraded coral reef in Pulau Badi, Indonesia (16) reported that transplanting wild-grown stony coral (mainly *Acropora* spp.) fragments onto artificial structures led to an increase in live coral cover on the structures and surrounding reef and, in one area, an increase in overall coral cover compared to before the structures were installed. After 24.5 years, the average cover of live coral on the structures ranged from 17%–89%.

Live coral cover on the natural substrate in one area was 7% before the structures were installed, whereas one year after transplanting, overall coral cover in the area was 48% of which 25% was on the structures. At the end of the study, average live coral cover in the oldest section (deployed in March 2013) was 21% on structures and 41% on the natural substrate. Data were not statistically analysed. Between March 2013 and September 2015, approximately 11,000 hexagonal steel-rod 'spider' structures (0.337 $m^2$) were placed 28 cm above the substrate across ~7,000 $m^2$ of degraded coral reefs. Eighteen stony coral fragments (~15 cm in length) were evenly spaced around the spider and attached using cable ties. Coral cover on the structures was recorded every four months for three years from 2014. Some spider structures were vandalized in 2014 and a large storm affected one section in 2017. **Costs (US$)**: Total cost of installing 11,000 spiders (including materials, construction labour, transport, coral attachment and installation labour) was US$174,000 (2015 values).

A replicated, controlled study in 2017–2019 at two reefs off Belize (17) found that some stony corals *Pseudodiploria strigosa* and *Siderastrea siderea* transplanted between offshore and nearshore reefs survived for at least 17 months and in three of four cases corals grew over that period. For transplants from an offshore to nearshore reef, survival after 17 months was 96% for both species. For transplants from a nearshore to offshore reef, survival after 17 months was 92% for *Pseudodiploria strigosa* and 32% for *Siderastrea siderea*. Survival of fragments placed back in their native reef was 100% for *Pseudodiploria strigosa* (offshore and nearshore) and 100% (offshore) or 72% (nearshore) for *Siderastrea siderea*. All transplanted *Pseudodiploria strigosa* had gained weight after 17 months (77–146% increase). For *Siderastrea siderea*, fragments transplanted to the nearshore reef gained weight (79% after 17 months) but those transplanted offshore did not (–3% after 17 months). Results on endosymbiont density, chlorophyll-a concentration and energy reserves were also reported. In 2017, colonies were collected from a nearshore and offshore reef (6 colonies/reef/species) and fragmented. Fragments were super-glued to plastic dishes with pre-drilled holes, attached to mesh nursery tables using cable ties and installed on the sea floor. Six fragments from each colony (12 colonies/species) were transplanted (nearshore to offshore, or offshore to nearshore) and six

were placed back in their native reef. A subset of fragments was collected after three (35 fragments), nine (37) or 17 months (46) to assess growth and survival.

A replicated study in 2016 at a coral reef site in Hawai'i, USA (18) found that transplanting stony corals *Pocillopora meandrina* onto artificial substrate, with or without beneficial invertebrates, resulted in growth over a six-month period. During the first eight weeks, coral growth was lower in corals with two beneficial invertebrate species (*Trapezia intermedia* and *Alpheus lottini*; 0.12% change/day) than for corals with just *T. intermedia* or no beneficial invertebrate species (0.15% change/day), and corals with *A. lottini* had similar growth to all other treatments (0.14% change/day). Over a period of six months, growth rates were similar for corals with or without beneficial invertebrate species (0.05–0.08% change/day). In May 2016, forty coral colonies (all hosting *T. intermedia*) were collected from a forereef habitat and assigned to one of four treatments (9–11 corals/treatment): transplanting with two beneficial invertebrate species (*A. lottini* and *T. intermedia*), with one (*A. lottini* or *T. intermedia*) or with none. All corals were attached to PVC plates and secured to cement blocks and randomly situated within an experimental grid (10 rows of four corals). Coral growth was assessed using buoyant weights for the first eight weeks and using aerial photographs for the subsequent six months.

A replicated, controlled study in 2020 on a reef off Florida, USA (19a) found that after transplanting wild-grown stony coral *Orbicella faveolata*, fragments placed adjacent to staghorn coral colonies *Acropora cervicornis* had lower predation mortality than those placed 25–50 cm away. Fragments placed adjacent to staghorn colonies had lower predation mortality (64% after four weeks) than corals located 25–50 cm away (86–92% after four weeks). Authors reported that corals placed 25–50 cm away had lower mortality than average predation rates in plots without staghorn colonies (100%), although this result was not tested for statistical significance. Predation mortality increased throughout the course of the experiment (1 week: 3–10%, 2 weeks: 27–61%, 4 weeks: 68–90%). Coral fragments were transplanted to three sites (10 m diameter, 36 fragments/plot) with staghorn coral (4 corals/$m^2$). Fragments were implanted in a cement mixture and placed 2–3 cm, 25 cm or 50 cm from the base of a staghorn coral colony. Every coral fragment was surveyed

visually one week, two weeks, and four weeks after transplanting. Mortality was also compared to fragments transplanted into additional plots that were >10 m from staghorn colonies.

A replicated, controlled study in 2020 on a reef off Florida, USA (19b) found that after transplanting wild-grown stony coral *Orbicella faveolata*, fragments protected with cages or spikes had lower predation mortality than those with no protection. Fragments protected by full cages had lower predation mortality after four weeks (0%) compared to those protected with open-top cages (75%), spikes (19%) or fragments with no protection (100%). One week after removing cages and spikes, 72–97% of the corals suffered complete mortality, and 96% of additional fragments that were transplanted at that time with no protection also suffered complete mortality. Predation mortality increased throughout the first month (1 week: 0–25%, 2 weeks: 0–87%, 4 weeks: 0–100%). Seventy-two coral fragments (5 $cm^3$) were transplanted to three reef plots (10 m diameter, 24 fragments/plot). At each plot, 12 fragments were protected by full cages and 12 by open-top cages. In addition, 24 cement "pucks" (10 cm diameter) were placed in each plot, 12 of which were fitted with steel spikes. A coral fragment was glued to the centre of each puck. Cages and spikes were removed after 1 month, and nine additional fragments with no protection were also transplanted at this time. Every coral fragment was surveyed visually one week, two weeks, and four weeks after transplant, and corals in the cage and spike treatments were also monitored one week after cage and spike removal.

A replicated, controlled study in 2020 on a reef off Florida, USA (19c) found that after transplanting wild-grown stony coral *Orbicella faveolata* onto artificial substrate, coral fragments transplanted as individuals suffered higher predation mortality than those transplanted in clusters. Individual coral fragments had higher predation mortality after four weeks (100%) compared to clusters of coral fragments (80%). Individual coral fragments also lost more tissue than clusters of coral fragments (data reported as statistical model results). Predation mortality increased throughout the course of the experiment (1 week: 0%, 2 weeks: 15–45%, 4 weeks: 80–100%). Coral fragments were transplanted onto a cement mixture as either an individual fragment (5 $cm^2$) or as a cluster of five fragments (25 $cm^2$). Three plots were established, with 12 individual fragments and five fragment clusters transplanted to each

plot. Fragments were placed haphazardly within plots, no closer than 50 cm from each other. Every coral fragment was surveyed visually one week, two weeks, and four weeks after transplant.

A replicated study in 2018–2019 at six coral reef sites off Mauritius (20) found that after stony coral *Acropora muricata* fragments were transplanted onto an artificial substrate next to existing stony coral *Porites lutea* colonies, *P. lutea* colonies with adjacent *A. muricata* fragments had similar bite density and surface area damage compared to colonies without adjacent *A. muricata* fragments. Data reported as statistical model outputs. In December 2018, at each of three sites, 40 *Acropora muricata* fragments were transplanted to 10 existing, isolated *Porites lutea* colonies (four fragments/ colony). Fragments consisted of a forked branch measuring approximately 30–40 cm in length. Transplanted *Acropora muricata* and existing *Porites lutea* were fixed together to concrete blocks with cement and string. Transplanted corals were monitored in February–March, April, and June 2019. In-situ *Porites lutea* (434 colonies) with no adjacent *Acropora muricata* were monitored across six sites every two months from September 2018 to June 2019.

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## 13.8. Change transplant attachment method

https://www.conservationevidence.com/actions/3995

- **Three studies** evaluated the effects of using different material to attach transplanted coral fragments to the substrate. One study was in the Phillipines[1], one in the British Virgin Islands[2] and one in the USA[3].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (3 STUDIES)**

- **Survival (3 studies)**: One replicated study in the Philippines[1] found that using marine epoxy or epoxy resin to attach transplanted coral fragments to the substrate resulted in a lower detachment rate and a shorter time to self-attachment compared to using cyanoacrylate glue (superglue). One replicated controlled study in the British Virgin Islands[2] found that attaching transplanted fragments to the substrate using adhesive cable-ties or cement led to a higher survival rate compared to fragments left unattached although there was no difference between attachment methods. One replicated, controlled study in the USA[3] found no difference in survival

between coral fragments attached using cement, epoxy, or cable ties and nails.

- **Condition** (**1 studies**): One replicated controlled study in the British Virgin Islands[2] found that using adhesive cable-ties or cement to attach transplanted fragments to the substrate led to a higher increase in live tissue growth compared to unattached fragments although there was no difference between attachment methods.

**Background**

Coral fragments can be naturally generated (e.g. broken off due to storm or wave action) or be taken from a donor colony for the purposes of transplanting to restore a degraded reef. Unattached fragments can be subjected to burial and abrasion (Dizon *et al*. 2008) and have a lower survival rate (Forrester *et al*. 2011). Therefore, fragments are usually attached to the substrate using man-made material most commonly an adhesive substance (such as marine epoxy or super-glue) is used to 'stick' the fragment to the substrate or the fragment is 'tied on' using cable ties, wire, string, etc.

This action specifically refers to the type of attachment method used to attach coral to the substrate. Studies describing transplanting wild-grown coral onto natural substrates or nursery-grown corals onto natural or artificial substrates are covered in *Transplant nursery-grown corals onto natural substrate; Transplant nursery-grown corals onto artificial substrate; Transplant wild-grown corals onto natural substrate*; and *Transplant wild-grown corals onto artificial substrate*.

Dizon R.M. Edwards A.J. & Gomez E.D. (2008) Comparison of three types of adhesives in attaching coral transplants to clam shell substrates. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 18, 1140–1148. https://doi.org/10.1002/aqc.944

Forrester G.E., O'Connell-Rodwell C., Baily P., Forrester L.M., Giovannini S., Harmon L., Karis R., Krumholz J., Rodwell T. & Jarecki L. (2011) Evaluating methods for transplanting endangered Elkhorn Corals in the Virgin Islands. *Restoration Ecology*, 19, 299–306. https://doi.org/10.1111/j.1526-100X.2010.00664.x

A replicated study in 2005 at three sites on a coral reef near Bolinao, north-western Philippines (1) found that using marine epoxy or epoxy putty to attach wild-grown coral nubbins (small fragments) to the natural substrate resulted in fewer nubbins becoming detached and a shorter time for nubbins to self-attach (naturally grow onto the substrate) than those attached using cyanoacrylate glue (superglue), but no difference in survival or the number of nubbins that self-attached. Detachment rates after five months were significantly lower for nubbins attached using marine epoxy (4/180) or epoxy putty (15/180) than those attached using cyanoacrylate glue (43/180). In addition, the time taken for nubbins to self-attach to the substrate was significantly shorter for marine epoxy (2.4 months) and epoxy putty (1.9 months) than for cyanoacrylate glue (2.9 months). The number of nubbins that self-attached did not differ significantly between adhesives (marine epoxy: 76, epoxy putty: 87, cyanoacrylate glue: 62). Similarly, survival rates did not vary between adhesive types (marine epoxy: 93, epoxy putty: 101, cyanoacrylate glue: 116). In June 2005 and August 2005, a total of 540 nubbins (fragments 2–3 cm in length) were collected from wild-grown colonies of 11 stony and one non-stony coral species at two donor sites. Substrates were created using 15 giant clam *Tridacna gigas* half-shells deployed at each of three sites at a depth of 2–4 m. Twelve nubbins (one/species – see original paper) were attached to each shell using one of the three adhesives (total 180 nubbins/adhesive). Five shells/adhesive type were deployed at each of three sites (15 shells/site). Monitoring took place every two weeks for five months (dates not given).

A replicated, controlled study in 2007–2009 at coral reef sites near Guana Island, British Virgin Islands (2), found that attaching transplanted storm-generated fragments of elkhorn *Acropora palmata* coral using adhesive, cable-ties or cement led to higher survival and increase in live tissue growth compared to unattached fragments, but there was no difference between attachment methods. After one year, survival of re-attached fragments was higher (2007: 56%; 2008: 62%)

compared to unattached fragments (2007: 3%; 2008: 12%). However, there was no difference in survival for fragments re-attached in 2007 (data not reported) or 2008 using different attachment methods (cable-ties: 56%; Z-spar epoxy: 50%; PC marine epoxy: 44%; cement: 66%). In addition, there was no significant difference in percentage increase in live tissue coverage between attachment types in 2007 (cable tie: 444%; Z-spar epoxy: 298%) or 2008 (cable tie: 153%; Z-spar epoxy: 83%; PC marine epoxy: 114%; cement: 97%). In July–August 2007 and 2008, eighty-six (2007) and 280 (2008) storm-generated fragments of elkhorn coral were collected from coral reefs. These were re-attached to the substrate (ensuring live tissue was in contact with the attachment surface) using cable ties (2007: 29 fragments; 2008: 103 fragments), Z-spar epoxy resin (2007: 28 fragments; 2008: 58 fragments), PC Marine Epoxy Putty (2008: 36 fragments), or cement (2008: 40 fragments), or left unattached at the collection site (2007: 29 fragments; 2008: 43 fragments). Survival was recorded after one year and growth measured after two and 12 months using scaled photographs.

A replicated, controlled study in 2019 at a coral reef restoration site off Florida, USA (3) found that attaching staghorn coral *Acropora cervicornis* fragments to natural substrate using a range of concrete mixtures or epoxy resulted in similar survival compared to when nails and cable ties were used. Transplants using a range of cement mixes or epoxy found average tissue mortality of 2% (cement) or 0% (epoxy) after eight days, with no additional mortality after one month and recovery after five months. Comparisons of the best performing cement and nails and cable ties found similar tissue mortality across all methods (0–27% partial mortality, 0–13% full mortality). Divers were able to transplant around 11 corals/dive using cement compared to six corals/dive using nails and cable ties (result was not tested for statistical significance). A total of 225 coral fragments were used to compare cement mixes and epoxy. Five bases (8–10 cm diameter) were deployed for each mix, and three fragments were placed in each base. Survival was assessed after eight days and then again at one and five months. A further 50 fragments were used to compare the best performing cement with the nail and cable tie method (25 fragments/method) and coral survival was assessed after one and four months. **Costs (US$)**: Transplanting nursery-grown coral in 2019 cost $0.05/coral when using cement, $0.47 when using epoxy

and $0.50 using the nail and cable tie method. Costs included materials only and did not include any shipping costs for materials.

