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Global kelp forest restoration: past lessons, present status, and future directions

Aaron M. Eger[1*], Ezequiel M. Marzinelli[2,3,4], Hartvig Christie[5], Camilla W. Fagerli[5], Daisuke Fujita[6], Alejandra P. Gonzalez[7], Seok Woo Hong[8], Jeong Ha Kim[8], Lynn C. Lee[9,10], Tristin Anoush McHugh[11,†], Gregory N. Nishihara[12], Masayuki Tatsumi[13], Peter D. Steinberg[1,3] and Adriana Vergés[1,3]

[1]*Centre for Marine Science and Innovation & Ecology and Evolution Research Centre, School of Biological, Earth and Environmental Sciences, The University of New South Wales, Sydney, NSW 2052*
[2]*The University of Sydney, School of Life and Environmental Sciences, Sydney, NSW 2006, Australia*
[3]*Sydney Institute of Marine Science, 19 Chowder Bay Rd, Mosman, NSW 2088, Australia*
[4]*Singapore Centre for Environmental Life Sciences Engineering, Nanyang Technological University, Singapore, 637551, Singapore*
[5]*Norwegian Institute for Water Research, Økernveien 94, Oslo, 0579, Norway*
[6]*University of Tokyo Marine Science and Technology, School of Marine Bioresources, Applied Phycology, Konan, Minato-ku, Tokyo, 108-8477, Japan*
[7]*Departamento de Ciencias Ecológicas, Facultad de Ciencias, Universidad de Chile, Las Palmeras 3425, Ñuñoa, Santiago, Chile*
[8]*Department of Biological Sciences, Sungkyunkwan University, Suwon, 2066, South Korea*
[9]*Gwaii Haanas National Park Reserve, National Marine Conservation Area Reserve, and Haida Heritage Site, 60 Second Beach Road, Skidegate, Haida Gwaii, BC, V0T 1S1, Canada*
[10]*Canada & School of Environmental Sciences, University of Victoria, 3800 Finnerty Road, Victoria, BC, V8P 5C2, Canada*
[11]*Reef Check Foundation, Long Marine Laboratory, 115 McAllister Road, Santa Cruz, CA 95060, U.S.A.*
[12]*Organization for Marine Science and Technology, Institute for East China Sea Research, Nagasaki University, 1551-7 Taira-machi, Nagasaki City, 851-2213, Japan*
[13]*Institute for Marine and Antarctic Studies, University of Tasmania, Hobart, TAS 7004, Australia*

ABSTRACT

Kelp forest ecosystems and their associated ecosystem services are declining around the world. In response, marine managers are working to restore and counteract these declines. Kelp restoration first started in the 1700s in Japan and since then has spread across the globe. Restoration efforts, however, have been largely disconnected, with varying methodologies trialled by different actors in different countries. Moreover, a small subset of these efforts are 'afforestation', which focuses on creating new kelp habitat, as opposed to restoring kelp where it previously existed. To distil lessons learned over the last 300 years of kelp restoration, we review the history of kelp restoration (including afforestation) around the world and synthesise the results of 259 documented restoration attempts spanning from 1957 to 2020, across 16 countries, five languages, and multiple user groups. Our results show that kelp restoration projects have increased in frequency, have employed 10 different methodologies and targeted 17 different kelp genera. Of these projects, the majority have been led by academics (62%), have been conducted at sizes of less than 1 ha (80%) and took place over time spans of less than 2 years. We show that projects are most successful when they are located near existing kelp forests. Further, disturbance events such as sea-urchin grazing are identified as regular causes of project failure. Costs for restoration are historically high, averaging hundreds of thousands of dollars per hectare, therefore we explore avenues to reduce these costs and suggest financial and legal pathways for scaling up future restoration efforts. One key suggestion is the creation of a living database which serves as a platform for recording restoration projects, showcasing and/or

* Address for correspondence (Tel: +610487523824; E-mail: aaron.eger@unsw.edu.au)
†Present address: The Nature Conservancy, 830 S Street, Sacramento, CA 95811, U.S.A.

re-analysing existing data, and providing updated information. Our work establishes the groundwork to provide adaptive and relevant recommendations on best practices for kelp restoration projects today and into the future.

Key words: kelp, restoration, marine, Laminariales, transplant, sea urchins, seaweed, costs, recovery, forest

CONTENTS

I. Introduction . . . 1450
(1) The need to restore kelp forests . . . 1450
(2) History of kelp forest management . . . 1451
(3) Motivations for restoring kelp forests in the 21st century . . . 1451
(4) Study objectives . . . 1452
II. Materials and methods . . . 1452
(1) Literature searches . . . 1452
(2) Data collection . . . 1453
(3) Factor analysis . . . 1454
III. Regional histories of restoration . . . 1454
(1) Overview of kelp forest restoration . . . 1454
(2) Japan . . . 1454
(3) Korea . . . 1456
(4) USA . . . 1456
(5) Canada . . . 1457
(6) Australia . . . 1458
(7) Europe . . . 1458
(8) Chile . . . 1459
IV. Analysis of the global database . . . 1459
(1) Overview . . . 1459
(2) Groups involved in restoration . . . 1459
(3) Project size . . . 1461
(4) Proximity to other kelp forests improves project success . . . 1462
(5) Environmental barriers to restoration success . . . 1462
(6) Ecological success in kelp forest restoration . . . 1462
(7) Kelp restoration in Japan: a qualitative assessment . . . 1463
V. Restoration methodologies . . . 1463
(1) Transplanting . . . 1463
(2) Seeding kelp populations . . . 1463
(3) Removing competitors . . . 1464
(4) Grazer control . . . 1464
(5) Artificial reefs . . . 1465
(6) Restoration methodologies in the future . . . 1466
VI. Socioeconomic considerations for restoration . . . 1466
(1) Financing restoration . . . 1466
(2) Legal frameworks for restoration . . . 1467
VII. Conclusions . . . 1468
VIII. Acknowledgements . . . 1469
IX. References . . . 1469
X. Supporting information . . . 1475

I. INTRODUCTION

(1) The need to restore kelp forests

Kelp forests, defined here as habitat-forming brown algae in the orders Laminariales, Fucales, and Desmarestiales (Wernberg & Filbee-Dexter, 2019), are globally distributed habitats which have declined around the world (Thibaut *et al.*, 2005; Fujita, 2011; Johnson *et al.*, 2011; Vásquez *et al.*, 2014a; Blamey & Bolton, 2018; Rogers-Bennett & Catton, 2019). The causes of these declines range from local stressors such as pollution to global impacts, such as climate change (Wernberg *et al.*, 2019). Early and persistent declines of kelp forests in the 1800s were linked to population expansion of sea urchins, most often facilitated by the removal of urchin predators from the ecosystem (Roberts, 2007). Subsequent kelp population declines in the 20th century were driven by threats such as direct harvest of kelp or high levels of water pollution from urban areas (Wilson & North, 1983; Vogt & Schramm, 1991; Coleman *et al*., 2008; Connell *et al*., 2008).

These stressors are still relevant to contemporary kelp ecosystem management but now interact with climate change, a phenomenon that has multiple consequences for kelp forests (Smale, 2020). Increasing water temperatures and marine heatwaves have resulted in large contractions of kelp populations as they are pushed past their physiological preferences and limits (Tegner & Dayton, 1991; Kang, 2010; Wernberg *et al*., 2016a; Arafeh-Dalmau *et al*., 2019; Rogers-Bennett & Catton, 2019). Warmer sea water temperatures have also facilitated the range expansion of herbivorous sea urchins which can overgraze entire forests and create urchin barrens, a phenomenon identified in most countries that contain kelp (Fujita, 2010; Filbee-Dexter & Scheibling, 2014; Ling *et al*., 2015). More recently, temperature-driven shifts in the ranges of herbivorous fishes are also causing similar declines in kelp forests near the warm edge of their distribution (Vergés *et al*., 2014; Zarco-Perello *et al*., 2017). Such extensive losses have dramatic ecological and economic impacts. For instance, kelp losses have caused the closure of lobster, abalone, sea urchin, and kelp fisheries in several regions around the globe (Steneck *et al*., 2013; Bajjouk *et al*., 2015; Rogers-Bennett & Catton, 2019).

(2) History of kelp forest management

Managing kelp forests and their declines has a lengthy global history. Traditionally, kelp forest management has been a passive activity whereby managers focused on improving environmental or physical conditions, for instance, by improving water quality (Foster & Schiel, 2010), limiting kelp harvest (Fujita, 2011; Frangoudes & Garineaud, 2015), or protecting species that facilitate kelp forests (Caselle *et al*., 2015). These methods can be successful, and low-level exploitation in Chile, Norway, Ireland, and France have ensured that sustainable kelp harvesting continues to exist in those countries (Werner & Kraan, 2004; Lorentsen, Sjotun & Gremillet, 2010; Buschmann *et al*., 2014; Frangoudes & Garineaud, 2015). Marine protected areas (MPAs) have also worked to increase populations of species that facilitate kelp forests and reduce human pressures (Caselle *et al*., 2015). For example New Zealand created the Cape Rodney to Okakari Point Marine Reserve (i.e. 'Leigh Reserve') in 1976 and this MPA now maintains healthy kelp forests (*Ecklonia radiata*, J. Agardh, and Fucales species) relative to areas outside the reserve, which are dominated by urchin barrens (Shears & Babcock, 2003).

Despite successes with other conservation objectives such as restoring predator populations, (Lester *et al*., 2009), many passive measures (i.e. those that do not manipulate kelp or their consumers) have failed to re-establish lost kelp populations (Wernberg *et al*., 2019). For instance, improvements in water quality in Sydney, Australia (Scanes & Philip, 1995) did not lead to the re-establishment of the locally extinct fucoid, crayweed (*Phyllospora comosa*, C. Agardh) (Coleman *et al*., 2008; Vergés *et al*., 2020). Transplant experiments demonstrated that while the environment was now suitable for *P. comosa*, propagule supply and/or post-settlement survival was likely insufficient for the species to re-establish populations naturally (Campbell *et al*., 2014). While other passive approaches like MPAs can succeed in restoring predator species and kelp forests (Eger & Baum, 2020), they can also fail to facilitate the re-establishment of a kelp forest (Leung, Yeung & Ang, 2014). As a result, managers are increasingly considering active restoration approaches in combination with removing or mitigating the causes of decline (Morris *et al*., 2020; Layton *et al*., 2020b).

Restoration is defined by the Society for Ecological Restoration (SER) as 'the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed' (SER, 2004, p. 3). Active restoration is attempted by introducing or removing biotic or abiotic materials from the environment. If kelp reproduction is limited, reproductive individuals are introduced, either by adding spores or gametophytes and/or by transplanting mature plants that act themselves as the spore source (Layton *et al*., 2021). If herbivory is an issue, it can be mitigated by culling, transporting, or harvesting grazers such as urchins or herbivorous fish (Fujita, 2010; Watanuki *et al*., 2010; Tracey *et al*., 2015; Strand *et al*., 2020; Lee *et al*., 2021). Thus, restoration as defined by SER requires that the activity improves or brings back previously existing species or habitats, regardless of the restoration methods used.

Restoration as defined above is distinguished from 'afforestation' (e.g. habitat offsetting) which is the process of creating new kelp habitat in areas that did not previously have kelp forests and is therefore not considered 'true' restoration. Artificial reef deployment is the most common form of afforestation, which creates kelp habitat by adding new rocky reef substrate that can enhance the settlement and growth of existent kelp propagules or can act as a base for transplanting or seeding (Schroeter, Reed & Raimondi, 2018; Shelamoff *et al*., 2020).