(1) Dizon R.M., Edwards A.J. & Gomez E.D. (2008) Comparison of three types of adhesives in attaching coral transplants to clam shell substrates. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 18, 1140–1148. https://doi.org/10.1002/aqc.944

(2) Forrester G.E., O'Connell-Rodwell C., Baily P., Forrester L.M., Giovannini S., Harmon L., Karis R., Krumholz J., Rodwell T. & Jarecki L. (2011) Evaluating methods for transplanting endangered Elkhorn Corals in the Virgin Islands. *Restoration Ecology*, 19, 299–306. https://doi.org/10.1111/j.1526-100X.2010.00664.x

(3) Unsworth J.D., Hesley D., D'Alessandro M. & Lirman D. (2021). Outplanting optimized: developing a more efficient coral attachment technique using Portland cement. *Restoration Ecology*, 29(1), e13299. https://doi.org/10.1111/rec.13299

## 13.9. Remove problematic species

https://www.conservationevidence.com/actions/3996

- **Six studies** evaluated the effects of removing problematic species. One study was in each of Indonesia[1], the Philippines[2], the British Virgin Islands[3], Belize[4], Menorca (Spain)[5], and the USA[6]

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (6 STUDIES)**

- **Abundance/Cover (3 studies)**: Two of three replicated studies (including one randomized, controlled, one controlled, and one before-and-after) in Indonesia[1], the Philippines[2], and Menorca[5], found that repeated removal of problematic soft coral led to an increased number of stony coral colonies[1], and removing nuisance algae led to a higher number of juvenile corals[5]. One study[2], found that using topshell snails to control nuisance algae around transplanted coral did not lead to an increase in coral recruitment.
- **Survival (4 studies)**: One randomized, replicated, controlled study in the USA[6] found that removing nuisance algae and zoanthids from stony coral fragments immediately after

transplanting led to greater survival than fragments on sites cleared monthly or not at all. Two of three replicated, controlled studies in the Philippines[2], the British Virgin Islands[3], and Belize[4] found that clearing nuisance algae from around transplanted fragments did not lead to higher survival for elkhorn coral[3] and led to a lower survival rate for mustard hill coral[4] compared to areas without clearance. One study[2] found that using topshell snails to control nuisance algae around the transplant site did not result in higher survival of transplanted coral compared to areas without topshell snails.

- **Condition (3 studies)**: One of three replicated studies (including one randomized, controlled) in the British Virgin Islands[3], Belize[4], and the USA[6] found that removing nuisance macroalgae from around transplanted fragments of elkhorn coral led to higher live tissue growth compared to fragments transplanted without algae clearance[3], and one study[4] found removing nuisance algae from the transplant site led to a lower bleaching rate for one of two transplanted species, but no effect on growth of either. One study[6] found mixed effects on growth of transplanted stony coral fragments when algae and zoanthids were removed from around the transplant site immediately, monthly, or not removed.

**Background**

The effect of soft corals, macroalgae, and other problematic species on coral growth is mixed. However, degraded coral reefs can transition from hard (stony) coral to soft coral or algae-dominated systems (Fox *et al*. 2003; Souter *et al*. 2021). These systems can inhibit growth of stony coral and can lack the complex structure that reef-building stony corals provide, which is essential to support high biodiversity and ecosystem services (Fox *et al*. 2003; Souter *et al*. 2021). In addition, the quantity of algae on a coral reef system is an indicator of stress in coral reefs and is associated with a global decline in stony corals (Souter *et al*. 2021).

This action specifically relates to the effect on stony corals of removing problematic species from around transplant or restoration sites. The action refers to nuisance macroalgae species and not the beneficial algae (zooxanthellae) that form part of the coral itself. Studies describing transplanting wild-grown coral onto natural substrates or nursery-grown corals onto natural or artificial substrates are covered in *Transplant nursery-grown corals onto natural substrate; Transplant nursery-grown coral fragments onto artificial substrate; Transplant wild-grown corals onto natural substrate*; and *Transplant wild-grown coral onto artificial substrate*.

Fox H.E., Pet J.S., Dahuri R. & Caldwell R.L. (2003) Recovery in rubble fields: long-term impacts of blast fishing. *Marine Pollution Bulletin*, 46, 1024–1031. https://doi.org/10.1016/S0025-326X(03)00246-7

Souter D., Planes S., Wicquart J., Logan M., Obura D. & Staub F. (Eds) (2021) *Status of Coral Reefs of the World: 2020*. Global Coral Reef Monitoring Network (GCRMN). Available from: https://gcrmn.net/2020-report/

A replicated, before-and-after study in 1999–2000 at a coral rubble site in Komodo National Park, Indonesia (1) found that repeated clearance of problematic soft coral from hard coral rubble led to an increase in the number of stony coral colonies, while soft coral increased in areas cleared only once. The average number of stony coral colonies increased in the repeatedly cleared plots, from 2.94/$m^2$ when soft coral was first cleared to 7.15/$m^2$ five months later. In plots where problematic soft coral was only cleared once, coverage of soft coral increased to 95–100% five months after the initial clearance. In November 1999, problematic soft coral colonies were cleared from fifteen 1 × 1 m plots on areas of coral rubble left by blast fishing. Eight of the 15 plots were re-cleared of soft coral every other month while the remaining seven plots were left to be re-colonized by soft coral. Plots were surveyed and photographed, and stony coral colonies were counted when the soft coral was first cleared and again five months later.

A replicated, controlled study in 2007 near a coral reef in Bolinao, northwestern Phillipines (2) found that adding topshell snails *Trochus niloticus* to control algal growth on artificial reef structures (some

with transplanted stony coral fragments attached) did not lead to an increase in coral recruitment or survival rate of fragments compared to structures without topshells. Five months after transplanting, there was no significant difference in the average density of coral spat (settled larvae) on structures with topshells (8–30 spat/$m^2$) and without topshells (16–22/$m^2$), and no significant difference in survival rate for fragments (data not reported). Overall, survival rate ranged from 51% (*Acropora muricata*) to 97% (*Montipora digitata*). A total of 2,189 coral spat were recorded: 85% pocilloporids, 8% poritids, 4% acroporids, and 6% unidentifiable. In January 2007, forty-two concrete pallet balls (1.2 m diameter, 0.9 m high) (see paper for full design) were placed 4–8 m deep on sandy substrate 3–5 m from a natural coral reef. Ten topshells were added to each of 21 balls. Each pallet ball also had zero, 25 (5/species) or 50 (10/species) nursery-reared stony coral *Pocillopora damicornis, Acropora muricata, Porites cylindrica, Montipora digitata*, and *Echinopora lamellosa* fragments attached. Coral spat was counted on each pallet ball after approximately five months. Survival was recorded after six months.

A replicated, controlled study in 2008–2009 at two coral reef sites near Guana Island, British Virgin Islands (3) found that removing macroalgae from the transplant site for storm-generated fragments of elkhorn *Acropora palmata* coral led to a higher increase in live tissue growth but no difference in survival compared to fragments transplanted without algae removal. One year after transplanting, the increase in live tissue surface area was higher on fragments where algae had been removed (160%) than fragments transplanted without algae clearance (68%). Survival of fragments after one year did not vary significantly (algae cleared: 52%; algae not cleared: 60% survival). In July–August 2008, a total of 237 storm-generated fragments of elkhorn coral were collected from a coral reef and prepared for transplantation either at the collection site or another site 0.4–3.6 km away. Fragments were attached to the reef substrate, or dead elkhorn coral skeletons, using cable ties, marine epoxy or cement and ensuring live tissue was in contact with the substrate. Once attached, macroalgae was scraped away from a circle of 20 cm radius around 117 of the 237 fragments. Growth (surface area of live tissue) was measured after two and 12 months, and survival was recorded after 12 months using photographs.

A replicated, controlled study in 2003–2004 of 16 patch reefs in Belize (4) found that removing algae from transplant sites for massive starlet *Siderastrea siderea* and mustard hill *Porites astreoides* corals had mixed effects on bleaching and survival rates, and no effect on growth. Eighteen months after transplanting, average bleaching rates for massive starlet coral were lower at sites with algae removed (0.7%) than without (1%), and there was no effect on growth or survival rates (data not reported). For mustard hill coral, sites with algae clearance had lower average survival rates (85%) than those without (90%), and there was no effect on growth or bleaching rates (data not reported). In January 2003, and monthly thereafter, algae were removed from eight of 16 patch reefs (each 25–50 $m^2$) using hedge clippers and wire brushes. Algae were left intact on the other eight reefs. Shortly after initial algae removal, six 'fist-sized' massive starlet and mustard hill corals were collected from 1–3 km away and attached to each of the 16 reefs using masonry cement. Bleached corals were counted monthly, and surviving corals measured every three months, until August 2004.

A randomized, replicated study in 2008 at two sites in Menorca, Balearic Islands, NW Mediterranean (5), found that removing nuisance turf algae from near temperate soft coral *Eunicella singularis* colonies led to a higher number of juvenile coral than areas without algae removal. Three months after turf algae was removed, the average number of juvenile soft coral was higher in areas where turf algae had been removed (Cap Roig: 14.7/$m^2$; Na Ponsa: 2.2/$m^2$) compared to areas without algae removal (Cap Roig: 1.5/$m^2$; Na Ponsa 0/$m^2$). In April 2008, forty 40 × 40 cm quadrats were randomly marked 15–20 m deep at two sites (Cap Roig and Na Ponsa). Turf algae was removed from within 20 quadrats at each site and 20 were left undisturbed. Three months later, in July 2008, quadrats were inspected in-situ and *Eunicella singularis* recruits (~3–5 mm high) were counted.

A replicated, randomized, controlled study in 2016–2017 at a coral reef in Florida, USA (6) found that removing algae, along with zoanthids *Palythoa caribaeorum*, from around colonies of three stony coral species immediately after transplanting resulted in greater overall survival compared to those with algae and zoanthids removed monthly or not removed, and there were mixed effects on growth. After 17–18 months, overall survival for three coral species combined was greater

at transplant sites where algae and zoanthids were initially removed than at sites where algae and zoanthids were removed monthly or not removed (data reported as statistical model results). Removing algae and zoanthids initially or monthly led to greater increases in volume of staghorn coral *Acropora cervicornis* colonies (990–1,409%), greater losses in surface area of great star coral *Montastraea cavernosa* colonies (–25 to –30%) and similar losses in surface area of mountainous star coral *Orbicella faveolata* colonies (–4 to –12%) compared to colonies that did not have algae and zoanthids removed (staghorn: 570%; great star: –8%; mountainous star: –12%). In March 2016, forty-five nursery-grown colonies from each of three coral species (staghorn: 66–575 $cm^3$; great star: 45–120 $cm^2$; mountainous star: 38–130 $cm^2$) were transplanted onto hard substrate on a reef. One of each of three treatments was applied to each colony: algae and zoanthids *Palythoa caribaeorum* removed from a circle of 25 cm radius at the time of transplanting or at monthly intervals, or algae not removed (15 colonies/species/treatment). Colony survival and growth (volume or surface area of live tissue) were recorded after three, six, nine, 13 and 17–18 months.

(1) Fox H.E., Pet J.S., Dahuri R. & Caldwell R.L. (2003) Recovery in rubble fields: long-term impacts of blast fishing. *Marine Pollution Bulletin*, 46, 1024–1031. https://doi.org/10.1016/S0025-326X(03)00246-7

(2) Villanueva R.D., Edwards A.J. & Bell J.D. (2010) Enhancement of grazing gastropod populations as a coral reef restoration tool: Predation effects and related applied implications. *Restoration Ecology*, 18, 803–809. https://doi.org/10.1111/j.1526-100X.2010.00742.x

(3) Forrester G.E., O'Connell-Rodwell C., Baily P., Forrester L.M., Giovannini S., Harmon L., Karis R., Krumholz J., Rodwell T. & Jarecki L. (2011) Evaluating methods for transplanting endangered Elkhorn Corals in the Virgin Islands. *Restoration Ecology*, 19, 299–306. https://doi.org/10.1111/j.1526-100X.2010.00664.x

(4) McClanahan T.R., Huntington B.E. & Cokos B. (2011) Coral responses to macroalgal reduction and fisheries closure on Caribbean patch reefs. *Marine Ecology Progress Series*, 437, 89–102. https://doi.org/10.3354/meps09285

(5) Linares C., Cebrian E. & Coma R. (2012) Effects of turf algae on recruitment and juvenile survival of gorgonian corals. *Marine Ecology Progress Series*, 452, 81–88. https://doi.org/10.3354/meps09586

(6) Lustic C., Maxwell K., Bartels E., Reckenbeil B., Utset E., Schopmeyer S., Zink I. & Lirman D. (2020) The impacts of competitive interactions on coral colonies after transplantation: A multispecies experiment from the Florida Keys, US. *Bulletin of Marine Science*, 96, 805–818. https://doi.org/10.5343/bms.2019.0086

# Other species management

## 13.10. Introduce larvae directly onto natural or artificial reefs to encourage settlement

https://www.conservationevidence.com/actions/3997

- **Three studies** evaluated the effects of releasing larvae directly onto natural or artificial reefs to encourage settlement. One study was in each of Australia[1], the USA[2], and Palau[3].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (3 STUDIES)**

- **Abundance (3 studies)**: Two of three replicated studies (including one controlled) in Australia[1], the USA[2], and Palau[3], found that settlement density was higher when larvae were released directly onto tiles on the reef compared to naturally settling on tiles[1] or the reef[3] nearby, whereas another study[2] found no difference between settlement rate of larvae released directly onto the different natural substrates. One of the studies[1] found that settlement rate was higher when larvae were released onto the reef at a slower rate.
- **Survival (1 study)**: One replicated study in the USA[2] found no difference in survival for larvae released directly onto different natural substrates.

**Background**

Introducing larvae directly onto natural or artificial reefs can be used with the aim of enhancing the natural settlement process to restore coral species on a reef. Wild coral spawning events often result in 'slicks' containing billions of egg/sperm bundles (Heyward *et al*. 2002). These are collected and allowed to transform into larvae in a nursery (either in-situ or ex-situ) before being released directly onto the natural or artificial reef (Heyward *et al*. 2002; Edwards *et al*. 2015). The release process usually involves covering an area of reef to prevent the larvae dispersing then introducing larvae through a hose or syringe into the covered area. The reef remains covered for approximately 24 hours to enable the larvae to settle onto the substrate.

This action describes introducing larvae onto a reef and once settled, allowing the larvae to develop without further intervention. Studies describing cultivating corals in in-situ or ex-situ nurseries are covered in *Cultivate coral larvae in an artificial nursery located in a natural habitat*; and *Cultivate corals in an ex-situ nursery*. Studies describing transplanting nursery-grown coral onto natural or artificial substrates or wild-grown corals onto natural or artificial substrates are covered in *Transplant nursery-grown corals onto natural substrate; Transplant nursery-grown coral fragments onto artificial substrate; Transplant wild-grown corals onto natural substrate*; and *Transplant wild-grown coral onto artificial substrate*.

Edwards A.J., Guest J.R., Heyward A.J., Villanueva R.D., Baria M.V., Bollozos I.S.F. & Golbuu Y. (2015) Direct seeding of mass-cultured coral larvae is not an effective option for reef rehabilitation. *Marine Ecology Progress Series*, 525, 105–116. https://doi.org/10.3354/meps11171

Heyward A.J., Smith L.D., Rees M. & Field S.N. (2002). Enhancement of coral recruitment by *in situ* mass culture of coral larvae. *Marine Ecology Progress Series*, 230, 113–118. https://doi.org/10.3354/meps230113

A replicated, controlled study in 1997 at an in-situ nursery and natural coral reef site in Coral Bay, Australia (1), reported that reefs re-seeded using stony coral larvae cultivated in an ex-situ nursery then pumped directly onto artificial settlement tiles on the reef had a higher settlement rate than larvae naturally settled on tiles placed on a nearby reef. In addition, pumping larvae over a longer time led to a higher settlement rate than larvae pumped onto the tiles quickly. Six weeks after re-seeding, there was an average of 236 coral spat (settled larvae)/tile (range 80–384) in areas where direct re-seeding took place over 12 hours, 23/tile (range 0.17–68) in areas re-seeded for 20 minutes, and 0.27/tile in areas with natural settlement. Data were not statistically tested. In March 1997, approximately four million egg/sperm bundles (gametes) were collected from a stony coral (Acroporid family) spawning slick on the ocean surface and transferred to four in-situ settlement ponds (~one million/pond) and cultured for seven days. Six terracotta settlement tiles (110 × 110 × 10 mm) were attached to the substrate at each of four sites around Coral Bay using a 10 cm roofing nail and PVC spacer pipe (4 cm diameter, 2 cm long). A mesh (1.8 × 1.0 m, 200 μm) was placed over the tiles at two sites and cultured larvae was pumped from the pond into one meshed area for 20 minutes and another for 12 hours using a hose. The other two sites were left to be naturally settled. All tiles were retrieved after six weeks and the number of coral spat on each counted.

A replicated study in 2006 at a coral reef in Biscayne National Park, Florida, USA (2) found that after larvae were released directly onto three natural substrates on a reef, there was no difference in settlement rate or survival of brooding coral *Porites asteroids* spat (settled larvae) growing on the different substrates. Two to three days after release, there was no difference in the proportion of released larvae that settled on different substrate types (6.1–8.9 % settled on crustose coralline algae; 4.7–8.8 % on turf algae, and 7.9–10.1 % on a mixture of crustose coralline algae, turf algae and bare substrate). Overall survival after five months was <1% and there was no difference in survival rate between spat on crustose coralline, turf, or mixed substrates (data reported as interval survival rate). In April and May 2006, larvae were collected from 20 wild-grown adult *Porites asteroides* colonies (see paper for methods). Ten seeding pods (see paper for design) were attached to individual 10 × 10 cm plots on each of three substrates (crustose coralline algae, turf algae,

and mixed crustose coralline algae, turf algae and bare substrate), 4 m deep. Coral larvae were injected into each seeding pod (approximately 100/pod in April, 75/pod in May). Once pods were removed after two (April) or three (May) days, coral spat settlement patterns were mapped and survival recorded using a fluorescence technique (see original paper). Monitoring was carried out at irregularly spaced intervals for five months.

A replicated, study in 2007–2008 at Iou Lukes reef, Palau (3), found that using nursery-cultivated stony coral *Acropora digitata* larvae to directly 'seed' artificial reef structures initially led to a higher density of *Acropora* spp. coral spat (settled larvae) on the structures compared to natural settlement, but there was no difference in density of coral spat over time. Average coral spat density after five weeks was significantly higher on settlement tiles seeded with larvae (205/0.1 $m^2$) than unseeded tiles (52). However, after 30 weeks, stony coral density on seeded tiles had declined significantly (60/0.1 $m^2$), and there was no longer a statistically significant difference compared to unseeded tiles (33/0.1 $m^2$). In January 2007, fourteen concrete/limestone 'pallet-balls' (1.2 × 0.9 m) were placed 3–5 m apart, 5–8 m deep on the seafloor adjacent to a natural reef. Fibre cement settlement tiles (10 × 10 × 0.6 cm) were attached to each ball in mid-January 2008 (4 tiles/ball). In April 2008, a tent with an inner 250 × 250 μm mesh was placed over each of seven randomly selected pallet balls, and 40,000–260,000 nursery-cultured stony coral larvae were poured onto each pallet-ball (density 54.6–459.8/0.1 $m^2$). Tents remained for 24 hours. Coral density was recorded on tiles retrieved five and 30 weeks after wild-growing coral colonies had spawned.

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(3) Edwards A.J., Guest J.R., Heyward A.J., Villanueva R.D., Baria M.V., Bollozos I.S.F. & Golbuu Y. (2015) Direct seeding of mass-cultured coral larvae is not an effective option for reef rehabilitation. *Marine Ecology Progress Series*, 525, 105–116. https://doi.org/10.3354/meps11171

## 13.11 Play reef 'soundscapes' to enhance settlement of coral larvae

https://www.conservationevidence.com/actions/4017

- We found no studies that evaluated the effects on corals of playing reef 'soundscapes' to enhance settlement of coral larvae.

*'We found no studies' means that we have not yet found any studies that have directly evaluated this action during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the action has any desirable or harmful effects.*

**Background**

Settlement of coral larvae is a vital process in the maintenance and restoration of reef habitats. Coral larvae respond to a range of environmental variables, and there is a growing understanding of the role of acoustic cues in the settlement process (Lillis *et al*. 2016), with larvae responding to the 'soundscapes' typical of healthy reef habitat with higher settlement rates (Lillis *et al*. 2018). Playing recordings of reefs soundscapes may therefore contribute to higher settlement, with beneficial consequences for expansion of existing reefs and restoration of degraded ones.

Lillis A., Bohnenstiehl D., Peters J.W. & Eggleston D. (2016) Variation in habitat soundscape characteristics influences settlement of a reef-building coral. *PeerJ*, 4, e2557. https://doi.org/10.7717/peerj.2557

Lillis A., Apprill A., Suca J.J., Becker C., Llopiz J.K. & Mooney T.A. (2018) Soundscapes influence the settlement of the common Caribbean coral *Porites astreoides* irrespective of light conditions. *Royal Society Open Science*, 5, 181358. https://doi.org/10.1098/rsos.181358

## 13.12 Use electrical current to stimulate coral growth

https://www.conservationevidence.com/actions/3998

- **Four studies** evaluated the effects of using electrical current to stimulate coral growth. Two studies were in the Philippines[2,3], and one study was in each of Jordan[1], and Indonesia[4].

COMMUNITY RESPONSE (0 STUDIES)

**POPULATION RESPONSE (4 STUDIES)**

- **Survival (4 studies)**: Four replicated studies (including one controlled and one randomized, controlled) in Jordan[1], the Philippines[2,3], and Indonesia[4], found that applying an electrical current had mixed effects on coral survival. Two of the studies[1,2] found that using an electrical current to stimulate coral growth led to higher survival for fragments of some species[1,2], compared to those without electrical stimulation[2]. One study[3] found that survival was higher for corals six months after the electrical current was turned off compared to fragments that did not receive electrical current.
- **Condition (4 studies)**: Four replicated studies (including one controlled one randomized, controlled) in Jordan[1], the Philippines[2,3], and Indonesia[4], found that applying an electrical current had mixed effects on coral growth and attachment success. One of the studies[2] found that electrical current led to greater girth growth, but not linear growth of fragments compared to fragments without electrical current, whereas another study[4] found that growth of fragments was lower in an electrical field with a cathode than without. One of the studies[1] found that electrical current led to all coral nubbins (small fragments) attaching to the substrate. One study[3] found no difference in growth for corals six months after the electrical current was turned off compared to fragments that did not receive electrical current.

**Background**

In the 1980s, a technique was developed using electrolysis to construct 'electric reefs' by passing an electrical current using an anode (that causes water to break up and oxygen to form) and a cathode (that causes water to break up and release hydrogen bubbles into the water). The smaller anode is made from electrically conductive material and often suspended in the water column. The cathode is also made from an electrically conductive material such as wire mesh and is attached to the seafloor (in a natural habitat) (Hilbertz 1975). Dissolved minerals (such as calcium carbonate) break down near the anode and recombine onto the cathode – a process known as mineral accretion – rapidly creating a hard limestone substrate (also known as 'biorock'). Electric reefs have been used with the aim of restoring areas of coral reef and providing a substrate for natural coral settlement. More recently, coral fragments have been transplanted onto the cathode so mineral accretion around the coral skeleton itself may improve the health and growth rate of corals or enable a more secure attachment to the substrate (Sabater & Yap 2002, Borell *et al*. 2010).

This action specifically relates to the use of electrical current on transplanted coral fragments. Studies describing the creation and use of biorock to provide habitat are described in *Use natural materials to restore/repair/create habitat for corals to encourage natural coral settlement*. Studies describing transplanting nursery-grown coral onto natural or artificial substrates or wild-grown corals onto natural or artificial substrates are covered in *Transplant nursery-grown corals onto natural substrate; Transplant nursery-grown coral fragments onto artificial substrate; Transplant wild-grown corals onto natural substrate*; and *Transplant wild-grown coral onto artificial substrate*.

Borell E.M., Romatzki S.B.C. & Ferse S.C.A. (2010) Differential physiological responses of two congeneric scleractinian corals to mineral accretion and an electric field. *Coral Reefs*, 29, 191–200. https://doi.org/10.1007/s00338-009-0564-y

Hilbertz W.H. (1975) Towards self-growing structures. *Industrialization Forum*, 2, 53–56.

Sabater M.G. & Yap H.T. (2002) Growth and survival of coral transplants with and without electrochemical deposition of CaCO3. *Journal of Experimental Marine Biology and Ecology*, 272, 131–146. https://doi.org/10.1016/S0022-0981(02)00051-5

A replicated study (year not given) in the Gulf of Aqaba, Jordan (1), reported that using an electrical current to stimulate mineral formation resulted in all transplanted stony coral nubbins (small fragments) attaching to the substrate, but survival rates were species, but not depth-dependent. Eight weeks after transplanting, all coral nubbins were fully attached to the steel mesh cathodes. After three months, survival rate for all nubbins was over 80%, except *Pocillopora damicornis* at 6 m deep (16%) and *Acropora squarrosa* (reported on graph as *Pocillopora damicornis*) at 12 m (72%). Survival after seven months ranged from 0% (*Pocillopora damicornis* at 6 m) to 96% (*Acropora variabilis* at 6 m and 18 m deep), and after 12 months (12 m depth only) survival ranged from 36% (*Acropora squarrosa*) to 72% (*Acropora variabilis*). Results were not statistically tested. A total of 400 nubbins from six stony coral species (*Acropora variabilis, Acropora squarrosa, Stylophora pistillata, Pocillopora damicornis, Montipora danae*, and *Pavona varians*) were collected from colonies near the experimental site. Four cathodes comprising 1 × 3 m non-galvanized 10 mm steel mesh squares were fixed horizontally to the natural coral rock at each of one, six, 12 and 18 m deep using steel wire. The anode (titanium mesh square) was suspended 20 cm above the cathode. Nubbins were attached to each square (25 from each of four of the six species/square; see paper for configuration). The cathode and anode were attached to a power supply and an electrical current was put through the squares for 18h/day for two months, then switched off. Attachment was recorded after two and eight weeks. Survival was recorded after three and seven months, and after 12 months just for 12 m depth.

A replicated, randomized, controlled study in 1999–2000 at a coral reef at Quezon Island, northern Philippines (2) found that using electrical

current to stimulate mineral accretion on stony coral *Porites cylindrica* nubbins (small fragments) resulted in a higher survival rate and greater girth growth than unstimulated nubbins but no difference in % linear growth. However, survival and linear growth of electrically stimulated nubbins were both lower than naturally growing coral. Six months after electrical stimulation started, survival of stimulated nubbins (86%) was higher than unstimulated (70%) but lower than adjacent natural colonies (95%). Average girth growth after six months was higher for stimulated (1.4–1.7 mm) than unstimulated (0.8–1.1 mm) nubbins. There was no difference in % linear growth between stimulated (38%) and unstimulated (36%) nubbins but both were lower than natural colonies (45%). Results for corallite development are presented in the original paper. In December 1999, two-hundred-and-sixty 'thumb-sized' nubbins were randomly collected from three patches of wild-growing stony coral colonies within the experiment site. An additional 40 branches on colonies on each patch were randomly tagged and left to grow naturally. Two 1 × 1 m galvanized steel mesh sheets were attached to the seabed at each of three locations 4–8 m deep. At each site, a PVC frame was constructed above each sheet with electricity supplied to the stimulated nubbins (see paper for methods). Survival, linear, and girth growth were measured every two months for six months.

A randomized, replicated, controlled study in 1999–2001 at a coral reef at Quezon Island, northern Philippines (3) found that six months after an electrical current to stimulate mineral accretion was switched off, survival of stony coral *Porites cylindrica* nubbins (small fragments) was higher for stimulated nubbins than unstimulated, but there was no difference in linear or girth growth. After six months, the number of surviving nubbins was higher for previously stimulated (63/66, 95%) than unstimulated (55/64, 86%) nubbins. There was no difference in linear growth between stimulated (2.4–4.0 mm/2 months) and unstimulated (2.5–4.4 mm/2 months) nubbins. Girth growth did not differ after six months between stimulated (2.2 mm) and unstimulated (2.0 mm) nubbins. In December 1999, two-hundred-and-sixty 'thumb-sized' nubbins were randomly collected from three patches of wild-growing *Porites cylindrica* colonies within the experiment site. Two 1 × 1 m galvanized steel mesh sheets were attached to the seabed at each of three locations 4–8 m deep. At each site, a PVC frame was constructed

above each sheet with electricity supplied to the stimulated nubbins (see study 2 for experimental set-up). After six months, the electrical current was switched off and 66 stimulated, and 64 unstimulated nubbins were left in-situ for a further six months. Survival and linear growth were measured every two months for six months, girth growth was measured after six months.

A replicated, controlled study in 2006 at an artificial nursery on coral rubble in North Sulawesi, Indonesia (4) found that cultivating stony coral *Acropora youngei* and *Acropora pulchra* fragments inside an electric field with a cathode led to lower survival for *Acropora youngei* and lower linear growth for both species than fragments cultivated inside an electric field without a cathode, or fragments cultivated outside the electric field. After four months, *Acropora yongei* fragments cultivated with a cathode had a lower survival rate (68%) than fragments without a cathode (99%) or outside the electric field (99%). However, there was no difference in survival for *Acropora pulchra* (with cathode: 83%, without cathode: 91%, outside electric field: 87%). Linear growth was lower for both species with cathode (*A. yongeii*: 10 mm, *A. pulchra*: 8 mm) compared to without cathode (*A. yongeii*: 22 mm, *A. pulchra*: 11 mm) and for *A. yongeii* outside the electric field (15 mm) but higher than *A. pulchra* outside the electric field (5 mm). There were mixed effects for chlorophyll fluorescence, chlorophyll *a*, and zooxanthellae density and concentrations (see original paper for results). In March 2006, three hundred and fifty fragments (6–8 cm) each from *A. yongei* and *A. pulchra* were collected from near the experiment site. Fourteen frames were placed on coral rubble substrate (five electric field with cathode, five electric field insulated from the cathode, four outside the electric field) (see original paper for methods). Twenty-five fragments from both species were glued to each frame. Monitoring took place every four weeks for four months. Final mortality rate and growth (linear skeletal extension) was measured after four months.

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# 14. Education and awareness

**Background**

Education is central to the success of any long-term efforts to conserve and restore coral reefs (Browning *et al*. 2006). However, human behaviour is complex and influenced by a great many factors. Providing education and awareness training may not be sufficient to change behaviour – particularly in the context of the considerable threats faced by coral reefs. Behaviour change to affect coral reef conservation will need to come from all those involved including politicians, policy-makers, fishers, tourists, local and indigenous people.

Studies do not always measure the outcome on coral reefs of education programmes and it may be necessary to work with social and behavioural scientists to consider how attitudes, values and social norms relate to coral reef conservation. In addition, Conservation Evidence does not systematically search specialist behavioural and social science journals and reports, therefore it is likely we have missed some relevant studies.