(3) Motivations for restoring kelp forests in the 21st century

Restoring kelp forests provides society with many benefits. Healthy kelp forests directly support United Nations Sustainable Development Goals 2 (zero hunger), 8 (work and economic growth), 13 (climate action), and 14 (life under water; Cormier & Elliott, 2017). By conserving and restoring kelp ecosystems, we maintain a foundational marine habitat and ensure access to key ecosystem services such as habitat provisioning (Teagle *et al*., 2017), nutrient cycling (Kim, Kraemer & Yarish, 2015) and carbon sequestration (Chung *et al*., 2013; Filbee-Dexter & Wernberg, 2020). Kelp forests also underpin harvest services, for example, supporting direct kelp harvest (Buschmann *et al*., 2014) or fisheries through the species that they support (Smale *et al*., 2013). The services provided by these underwater forests are currently estimated at millions of dollars per km of coastline and billions of dollars per country (Smale *et al.*, 2013; Vásquez *et al.*, 2014a; Bennett *et al.*, 2016; Blamey & Bolton, 2018; Eger *et al.*, 2021), and provide livelihoods for coastal communities around the world. In addition to their economic values, kelp forests also hold significant cultural and aesthetic value to their local community (Thurstan *et al.*, 2018; Turnbull *et al.*, 2020).

International interest and recognition of marine ecosystem restoration is increasing, yet kelp forests are often excluded from these agendas despite their potential contributions to international goals and targets (Feehan, Filbee-Dexter & Wernberg, 2021). The largest initiatives are led by the United Nations (UN), which has declared 2021–2030 as the 'Decade of Ecosystem Restoration' as well as the 'Decade of Ocean Science for Sustainable Development'. These independent but complementary initiatives are calling for a global focus on renewing marine and other ecosystems (Waltham *et al.*, 2020), while also providing needed ecosystems services, helping combat climate change and safeguarding biodiversity and food security (Claudet *et al.*, 2020). Kelp forest restoration has the potential to meet the objectives of both UN initiatives. If carbon credits are verified and established, kelp forest restoration also provides a means for countries to work toward their 'Nationally Determined Contribution' (NDC) to mitigate carbon emissions under the Paris Agreement, in addition to European Union agreements to restore set amounts of habitat. These contributions could then also be commodified as carbon credits, while other services such as nutrient removal could also be commodified and provide further incentives to restore kelp forests (Platjouw, 2019; Seddon *et al.*, 2019; Vanderklift *et al.*, 2019).

While there are clear benefits from restoring kelp forests and global interest is accelerating, the path forward is uncertain. This uncertainty is in part because despite similarities in the causes of decline and restoration methodologies, very little information has been shared between projects within and among countries. The most recent analyses provide useful qualitative assessments of past restoration projects, but focus on work published in English-speaking countries and in the peer-reviewed literature (Morris *et al.*, 2020; Layton *et al.*, 2020b). Most restoration projects, however, are not formally published in peer-reviewed journals and occur in non-English speaking countries (Bayraktarov *et al.*, 2020; Eger *et al.*, 2020c). As a result, projects have typically learned and applied methodologies independently. Addressing this limitation will help ensure that lessons learned from 60 to 300 years of history in kelp restoration contribute to a more rapid rate of restoration successes.

(4) Study objectives

This review aims to provide a comprehensive history of kelp forest restoration, assess the current state of the field, and provide recommendations for how this field can advance. We achieve this by reviewing the global history of kelp restoration, analysing past projects, examining the determinants of success, and describing solutions to barriers to future restoration projects. This comprehensive, multi-language project first reviews the history of kelp restoration in independent geographic clusters around the world. Following this qualitative overview, we present the results of a new kelp restoration project database (kelpforestalliance.com) and describe the global state of the field, what factors have resulted in success, and which in failure. Finally, we discuss the methodologies, costs, motivations, and legal frameworks currently related to kelp restoration and how we can enhance the factors that can lead to success in restoration and mitigate potential barriers in future.

II. MATERIALS AND METHODS

(1) Literature searches

To find published literature on kelp restoration, we conducted a search using the *Web of Science* on December 7th, 2018 using the following terms: 'restor* OR rehabilitat* OR green engineering OR ecoengineering OR ecological engineering OR return* OR recov* OR afforest*' AND 'kelp* OR seaweed* OR macroalga* OR Laminariales OR Fucales OR Desmarestiales'. The search returned 1431 results (see online Supporting Information, Appendix S1). We reviewed the titles and abstracts of the returned results and selected 156 publications that appeared to reference a kelp restoration project for additional screening. These 156 publications were reviewed to determine if they met our study's inclusion criteria. These criteria were to identify studies that: (*i*) focused on canopy-forming algae from either the Laminariales, Fucales, or Desmarestiales; and (*ii*) aimed at enhancing kelp ecosystems, *in-situ*, for non-commercial purposes (e.g. not aquaculture or mariculture). Relevant methods included transplanting, seeding, grazer control, installing artificial reefs, and others. Of these initial 156 publications, 51 met our criteria for data extraction. After the first literature search, a publication alert with the same terms was set up to collect relevant new records until March 29th, 2021.

We collected data on both restoration and afforestation projects and tested (see Section II.3) for differences in project success but found none between these two approaches (see Section IV.4). Thus, we combined data from restoration and afforestation approaches in subsequent analyses. Individual projects are specifically referred to as restoration or afforestation, while collective projects (e.g. across a country or across years) are referred to under the umbrella term 'restoration'.

To find kelp restoration projects that may not be in the scientific literature, we conducted similar searches by country or geographic region using English, Spanish, or French search terms as relevant, using the *Google Search* engine with simplified terms to query only "kelp restor*" and a location (e.g. Norway or California; see Appendix S1). We included all countries where kelp is known to occur (Wernberg *et al.*, 2019), and ran searches between 11/10/2019 and

12/12/2019 (Appendix S1). We reviewed between 30 and 100 search results per regional search and compiled a list of groups potentially conducting kelp restoration. We then contacted each group individually to inquire if they could contribute information on their restoration efforts. We supplied each group with a data template for them to complete (Appendix S2).

To find Japanese kelp restoration literature, we conducted an internet search using *JStage* on November 27th, 2019, and returned 616 results, 150 of which were identified for further screening. The search term was磯焼け – the Japanese word *isoyake* – a commonly used term for kelp forest degradation in Japan. A fluent Japanese speaker (M.T.) then reviewed the documents to assess their eligibility. If a paper met the criteria described above, the relevant information was extracted and translated into English. We also translated the database used to inform the second Isoyake Guidelines (Fujita, 2019) and obtained descriptive information about restoration projects. This database was compiled with studies from the Tokyo University of Marine Science and Technology Library and covered the years 1970–2014. Ultimately, the Isoyake Guidelines database contained no information about the outcomes of the restoration projects and our published Japanese literature search found few studies with quantitative or semi-quantitative data. We therefore considered the Japanese studies from a qualitative perspective only and did not use them in the quantitative analyses.

To find Korean kelp restoration literature, we conducted the Korean literature search using *Google Scholar* and *RISS* on November 27th, 2019 and returned 600 results for *Google Scholar* and 60 for *RISS*. The search terms were회복, 복원, 해조류–, the Korean words for 'recovery', 'restoration', and 'marine algae'. A fluent Korean speaker (S.W.H.) then reviewed the papers to assess their eligibility. If the paper met the previously described selection criteria, the relevant information was extracted and translated into English.

(2) Data collection

We extracted data from each paper using the *metaDigitise* package (Pick, Nakagawa & Noble, 2019) in the R programming language (R Core Team, 2019). If the required data were not included in the paper, we contacted the corresponding author to provide any missing information. See the data template (Appendix S2) for the full suite of parameters that were collected.

We used snowball sampling (Biernacki & Waldorf, 1981) in all languages to accumulate contacts for other reports, persons, or groups conducting kelp restoration across the world. We compiled two language-specific project lists using this method in Norway and Chile. A personal contact list is maintained but will not be published for privacy reasons.

Data identifier: we assigned each study a reference number, event number, and an observation number. The reference number was unique to each report or reported project. The event number was unique to a restoration event or action. For example, entries for two artificial reefs contained in the same report but set in different locations would have the same reference number and different event numbers. The same observation number indicated different measurements of the same event, for example, if two species were transplanted together but recorded individually. We used different unique identifiers related to the reference level, event level, and project level when creating the different graphs (Appendix S3).

Cost data: we collected cost information either directly from the publication or report, or through personal communication with the authors. As far as possible, we divided costs into capital, operating, construction, in-kind, and monitoring categories, and recorded the year currency of the value. To allow for accurate cost comparisons between currencies and years, we converted all dollars into USD for the year 2010. First, using the Penn Table (Feenstra, Inklaar & Timmer, 2015), we converted the local currency to USD based on the exchange rates during the year of reporting. Afterwards, we indexed costs for inflation to year 2020 using the Consumer Price Index (The World Bank, 2019). These values only consider the costs of the restoration actions, not of planning or monitoring.

Area extent: while most studies that reported area typically gave only the starting size, when possible, we recorded size (area) as the largest measurement recorded for the project, including expansion after restoration. Therefore, if a study transplanted kelps over 10 m^2 and after monitoring for 2 years discovered the patch had grown to 100 m^2, we recorded 100 m^2 as the area extent. Conversely, if a patch shrank from 10 m^2 to 1 m^2, we recorded 10 m^2 in our database. Methods used to measure area extent differed depending on the study, and included aerial surveys, vessel-based monitoring, and underwater video footage.

Duration: we recorded duration as the day from the first restoration action to the day of the last observation or action recorded. We always used the last available time point to record our data.

Year: we recorded the year in which the first restoration action was initiated, rather than the year of the publication.

Location: we either extracted the geographic coordinates from the reports themselves or obtained approximate coordinates from Google Earth Pro®.

Group involved: we classified the groups involved in the restoration process as being: (*i*) academic (university or research institute); (*ii*) government (municipal, indigenous, state, or federal management body); (*iii*) non-government organisation (NGO; registered non-profit); (*iv*) industry (environmental consultants, aquaculture, energy development); and (*v*) community (organised local group, not registered as non-profit).

Motivation: while reading each report, we searched the text to determine the motivation for each restoration project and classified the primary, secondary, or tertiary motivation into one of the following seven categories (Bayraktarov *et al*., 2019): (*i*) improve restoration approach, technology, methods; (*ii*) restoration after environmental impact (e.g. shipgrounding, mining, oil spill, hurricane); (*iii*) biodiversity enhancement (e.g. native vegetation, habitat creation, ecosystem connectivity, ecological resilience); (*iv*) answer ecological research questions; (*v*) enhance ecosystem services (e.g. fisheries production); (*vi*) biodiversity offset (e.g. threatened species, threatened ecological community); (*vii*) social reasons (e.g. community involvement, job creation, nature education, environmental outreach).

Variables measured: we recorded the project outcomes in several formats (Appendix S2) and several different assessment structures depending on individual project design. Projects were either assessed as the same site over time, a restored site in comparison to a reference site(s), or a restored site in comparison to a degraded site(s). The end variables quantified were area, density, count, growth, survival (1/0), percentage survival, percentage cover, or growth measures. If a project reported on a site over time, we recorded the first measure at the beginning of the project and the last measure as the last available data point.

Success score: the information related to the outcome of the restoration attempt was reported in several different formats using a variety of values (Appendix S2). This mix of reporting standards and units made it difficult uniformly to analyse the success scores. We overcame this issue by using the simplest available metric, a binary survival score. The binary success score was set as 1 if any kelp remained at the time of the last report and 0 if none remained. There were insufficient sample sizes for the other reporting styles (e.g. those with before–after control–impact designs) to conduct additional analyses using these metrics.

(3) Factor analysis

To evaluate the effect of each covariate (fixed effect) on binary success scores, we used generalised linear mixed-effects models with a binomial distribution. Because very few projects had data for all the covariates, we evaluated each factor individually and were therefore not able to evaluate the relative importance of each covariate. We analysed the effects of the following covariates: publication type, to test for publication bias; latitude, to assess the role of biogeography; genus, to determine if some species were easier to restore than others; the method used, to test the efficacy of each method; the area of the project, to see if larger projects were more successful; whether the restored site was in a protected area, to assess potential benefits from that protection; the impacts of disturbances on restoration projects if a disturbance was reported; whether site selection criteria were in place, to see if that selection contributed to success; how close the project was to a kelp bed of the same species, to help determine if natural adjacent populations assisted in population restoration; whether the project specifically mitigated a stressor; the project duration, to see if longer projects were more successful; and whether a project was restoration or afforestation.

To account for multiple data points contained in some reports (Appendix S4), we used mixed-effects models with the study/project reference number as the random effect to account for the correlation between data points in the same study. The generalised mixed-effects models were fitted in R using the *lme4* package (Bates *et al*., 2015) and we used the *lmerTest* package, which applies Satterthwaite's degrees of freedom estimations and the *F*-statistic to assess significance (Kuznetsova, Brockhoff & Christensen, 2017). We then used these models to predict the probability of success using the *predict* function in R, while the error was calculated using the *predictInterval* function in the *merTools* package in R. This function creates a sampling distribution for the fixed and random effects and then draws the range of values from that distribution.

All analyses and graphing were conducted using the R programming language (R Core Team, 2019).

III. REGIONAL HISTORIES OF RESTORATION

(1) Overview of kelp forest restoration

Our review of the history of kelp forest restoration revealed a global field dating back decades to centuries. While many different species have been targeted for restoration, relatively similar approaches to restoration have been developed in each region. Despite their methodological similarities, the social contexts in which restoration has occurred have been very different. To understand these contexts better, we first qualitatively review the regional histories of restoration individually (Fig. 1) and later evaluate the new global restoration database (see Section IV). A few Korean and Japanese projects discussed in the regional review were not captured in the global database because they were not returned in the searches for those regions.

(2) Japan

Japan has the world's longest and richest history of kelp forest management over hundreds of years, including over 700 recorded restoration projects since the 1970s. *Saccharina* species (Kombu in Japanese) are popular food items and are the most commercially important kelp. This genus is found in the cold temperate waters of Japan (Hokkaido to NE Honshu; Fujita, 2011). Starting in the 14th century, Kombu was harvested by Hokkaidoan fishers and exported by ship to central and southern Japan, then later exported to China. The domestic market persists today, and Japan produced 79000 metric tons in 2019 (FishstatJ, 2020). While economically productive, this harvest has previously led to kelp population declines (Fujita, 2011).

The early efforts in Japan fell were both restoration and afforestation. The first recorded restoration project was in 1718 when a monk, Saint Teiden, instructed fishers to throw stones into coralline barrens to encourage kelp regrowth in NW Honshu (Ueda, Iwamoto & Miura, 1963). A local fisher then led a larger afforestation project and installed 317000 stone blocks onto a sandy seabed off SE Hokkaido between 1863 and 1868, increasing his yearly kelp yield from 7 to

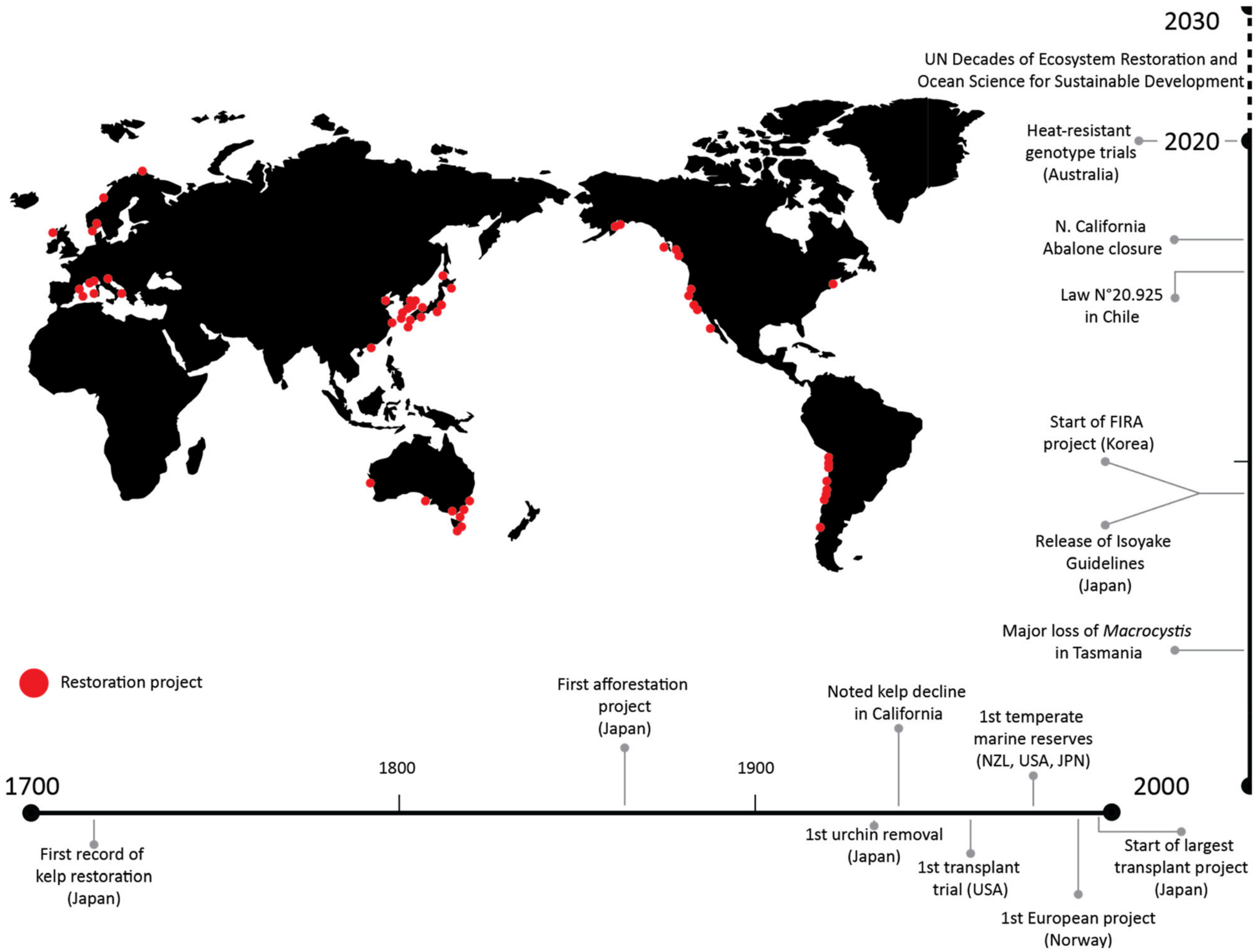

Fig. 1. Location and timeline of important global kelp restoration-related events.

20 tons (Ueda *et al*., 1963). Thereafter, afforestation *via* reef construction (*tsuki-iso*) became increasingly common in northern Japan and an additional 300 ha of reefs were installed from 1921 to 1950 (Kuroda *et al*., 1957). While these efforts were extensive, they were not always successful, and sedimentation commonly led to restoration failure (Kinoshita, 1947). The second common method to enhance kelp populations during this time was the clearing of competitors such as turf algae from the benthos, either by hand or with mechanical scrapers (170 ha from 1921 to 1950; Kuroda *et al*., 1957).

Fishers in NW Hokkaido also noticed that sea urchins would graze on their kelp stocks and began to remove urchins to protect the kelp. A local cooperative first realised these 'pests' could be of potential value and started to purchase the removed urchins, process them, and ship them to Honshu (main island of Japan) in 1932 (Kinoshita, 1947). The demand for Kombu as a food and as a feedstock continued to increase and more structured fisheries management systems formed in the 1950s and 1960s (Fujita, 2011). National and prefectural governments continued to focus on deploying artificial reefs, now using manufactured concrete blocks (Tokuda *et al*., 1994). Concurrently, the urchin-culling efforts also expanded to NE Honshu and SW Hokkaido, as did clearing the benthos of competitors (Fujita, Machiguchi & Kuwahara, 2008a). Sea urchin removal and artificial reef placements have had few changes to their approaches. Scraping the benthos, however, has advanced to include chains moved by wave action, boat-operated rotators, and even remotely controlled underwater excavators (Japanese Fisheries Agency, 2021).

Restoration attempts for *Ecklonia* and *Eisenia* species in Japan's warmer central and southern waters started in the 1980s (Arai, 2003). These genera are eaten locally by people and are an important habitat for abalone and lobster populations that support major coastal fisheries in Japan. In contrast to northern Japan, these restoration efforts have focused on transplantation and grazer control of not only urchins, but also herbivorous fishes [*Siganus fuscescens*, *Calotomus japonicus*, *Kyphosus* spp. (Fujita, Noda & Kuwahara, 2008b; Fujita, 2010)]. Managers in NE Kyushu (southernmost main island) repeatedly found that consistent removal of these grazers was key to kelp restoration success, as short-term control using cages or gillnets would result in a period of kelp regrowth, but eventually failed when managers removed the cages and the herbivores ate the transplants (Fujita, 2011).

These lessons were all applied in what is now the largest successful kelp restoration project in Japan. Starting in 1999, the Shizuoka Prefectural Government placed small concrete blocks in healthy kelp forests, allowed spores to settle on them, and then transported them to barrens to restore *Ecklonia* forests in a deforested area (Izu Peninsula, central-east Japan; Eger *et al.*, 2020c). Local fisheries cooperatives, municipal, and prefectural government groups joined these actions for a second phase that ran from 2002 to 2010. As of 2018, ~870 ha of *Ecklonia* has been restored, leading to such a marked recovery of abalone populations that managers are considering the re-opening of a closed abalone fishery (Eger *et al.*, 2020c).

Given the numerous projects conducted in Japan, there have been many opportunities to learn from their outcomes. Indeed, these efforts were reviewed in 2009, 2015, and 2021 by the federal Fisheries Agency to provide detailed guidelines for future projects. The Isoyake Taisaku Guidelines (Japanese Fisheries Agency, 2009, 2015, 2021) were launched alongside a funding initiative to promote reforestation of algae forests. This initiative, known as the Fisheries Multiple-function Demonstration Project (FMDP), operated from 2009 to present and funds fishing cooperatives and NGOs to control herbivores, transplant kelp, maintain herbivore exclusions, clear the benthos, remove sediments, and improve upstream water quality (Sekine, 2015). The national government provides half the required funds, the prefectural government provides a quarter, and applicants fund the last quarter (Sekine, 2015). In addition to funding, the project provides access to experts to guide the restoration process. Approximately 300 thousand yen (~\$2540 USD 2010) per hectare is invested in this process. Despite 288 groups accessing the funds and support, <100 ha of algae has been restored since its inception (Y. Sekine, personal communication). The limited success of this initiative has been attributed to increased herbivory, increased water temperatures, reduced nutrients, increased frequency and strength of typhoons and flooding, increasingly armoured and industrialised coastlines, and the end of project funding (Fujita, 2019).