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# Appendix 1: English language journals (and years) searched

Journals (and years) searched and for which relevant papers have been added to the Conservation Evidence discipline-wide literature database. An asterisk indicates the journals most relevant to this synopsis.

| **Journal** | **Years searched** | **Topic** |
|---|---|---|
| *Acrocephalus* | 2009–2018 | All biodiversity |
| *Acta Chiropterologica* | 1999–2019 | All biodiversity |
| *Acta Herpetologica* | 2006–2018 | All biodiversity |
| *Acta Oecologica* | 1990–2018 | All biodiversity |
| *Acta Theriologica* | 1977–2000 | All biodiversity |
| *African Bird Club Bulletin* | 1994–2017 | All biodiversity |
| *African Journal of Aquatic Science* | 2000–2022 | All biodiversity |
| *African Journal of Ecology* | 1963–2016 | All biodiversity |
| *African Journal of Herpetology* | 1990–2018 | All biodiversity |
| *African Journal of Marine Science* | 1983–2018 | All biodiversity |
| *African Primates* | 1995–2012 | All biodiversity |
| *African Sea Turtle Newsletter* | 2014–2018 | All biodiversity |
| *African Zoology* | 1979–2013 | All biodiversity |
| *Agriculture, Ecosystems & Environment* | 1983–2021 | All biodiversity |
| *Ambio* | 1972–2019 | All biodiversity |
| *American Journal of Primatology* | 1981–2019 | All biodiversity |
| *American Naturalist* | 1867–2019 | All biodiversity |
| *Amphibia-Reptilia* | 1980–2012 | Amphibian Conservation |
| *Amphibia-Reptilia* | 2013–2014 | Reptile Conservation |

| Journal | Years searched | Topic |
|---|---|---|
| *Amphibia-Reptilia* | 1980–2018 | All biodiversity |
| *Amphibian & Reptile Conservation* | 1996–2006 | Amphibian Conservation |
| *Amphibian & Reptile Conservation* | 1996–2018 | All biodiversity |
| *Animal Biology* | 2003–2013 | All biodiversity |
| *Animal Conservation* | 1998–2021 | All biodiversity |
| *Animal Nutrition* | 2015–2019 | All biodiversity |
| *Animal Welfare* | 1992–2019 | All biodiversity |
| *Animals* | 2011–2019 | All biodiversity |
| *Annales Zoologici Fennici* | 1964–2013 | All biodiversity |
| *Annales Zoologici Societatis Zoologicae Botanicae Fennicae Vanamo* | 1932–1963 | All biodiversity |
| *Annual Review of Ecology, Evolution, and Systematics* (*formerly Annual Review of Ecology and Systematics 1970–2002*) | 1970–2021 | All biodiversity |
| *Annual Review of Entomology* | 2000–2019 | All biodiversity |
| *Antarctic Science* | 1980–2018 | All biodiversity |
| *Anthrozoos* | 1987–2019 | All biodiversity |
| *Apidologie* | 1958–2009 | All biodiversity |
| *Applied Animal Behaviour Science* | 1984–2019 | All biodiversity |
| *Applied Herpetology* | 2003–2009 | All biodiversity |
| *Applied Vegetation Science* | 1998–2022 | All biodiversity |
| *Aquarium Sciences and Conservation* | 1997–2001 | All biodiversity |
| *Aquatic Biology* | 2007–2022 | All biodiversity |
| *Aquatic Botany* | 1975–2022 | All biodiversity |
| *Aquatic Conservation: Marine and Freshwater Ecosystems* | 1991–2022 | All biodiversity |
| *Aquatic Ecology* | 1968–2023 | All biodiversity |
| *Aquatic Ecosystem Health & Management* | 1998–2022 | All biodiversity |
| *Aquatic Invasions* | 2006–2022 | All biodiversity |
| *Aquatic Living Resources* | 1988–2018 | All biodiversity |
| *Aquatic Mammals* | 1972–2018 | All biodiversity |
| *Arid Land Research and Management* (*formerly Arid Soil Research and Rehabilitation 1987–2000*) | 1987–2013 | All biodiversity |
| *Asian Herpetological Research* | 2010–2018 | All biodiversity |
| *Asian Primates* | 2008–2012 | All biodiversity |
| *Asiatic Herpetological Research* | 1993–2008 | All biodiversity |
| *Auk* | 1980–2016 | All biodiversity |
| *Austral Ecology* | 1977–2019 | All biodiversity |

| **Journal** | **Years searched** | **Topic** |
|---|---|---|
| *Austral Entomology* | 2014–2019 | All biodiversity |
| *Australasian Journal of Herpetology* | 2009–2012 | All biodiversity |
| *Australian Mammalogy* | 2000–2019 | All biodiversity |
| *Avian Conservation and Ecology* | 2005–2016 | All biodiversity |
| *Basic & Applied Herpetology* | 2011–2018 | All biodiversity |
| *Basic and Applied Ecology* | 2000–2021 | All biodiversity |
| *Behavioral Ecology* | 1990–2013 | All biodiversity |
| *Behaviour* | 1948–2013 | All biodiversity |
| *Biawak* | 2001–2017 | All biodiversity |
| *Bibliotheca Herpetologica* | 1999–2017 | All biodiversity |
| *BioControl (formerly Entomophaga until 1998)* | 1956–2016 | All biodiversity |
| *Biocontrol Science and Technology* | 1991–1996 | All biodiversity |
| *Biodiversity* | 2000–2018 | All biodiversity |
| *Biodiversity and Conservation* | 1994–2021 | All biodiversity |
| *Biological Conservation* | 1981–2021 | All biodiversity |
| *Biological Control* | 1991–2017 | All biodiversity |
| *Biological Invasions* | 1999–2017 | All biodiversity |
| *Biology and Environment: Proceedings of the Royal Irish Academy* | 1993–2017 | All biodiversity |
| *Biology Letters* | 2005–2018 | All biodiversity |
| *Biotropica* | 1990–2019 | All biodiversity |
| *Bird Conservation International* | 1991–2016 | All biodiversity |
| *Bird Study* | 1980–2016 | All biodiversity |
| *Boreal Environment Research* | 1996–2014 | All biodiversity |
| *Bulletin Français de la Pêche et de la Pisciculture* | 1986–2007 | All biodiversity |
| *Bulletin of Marine Science* | 2000–2021 | All biodiversity |
| *Bulletin of the Chicago Herpetological Society* | 1990–2018 | All biodiversity |
| *Bulletin of the Maryland Herpetological Society* | 1980–2015 | All biodiversity |
| *Canadian Journal of Fisheries and Aquatic Sciences* | 1901–2023 | All biodiversity |
| *Canadian Journal of Forest Research* | 1971–2018 | All biodiversity |
| *Caribbean Herpetology* | 2010–2018 | All biodiversity |
| *Caribbean Journal of Science* | 1961–2013 | All biodiversity |
| *CCAMLR Science* | 1985–2016 | All biodiversity |
| *CEE (Collaboration for Environmental Evidence) Systematic Reviews* | 2004–2016 | All biodiversity |
| *Chelonian Conservation and Biology* | 1993–2018 | All biodiversity |
| *Chelonian Research Monographs* | 1996–2017 | All biodiversity |
| *Coastal Engineering* | 2000–2018 | All biodiversity |

| **Journal** | **Years searched** | **Topic** |
|---|---|---|
| *Collinsorum* (*formerly Journal of Kansas Herpetology*) | 2012–2018 | All biodiversity |
| *Colonial Waterbirds* | 1983–1998 | All biodiversity |
| *Community Ecology* | 2000–2012 | All biodiversity |
| *Conservation Biology* | 1987–2021 | All biodiversity |
| *Conservation Evidence* | 2004–2021 | All biodiversity |
| *Conservation Genetics* | 2000–2013 | All biodiversity |
| *Conservation Letters* | 2008–2021 | All biodiversity |
| *Contemporary Herpetology* | 1998–2009 | All biodiversity |
| *Contributions to Primatology* | 1974–1991 (final published volume) | All biodiversity |
| *Copeia* | 1910–2018 | All biodiversity |
| *Copeia* | 2004–2016 | Reptile Conservation |
| *Coral Reefs* | 2000–2021 | All biodiversity |
| *Cunninghamia* | 1981–2016 | All biodiversity |
| *Current Herpetology* (*formerly Acta Herpetologica Japonica 1964–1971 and Japanese Journal of Herpetology 1972–1999*) | 1964–2018 | All biodiversity |
| *Dodo* | 1977–2001 | All biodiversity |
| *Ecological and Environmental Anthropology* | 2005–2008 | All biodiversity |
| *Ecological Applications* | 1991–2021 | All biodiversity |
| *Ecological Entomology* | 1985–2018 | All biodiversity |
| *Ecological Indicators* | 2001–2007 | All biodiversity |
| *Ecological Management & Restoration* | 2000–2019 | All biodiversity |
| *Ecological Restoration* | 1981–2021 | All biodiversity |
| *Ecological Solutions and Evidence* | 2020–2021 | All biodiversity |
| *Ecology* | 1936–2021 | All biodiversity |
| *Ecology Letters* | 1998–2019 | All biodiversity |
| *Ecology of Freshwater Fish* | 1992–2023 | All biodiversity |
| *Ecosystems* | 1998–2013 | All biodiversity |
| *Emu* | 1980–2016 | All biodiversity |
| *Endangered Species Bulletin* | 1966–2003 | All biodiversity |
| *Endangered Species Research* | 2004–2019 | All biodiversity |
| *Entomologia Experimentalis et Applicata* | 2015–2018 | All biodiversity |
| *Environmental Conservation* | 1974–2021 | All biodiversity |
| *Environmental Entomology* | 1990–2018 | All biodiversity |
| *Environmental Evidence* | 2012–2021 | All biodiversity |

| **Journal** | **Years searched** | **Topic** |
|---|---|---|
| *Environmental Management* | 1977–2021 | All biodiversity |
| *Environmentalist* | 1981–1988 | All biodiversity |
| *Estuaries and Coasts* | 2013–2022 | All biodiversity |
| *Ethology Ecology & Evolution* | 1989–2014 | All biodiversity |
| *European Journal of Soil Science* | 1950–2012 | Soil Fertility |
| *European Journal of Wildlife Research* (*formerly Zeitschrift für Jagdwissenschaft 1955–2003*) | 2004–2021 | All biodiversity |
| *Evolutionary Anthropology* | 1992–2014 | All biodiversity |
| *Evolutionary Ecology* | 1987–2014 | All biodiversity |
| *Evolutionary Ecology Research* | 1999–2014 | All biodiversity |
| *Fire Ecology* | 2005–2016 | All biodiversity |
| *Fish and Fisheries* | 2000–2023 | All biodiversity |
| *Fisheries* | 2017–2018 | All biodiversity |
| *Fisheries Management and Ecology* | 1990–2023 | All biodiversity |
| *Fisheries Oceanography* | 1992–2018 | All biodiversity |
| *Fisheries Research* | 1990–2018 | All biodiversity |
| *Flora* | 1991–2017 | All biodiversity |
| *Folia Primatologica* | 1963–2014 | All biodiversity |
| *Folia Zoologica* | 1959–2013 | All biodiversity |
| *Forest Ecology and Management* | 1976–2019 | All biodiversity |
| *Freshwater Biology* | 1975–2022 | All biodiversity |
| *Freshwater Science* (*formerly Freshwater Invertebrate Biology; then Journal of the North American Benthological Society*) | 1982–2023 | All biodiversity |
| *Frontiers in Marine Science* | 2017–2018 | All biodiversity |
| *Frontiers in Psychology* | 2019 | All biodiversity |
| *Functional Ecology* | 1987–2013 | All biodiversity |
| *Genetics and Molecular Research* | 2002–2013 | All biodiversity |
| *Geoderma* | 1967–2012 | Soil Fertility |
| *Gibbon Journal* | 2005–2011 | All biodiversity |
| *Global Change Biology* | 1995–2017 | All biodiversity |
| *Global Ecology and Biogeography* | 1991–2014 | All biodiversity |
| *Global Ecology and Conservation* | 2014–2018 | All biodiversity |
| *Grass and Forage Science* | 1980–2017 | All biodiversity |
| *Herpetofauna* | 2003–2007 | All biodiversity |
| *Herpetologica* | 1936–2018 | All biodiversity |
| *Herpetologica* | 2013–2016 | Reptile Conservation |
| *Herpetological Conservation and Biology* | 2006–2012 | Amphibian Conservation |

| **Journal** | **Years searched** | **Topic** |
|---|---|---|
| *Herpetological Conservation and Biology* | 2006–2012 | Reptile Conservation |
| *Herpetological Conservation and Biology* | 2006–2018 | All biodiversity |
| *Herpetological Monographs* | 1982–2018 | All biodiversity |
| *Herpetological Review* | 1967–2018 | All biodiversity |
| *Herpetology Notes* | 2008–2018 | All biodiversity |
| *Herpetozoa* | 1988–2018 | All biodiversity |
| *Human Wildlife Interactions* | 2007–2021 | All biodiversity |
| *Hydrobiologia* | 1995–2018 | All biodiversity |
| *Hystrix, the Italian Journal of Mammalogy* (*English, 1994-*) | 1994–2019 | All biodiversity |
| *Ibis* | 1980–2016 | All biodiversity |
| *ICES Journal of Marine Science* | 1990–2018 | All biodiversity |
| *iForest* | 2008–2016 | All biodiversity |
| *Inland Waters* | 2011–2022 | All biodiversity |
| *Insect Conservation and Diversity* | 2008–2018 | All biodiversity |
| *Integrative Zoology* | 2006–2013 | All biodiversity |
| *International Journal of Pest Management* (*formerly PANS Pest Articles & News Summaries 1969 - 1975, PANS 1976–1979 & Tropical Pest Management 1980–1992*) | 1969–1979 | All biodiversity |
| *International Journal of Primatology* | 1980–2019 | All biodiversity |
| *International Journal of the Commons* | 2007–2016 | All biodiversity |
| *International Journal of Wildland Fire* | 1991–2016 | All biodiversity |
| *International Wader Studies* | 1986–2006 | All biodiversity |
| *International Zoo Yearbook* | 1960–2019 | All biodiversity |
| *Invasive Plant Science and Management* | 2008–2016 | All biodiversity |
| *Israel Journal of Ecology & Evolution* | 1963–2013 | All biodiversity |
| *Italian Journal of Zoology* (*subsequently European Zoological Journal*) | 1978–2013 | All biodiversity |
| *Journal for Nature Conservation* | 2002–2021 | All biodiversity |
| *Journal of Animal Ecology* | 1932–2021 | All biodiversity |
| *Journal of Apicultural Research* | 1962–2009 | All biodiversity |
| *Journal of Applied Animal Nutrition* | 2012–2019 | All biodiversity |
| *Journal of Applied Animal Welfare Science* | 1998–2019 | All biodiversity |
| *Journal of Applied Ecology* | 1964–2021 | All biodiversity |
| *Journal of Aquatic Plant Management* (*formerly Hyacinth Control Journal 1962–1975*) | 1962–2022 | All biodiversity |
| *Journal of Arid Environments* | 1993–2017 | All biodiversity |

| **Journal** | **Years searched** | **Topic** |
|---|---|---|
| *Journal of Avian Biology* (*formerly Ornis Scandinavica 1970–1993*) | 1994–2016 | All biodiversity |
| *Journal of Cetacean Research and Management* | 1999–2018 | All biodiversity |
| *Journal of Coastal Research* | 2015–2018 | All biodiversity |
| *Journal of Ecology* | 1933–2021 | All biodiversity |
| *Journal of Ecology & Natural Resources* | 2017–2019 | All biodiversity |
| *Journal of Environmental Management* | 1973–300) | All biodiversity |
| *Journal of Experimental Marine Biology and Ecology* | 2000–2018 | All biodiversity |
| *Journal of Field Ornithology* | 1980–2016 | All biodiversity |
| *Journal of Forest Research* | 1996–2019 | All biodiversity |
| *Journal of Great Lakes Research* | 1975–2017 | All biodiversity |
| *Journal of Herpetological Medicine and Surgery* | 2009–2018 | All biodiversity |
| *Journal of Herpetology* | 1968–2018 | All biodiversity |
| *Journal of Insect Conservation* | 1997–2018 | All biodiversity |
| *Journal of Insect Science* | 2003–2018 | All biodiversity |
| *Journal of Kansas Herpetology* | 2002–2011 | All biodiversity |
| *Journal of Mammalian Evolution* | 1993–2014 | All biodiversity |
| *Journal of Mammalogy* | 1919–2019 | All biodiversity |
| *Journal of Mountain Science* | 2004–2016 | All biodiversity |
| *Journal of Negative Results: Ecology & Evolutionary Biology* | 2004–2016 | All biodiversity |
| *Journal of North American Herpetology* | 2014–2017 | All biodiversity |
| *Journal of Ornithology* (*formerly Journal für Ornithologie to 2004*) | 2004–2018 | All biodiversity |
| *Journal of Primatology* | 2012–2013 | All biodiversity |
| *Journal of Range Management* | 1948–2004 | All biodiversity |
| *Journal of Raptor Research* | 1966–2016 | All biodiversity |
| *Journal of Sea Research* (*formerly Netherlands Journal of Sea Research*) | 1961–2018 | All biodiversity |
| *Journal of the Marine Biological Association of the United Kingdom* | 1887–2018 | All biodiversity |
| *Journal of Tropical Ecology* | 1986–2021 | All biodiversity |
| *Journal of Vegetation Science* | 1990–2022 | All biodiversity |
| *Journal of Wetlands Ecology* | 2008–2012 | All biodiversity |
| *Journal of Wetlands Environmental Management* | 2012–2016 | All biodiversity |
| *Journal of Wildlife Diseases* | 1965–2012 | All biodiversity |
| *Journal of Wildlife Management* | 1945–2021 | All biodiversity |
| *Journal of Zoo and Aquarium Research* | 2013–2019 | All biodiversity |
| *Journal of Zoo and Wildlife Medicine* | 1970–2019 | All biodiversity |

| Journal | Years searched | Topic |
|---|---|---|
| *Journal of Zoology* | 1966–2021 | All biodiversity |
| *Kansas Herpetological Society Newsletter* | 1974–2001 | All biodiversity |
| *Knowledge and Management of Aquatic Ecosystems* | 2008–2023 | All biodiversity |
| *Lake and Reservoir Management* | 1984–2022 | All biodiversity |
| *Lakes & Reservoirs* | 2016–2022 | All biodiversity |
| *Land Degradation and Development* | 1989–2016 | All biodiversity |
| *Land Use Policy* | 1984–2012 | Soil Fertility |
| *Latin American Journal of Aquatic Mammals* | 2002–2018 | All biodiversity |
| *Lemur News* | 1993–2012 | All biodiversity |
| *Limnologica* | 1999–2022 | All biodiversity |
| *Mammal Research* (*formerly Acta Theriologica*) | 2001–2019 | All biodiversity |
| *Mammal Review* | 1970–2019 | All biodiversity |
| *Mammal Study* | 2005–2019 | All biodiversity |
| *Mammalia* | 1937–2019 | All biodiversity |
| *Mammalian Biology* | 2002–2018 | All biodiversity |
| *Mammalian Genome* | 1991–2013 | All biodiversity |
| *Management of Biological Invasions* | 2010–2016 | All biodiversity |
| *Mangroves and Salt Marshes* | 1996–1999 | All biodiversity |
| *Marine and Freshwater Research* | 1980–2018 | All biodiversity |
| *Marine Ecology* | 1980–2018 | All biodiversity |
| *Marine Ecology Progress Series* | 2000–2018 | All biodiversity |
| *Marine Environmental Research* | 1978–2018 | All biodiversity |
| *Marine Mammal Science* | 1985–2019 | All biodiversity |
| *Marine Pollution Bulletin* | 2010–2018 | All biodiversity |
| *Marine Turtle Newsletter* | 1976–2018 | All biodiversity |
| *Marsh Bulletin* | 2010–2020 | All biodiversity |
| *Mesoamerican Herpetology* | 2014–2017 | All biodiversity |
| *Mires and Peat* | 2006–2016 | All biodiversity |
| *Natural Areas Journal* | 1992–2017 | All biodiversity |
| *Nature Conservation* | 2012–2019 | All biodiversity |
| *NeoBiota* | 2011–2017 | All biodiversity |
| *Neotropical Entomology* | 2004–2018 | All biodiversity |
| *Neotropical Primates* | 1993–2012 | All biodiversity |
| *New Journal of Botany* | 2011–2013 | All biodiversity |
| *New Zealand Journal of Marine and Freshwater Research* | 1967–2022 | All biodiversity |
| *New Zealand Journal of Zoology* | 1974–2021 | All biodiversity |
| *New Zealand Plant Protection* | 2000–2022 | All biodiversity |

| **Journal** | **Years searched** | **Topic** |
|---|---|---|
| *North American Journal of Fisheries Management* | 1994–2023 | All biodiversity |
| *Northwest Science* | 2007–2016 | All biodiversity |
| *Oecologia* | 1969–2021 | All biodiversity |
| *Oikos* | 1949–2021 | All biodiversity |
| *Ornis Scandinavica* | 1980–1993 | All biodiversity |
| *Ornitología Neotropical* | 1990–2018 | All biodiversity |
| *Oryx* | 1950–2021 | All biodiversity |
| *Ostrich* | 1980–2016 | All biodiversity |
| *Pacific Conservation Biology* | 1993–2021 | All biodiversity |
| *Pakistan Journal of Zoology* | 2004–2013 | All biodiversity |
| *Phyllomedusa* | 2002–2018 | All biodiversity |
| *Plant Ecology (formerly Vegetatio 1948–1996)* | 1948–2007 | All biodiversity |
| *Plant Protection Quarterly* | 2008–2016 | All biodiversity |
| *Polish Journal of Ecology* | 2002–2013 | All biodiversity |
| *Population Ecology* | 1952–2013 | All biodiversity |
| *Preslia* | 1973–2017 | All biodiversity |
| *Primate Conservation* | 1981–2014 | All biodiversity |
| *Primates* | 1957–2013 | All biodiversity |
| *Rangeland Ecology & Management (previously Journal of Range Management 1948–2004)* | 2005–2016 | All biodiversity |
| *Raptors Conservation* | 2005–2016 | All biodiversity |
| *Regional Studies in Marine Science* | 2015–2018 | All biodiversity |
| *Reptile Rap - Newsletter of the South Asian Reptile Network (SARN)* | 1999–2016 | All biodiversity |
| *Restoration Ecology* | 1993–2021 | All biodiversity |
| *Riparian Ecology and Conservation* | 2013–2017 | All biodiversity |
| *River Research and Applications* | 1987–2022 | All biodiversity |
| *Russian Journal of Ecology (Springer - translated version)* | 1993–2013 | All biodiversity |
| *Russian Journal of Herpetology* | 1994–2018 | All biodiversity |
| *Russian Journal of Theriology* | 2013–2017 | All biodiversity |
| *Salamandra (English 2005+)* | 2005–2018 | All biodiversity |
| *Slovak Raptor Journal* | 2007–2016 | All biodiversity |
| *Small Ruminant Research* | 1988–2017 | All biodiversity |
| *Soil Biology & Biochemistry* | 1969–2012 | Soil Fertility |
| *South African Journal of Botany* | 1982–2018 | All biodiversity |
| *South African Journal of Wildlife Research* | 1971–2014 | All biodiversity |
| *South American Journal of Herpetology* | 2006–2018 | All biodiversity |
| *Southern Forests* | 2008–2018 | All biodiversity |

| Journal | Years searched | Topic |
|---|---|---|
| *Testudo* | 1978–2017 | All biodiversity |
| *The Canadian Field-Naturalist* (*formerly Ottawa Naturalist*) | 1887–2019 | All biodiversity |
| *The Condor* | 1980–2009 | All biodiversity |
| *The Herpetological Bulletin* | 1980–2003 | Amphibian Conservation |
| *The Herpetological Bulletin* | 2003–2013 | Reptile Conservation |
| *The Herpetological Bulletin* | 2008–2018 | All biodiversity |
| *The Herpetological Journal* | 1985–2016 | All biodiversity |
| *The Open Ornithology Journal* | 2008–2016 | All biodiversity |
| *The Rangeland Journal* | 1976–2016 | All biodiversity |
| *The Southwestern Naturalist* | 1956–2018 | All biodiversity |
| *The Wilson Bulletin* | 1980–2005 | All biodiversity |
| *The Wilson Journal of Ornithology* (*formerly The Wilson Bulletin*) | 2006–2016 | All biodiversity |
| *Trends in Ecology and Evolution* | 1986–2021 | All biodiversity |
| *Tropical Conservation Science* | 2008–2018 | All biodiversity |
| *Tropical Ecology* | 1960–2018 | All biodiversity |
| *Tropical Grasslands* | 1967–2010 | All biodiversity |
| *Tropical Zoology* | 1988–2018 | All biodiversity |
| *Turkish Journal of Zoology* | 1996–2014 | All biodiversity |
| *Ursus* | 1968–2019 | All biodiversity |
| *Vietnamese Journal of Primatology* | 2007–2009 | All biodiversity |
| *Wader Study Group Bulletin* | 1970–1972 | All biodiversity |
| *Waterbirds* (*formerly Colonial Waterbirds*) | 1999–2016 | All biodiversity |
| *Weed Biology and Management* | 2001–2016 | All biodiversity |
| *Weed Research* | 1961–2017 | All biodiversity |
| *West African Journal of Applied Ecology* | 2000–2016 | All biodiversity |
| *Western North American Naturalist* | 2000–2017 | All biodiversity |
| *Wetlands* | 1981–2016 | All biodiversity |
| *Wetlands Ecology and Management* | 1989–2022 | All biodiversity |
| *Wildfowl* | 1948–2018 | All biodiversity |
| *Wildlife Biology* | 1995–2013 | All biodiversity |
| *Wildlife Monographs* | 1958–2013 | All biodiversity |
| *Wildlife Research* | 1956–2012 | Bat Conservation |
| *Wildlife Research* | 1974–2019 | All biodiversity |
| *Wildlife Society Bulletin* | 1973–2019 | All biodiversity |
| *Zhurnal Obshchei Biologii* | 1972–2013 | All biodiversity |

| **Journal** | **Years searched** | **Topic** |
|---|---|---|
| *Zoo Biology* | 1982–2019 | All biodiversity |
| *ZooKeys* | 2008–2013 | All biodiversity |
| *Zoologica Scripta* | 1971–2014 | All biodiversity |
| *Zoological Journal of the Linnean Society* | 1856–2013 | All biodiversity |
| *Zootaxa* | 2004–2014 | All biodiversity |

# Appendix 2: Non-English language journals (and years) searched

Non-English language journals (and years) searched and for which relevant papers have been added to the Conservation Evidence discipline-wide literature database.

| **Journal** | **Years searched** | **Language** |
|---|---|---|
| *Journal of King Abdulaziz University: Marine Sciences*<br>مجلة جامعة الملك عبدالعزيز: علوم البحار | 2000–2018 | Arabic |
| *Afak Ilmia Journal*<br>مجلة آفاق علمية | 2017–2020 | Arabic |
| *The Arab Journal for Arid Environments*<br>المجلة العربية للبيئات الجافة | 2009–2018 | Arabic |
| *Baghdad Science Journal*<br>مجلة بغداد للعلوم | 2004–2020 | Arabic |
| *Tishreen University Journal for Research and Scientific Studies: Biological Sciences Series*<br>مجلة جامعة تشرين للبحوث والدراسات العلمية _ سلسلة العلوم البيولوجية | 2001–2020 | Arabic |
| *Journal of Plant Protection*<br>مجلة وقاية النبات العربية | 1993–2019 | Arabic |
| *Journal of King Abdulaziz University: Economics and Administration*<br>مجلة جامعة الملك عبدالعزيز: الاقتصاد والإدارة | 2015–2020 | Arabic |
| *Journal of Agricultural, Environmental and Veterinary Sciences*<br>مجلة العلوم الزراعية والبيئية والبيطرية | 2018–2020 | Arabic |
| *Journal of Thi-Qar Science*<br>مجلة علوم ذي قار | 2014–2018 | Arabic |

| Journal | Years searched | Language |
|---|---|---|
| *Journal of Marine Sciences and Environmental Techniques*<br>مجلة علوم البحار والتقنيات البيئية | 2016–2019 | Arabic |
| *Journal of King Abdulaziz University: Environmental Design Science*<br>مجلة جامعة الملك عبد العزيز: علوم تصاميم البيئة | 2003–2017 | Arabic |
| *Naturae* | 2017–2020 | French |
| *Écoscience*<br>*Ecoscience* | 1994–2019 | French |
| *Biotechnologie, Agronomie, Société et Environnement*<br>*Biotechnology, Agronomy, Society and Environment* | 2008–2020 | French |
| *Bois et Forêts des Tropiques*<br>*Tropical Woodlands and Forests* | 2009–2020 | French |
| *Courrier Scientifique du Parc Naturel Régional du Luberon et de la Réserve de Biosphère Luberon-Lure*<br>*Scientific Letters from the Regional Natural Park of Luberon et and the Biosphere Reserve Luberon-Lure* | 1997–2016 | French |
| *VertigO* | 2009–2019 | French |
| *Ecologia Mediterranea*<br>*Ecologia Mediterranea: International Journal of Mediterranean Ecology* | 2000–2019 | French |
| *Travaux Scientifiques du Parc National de Port-Cros*<br>*Scientific Reports of the Port-Cros National Park* | 2000–2019 | French |
| *Travaux Scientifiques du Parc National de la Vanoise*<br>*Scientific Reports of the Vanoise National Park* | 1986–2009 | French |
| *Alauda* | 2000–2005 | French |
| *Revue d'Écologie* (*La Terre et La Vie*)<br>*Earth and Life* | 2006–2018 | French |
| *Bulletin de la Société Zoologique de France*<br>*Bulletin of the French Zoology Society* | 1973–2015 | French |
| *Le Naturaliste Canadien*<br>*The Canadian Naturalist* | 2008–2018 | French |
| *Salamandra* (*German 1965–2004*) | 1965–2004 | German |
| *Hercynia* | 1963–2017 | German |
| *Ornithologischer Anzeiger*<br>*Ornithological Journal* | 1951–2017 | German |
| *Pulsatilla: Zeitschrift für Botanik und Naturschutz*<br>*Pulsatilla: Journal of Botany and Nature Conservation* | 2000–2007 | German |
| *Archiv für Forstwesen und Landschaftsökologie*<br>*Archive for Forestry and Landscape Ecology* | 2013 | German |

| **Journal** | **Years searched** | **Language** |
|---|---|---|
| *Naturschutz und Landschaftsplanung*<br>*Conservation and Landscape Planning* | 2003–2017 | German |
| *Allgemeine Forst- und Jagdzeitung*<br>*Journal for Forestry and Forest Science* | 2000–2016 | German |
| *Nyctalus: Internationale Fledermaus-Fachzeitschrift*<br>*Nyctalus: International Bat Journal* | 2005–2017 | German |
| *Die Orchidee*<br>*The Orchid* | 1949–2016 | German |
| *Botanik und Naturschutz in Hessen*<br>*Botany and Nature Conservation in Hessen* | 1987–2018 | German |
| *ANLiegen Natur: Zeitschrift für Naturschutz und Angewandte Landschaftsökologie*<br>*Concerning Nature: Journal for Nature Conservation and Applied Landscape Ecology* | 2006–2017 | German |
| *Die Vogelwelt: Beiträge zur Vogelkunde*<br>*Bird Life: Contributions to Ornithology* | 2005–2017 | German |
| *Waldökologie Online (until 2008)*<br>*Forest Ecology Online* | 2004–2008(6) | German |
| *Der Ornithologische Beobachter*<br>*Ornithological Observer* | 1950–2017 | German |
| *Telma* | 1971–2019 | German |
| *Gesunde Pflanzen: Pflanzenschutz, Verbraucherschutz, Umweltschutz*<br>*Healthy Plants: Crop Protection, Consumer Protection, Environment Protection* | 2002–2017 | German |
| *Die Bodenkultur: Journal of Land Management, Food and Environment*<br>*Soil Culture: Journal for Land Management, Food and Environment* | 2016–2017 | German |
| *Journal für Ornithologie (German: up to 2004)*<br>*Journal of Ornithology (German: up to 2004)* | 1959–2003 | German |
| *Die Erde*<br>*The Earth* | 1952–2004 | German |
| *Freiberg Online Geoscience - FOG* | 1998–2017 | German |
| *RANA - Mitteilungen für Feldherpetologie und Ichthyofaunistik*<br>*RANA - Communications for Field Herpetology and Ichthyofauna* | Vol1(1983)–Vol17(2016) excluding special issues | German |
| *Vogelwarte: Zeitschrift für Vogelkunde*<br>*Bird Observatory: Ornithology Journal* | 2005–2017 | German |