(3) Korea

The Korean peninsula is bounded by three seas and Korea has a long history as a maritime nation that harvests fish, invertebrates, and seaweeds. The decline of over 10000 ha of seaweed forests during the 20th century (Sondak & Chung, 2015) has put this relationship at risk. Following the Korean War (1953), the South Korean government has worked to increase the availability and access to the marine resources within their own Exclusive Economic Zone (EEZ). Their management strategies focus on modifying the ocean with artificial materials while also working to enhance the biomass of harvestable species (Sánchez-Velasco, Oriol & Valiente, 2020). Construction of these artificial reefs started in 1971 and was targeted at enhancing coastal fisheries in depths of 20–40 m. Under this initiative, the installation of eight different types of reefs continued until 1990 with a sum cost of \$61 million USD (FIRA, 2020).

These reefs gave rise to the concept of marine ranching, which cultures species in the ocean for consumption. A pilot ranching project took place from 1982 to 1989 and resulted in the Near-shore fisheries Marine Ranching Master plan in 1994 (Park *et al.*, 1995). The National Institute of Fisheries Science (NIFS) ran this program from 1998 to 2010 and worked to enhance fisheries and create or restore kelp forests in multiple areas along the Korean coastline. NIFS worked with kelp genera that were amenable to cultivation, focusing on dasima (*Saccharina japonica* C.E. Lane, C. Mayes, Druehl & G.W. Saunders), *Ecklonia* spp., miyeok (*Undaria pinnatifida* Suringar), and *Sargassum* spp. Once the kelps were successfully cultivated, they were typically transplanted on the artificial reefs using ropes containing juveniles or seeded using spore bags (Park *et al.*, 2019).

Following the initial NIFS projects, the Korea Fisheries Resource Agency (FIRA) was established in 2009 and took over marine ranching, kelp restoration, and afforestation projects in Korea. This date marked the start of the world's largest kelp forest afforestation and restoration program. The project is running until 2030 with a yearly budget of \$29 million USD (2019) (FIRA, 2020) and aims to create or restore 50000 ha of kelp forests, with >20000 ha installed at 173 sites as of 2019 (Lee, 2019).

FIRA initially followed similar protocols as previous work, using transplants or seeds on artificial reefs. However, they are now focusing on urchin control and the best ways to restore kelp on rocky reefs that once supported kelp forests (Yang *et al.*, 2019). The projects in Korea have been led largely by the federal government with considerable input from local universities, which research different restoration techniques, provide historical baselines and targets, and advise ongoing management efforts (Hong *et al.*, 2021). For the foreseeable future it appears that most kelp restoration work in Korea will occur under the FIRA program with input from university researchers. Although community groups do not themselves work to restore kelp forests in Korea, the government projects are generally well supported by Koreans, who are indeed seafood and seaweed lovers (Han, 2010). In some instances, projects were initiated in response to public pressure (Kang, 2018). Within Korea, there are seaweed festivals and even a day known as 'Marine Gardening Day' which celebrates the ties between people and the ocean, and encourages responsible stewardship and restoration of the sea.

(4) USA

Kelp in southern California, notably giant kelp (*Macrocystis pyrifera*, C. Agardh, herein *Macrocystis*), has been an important source of materials such as alginates, potash, and acetone since the early 1900s (Barksy *et al.*, 2003), and has an extended management history. When kelp populations declined due to poor water quality and overharvesting

(Wilson, Haaker & Hanan, 1977), the first restoration trials were motivated by a desire to restore these resources. The first recorded North American trials transplanted *Macrocystis* in southern California in 1958 (North, 1958). These efforts were soon combined with the manual or chemically induced mortality of grazing fishes and urchins (Wilson & North, 1983).

Academics, fishery managers, and industry groups soon led repeated initiatives to restore *Macrocystis* with transplants, seeding, and urchin culling during the 1960s and 1970s (Wilson *et al*., 1977; Wilson & North, 1983). Most commonly, projects succeeded in restoring tens to hundreds of hectares of kelp while others failed due to heatwaves, urchin incursions, or storms (Wilson *et al*., 1977; Wilson & North, 1983); following these efforts, the number of projects remained low until after the year 2000. During this decade, several community groups, notably those under the banner of the California Coast Keepers organisation, became interested in restoring their local marine environment. Noticing correlations between increased urchins and decreased kelp forests, these groups led initiatives to remove urchins and transplant kelp individuals (House *et al*., 2018; Williams *et al*., 2021).

Afforestation through the installation of artificial reefs has been of notable interest in California. Early attempts used available materials (e.g. disused trams) to establish kelp forests (Carlisle, Turner & Ebert, 1964), but later developed into more robust strategies using rocky materials. In an attempt to increase the stock of sport fish during the mid-1980s and early 1990s, the California Department of Fish and Wildlife (CDFW; Carter *et al*., 1985) created a series of artificial reefs throughout California. Later in the 1990s, the California government mandated installation of what is now a 172 ha artificial reef to offset a *Macrocystis* forest that was destroyed by warm water outflow from a nuclear power plant (Reed *et al*., 2006). Similarly, municipal governments in Seattle, Washington, and Vancouver, British Columbia, have led efforts to build new reefs to offset industrial projects which destroyed kelp forest habitat (Cheney *et al*., 1994; Fehr, Thompson & Barron, 2011).

In northern California, recent restoration efforts for bull kelp, *Nereocystis luetkeana*, have ensued due to rapid and extensive losses (McHugh, Abbott & Freiwald, 2018; Hohman *et al*., 2019). In just under a decade, multiple stressors, such as the loss of apex predators, high urchin grazer recruitment, and prolonged warm water events have resulted in a net loss of >95% of *N. luetkeana* forests, and subsequent lack of recovery, along 350 km of coastline in just under a decade (Rogers-Bennett & Catton, 2019; McPherson *et al*., 2021). The kelp forest collapse negatively impacted ecosystem, economic, and social health of northern California coastal communities.

As a consequence, interest is growing in California ocean users to safeguard the iconic and vitally important kelp forest ecosystem *via* monitoring, and if appropriate, through restorative actions. Further, California policy makers plan to develop comprehensive ecosystem-based management and restoration strategies moving forward to protect coastal and marine biodiversity and ensure the continued delivery of ecosystem services (Ocean Protection Council, 2021). The involvement of the State has provided fiscal, regulatory, and institutional support for research and pilot kelp restoration projects being led by key community members, NGOs (e.g. Reef Check California, Greater Farallones Association, and The Nature Conservancy) and academics (Ocean Protection Council, 2021). Some of the projects currently being explored in northern California include: developing regulatory pathways and methods to reduce urchin grazing pressure through recreational and commercial diver efforts; using occupied and unoccupied aircraft imagery to understand *N. luetkeana* canopy coverage over time; evaluating a variety of *N. luetkeana* culturing and out-planting procedures, leveraging conservation genomics and gametophyte banking to preserve the genetic diversity of *N. luetkeana*; investigating the dynamics of urchin recruitment and reproduction; kelp farming; developing *N. luetkeana* spore dispersal models; exploring the feasibility of predator (sunflower sea star *Pycnopodia helianthoides*) restoration; and outreach and education (Ocean Protection Council, 2021). An increase in the frequency and duration of conditions that are stressful to kelp will likely result in localised and regional future kelp forest degradation, reinforcing the necessity for developing climate-resilient solutions to ensure ecosystem health (Hohman *et al*., 2019; Gleason *et al*., 2021).

Elsewhere, kelp restoration efforts in Washington and Oregon are now emerging through groups such as The Northwest Straits Commission (nwstraits.org/our-work/kelp-recovery), the Oregon Kelp Alliance (oregonkelp.com), and the Elakha Alliance (elakhaalliance.org/). These groups are trialling and exploring transplantation, urchin culling, and sea otter reintroduction as restoration strategies.

(5) Canada

Kelp restoration projects have taken place on a limited scale in recent decades in British Columbia (BC), although the anticipated negative impacts of climate change (Krumhansl, Bergman & Salomon, 2017) and urchin barrens have increased interest in the subject. In response to extensive urchin barrens limiting kelp distribution, the A-Tlegay Fisheries Society, Gwaii Haanas National Park Reserve, National Marine Conservation Area Reserve, and Haida Heritage Site (hereafter Gwaii Haanas; cooperatively managed by the Haida Nation and Government of Canada), and the Pacific Urchin Harvesters Association are trialling increased quotas and/or opening closed areas for commercial fishing of red sea urchins (*Mesocentrotus franciscanus*; Department of Fisheries and Oceans Canada, 2020). Elsewhere, interest is growing in restoring or farming kelp as a climate solution on Vancouver Island (Ocean Wise Seaforestation Initiative – ocean.org). Prior small-scale *Nereocystis* restoration projects have taken place in southern BC (similar for northern Washington State), focused on seeding to start new populations in response to general declines (Heath, Zielinski & Zielinski, 2015).

In Gwaii Haanas in northern BC, cooperative management partners – Council of the Haida Nation, Parks Canada, and Fisheries and Oceans Canada – initiated a larger-scale kelp forest restoration project over 20 ha of shallow subtidal rocky reef (Lee *et al.*, 2021). This work was motivated to restore ecosystem balance by mimicking sea otter predation (historically extirpated; see Bodkin, 2015) on urchins where sea otters have not yet returned. Restoration work was initiated in 2018–19 with pre- and post-restoration monitoring and research funding over 5 years. This project involves close collaborations among Gwaii Haanas management partners as well as the commercial urchin fishing industry and multiple academic institutions. Due to this diverse partnership and engagement with Haida Gwaii communities, cultural and social considerations are as important to the project as ecological gains (Lee *et al.*, 2021). Provision of urchin roe for food in the communities, working with Haida divers in monitoring and research, as well as employing Haida and commercial divers to remove, crush and maintain low urchin densities at the sites, are all key components of the project.

(6) Australia

The focus on kelp restoration in Australia is recent, and efforts have focused on urchin culling and/or removal in *E. radiata* forests, on restoring giant kelp (*Macrocystis* spp.) populations in Tasmania, or on restoring the locally extinct fucoid crayweed (*P. comosa*). Urchin removals have most often been done by abalone and urchin fishery organisations that are working to restore kelp habitat and create more biomass of abalone and/or urchin in the states of New South Wales, Victoria, South Australia, and Tasmania (Worthington & Blount, 2003; Gorfine *et al.*, 2012). The Tasmanian government subsidises the local urchin fishery to remove invasive urchins, which have expanded their range south from continental Australia (Ling *et al.*, 2009), including for urchins that might not otherwise be profitable to harvest (Larby, 2020).

There have been three main efforts to restore specific taxa *via* transplantation in Australia. First, SeaCare Inc. installed small patches of *Macrocystis* in Tasmania from 1997 to 2001. However, the efforts were not sustained and they did not achieve long-term success (Sanderson, 2003). While currently in early development, researchers from the University of Tasmania are working to select thermally tolerant kelp from the remnant populations of *Macrocystis* and are trialling outplants back into the ocean (Layton & Johnson, 2021). The other main project is Operation Crayweed which has been working since 2011 to restore *P. comosa* and associated biota along the coast of the Sydney metropolitan area (Campbell *et al.*, 2014; Marzinelli *et al.*, 2016). Operation Crayweed is notable for their work with community groups, schools, and artists to connect people to their restoration projects (Vergés *et al.*, 2020), as well as their work on genetic mixing of transplant populations and the identification of genotypes for future-proofing against climate change (Wood *et al.*, 2021).