| Journal | Years searched | Language |
|---|---|---|
| *Mitteilungen des Badischen Landesvereins für Naturkunde und Naturschutz*<br>*Communications of the Baden Association for Natural History and Nature Conservation* | 1953–2015 | German |
| *Auenmagazin* (*Magazin des Auenzentrums Neuburg a. d. Donau*)<br>*Floodplains Journal* (*Magazine of the Auenzentrums Neuburg a. d. Danube*) | 2010–2017 | German |
| *Biodiversität und Naturschutz in Ostösterreich*<br>*Biodiversity and Conservation in Eastern Austria* | 2015–2018 | German |
| *Natur und Landschaft: Zeitschrift fur Naturschutz und Landschaftspflege*<br>*Nature and Landscape: Journal for Nature Conservation and Landscape Management* | 1990–2017 | German |
| *Mertensiella* | 1988–2017 | German |
| *Zeitschrift für Jagdwissenschaft*<br>*Journal of Hunting Science* [*Became European Journal of Wildlife Research* (*Springer*) *in 2004*] | 1955–2003 | German |
| *Bulletin de la Société des Naturalistes Luxembourgeois*<br>*Bulletin of the Luxemburgian Naturalist Society* | 1950–2017 | German |
| *Arachnologische Mitteilungen*<br>*Arachnological Letters* | 1991–2017 | German |
| *Silva Fera: Wissenschaftliche Nachrichten aus dem Wildnisgebiet Dürrenstein*<br>*Silva Fera: Scientific News from the Dürrenstein Wilderness Area* | 2012–2017 | German |
| *Forstarchiv*<br>*Forestry Archive* | 2007–2017 | German |
| *Zeitschrift für Feldherpetologie*<br>*Journal for Field Herpetology* | 1994–2017 | German |
| *Libellula* | 1982–2016 | German |
| *Inatura Forschung Online*<br>*Inatura Research Online* | 1996–2007 | German |
| *ABU-Info* (*Arbeitsgemeinschaft Biologischer Umweltschutz im Kreis Soest e.V.*)<br>*ABU-Info* (*Working Group for Biological Environmental Protection in Soest District*) | 2006–2017 | German |
| *Fachzeitschrift für Waldökologie, Landschaftsforschung und Naturschutz* (*formerly Waldökologie Online*)<br>*Journal for Forest Ecology, Landscape Research and Nature Conservation* | 2008–2016 | German |

| **Journal** | **Years searched** | **Language** |
|---|---|---|
| *Insecta* | 1992–2014 | German |
| *Der Zoologische Garten: Zeitschrift für die Gesamte Tiergärtnerei (Neue Folge)*<br>*The Zoological Garden: Journal for the Entire Zoo* | 2007–2017 | German |
| *Tuexenia* | 1981–2016 | German |
| *Állattani Közlemények*<br>*Journal of Zoology* | 2010–2019 | Hungarian |
| *Botanikai Közlemények*<br>*Journal of Botany* | 2010–2020 | Hungarian |
| *Természetvédelmi Közlemények*<br>*Journal of Nature Conservation* | 2010–2019 | Hungarian |
| *Tájökológiai Lapok*<br>*Journal of Landscape Ecology* | 2010–2019 | Hungarian |
| *Jurnal Primatologi Indonesia* | 2009 | Indonesian |
| *Forest@ - Rivista di Selvicoltura ed Ecologia Forestale*<br>*Forest @ - Journal of Silviculture and Forest Ecology* | 2004–2020 | Italian |
| *Alula*<br>*Alula* | 1992–2019 | Italian |
| *Biologia Ambientale*<br>*Environmental Biology* | 1994–2018 | Italian |
| *Hystrix, the Italian Journal of Mammalogy (Italian 1986–1993)* | 1986–1993 | Italian |
| *Avocetta* | 2000–2013 | Italian |
| *Rivista Italiana di Ornitologia*<br>*Research in Ornithology* | 2010–2019 | Italian |
| *Picus* | 2004–2018 | Italian |
| *Wildlife Conservation Japan*<br>野生生物保護 | 1995–2013 | Japanese |
| *Journal of the Mammalogical Society of Japan*<br>哺乳動物学雑誌 | 1959–1986 | Japanese |
| *Bulletin of the Herpetological Society of Japan*<br>爬虫両棲類学会報 | 1999–2008 | Japanese |
| *Journal of the Japanese Institute of Landscape Architecture (1994+)*<br>ランドスケープ研究 | 1994–2017 | Japanese |
| *Bulletin of the International Association for Landscape Ecology-Japan*<br>国際景観生態学会日本支部会報 | 2002–2003 | Japanese |
| *Strix*<br>ストリクス | 1982–2017 | Japanese |

| Journal | Years searched | Language |
|---|---|---|
| *Japanese Journal of Ecology*<br>日本生態学会誌 | 1954–2017 | Japanese |
| *Journal of the Japanese Forest Society* (*2005+*)<br>日本森林学会誌 | 2005–2017 | Japanese |
| *Mammalian Science*<br>哺乳類科学 | 1961–2016 | Japanese |
| *The Journal of the Japanese Landscape Architectural Society*<br>造園学雑誌 | 1925–1927 | Japanese |
| *Journal of the Japanese Institute of Landscape Architects* (*1934–1994*)<br>造園雑誌 | 1934–1994 | Japanese |
| *Landscape Ecology and Management*<br>景観生態学 | 2005–2016 | Japanese |
| *Japanese Journal of Ornithology*<br>日本鳥学会誌 | 1917–2015 | Japanese |
| *Journal of the Japanese Forestry Society* (*1919–2004*)<br>日本林学会誌 | 1985–2004 | Japanese |
| *Japanese Journal of Conservation Ecology*<br>保全生態学研究 | 1996–2016 | Japanese |
| *Doubutsugaku zasshi*<br>動物学雑誌 | 1888–1983 | Japanese |
| *Landscape Research Japan Online*<br>ランドスケープ研究(オンライン論文集) | 2008–2017 | Japanese |
| *Wildlife and Human Society*<br>野生生物と社会 | 2013–2017 | Japanese |
| *Ecology and Civil Engineering*<br>応用生態工学 | 1998–2017 | Japanese |
| *Reintroduction*<br>野生復帰 | 2011–2019 | Japanese |
| *Korean Journal of Ornithology*<br>한국조류학회지 | 1994–2020 | Korean |
| *Journal of Korean Society of Forest Science*<br>한국산림과학회지(한국임학회지) | 2002–2020 | Korean |
| *Korean Journal of Environmental Biology*<br>환경생물 | 2002–2020 | Korean |
| *Korean Journal of Environment and Ecology*<br>한국환경생태학회지 | 2001–2020 | Korean |
| *Journal of Wetlands Research*<br>한국습지학회지 | 1999–2020 | Korean |

| **Journal** | **Years searched** | **Language** |
|---|---|---|
| *Journal of Environmental Sciences*<br>علوم محیعلوم محیطیطی | 2004–2017 | Persian |
| *Journal of Animal Environment*<br>فصلنامه محیط زیست جانوری | 2014–2017 | Persian |
| *Iranian Journal of Applied Ecology*<br>بوم شناسی کاربردی | 2012–2017 | Persian |
| *Iranian Journal of Natural Resources*<br>مجله منابع طبیعی ایران | 2002–2009 | Persian |
| *Journal of Environmental Studies*<br>محیط شناسی | 2009–2017 | Persian |
| *Environmental Researches*<br>پژوهش های محیط زیست | 2010–2017 | Persian |
| *Journal of Natural Environment*<br>نشریه محیط زیست طبیعی | 2010–2017 | Persian |
| *Experimental Animal Biology*<br>زیست شناسی جانوری تجربی | 2012–2017 | Persian |
| *Journal of Animal Researches*<br>پژوهش های جانوری | 2013–2017 | Persian |
| *Kulon*<br>*Stone Curlew* | 1996–2018 | Polish |
| *Parki Narodowe i Rezerwaty Przyrody*<br>*National Parks and Nature Reserves* | 2009–2015 | Polish |
| *Chrońmy Przyrodę Ojczystą*<br>*Let's Protect Our Indigenous Nature* | 2004–2019 | Polish |
| *Ornis Polonica* | 2010–2020 | Polish |
| *Nature Conservation* (*English language Vol58 2001+; formerly in Polish as Ochrona Przyrody 1920–2000*) | 2001–2008 | Polish |
| *Studia Naturae*<br>*Studia Naturae / Nature Studies* | 1987–2013 | Polish |
| *Notatki Ornitologiczne*<br>*Ornithological Notes* | 1989–2009 | Polish |
| *Przegląd Przyrodniczy*<br>*Nature Review* | 2010–2019 | Polish |
| *Naturalia* | 2012–2016 | Polish |
| *Nietoperze*<br>*Bats* | 2000–2011 | Polish |
| *Iheringia: Série Zoologia*<br>*Iheringia: Zoology Series* | 2000–2018 | Portuguese |
| *Biodiversidade* (*UFMT*)<br>*Biodiversity* | 2007–2019 | Portuguese |

| Journal | Years searched | Language |
|---|---|---|
| *Bioikos* | 1987–2016 | Portuguese |
| *Revista CEPSUL - Biodiversidade e Conservação Marinha*<br>*CEPSUL Magazine - Marine Biodiversity and Conservation* | 2010–2017 | Portuguese |
| *Boletim do Museu de Biologia Mello Leitão*<br>*Bulletin of the Mello Leitão Biology Museum* | 2013–2018 | Portuguese |
| *Neotropical Biology and Conservation* | 2006–2017 | Portuguese |
| *Natureza & Conservação*<br>*Brazilian Journal of Nature Conservation* | 2003–2009 | Portuguese |
| *Boletim da Sociedade Brasileira de Mastozoologia*<br>*Bulletin of the Brazilian Society of Mastozoology (mammalogy)* | 1985–2017 | Portuguese |
| *Revista de Gestão Costeira Integrada*<br>*Journal of Integrated Coastal Zone Management* | 2007–2019 | Portuguese |
| *Arquipelago - Life and Marine Sciences* | 1980–2020 | Portuguese |
| *Acta Amazônica*<br>*Amazon Record/Journal* | 1971–2019 | Portuguese |
| *Ambiência* | 2005–2019 | Portuguese |
| *Floresta* | 1969–2017 | Portuguese |
| *Revista Brasileira de Ecologia*<br>*Brazilian Journal of Ecology* | 1997–2009 | Portuguese |
| *Revista Nordestina de Biologia*<br>*Northeastern Journal of Biology* | 1978–2016 | Portuguese |
| *Biota Neotropica*<br>*Neotropical Biodiversity* | 2001–2011 | Portuguese |
| *Chiroptera Neotropical*<br>*Neotropical Chiroptera* | 1995–2015 | Portuguese |
| *Biodiversidade Brasileira*<br>*Brazilian Biodiversity* | 2011–2016 | Portuguese |
| *Portugaliae Acta Biologica*<br>*Portugal - Biological Journal* | 2000–2003 | Portuguese |
| *Evolução e Conservação da Biodiversidade*<br>*Evolution and Conservation of Biodiversity* | 2010–2011 | Portuguese |
| *FLORAM - Revista Floresta e Ambiente*<br>*Brazilian Journal of Forestry and Environment* | 1994–2020 | Portuguese |
| *Biotemas* | 1988–2018 | Portuguese |
| *Megadiversidade*<br>*Megadiversity* | 2005–2009 | Portuguese |

| **Journal** | **Years searched** | **Language** |
|---|---|---|
| *Ciência & Ambiente*<br>*Science and Environment* | 1990–2015 | Portuguese |
| *Revista Brasileira de Gestão Ambiental e Sustentabilidade*<br>*The Brazilian Journal of Environmental Management and Sustainability* | 2014–2017 | Portuguese |
| *Revista de Biologia Neotropical*<br>*Journal of Neotropical Biology* | 2004–2018 | Portuguese |
| *MG Biota* | 2008–2016 | Portuguese |
| *Biota Amazônica*<br>*Amazonian Biota* | 2011–2018 | Portuguese |
| *Revista de Ciências Agrárias* (*SCAP*)<br>*Journal of Agricultural Sciences* (*SCAP*) | 2007–2019 | Portuguese |
| *Zoologicheskiĭ Zhurnal* (*Russian Journal of Zoology*)<br>Зоологический журнал | 1939–2020(8) | Russian |
| *Ekologiya* (*Russian Journal of Ecology*)<br>Экология | 2000–2020(4) | Russian |
| *Current Studies in Herpetology*<br>Современная герпетология | 2000–2019 | Russian |
| *Russian Journal of Ornithology*<br>Русский орнитологический журнал | 1993–2020 | Russian |
| *Bulletin of Moscow Society of Naturalists: Biological Series*<br>Бюллетень МОИП, серия биологическая | 1935–2020 | Russian |
| *Biological Bulletin of the Russian Academy of Science*<br>Известия РАН, серия биологическая | 1957–2020 | Russian |
| *Povolzhsky Journal of Ecology*<br>Поволжский экологический журнал | 2002–2020 | Russian |
| *Herald of Game Management*<br>Вестник охотоведения | 2007–2020(2) | Russian |
| *Steppe Bulletin*<br>Степной бюллетень | 1998–2020 | Russian |
| *Contemporary Problems of Ecology*<br>Сибирский экологический журнал | 1994–2020 | Russian |
| *Journal of Ichthyology*<br>Вопросы ихтиологии | 1961–2020 | Russian |
| *Nature Conservation Research*<br>Заповедная наука | 2016–2020(No.3) | Russian |
| *Shanghai Environmental Science*<br>上海环境科学 | 1982–2017 | Simplified Chinese |

| Journal | Years searched | Language |
|---|---|---|
| *Chinese Bulletin of Life Sciences*<br>生命科学 | 1988–2017 | Simplified Chinese |
| *Journal of Soil and Water Conservation*<br>水土保持学报 | 1987–2017 | Simplified Chinese |
| *Acta Ecologica Sinica*<br>生态学报 | 1981–2016 | Simplified Chinese |
| *Chinese Journal of Biological Control*<br>中国生物防治学报 | 1985–2017 | Simplified Chinese |
| *Journal of Plant Resources and Environment*<br>植物资源与环境学报 | 1992–2016 | Simplified Chinese |
| *Marine Environmental Science*<br>海洋环境科学 | 1982–2017 | Simplified Chinese |
| *Marine Sciences*<br>海洋科学 | 1977–2017 | Simplified Chinese |
| *Chinese Bulletin of Botany*<br>植物学报 | 2006–2016 | Simplified Chinese |
| *Zoological Systematics*<br>动物分类学报 | 1964–2017 | Simplified Chinese |
| *Chinese Journal of Applied and Environmental Biology*<br>应用与环境生物学报 | 1995–2017 | Simplified Chinese |
| *Sichuan Journal of Zoology*<br>四川动物 | 1996–2016 | Simplified Chinese |
| *Biodiversity Science*<br>生物多样性 | 1993–2016 | Simplified Chinese |
| *Wetland Science*<br>湿地科学 | 2003–2017 | Simplified Chinese |
| *Journal of Lake Sciences*<br>湖泊科学 | 1989–2017 | Simplified Chinese |
| *Scientia Silvae Sinicae*<br>林业科学 | 1955–2017 | Simplified Chinese |
| *Journal of Agro-Environment Science*<br>农业环境科学学报 | 1981–2017 | Simplified Chinese |
| *Journal of Biology*<br>生物学杂志 | 1983–2016 | Simplified Chinese |
| *Acta Theriologica Sinica*<br>兽类学报 | 1981–2018 | Simplified Chinese |
| *Acta Botanica Boreali-Occidentalia Sinica*<br>西北植物学报 | 2012–2016 | Simplified Chinese |
| *Journal of China Agricultural University*<br>中国农业大学学报 | 1955–2017 | Simplified Chinese |