(7) Europe

Kelp populations inhabit the coastlines of ~20 countries in the Europe–Mediterranean region, with records of kelp restoration focused on Norway, Spain, and Italy.

In Norway, urchin grazing has been a major driver of kelp declines since at least the 1970s (Sivertsen, 1997; Norderhaug & Christie, 2009). As an experimental study, scientific divers crushed urchins with hammers over 10 diver-days in Central Norway in 1988. While the reduction in urchins allowed the canopy (mainly sugar kelp, *Saccharina latissima* Druehl & G.W. Saunders) to recover rapidly (Leinaas & Christie, 1996) and subsist for almost a decade, later surveys showed the urchins had returned and the kelp disappeared (Norderhaug & Christie, 2009). Following these initial trials, researchers remained interested in restoration but government bodies did not fund further projects due to perceived challenges and lack of interest. Kelp restoration work was not initiated again until 2003 when the 'Sugar Kelp Project' (2003–2008) trialled different small-scale methods, including scraping the benthos to remove competitors, transplanting adult and juvenile kelp on either hard substrate or ropes, and seeding (Moy *et al.*, 2008; Moy & Christie, 2012).

Although the project failed when turf algae outcompeted the kelps, this project marked the start of renewed interest by the Norwegian Institute for Water Resources (NIVA) and similar groups to restore kelp. NIVA then trialled artificial reefs in northern Norway in 2006 which was successful over a five-year period, but ultimately failed as urchins overgrazed the kelps (Christie *et al.*, 2019a). In 2011–18, both NIVA and the Institute of Marine Research (IMR) tested various restoration techniques, focused on either manually crushing and excluding urchins, outplanting or transplanting *Saccharina* and *Laminaria* (Fraschetti *et al.*, 2017; Fredriksen *et al.*, 2020) and chemically killing urchins using quicklime (Strand *et al.*, 2020). The fast-recovering species in these studies were *S. latissima*, *Alaria esculenta* and the arctic *Saccorhiza dermatodea*. The quicklime efforts are notable because they had lower co-mortality rates than the previous quicklime projects in the early 1960s in California (Wilson & North, 1983) and 1980s in eastern Canada (Bernstein & Welsford, 1982). Recently, researchers and entrepreneurs are collaborating to develop market-based solutions to overabundances of urchins. Starting with a small-scale pilot project in 2018–19, NIVA, a business (Urchinomics®), and a community group (www.tarevoktere.org) have been exploring either directly harvesting urchins or collecting them, transporting them on land, and growing them for the food market (Verbeek *et al.*, 2021).

Interestingly, natural recovery of *L. hyperborea* and *S. latissima* populations in mid-Norway have been occurring without any intervention over the last couple of decades (Fagerli, Norderhaug & Christie, 2013). Increases in sea surface temperature reduced the survivorship of the green urchin (*Strongylocentrotus droebachiensis*) and facilitated the expansion of the edible predatory crab (*Cancer pagarus*), which has reduced urchin populations (Christie *et al.*, 2019b). Neither of these actions was intentional but they demonstrated that novel warmer conditions may enhance kelp recovery and/or restoration in some higher latitude reefs (Filbee-Dexter *et al.*, 2019), while they may impede restoration and accelerate declines at lower latitudes (Vergés *et al.*, 2016; Wernberg *et al.*, 2016a; Qiu *et al.*, 2019).

Restoration of kelp in the Mediterranean has largely focused on the fucoid genus *Cystoseira*. Anthropogenic pressures in the Mediterranean basin are intense with a long and sustained history of coastal development (Gibson, Atkinson & Gordon, 2007). As a result, populations of *Cystoseira* have declined throughout the region (Thibaut *et al.*, 2005). Universities and research institutes, primarily in Italy and Spain, worked on the initial restoration efforts. These projects focused on trialling small-scale culturing and outplanting (Verdura *et al.*, 2018; De La Fuente *et al.*, 2019; Tamburello *et al.*, 2019), and have also considered urchin removal, which was identified as a barrier to success (Guarnieri *et al.*, 2014). Following these initial trials, the Marine Ecosystem Restoration in Changing European Seas (MERCES) project was created with European Union funding and ran from 2016 to 2020 (Fabbrizzi *et al.*, 2020). This project included kelps among other marine habitats and expanded the scope of past restoration efforts; it has trialled methods to outplant *Cystoseira* in Italy, Albania, Tunisia, and Spain (Iveša, Djakovac & Devescovi, 2016; MERCES, 2020).

(8) Chile

Macrocystis and *Lessonia* are foundational species along the Chilean coastline and are important commodities and habitats for fisheries species. Wild harvest of *Macrocystis* has a long history in Chile and is now one of the few remaining wild kelp harvests in the world (Buschmann *et al.*, 2014). The fishery annually harvests 400000 dry tonnes and provides 10% of the world's alginate (Buschmann *et al.*, 2014). This harvest has reduced portions of the wild kelp populations with an associated reduction in ecosystem services, currently valued at $54 USD million (Vásquez *et al.*, 2014a). To help manage the diminishing populations, the federal government established a management program (Law N°20.925) that provided funds to encourage the cultivation as well as restoration of seaweeds (Biblioteca del Congresso Nacional de Chile, 2020). The primary focus of projects stemming from the program has been the long-line cultivation of *Macrocystis* with less work on restoring either genera or cultivating *Lessonia*.

Lessonia restoration projects in Chile are often supported by regional or national funding agencies. The projects are typically partnerships between academia and fishery cooperatives, and usually work with transplants. Transplantation methods include attaching juvenile plants onto existing holdfasts (Westermeier *et al.*, 2016), or adding mature plants to artificial substrates, which are then secured onto the benthos (Correa *et al.*, 2006). Although these projects have demonstrated that transplants can indeed survive and grow, considerable variation was shown in the density, biomass, and length of plants among projects, both by methodology and planting season.

Lessonia restoration projects have had limited success in Chile. The first restoration attempts for *L. berteroana* occurred in response to increased herbivory and enhanced El Niño–Southern Oscillation (ENSO) cycles in 1990 (Vásquez & Tala, 1995). These projects combined the outplanting of spores, juveniles, and reproductive adults fixed to the substrate using epoxy and anchored boulders (Vásquez & Tala, 1995; Correa *et al.*, 2006; Westermeier *et al.*, 2016). Early survival rates for these methods averaged around 50% and plants showed similar growth rates to natural populations. However, the projects were only maintained over short timescales and small spatial extents.

Building off this work, researchers are now testing whether increasing genetic diversity can increase restoration success rates. Researchers are grafting plants together, creating chimeric individuals of *L. berteroana* (Montagne) and *L. spicata* (Santelices), and transplanting them over larger areas than previously attempted. As a result, the transplanted individuals have the DNA of the two donor plants, ideally improving tolerance to stressors such as temperature. The work has been patented (Patent CL201701827) and conducted in collaboration with three universities, government funds, and a private company. If this method is successful, it will be an important step in Chilean kelp restoration, as local kelp forests are vulnerable to physiological stress caused by warmer sea temperatures (Vásquez *et al.*, 2014a).

IV. ANALYSIS OF THE GLOBAL DATABASE

(1) Overview

Our database (Appendix S4) collated 259 kelp restoration and afforestation efforts that provide quantitative insights into the characteristics of restoration projects and determinants of success. Recorded projects started in 1957 and the number of projects per decade has consistently increased since then (Fig. 2; Appendix S4; Eger *et al.*, 2020b). Of these projects, most work has been done in Japan and the USA, particularly California (Fig. 1). As a result, efforts have been focused on the restoration or afforestation of the genera within these countries (*Macrocystis* in USA and *Laminaria* spp. in Japan; Figs 2 and 4). While projects occurred in 12 other countries, many countries have no recorded restoration or afforestation projects (Fig. 1). Notably, the UK, Ireland, France, Russia, Iceland, and China have significant kelp populations and management histories, but we found no recorded projects. This result suggests that restoration and/or afforestation is not a focus in these countries, that local actors do not prioritise the restoration of kelp ecosystems, or that information regarding restoration efforts is difficult to access. Given that restoration projects have not been conducted in many countries with kelp habitat, it is not surprising that kelp restoration projects are less common than those for other marine habitats (Saunders *et al.*, 2020).

(2) Groups involved in restoration

Academic researchers have been most commonly involved in kelp ecosystem restoration (Appendix S5). Relatively few

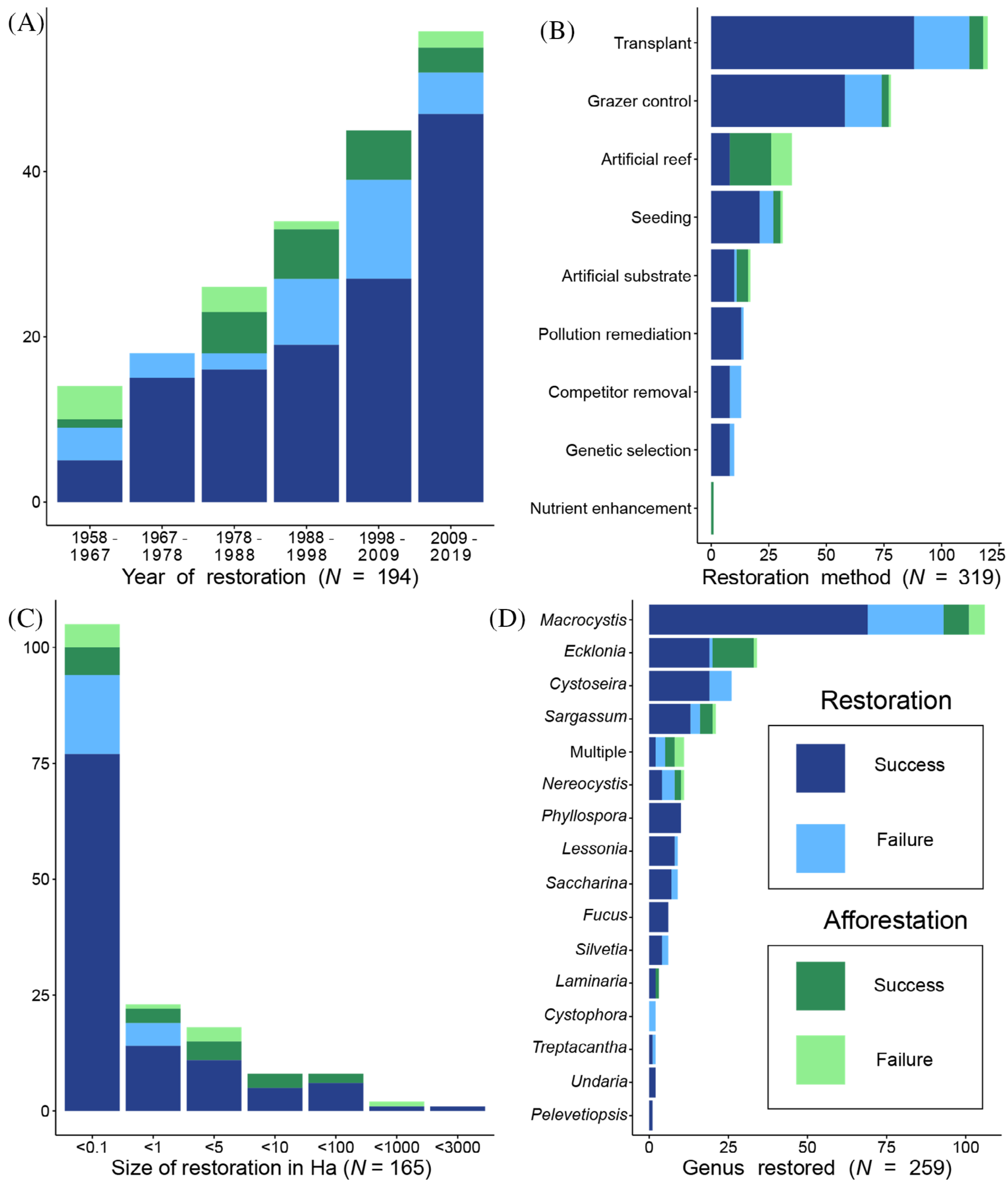

Fig. 2. Descriptive results showing ecological success (darker shade) or failure (lighter shade) of kelp restoration (blue) and afforestation (green) projects completed to date (*N*) by: (A) year the restoration project commenced; (B) main method used for restoration; (C) size of restoration project; and (D) genus restored. Full details of the included studies are provided in Appendix S4.