| **Journal** | **Years searched** | **Language** |
|---|---|---|
| *Acta Pedologica Sinica*<br>土壤学报 | 1948–2017 | Simplified Chinese |
| *Ecological Science*<br>生态科学 | 1982–2016 | Simplified Chinese |
| *Acta Prataculturae Sinica*<br>草业学报 | 2008–2017 | Simplified Chinese |
| *Plant Protection*<br>植物保护 | 1963–2016 | Simplified Chinese |
| *World Forestry Research*<br>世界林业研究 | 1988–2017 | Simplified Chinese |
| *Chinese Journal of Microecology*<br>中国微生态学杂志 | 1989–2017 | Simplified Chinese |
| *Chinese Journal of Plant Ecology* (*formerly Acta Phytoecologica Sinica, Acta Phytoecologica et Geobotanica Sinica, Journal of Plant Ecology*)<br>植物生态学报 | 1963–2016 | Simplified Chinese |
| *Urban Environment & Urban Ecology*<br>城市环境与城市生态 | 1988–2016 | Simplified Chinese |
| *Acta Zoologica Sinica* (*subsequently Current Zoology from 2008*)<br>动物学报 | 1935–2008 | Simplified Chinese |
| *Chinese Journal of Wildlife*<br>野生动物学报 | 1979–2016 | Simplified Chinese |
| *Resources Science*<br>资源科学 | 1977–2016 | Simplified Chinese |
| *Chinese Journal of Zoology*<br>动物学杂志 | 1957–2016 | Simplified Chinese |
| *Journal of Natural Resources*<br>自然资源学报 | 1986–2016 | Simplified Chinese |
| *Acta Hydrobiologica Sinica*<br>水生生物学报 | 1997–2017 | Simplified Chinese |
| *Journal of Arid Land Resources and Environment*<br>干旱区资源与环境 | 1987–2017 | Simplified Chinese |
| *Journal of Mountain Science/Research*<br>山地学报 | 1983–2017 | Simplified Chinese |
| *Resources and Environment in the Yangtze Basin*<br>长江流域资源与环境 | 1992–2017 | Simplified Chinese |
| *Plant Diversity and Resources*<br>植物分类与资源学报杂志 | 1975–2017 | Simplified Chinese |

| Journal | Years searched | Language |
|---|---|---|
| *Journal of Ecology and Rural Environment* (*formerly Rural Eco-Environment*)<br>生态与农村环境学报 | 1985–2017 | Simplified Chinese |
| *Life Science Research*<br>生命科学研究 | 1997–2016 | Simplified Chinese |
| *Pratacultural Science*<br>草业科学 | 1984–2017 | Simplified Chinese |
| *Chinese Journal of Applied Ecology*<br>应用生态学报 | 1990–2016 | Simplified Chinese |
| *Journal of Hydroecology* (*formerly Reservoir Fisheries*)<br>水生态学杂志 | 1981–2017 | Simplified Chinese |
| *Bulletin of Soil and Water Conservation*<br>水土保持通报 | 1981–2017 | Simplified Chinese |
| *Chinese Journal of Grasslands* (*formerly Grassland of China*)<br>中国草地学报 | 1979–2016 | Simplified Chinese |
| *Advances in Marine Science*<br>海洋科学进展 | 1983–2017 | Simplified Chinese |
| *Journal of Tropical and Subtropical Botany*<br>热带亚热带植物学报 | 1992–2016 | Simplified Chinese |
| *China Environmental Science*<br>中国环境科学 | 1981–2017 | Simplified Chinese |
| *Ecology and Environmental Sciences* (*formerly Ecology and Environment*)<br>生态环境学报 | 1992–2016 | Simplified Chinese |
| *Asian Journal of Ecotoxicology*<br>生态毒理学报 | 2006–2017 | Simplified Chinese |
| *Journal of Fisheries of China*<br>水产学报 | 1965–2017 | Simplified Chinese |
| *Zoological Research*<br>动物学研究 | 1980–2016 | Simplified Chinese |
| *Chinese Journal of Ecology*<br>生态学杂志 | 1982–2016 | Simplified Chinese |
| *Journal of Desert Research*<br>中国沙漠 | 1981–2017 | Simplified Chinese |
| *Environmental Science*<br>环境科学 | 1976–2017 | Simplified Chinese |
| *Acta Agrestia Sinica*<br>草地学报 | 1989–2017 | Simplified Chinese |

| **Journal** | **Years searched** | **Language** |
|---|---|---|
| *Acta Phytophylacica Sinica*<br>植物保护学报 | 1962–2017 | Simplified Chinese |
| *Bulletin of Botanical Research*<br>植物研究 | 1959–2017 | Simplified Chinese |
| *Chinese Journal of Eco-Agriculture*<br>中国生态农业学报 | 1993–2017 | Simplified Chinese |
| *Soils*<br>土壤 | 1958–2017 | Simplified Chinese |
| *Edentata*<br>*Edentata* | 1994–2018 | Spanish |
| *Centros: Revista Cientifica Universitaria*<br>*Centros: Scientific Journal of the University* | 2012–2018 | Spanish |
| *Mastozoología Neotropical*<br>*Neotropical Mammalogy* | 1994–2018 | Spanish |
| *Biodiversity and Natural History (formerly Boletín de Biodiversidad de Chile)*<br>*Biodiversity and Natural History (formerly Boletín de Biodiversidad de Chile)* | 2015–2017 | Spanish |
| *Journal of Bat Research and Conservation (formerly known as Barbastella)* | 2017–2019 | Spanish |
| *Bioma (El Salvador)* | 2012–2016 | Spanish |
| *Revista de Biología Tropical*<br>*International Journal of Tropical Biology and Conservation* | 1976–2018 | Spanish |
| *Butlletí del Grup Català d'Anellament*<br>*Bulletin of the Catalan Ring Group* | 1981–2001 | Spanish |
| *Agrociencia Uruguay*<br>*Agroscience Uruguay* | 1997–2017 | Spanish |
| *Galemys* | 1997–2017 | Spanish |
| *Zoologica Baetica* | 1990–2015 | Spanish |
| *Therya* | 2010–2018 | Spanish |
| *Etología*<br>*Ethology* | 1989–2003 | Spanish |
| *Boletín Chileno de Ornitología*<br>*Chilean Ornithology Bulletin* | 1994–2015 | Spanish |
| *Acta Zoológica Mexicana*<br>*Mexican Zoological Record/Journal* | 1984–2019 | Spanish |
| *Ecología Aplicada*<br>*Applied Ecology* | 2002–2018 | Spanish |

| Journal | Years searched | Language |
|---|---|---|
| *Huitzil: Revista Mexicana de Ornitología*<br>*Huitzil: Journal of Mexican Ornithology* | 2000–2018 | Spanish |
| *Orinoquia* | 2003–2018 | Spanish |
| *Boletín de la Sociedad Argentina de Botánica*<br>*Bulletin of the Botanical Society of Argentina* | 2013–2018 | Spanish |
| *Arxius de Miscel·lània Zoològica*<br>*Arxius de Miscel·lània Zoològica* | 2003–2019 | Spanish |
| *Quebracho: Revista de Ciencias Forestales*<br>*Quebracho: Journal of Forest Sciences* | 2008–2018 | Spanish |
| *Ardeola* | 1954- 2019 | Spanish |
| *Boletín de la Asociación Herpetológica Española*<br>*Bulletin of the Spanish Herpetological Association* | 2004–2018 | Spanish |
| *Historia Natural*<br>*Natural History* | 2011–2018 | Spanish |
| *Revista Chilena de Historia Natural*<br>*Chilean Journal of Natural History* | 1897–2018 | Spanish |
| *Cuadernos de Herpetología*<br>*Herpetology notebooks* | 2010–2018 | Spanish |
| *Barbastella* | 2000–2016 | Spanish |
| *Grupo Jaragua* | 1997–2011 | Spanish |
| *Cedamaz* | 2014–2018 | Spanish |
| *Revista Mexicana de Mastozoología*<br>*Mexican Journal of Mastozoology* | 1995–2017 | Spanish |
| *Boletín de Biodiversidad de Chile*<br>*Bulletin of Biodiversity of Chile* | 2009–2014 | Spanish |
| *Revista Internacional de Contaminación Ambiental*<br>*International Journal of Environmental Pollution* | 1985–2018 | Spanish |
| *Folia Amazónica* | 1988–2018 | Spanish |
| *Studia Oecológica* | 1981–1995 | Spanish |
| *El Hornero: Revista de Ornitología Neotropical* | 2003–2017 | Spanish |
| *Ecosistemas: Revista Científica de Ecología y Medio Ambiente*<br>*Ecosystems: Scientific Journal of Ecology and Environment* | 2001–2018 | Spanish |
| *Revista Mexicana de Ciencias Forestales*<br>*Mexican Journal of Forestry Sciences* | 2010–2018 | Spanish |
| *Caldasia* | 1940–2019 | Spanish |
| *Ecosistemas y Recursos Agropecuarios*<br>*Ecosystems and Agricultural Resources* | 1994–2018 | Spanish |
| *Colombia Forestal* | 2000–2018 | Spanish |

| Journal | Years searched | Language |
|---|---|---|
| *Animal Biodiversity and Conservation* (*Museu de Ciències Naturals de Barcelona*) | 2001–2019 | Spanish |
| *Revista Española de Herpetologia*<br>*Spanish Journal of Herpethology* | 2003–2007 | Spanish |
| *BioScriba*<br>*BioScriba* | 2008–2017 | Spanish |
| *Notulas Faunisticas* | 2008–2018 | Spanish |
| *Novitates Caribaea* | 1999–2019 | Spanish |
| *Gestión Ambiental* | 1999–2017 | Spanish |
| *Revista Nicaragüense de Biodiversidad*<br>*Nicaraguan Journal of Biodiversity* | 2015–2019 | Spanish |
| *Bosques Latitud Cero*<br>*Forests Latitude Zero* | 2014–2018 | Spanish |
| *Ocelotlán* | 2003–2012 | Spanish |
| *Revista Chilena de Ornitología* (*formerly Boletín Chileno de Ornitología*)<br>*Chilean Journal of Ornithology* | 2016–2018 | Spanish |
| *Mammalogy Notes* | 2014–2017 | Spanish |
| *Mediterránea: Serie de Estudios Biológicos*<br>*Mediterranean: Biological Studies Series* | 1982–2015 | Spanish |
| *A Carriza: Sociedade Galega de Ornitoloxia* | 2001–2009 | Spanish |
| *Hidrobiológica*<br>*Hydrobiology* | 1991–2018 | Spanish |
| *Revista Peruana de Biología*<br>*Peruvian Journal of Biology* | 1974–2019 | Spanish |
| *Ecología Austral*<br>*Austral Ecology* | 2001–2018 | Spanish |
| *Boletín de la Real Sociedad Española de Historia Natural: Sección Biológica*<br>*Bulletin of the Royal Spanish Society of Natural History: Biological Section* | 2003–2017 | Spanish |
| *Madera y Bosques*<br>*Wood and Forests* | 1995–2018 | Spanish |
| *Boletín Científico Centro de Museos: Museo de Historia Natural*<br>*Scientific Journal of the Museum Center: Natural History* | 1996–2019 | Spanish |
| *Revista Mexicana de Biodiversidad*<br>*Mexican Journal of Biodiversity* | 2005–2018 | Spanish |
| *Anales de Biología* | 1984–2019 | Spanish |

| Journal | Years searched | Language |
|---|---|---|
| *Revista Catalana d'Ornitologia*<br>*Catalan Journal of Ornithology* | 2002–2018 | Spanish |
| *Semiárida* | 2013–2018 | Spanish |
| *Quarterly Journal of Chinese Forestry* (*Taiwan*)<br>中華林學季刊 | 2004–2019 | Traditional Chinese |
| *Journal of the National Taiwan Museum*<br>國立臺灣博物館學刊 | 2005–2019 | Traditional Chinese |
| *Raptor Research of Taiwan*<br>台灣猛禽研究 | 2003–2016 | Traditional Chinese |
| *Bio Formosa* (*Taiwan*)<br>生物學報 | 1966–2014 | Traditional Chinese |
| *Journal of the Experimental Forest of National Taiwan University*<br>臺灣大學生物資源暨農學院實驗林研究報告 | 1987–2019 | Traditional Chinese |
| *Taiwan Journal of Biodiversity*<br>台灣生物多樣性研究 | 1999–2019 | Traditional Chinese |
| *Notes and Newsletter of Wildlifers* (*Taiwan*)<br>野生動物保育彙報及通訊 | 2005–2012 | Traditional Chinese |
| *Journal of Ecology and Environmental Sciences* (*Taiwan*)<br>環境與生態學報 | 2008–2012 | Traditional Chinese |
| *Fungal Science* (*Taiwan*) | 1995–2019 | Traditional Chinese |
| *Chinese Bioscience* (*Taiwan*)<br>生物科學 | 2003–2014 | Traditional Chinese |
| *Journal of National Park* (*Taiwan*)<br>國家公園學報 | 1989–2019 | Traditional Chinese |
| *Taipei Zoo Bulletin*<br>動物園學報 | 1989–2013 | Traditional Chinese |
| *Journal of Agriculture and Forestry* (*Taiwan*)<br>農林學報 | 2000–2018 | Traditional Chinese |
| *Taiwan Journal of Forest Science*<br>臺灣林業科學 | 1986–2020 | Traditional Chinese |
| *Doğanın Sesi*<br>*Journal of Nature's Voice* | 2018–2019 | Turkish |
| *Dicle Üniversitesi Fen Bilimleri Enstitüsü Dergisi*<br>*Journal of Dicle University Natural Sciences Enstitute* | 2019 | Turkish |
| *Türk Tarım - Gıda Bilim ve Teknoloji Dergisi*<br>*Turkish Journal of Agriculture - Food Science and Technology* | 2014–2019 | Turkish |

| Journal | Years searched | Language |
|---|---|---|
| *Anadolu Orman Araştırmaları Dergisi*<br>*Anatolia Journal of Forest Research* | 2015–2019 | Turkish |
| *Toprak Bilimi ve Bitki Besleme Dergisi*<br>*Journal of Soil Science and Plant Nutrition* | 2012–2019 | Turkish |
| *Türkiye Ormancılık Dergisi*<br>*Journal of Turkey Forestry* | 2000–2019 | Turkish |
| *Iğdır Üniversitesi Fen Bilimleri Enstitüsü Dergisi*<br>*Journal of Iğdır University Institute of Science* | 2019–2020 | Turkish |
| *Bartın Orman Fakültesi Dergisi*<br>*Journal of Bartin Faculty of Forestry* | 2000–2019 | Turkish |
| *Su Ürünleri Dergisi*<br>*Journal of Fisheries* | 2000–2019 | Turkish |
| *Ege Üniversitesi Ziraat Fakültesi Dergisi*<br>*Journal of Ege University Faculty of Agriculture* | 2014–2019 | Turkish |
| *Kommagene Biyoloji Dergisi*<br>*Commagene Journal of Biology* | 2017–2019 | Turkish |
| *Zeugma Biyolojik Bilimler Dergisi*<br>*Zeugma Biological Science* | 2020 | Turkish |
| *Türk Coğrafya Dergisi*<br>*Turkish Geographical Review* | 2000–2019 | Turkish |
| *Akdeniz Üniversitesi Ziraat Fakültesi Dergisi*<br>*Mediterranean Agricultural Sciences* | 2009–2019 | Turkish |
| *Deniz Bilimleri ve Muhendisligi Dergisi*<br>*Aquatic Sciences and Engineering* | 2007–2020 | Turkish |
| *Kastamonu Üniversitesi Orman Fakültesi Dergisi*<br>*Journal of Kastamonu University Faculty of Forestry* | 2001–2019 | Turkish |
| *Bağbahçe Bilim Dergisi*<br>*Journal of Bagbahce Science* | 2019 | Turkish |
| *Uluslararasi Doga Bilimleri be Biyoteknoloji Dergisi*<br>*International Journal of Life Sciences and Biotechnology* | 2018–2019 | Turkish |
| *Doğu Coğrafya Dergisi*<br>*Journal of Eastern Geography* | 2010–2019 | Turkish |
| *Uluslararası Doğu Anadolu Fen Mühendislik ve Tasarım Dergisi*<br>*Journal of International East Anatolia Science Engineering and Design* | 2019 | Turkish |
| *Atatürk Üniversitesi Ziraat Fakültesi Dergisi*<br>*Atatürk University Journal of Agricultural Faculty* | 2008–2020 | Turkish |