projects outside of Japan and Korea have been led by governments, NGOs, industry, or community groups. This imbalance perhaps reflects the nascent nature of kelp restoration as practitioners are still working to research and refine methodologies as opposed to attempting restoration on a large scale (Appendix S6). Further, restoration projects are currently expensive (see Section VI.1) and these costs may preclude large-scale restoration initiatives (Eger *et al*., 2020c). While there are some partnerships between academic restoration practitioners and other sectors of society (such as the Gwaii Haanas initiative; Lee *et al*., 2021), they are less common in the English-speaking world. Bridging this gap will be important for future restoration efforts. Academics can provide scientific knowledge on kelp ecosystem ecology and advice on methodology whereas other sectors can provide local and ecological knowledge, funding, social license, and the people required to complete the work at scale (Eger *et al*., 2020c; Lee *et al*., 2021). Such partnerships are already common in Japan and Korea, and it may be beneficial to replicate them elsewhere.

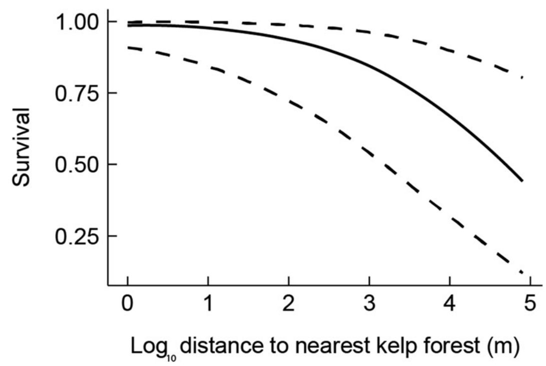

Fig. 3. Relationship between kelp survival and project proximity to an existing kelp forest including the same species.

(3) Project size

Perhaps because most restoration efforts have been experiments by academics, we found that 78% of projects were less than 1 ha in size (Fig. 2C). Only 37 projects attempted kelp restoration at areas greater than 1 ha, and only three of these were greater than 100 ha. Of those 37 projects, 13 were afforestation projects. The one recorded afforestation project >100 ha failed, and thus most of the few large-scale project successes are from restoration projects. Note that the FIRA afforestation collective project is not recorded as a line entry in the database. Tellingly, the main motivation for restoration was to improve methodologies (41% of recorded responses; Appendix S6). In our data compilation we also recorded the largest area of kelp forest achieved for each project (e.g. a project that planted 100 m^2 of kelp forest which subsequently shrank to 10 m^2 was recorded as 100 m^2; see Section II.2) and therefore the area size in our database may be an overestimation in some cases.

These findings show that kelp restoration remains an emerging field that has mostly focused on experimental and theoretical approaches to restoration. We anticipate this status will change as interest in kelp restoration grows, providing the opportunity to use information gained from the previous

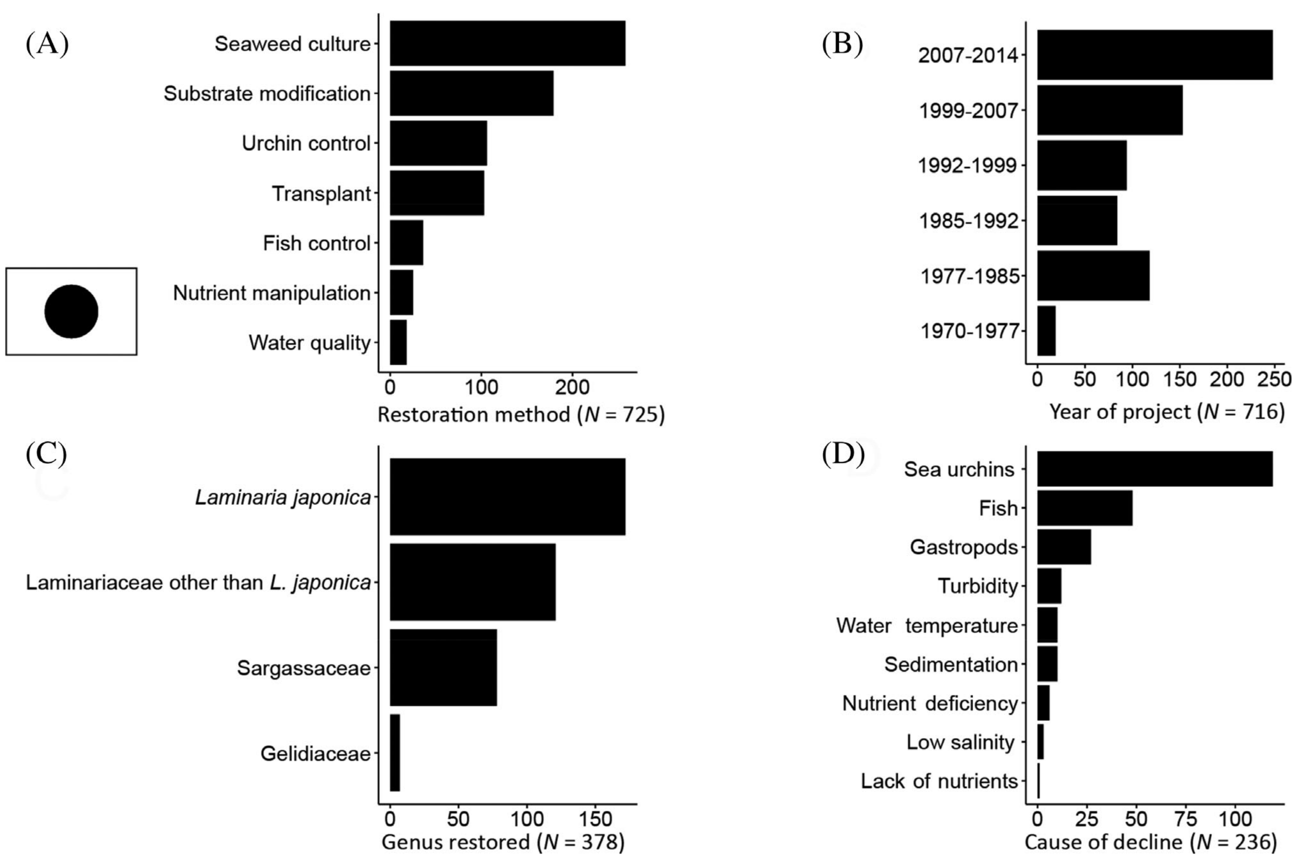

Fig. 4. Descriptive results of projects identified in the Japanese literature search: (A) main method used for restoration; (B) year the restoration project commenced; (C) taxon restored; and (D) initial cause of decline. No information about project outcomes was available. Sample sizes differ as not all data were recorded for each entry.

small-scale projects to inform the larger-scale ecosystem restoration projects expected in the future.

(4) Proximity to other kelp forests improves project success

We examined the factors affecting the survival of a kelp population at the end of the monitoring period using generalised linear mixed-effects models (see Section III.3). The best predictor of project success was the site's proximity to an existing kelp population (Fig. 3), suggesting that this is a key factor to consider in future restoration efforts. The only other significant predictor was whether there was a disturbance during the project, with success being less common following such events (e.g. heat wave, pollution, urchin ingression). The other covariates including kelp genus restored, year the project was conducted, project size, afforestation *versus* restoration, or the primary method of restoration did not significantly predict success. When more consistent metrics are available for projects in future, a more detailed multivariate assessment of success and varying definitions of success may yield different results.

The significant result suggests that restoration projects may benefit from a supply of propagules from nearby populations, suitable environmental conditions for restoration, and/or existing populations that facilitate the establishment and survival of new generations (Eger *et al*., 2020a). Notably, this finding is consistent at the regional level: projects that restored kelp at an ecologically meaningful scale were in locations where kelp had declined but not disappeared. For example, the large-scale FIRA afforestation project in Korea has created ~18000 ha of kelp through a combination of artificial reefs, transplants, and seeding, where kelp decline has been recorded at 10–30% (FIRA, 2020). Although significant, the decline in Korea is much less than the 90–95% declines reported in Tasmania and northern California. Other large-scale projects have shown similar patterns: successful restoration projects in eastern Japan, northern Norway, and southern California have all been in regions with remnant kelp populations (Eger *et al*., 2020c). Conversely, restoration projects in the Mediterranean, Australia, and northern California without substantial healthy populations of the target species nearby have not resulted in large-scale success to date. Of interest, while not a restoration project, there has been a recent rapid unassisted recovery of kelp species in Norway following large-scale declines (Leinaas & Christie, 1996; Christie *et al*., 2019a; Strand *et al*., 2020).

Future projects that work to restore areas near existing kelp populations of the target species, or of other co-occurring species which may facilitate recruitment (Eger *et al*., 2020a), or that work to enhance existing kelp populations before they decline (Coleman *et al*., 2020), may be more likely to succeed in restoring kelp. Past work has shown that once a kelp bed has shifted to an alternative state, it is difficult to reverse that shift (Filbee-Dexter & Wernberg, 2018). Accordingly, enhancing declining but existing kelp populations maybe the most cost-effective approach and should be prioritised in future management plans. Managers could achieve this goal, for example, by managing urchin populations before they cause barrens, or by transplanting or seeding kelp into or directly adjacent to existing kelp forests. In scenarios where kelp restoration is desired but no nearby populations exist, projects may be more likely to succeed if multiple areas are restored to support each other, or a single larger area is restored that can become self-sustaining. Such spatial approaches are already common in the design of MPA networks (Palumbi, 2003; Almany *et al*., 2009) and could be applied to restoration.

(5) Environmental barriers to restoration success

Across projects, we found several recurring ecological issues that prevented the long-term success of kelp restoration. The most common barrier to restoration success was the incursion of grazing species such as sea urchins and herbivorous fishes. Grazing by urchins has hampered restoration projects in Norway, California, Australia, Japan, and Korea (Wilson & North, 1983; Fujita, 2019; Layton *et al*., 2020b). While fish grazing is a less common barrier globally, it has been problematic in Australia, Japan, and Korea (Lee *et al*., 2014; Yoon, Sun & Chung, 2014; Vergés *et al*., 2020). Sedimentation and water pollution has caused problems in southern California and Washington in the USA, and Japan and Korea (Wilson & North, 1983; Carney *et al*., 2005; Kang, 2010; Fujita, 2011). Finally, extreme events such as storms, consistently warmer sea temperatures, and marine heat waves have caused transplants to die in southern California, Chile, and Australia (Wilson & North, 1983; Camus, 1994; Sanderson, 2003; Wernberg *et al*., 2016b). Finding ways to mitigate these barriers to success will be key to progressing the field of kelp restoration. Social barriers to restoration are not discussed in full in this review but see Section VI.

(6) Ecological success in kelp forest restoration

Defining and predicting ecological success in ecosystem restoration projects is a consistent challenge and one that we encountered in our analysis. None of the categorical variables (genus, year, project size, restoration group, duration) were significant predictors of restoration success. The predictive ability of these models may become more resolved as more nuanced metrics are success are used, as opposed to the binary categories we used herein.

Indeed, the high proportion of studies with successful outcomes (Fig. 2) masks the fact that most projects have been very small scale and do not correspond to the scale of previous and on-going degradation. Therefore, while percentage survival of kelp is a potential metric to define success, it may be misleading. Other analyses (van Katwijk *et al*., 2016) have attempted to overcome such barriers by creating subjective metrics of success, or 'success scores', but are limited by qualitative cut offs and confound different variables by combining factors such as survival, size, and project duration, while typically ignoring the specific goals of each project. A potential solution is using effect sizes from replicated, before–after control–impact research frameworks where goals are clearly defined (Underwood, 1992). However, very few studies in our synthesis used these designs and thus we were unable to use such effect sizes for analysis. For this field to progress further, future projects should include rigorous measurements of outcome and attempts should be made to standardise recording approaches across projects.

(7) Kelp restoration in Japan: a qualitative assessment

The Japanese literature database lacks quantitative information on restoration outcomes but provides insights into the state of restoration within the country (Fig. 4). Restoration and afforestation work in Japan focused on culturing programs, modifying the substrate with artificial materials, controlling sea urchins, and transplanting kelp (Fig. 4A). Several projects have also experimented with controlling grazing fish populations, a method that is not commonly used elsewhere in the world. Restoration in Japan (in addition to Korea) therefore appears to use more manipulative techniques than elsewhere in the world. Most projects outside of Japan relied on wild harvest of kelp plants, whereas in Japan culture or breeding programs provided source plants, likely linked to the fact that Japan is one of the largest producers of seaweed in the world (Nayar & Bott, 2014) and can adapt seaweed farming technology. Similarly, it appears much more common for projects to deploy artificial substrates in Japan (Tokuda *et al.*, 1994), a practice that while also common in Korea, is often opposed in other countries (Thierry, 1988; Tickell, Sáenz-Arroyo & Milner-Gulland, 2019). The Japanese coastline is heavily urbanised and artificial reefs are often used to offset these developments. As elsewhere (Benabou, 2014), offsetting practices may not truly replace the biodiversity that has been lost and may give license to further detrimental development.

Restoration projects increased between 2007 and 2014 (Fig. 4B), likely in response to the government program for incentivising restoration (Fujita, 2019). The most common cause of decline was grazing by sea urchins and fishes while increased water temperatures, sedimentation, nutrient deficiencies, and low salinity were also responsible for kelp decline in the database (Fig. 4D). The greatest number of projects were conducted in Hokkaido (Appendix S7), perhaps reflecting its large size and also its long history of marine and kelp management. Across the rest of the country, most areas had a similar number of restoration projects. Kelp restoration in Japan appears to have a globally unique trajectory where, in addition to having conducted the most restoration projects of any country, many different species and methods have been trialled. Given this broad experience, Japan can provide many lessons about the positive and negative aspects of different restoration techniques, including those less-commonly practiced elsewhere around the world, such as the culture of kelps for restoration, fish control, and substrate manipulation.

V. RESTORATION METHODOLOGIES

We found four main methods used actively to restore kelp populations: transplanting, seeding, grazer control, and artificial reefs (Figs 2B and 5), with the choice of method largely dictated by the cause of decline. Since the 20th century, the premise behind each method has not substantially changed but our review revealed different lessons learned from each method.

(1) Transplanting

Transplanting kelp typically involves adhering the holdfast to some artificial material and then adding that to the sea floor with the intention that the holdfast migrates to the benthos or the plant acts as a seed source and provides a suitable environment for new plants. Restorationists have trialled many different methods, including gluing holdfasts to the rock (Susini *et al.*, 2007), attaching them to small concrete blocks or stones (Oyamada *et al.*, 2008; Fredriksen *et al.*, 2020), tying them to ropes (North, 1976), attaching them to existing holdfasts (Hernandez-Carmona *et al.*, 2000), and attaching them to mesh mats, themselves anchored to the seafloor (Campbell *et al.*, 2014) or to artificial substrata (Marzinelli *et al.*, 2009).

The key limitation with each of these techniques is scalability and how well the plant can attach to the seafloor. Physical transplantation of kelp is a laborious process and manual installation will likely prove cost prohibitive for large-scale restoration projects. A new method termed 'green gravel' is being developed that reduces deployment time by removing the need for divers and increases scalability by using laboratory-cultured gametophytes that are attached to small stones (i.e. gravel), grown in the laboratory and then dispersed into the ocean (Fredriksen *et al.*, 2020). The method has demonstrated some success and a working group (greengravel.org) is trialling the approach in new locations and conditions (e.g. high wave-exposure sites). The benefit of transplanting is that it immediately introduces plants into the environment and these plants can create conditions more suitable for new recruits (Layton *et al.*, 2019; Japanese Fisheries Agency, 2021). Transplanting may therefore be a necessary first step that can establish source populations that then self-propagate. However, our results show that these transplanted patches need to be close to other existing kelps to survive (Eger *et al.*, 2020a; Layton *et al.*, 2020a).

(2) Seeding kelp populations

Broadly defined, seeding involves dispersing and/or growing the juvenile life stage (i.e. seeds, gametophytes, propagules, zoospores) of the kelp in the ocean. Seeding kelp populations has received much less attention than transplantation. This

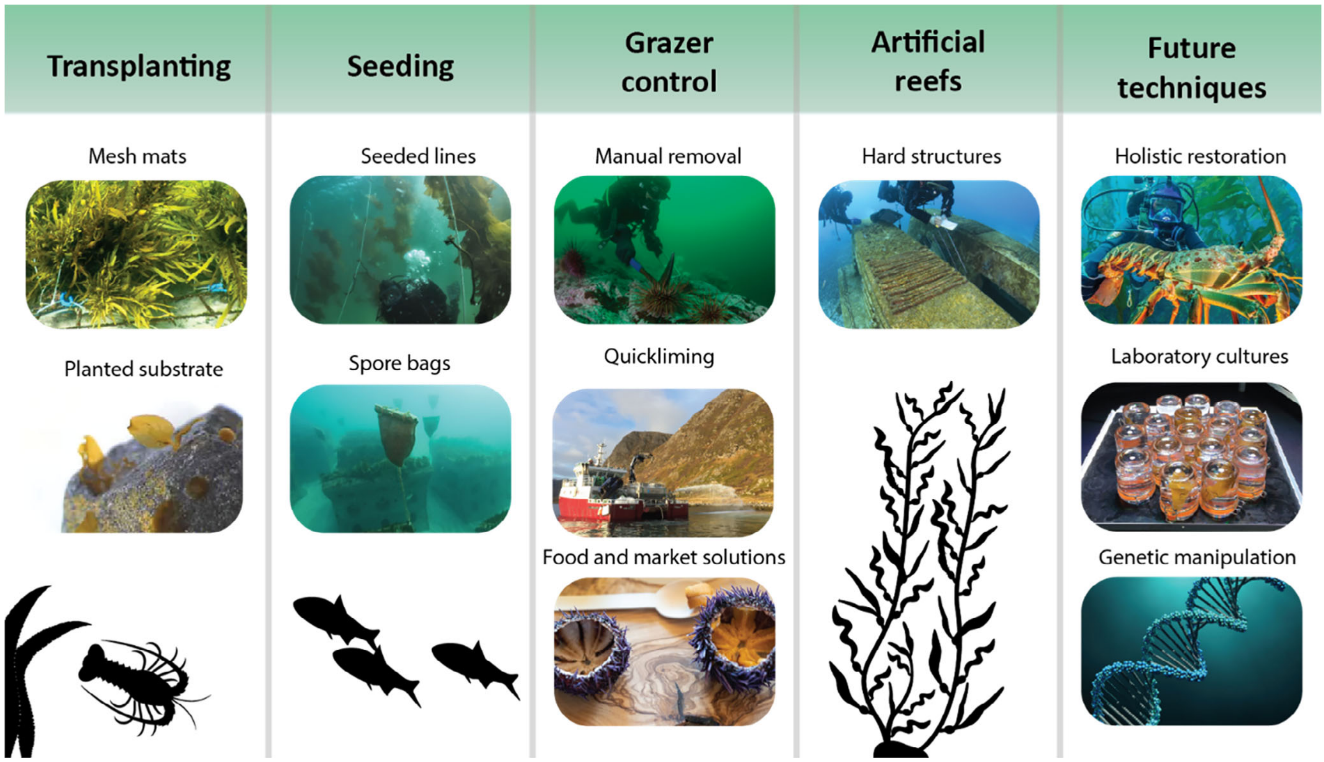

Fig. 5. Methods used in kelp forest restoration (photograph credits, left to right, top to bottom: Operation Crayweed, FIRA, Ryan Miller, FIRA, NOAA, Green Gravel, FIRA, NIVA, University of Tasmania, Urchinomics, Pixabay).

may be due to the extremely high mortality of kelp propagules (Schiel & Foster, 2006) and the perceived advantage of focusing on sporophytes where survival is many orders of magnitude higher. Projects that have used seeding have usually weighted mesh bags filled with fertile kelp blades to the bottom on the sea floor, allowing the propagules to settle on the sea floor (Westermeier *et al.*, 2014). Such projects have had limited success and remained time intensive as divers were used to install and remove the bags from the ocean. Restorationists in coral reef ecosystems are trialling the use of ships to disperse coral propagules into the ocean (Doropoulos *et al.*, 2019) and a similar approach could be trialled for kelp that would likely be more cost effective. Such seeding methods have promise because if successful, they are applicable at a much larger scale at relatively low cost, and allow genetic selection and manipulation to be more easily applied (Saunders *et al.*, 2020; Vanderklift *et al.*, 2020).

(3) Removing competitors

Removing kelp competitors from the sea floor has received very little attention outside of Japan, where they have developed a suite of techniques for clearing the rock bare (Japanese Fisheries Agency, 2015, 2021). Some of these methods can be maintained without continued input, for example, a chain spun by wave action, but others such as manual or mechanical removal are much more labour intensive. Regardless of the approach, large-scale scraping of the benthos is likely untenable in most countries and locations, thus this approach will likely be limited to small-scale transplant sites where removing competitors may help to establish the desired kelp population.

(4) Grazer control

Controlling grazers relies on manual removal or exclusion of the animal from the targeted restoration area. For sea urchins this can entail crushing them (Leinaas & Christie, 1996), relocating them (Mead, 2021), harvesting them (Piazzi & Ceccherelli, 2019), or killing them with quicklime (Bernstein & Welsford, 1982). These methods are also restricted by their labour costs (Fig. 6) and feasibility varies by location. One cost–benefit analysis of *Centrostephanus rodgersii* removal in Tasmania, Australia, by physically killing or removing the urchins estimated approximately 13 dive days per hectare per diver (Tracey *et al.*, 2015), though the exact removal rate of urchins is dictated by urchin density, depth, water conditions, and topography.