| Journal | Years searched | Language |
|---|---|---|
| *Trakya University Journal of Natural Sciences*<br>*Trakya University Journal of Natural Sciences* | 2000–2019 | Turkish |
| *Dumlupınar Üniversitesi Fen Bilimleri Enstitüsü Dergisi*<br>*Journal of Dumlupınar University Institute of Science* | 2000–2019 | Turkish |
| *Artvin Çoruh Üniversitesi Orman Fakültesi Dergisi*<br>*Artvin Coruh University Journal of Forestry Faculty* | 2000–2020 | Turkish |
| *Orman Bilimleri Dergisi*<br>*Turkish Journal of Forest Science* | 2017–2019 | Turkish |
| *İstanbul Üniversitesi Orman Fakültesi Dergisi* (*1951–2017; continues in English as Forestist from 2018*)<br>*Journal of the Faculty of Forestry Istanbul University* (*continues in English as Forestsist from 2018*) | 2009–2019 | Turkish |
| *Akademik Ziraat Dergisi*<br>*Journal of Academic Agriculture* | 2012–2019 | Turkish |
| *Nature Conservation* (*2013–2016*) [*formerly Nature Reserves in Ukraine* (*1995–2012*)]<br>Заповідна справа (*2013–2016*) [Заповідна справа в Україні (*1995–2012*)] | 2013–2016 | Ukrainian |
| *Problems of Bioindication and Ecology*<br>Питання біоіндикації та екології | 2008–2019 | Ukrainian |
| *Nature Reserves in Ukraine* (*1995–2012*) [*changed to Nature Conservation* (*2013–2016*)]<br>Заповідна справа в Україні (*1995–2012*)<br>[Заповідна справа (*2013–2016*)] | 1995–2012 | Ukrainian |
| *Visnyk of Lviv University: Biological Series*<br>Вісник Львівського університету: Серія біологічна | 2005–2019 | Ukrainian |

# Appendix 3: Reports (and years) searched

Conservation reports searched for the discipline-wide Conservation Evidence database. An asterisk indicates the reports most relevant to this synopsis.

| | | |
|---|---|---|
| Agreement on the Conservation of Cetaceans of the Black Sea Mediterranean Sea and contiguous Atlantic area (ACCOBAMS) | 45 numbered documents | Resolutions - Conservation actions (45 documents numbered but not in order). Official reports not searched (http://www.accobams.org/documents-resolutions/official-reports/) |
| Amphibian and Reptile Conservation (ARC) | 2021 | Dated reports 2012–2021 at https://www.arc-trust.org/technical-reports |
| Amphibian Survival Alliance | 1994–2012 | "Froglog (Bulletin of the Amphibian Survival Alliance)" magazine: Vol 9 - Vol 104 |
| Back from the Brink: Shifting Sands | x5 documents dated 2021 | All docs (x5 dated 2021) at this URL https://naturebftb.co.uk/the-projects/shifting-sands/ |
| British Trust for Ornithology | 1981–2016 | BTO Research Reports: 1–687 |
| Convention on the Conservation of Migratory Species of Wild Animals (CMS) | 1998–2018 | All documents 1998–2018 inclusive including Techincal Series reports TS no. 1–38 (some numbers missing: 628-303637) |

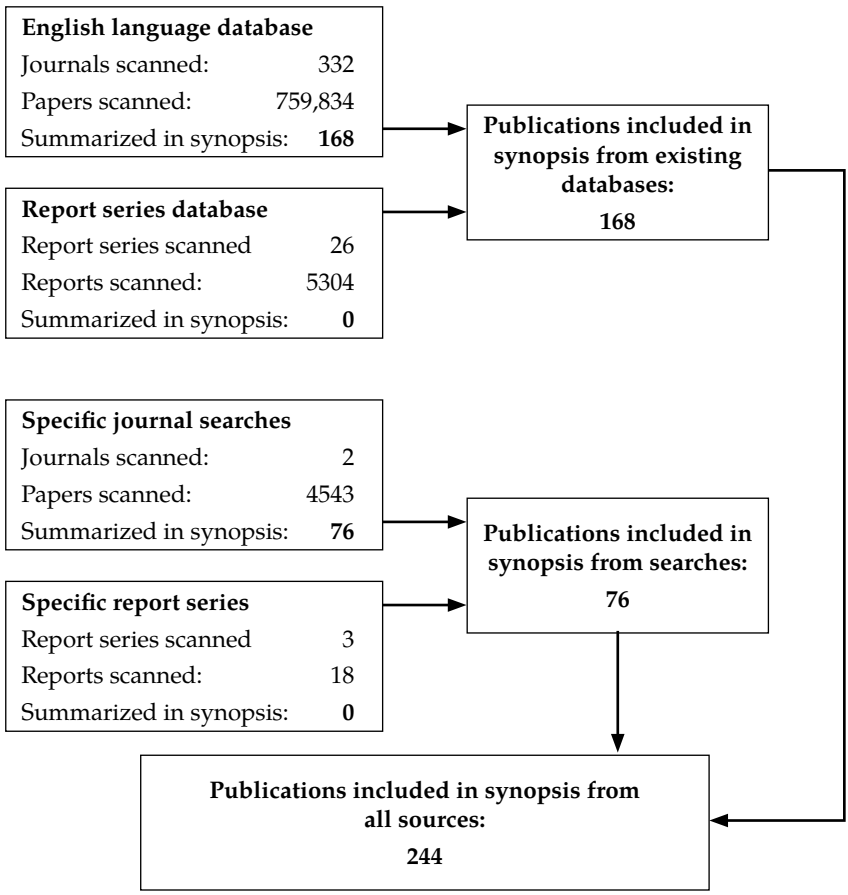
| | | |
|---|---|---|
| Environment Agency | 1996–2023 | Environment Agency - Environment Research Reports - Dated UK reports under the heading 'Research' and topic 'Environment', and Organisation 'Environment Agency' at: https://www.gov.uk/search/research-and-statistics |
| International Council for the Exploration of the Sea (ICES)* | 2011–2018 | ICES Working Group on Bycatch of Protected Species (WGBYC) Expert Reports: 2011–2018 inclusive (www.ices.dk/publications/our-publications/Pages/Expert-Group-Reports.aspx) |
| International Council for the Exploration of the Sea (ICES)* | 2003–2018 | ICES Working Group on Marine Mammal Ecology (WGMME) Expert Reports: 2003–2018 inclusive (www.ices.dk/publications/our-publications/Pages/Expert-Group-Reports.aspx) |
| International Society for Mangrove Ecosystems | 1993–2014 | Occasional Papers and Technical Reports dated 1993–2014 searched at http://www.mangrove.or.jp/english/subpage/publications.html |
| IUCN-SSC Anguillid Eel Specialist Group | 2016–2021 | IUCN-SSC Anguillid Eel Specialist Group Reports - Date reports at: https://www.iucn.org/ourunion/commissions/group/iucn-ssc-anguillid-eel-specialist-group |
| IUCN-SSC Cetacean Specialist Group | 1989–2018 | Cetacean Specialist Group Reports. Dated reports at https://iucn-csg.org/downloads/ |
| IUCN-SSC Crocodile Specialist Group | 2006–2018 | Crocodile Specialist Group Articles. Dated articles at http://www.iucncsg.org/pages/Publications.html |
| IUCN-SSC Crocodile Specialist Group | 2005–2017 | Crocodile Specialist Group Reports. Dated reports at http://www.iucncsg.org/pages/Publications.html |
| IUCN-SSC Freshwater Plant Specialist Group | 2016–2018 | IUCN-SSC Freshwater Plant Specialist Group Reports at https://www.iucn.org/commissions/ssc-groups/plants-fungi/plants/plants-a-g/freshwater-plant |
| IUCN-SSC Invasive Species Specialist Group | 1995–2013 | Aliens: The Invasive Species Bulletin (IUCN) Vol 1 - Vol 33 |

| | | |
|---|---|---|
| IUCN-SSC Marine Mammal Protected Area Specialist Group | 2017–2018 | Marine Mammal Protected Area Specialist Group Reports. Dated documents at https://www.marinemammalhabitat.org/downloads/ |
| Joint Nature Conservation Committee (JNCC)* | 1991–2018 | Report no.s 1–627 |
| MedWet | 1994–2017 | All publications dated 1994–2017 at https://medwet.org/publications/ |
| National Oceanic and Atmospheric Administration (NOAA)* | 1962–2018 | Fisheries Science & Data Resource Reports. Science & Data>Research and Survey Resources (dated) for species categories: whales dolphins and porpoises seals and sea lions i.e. not all reports at this link checked (https://www.fisheries.noaa.gov/resources/all-science?title=&species%5B54%5D=54&species%5B1000000066%5D=1000000066&species%5B53%5D=53&field_species_vocab_target_id=&sort_by=created) |
| Natural England | 1991–2018 | Reports dated 1991–2018 listed at http://publications.naturalengland.org.uk/category/7002 & http://publications.naturalengland.org.uk/category/10002 at Sep 2019. Records about... Habitat and species group sub-categories; Records about... Species; Terrestrial habitats; Farming & land management; Coastal Freshwater Marine |
| NatureScot | 2016–2018 | Reports 1–945 (2004–2018) |
| North Atlantic Marine Mammal Commission | 1998–2018 | NAMMCO outputs (Scientific publication series Vol1(1998)–10(2018) at https://nammco.no/library/ |
| Ramsar | 1998–2017 | Documents dated 1998–2017 at https://www.ramsar.org/search |
| Scientific Committee on Antarctic Research (SCAR) | 2004–2018 | 4 dated reports (2014–2018) and list of 7 selected publications (https://www.scar.org/science/eg-bamm/) |

| | | |
|---|---|---|
| Sea Mammal Research Unit (SMRU) | 2012–2018 | Marine Mammal Scientific Support to Scottish Government reports at http://www.smru.st-andrews.ac.uk/research-policy/reports-to-scottish-government/ |
| Sea Mammal Research Unit (SMRU) | 1990–2018 | SMRU reports for funders at http://www.smru.st-andrews.ac.uk/reports/ |
| Wetlands International | 1980–2017 | Publications Case Studies dated 1980–2017 (including "Flamingo: Bulletin of the IUCN-SSC/Wetlands International Flamingo Specialist Group" magazine) at https://www.wetlands.org/resources/ |
| Whale and Dolphin Conservation (WDC) | 2001–2018 | Dated reports 2001 - 2018 at https://uk.whales.org/policy/wdc-publications-and-reports/ |

# Appendix 4: Literature reviewed for the Coral Conservation Synopsis

The diagram below shows the total numbers of journals and report series searched for this synopsis, the total number of publications searched (title and abstract) within those, and the number of publications that were summarized from each source of literature



# Index



* stony/hard/scleractinian coral; ** soft coral.

# About the Team

Alessandra Tosi was the managing editor for this book.

This book was proofread by Tricia De Souza and indexed by Adèle Kreager.

Jeevanjot Kaur Nagpal designed the cover. The cover was produced in InDesign using the Fontin font.

Jeremy Bowman typeset the book in InDesign and produced the paperback and hardback editions. The main text font is Tex Gyre Pagella and the heading font is Californian FB. Jeremy also created the EPUB. Cameron Craig produced the PDF edition.

The conversion to the HTML edition was performed with epublius, an open-source software which is freely available on our GitHub page at https://github.com/OpenBookPublishers

Raegan Allen was in charge of marketing.

This book was peer-reviewed by anonymous referees. Experts in their field, these readers give their time freely to help ensure the academic rigour of our books. We are grateful for their generous and invaluable contributions.

# This book need not end here...

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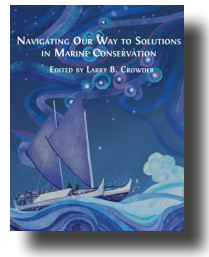


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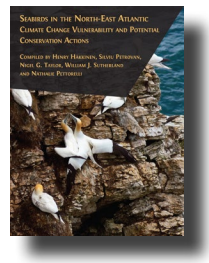


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*William J. Sutherland (editor)*

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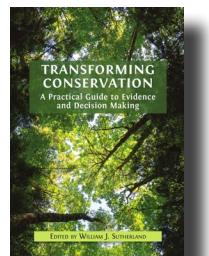
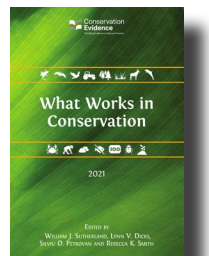



## What Works in Conservation

### 2021

*William J. Sutherland, Lynn V. Dicks, Silviu O. Petrovan & Rebecca K. Smith (editors)*

https://doi.org/10.11647/OBP.0267



# Coral Conservation
## Global evidence for the effects of actions

Ann Thornton, William H. Morgan, Eleanor K. Bladon, Rebecca K. Smith, and William J. Sutherland

*I am extremely impressed with both the scope and detail of this project. I think it will be immensely valuable.*

John Bruno, University of North Carolina

*Coral Conservation* provides an essential resource for anyone dedicated to conserving or restoring corals. This comprehensive synthesis of global scientific evidence examines the effectiveness of conservation and restoration actions targeting stony, soft and cold-water coral species inhabiting a diverse range of marine habitats in tropical, temperate and arctic waters from shallow coasts to the deep sea.

Addressing the urgent threats posed by climate change, invasive species, overfishing, and habitat destruction, this work summarizes evidence from actions in three core themes: protecting healthy reefs, mitigating human impacts, and undertaking active restoration. From establishing Marine Protected Areas to innovative techniques like coral gardening, the synopsis summarizes the evidence for practical actions and offers insights into their outcomes and applicability.

Designed to guide decision-makers—resource managers, conservationists, policymakers, and local advocates—as well as those curious to learn about actions that could help corals, this accessible guide provides succinct information to support evidence-based conservation.

By identifying the existing evidence and highlighting gaps in the knowledge, this volume can support practitioners and policymakers to allocate resources effectively by prioritising actions that work. By doing more of what works, we can reverse the loss of coral species and restore these vital habitats for the benefit of current and future generations.

The authors consulted an international group of coral experts and conservationists to produce this synopsis. Research funding was provided by A.G. Leventis Foundation and Oceankind.

This is the 25th publication in the Conservation Evidence Series Synopses, and is freely available from the online Conservation Evidence database (www.conservationevidence.com) ensuring that users have ongoing access to updated research and assessments. As with all Open Book publications, this entire book is also available to download for free on the publisher's website. Printed and digital editions, together with supplementary digital material, can also be found at: http://www.openbookpublishers.com.

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