While urchin management is more scalable than transplanting, it still requires substantial resources (Fig. 6). Urchins have been successfully baited to concentrate them in space and therefore make removal more efficient (Japanese Fisheries Agency, 2015, 2021; James *et al.*, 2017). Another solution to scaling up is potentially addressed by the use of quicklime (CaO) over urchin barrens (Strand *et al.*, 2020). In areas where urchin barrens are relatively

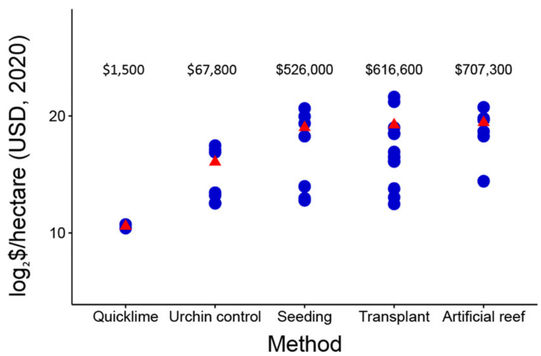

Fig. 6. Reported costs per hectare of restoring kelp populations according to the method used. Note that the *y*-axis is plotted on a logarithmic scale. Red triangles are mean values, which are also given in USD (2020) above each column.

depauperate of other species, collateral damage may be minimal, although other echinoderms and juvenile abalone can be damaged or killed (Strand *et al.*, 2020; Keane, 2021), thus local assessments of ecosystem effects are warranted. The trade-offs involved in this approach are beyond the scope of this review, but from a technical perspective, it can be used over large areas (Strand *et al.*, 2020).

Another challenge associated with urchin removal is to maintain the sites where they have been removed. Many projects have demonstrated that if sites are not maintained, urchins will often return and continue to graze kelp transplants or recruits (Carlisle *et al.*, 1964; North, 1978; Carney *et al.*, 2005; Yoon *et al.*, 2014). Current evidence suggests that sea urchin biomass needs to be <70 g urchins per m^2 and, in some cases, closer to 0 (Ling *et al.*, 2015). The exact number of urchins able to sustain a barren will depend on the species and grazer type (e.g. scraper *versus* grazer) and availability of alternative food (Byrnes *et al.*, 2013). As an addition or an alternative to continual site maintenance, restoring healthy predator populations alongside kelp forests that can keep sea urchin numbers low may also help to create a self-sustaining ecosystem (Eger *et al.*, 2020a). Regardless of the solution, restorationists will need to address this problem to ensure long-term viability.

Alternative solutions for managing grazer populations include the establishment of a fishery or ranching program which removes the animals from the ocean for food and/or profit (Lee *et al.*, 2021; Verbeek *et al.*, 2021). These market-based solutions have the added benefit of providing employment and increasing the perceived value of the kelp forests, hopefully spurring further conservation. A limited number of organisations are currently exploring these solutions in Norway, California, Australia, and Japan (Larby, 2020; Urchinomics, 2020). Restoration of natural sea urchin predators could be achieved either through marine reserves which may allow them to recover without further intervention (Eger & Baum, 2020), or through planned reintroductions/range expansions where key predators are missing (Eger *et al.*, 2020a). Managers could combine reserves and reintroductions with active restoration efforts to maximise chances of success.

Destructive grazing of kelps by fishes is less common than by urchins but is a consistent issue in some areas such as southern California, southern Japan, and some regions of Australia (Vergés *et al.*, 2019). There is likely to be an increase in interactions between kelp and range-expanding herbivorous fishes as sea temperatures rise (Vergés *et al.*, 2019). The same issues and potential solutions apply to controlling grazing fish populations as described above for urchins. In addition, increasing kelp abundance and density through successful restoration efforts could help mitigate grazer damage by distributing fish grazing pressure over many plants as opposed to a few. Focusing restoration efforts during times of the year when herbivores are less active or less abundant could also enhance kelp survival (Carney *et al.*, 2005). Future restoration projects should therefore aim to create large populations as opposed to small patches where grazing may be concentrated and should also consider seasonal variations in herbivory.

(5) Artificial reefs

Artificial reefs are another common approach although they are used more often in afforestation than in restoration projects. While they are often not well documented, artificial reefs have an extensive history, and the materials used range from rocks, tram cars (Carlisle *et al.*, 1964), bombs and ships (Tickell *et al.*, 2019), to materials designed to enhance algal growth (Fujita *et al.*, 2017). As discussed above, if artificial reefs are placed in habitats that did not contain kelp (e.g. on a sandy substratum, as is common), the approach is considered afforestation as opposed to true habitat restoration. The use of artificial reefs for afforestation is common in Japan and Korea (Lee *et al.*, 2017) but faces greater resistance elsewhere (l'vfeier, 1989; Tickell *et al.*, 2019).

The trade-offs between adding artificial materials to the ocean and leaving the naturally occurring habitat unaltered (often replacing sand or unconsolidated substrate habitats), remains a societal decision that may be increasingly important (Paxton *et al.*, 2020). A key benefit of artificial reefs is that managers can place them where they are easily maintained, and kelp transplants can be attached more easily than to the natural sea floor. New materials for artificial reefs include those that structure the concrete to enhance rugosity and provide additional settlement area (Ishii *et al.*, 2013; Bishop *et al.*, 2017), as well as infusing the concrete with iron, nitrates, and other growth-enhancing materials that are slowly released over time (Oyamada *et al.*, 2008). The materials required to build artificial reefs remain very expensive (~$707300 USD, 2020/hectare, Fig. 6) and require substantial investment, which has typically been provided by governments.

Kelp restoration projects can use a combination of methodologies which may improve the chances of success. For instance, restorationists can install a reef with transplants, clear the benthos and then seed, or as is most common, seed or transplant kelp and control grazer populations. None of these methods are mutually exclusive and the use of multiple methods may enhance growth of emerging kelp populations in different ways; for example, transplanted kelps could make the environment more amenable for the growth of seeded propagules. Removing competitors, controlling grazers, and/or adding substrate alone all rely on the local availability of propagules; if no local populations or existing gametophytes are available to act as seed sources, kelp will be unable to re-establish naturally at the restoration site. Therefore, restorationists need to consider the local conditions when applying any combination of these methods.

(6) Restoration methodologies in the future

Despite a relatively static past, future restoration may be required to change substantially to match the accelerated rate of environmental change (Wood *et al.*, 2019). For example, there may be important advantages to selecting certain kelp genotypes for restoration, either through selective breeding, direct genetic manipulation (Coleman *et al.*, 2020), or by using kelps that have survived extreme events (Coleman & Wernberg, 2020). With careful consideration of any unintended consequences, restorationists could select such individuals for their increased tolerance to warming sea temperatures or ability to ward off grazers, although selection for one trait could lower fitness in another (e.g. increased thermal tolerance may make individuals more susceptible to grazing; Coleman & Goold, 2019). In addition, as populations are rapidly being lost, the creation of seed banks on land that can preserve genetic material that may otherwise disappear is being considered (Layton & Johnson, 2021). Future restoration efforts should also consider the critical associations between a kelp 'host' and its microbiome, which is essential for host health and functioning (Egan *et al.*, 2013). Enhancing kelp microbiomes with beneficial microorganisms may also increase kelp resilience to stressors and enhance restoration success (Trevathan-Tackett *et al.*, 2019; Wood *et al.*, 2019; Dittami *et al.*, 2021). More generally, enhancing positive interactions between kelps and other organisms may be critical for success (Eger *et al.*, 2020a).

The question of scale may be addressed by borrowing techniques from the aquaculture industry which cultures spores on rope, suspends them in the ocean, and grows kelps free from the pressure of sea urchin grazing (Eger *et al.*, 2020a). These seeded lines could then be directly installed on the sea floor or suspended mid-water to act as a source population (Camus, Infante & Buschmann, 2019). Adding any foreign materials in the ocean requires careful consideration but given the scale at which we can grow kelp for food, it is plausible that we can use similar methods to help restore wild populations.

Changes to future management of fisheries for urchins and herbivorous fishes also offer potential practical long-term solutions for assisting in recovery of overgrazed populations (Larby, 2020). Such fisheries could be carefully integrated into protected areas and management zones, allowing for selective removal from the area (Bengtsson *et al.*, 2021). Further, while currently only a concept, the use of autonomous robots, such as those designed to kill crown-of-thorns sea stars on the Great Barrier Reef, could work continually to remove urchins over large spatial scales (https://balancedoceans.com/). However, consideration of any automated and remote methods must be carefully balanced against potential risks to other ecosystem components, including species at risk (e.g. abalone).

At the policy level, if we are to invest in restoring kelp forests, this will mean working to address their causes of decline. Specifically, future management policies must reduce overfishing of key species, reduce sedimentation and pollution rates, and ultimately work to slow or even reverse greenhouse gas emissions that are warming the oceans past some species' physiological tolerances (Gann *et al.*, 2019; Wood *et al.*, 2019). Each of these restoration strategies should be considered together with the potential risks, benefits, and societal willingness to engage with different methods (Coleman *et al.*, 2020).

Evaluating the causes of ecological success and failure will be a key step for advancing the field of kelp restoration. Although this review represents a beginning, this field is advancing rapidly and continued efforts to compile information in a central site as progress is made will be important to promote sharing and collective learning from individual project experiences. One potential avenue to achieve this is a collaborative project called the Kelp Forest Alliance, which includes a website (www.kelpforestalliance.com) that will freely host the database used for this review and can provide a framework for future restorationists to contribute appropriate data about their projects. The Kelp Forest Alliance intends to work as a nexus for information on kelp restoration projects that links together peoples from around the world, while also helping to advance research and resources for restoration projects.

VI. SOCIOECONOMIC CONSIDERATIONS FOR RESTORATION

(1) Financing restoration

Reported costs of kelp restoration vary substantially among and within methodologies and projects. Methods for the control of sea urchins have the lowest costs, with quickliming costing an average ~\$1500/ha (USD 2020) and manual removal averaging ~\$67800/ha. The other methods, transplanting, seeding, and building artificial reefs, range between \$526000 and \$707000/ha (Fig. 6). These values were calculated from studies employing a single method; multi-method projects may have similar or lower costs. For example, transplanting on artificial reefs can have lower costs than transplanting on natural ocean substrate. Interestingly, despite being easier to access, intertidal transplants were more costly than subtidal transplants, potentially due to a longer history of subtidal work and more refined methods; in addition, intertidal restoration project areas have been exceptionally small, and scaling costs per hectare based on a 1 m^2 plot can lead to overestimates as the marginal cost for each additional m^2 plot is unlikely to be linear.

The sample size used to collect data for the costs of kelp restoration was very low as most projects did not report costs, however the magnitude of difference suggests that kelp restoration can cost substantially more than restoration in other marine ecosystems (coral ~\$196000, seagrass ~\$126000, mangroves ~\$11000, saltmarsh ~\$80000, per ha, USD, 2020; Bayraktarov *et al.*, 2016). While not considering projects in Japan, for which cost data were mostly unavailable, relatively few kelp restoration projects have taken place compared to restoration of other marine systems (Saunders *et al.*, 2020). More extensive experience and refinement of methods may be contributing to lower costs per area restored in other systems. If this is the case, the expected costs for kelp restoration should decline as we gain further experience, methods are refined, and efficiency is improved. Economies of scale should also result in reduced cost per hectare for larger projects (Turner & Boyer, 1997). Indeed, reports from two large-scale kelp restoration and afforestation projects in Japan and Korea have reported costs of \$10–20000 per ha (Eger *et al.*, 2020c).

Ecological restoration is currently very expensive, yet societal economic benefits from investing in kelp restoration can be substantial. Preliminary analysis of *Ecklonia*, *Nereocystis*, *Macrocystis*, and *Laminaria* forests and the services they provide through fisheries, carbon sequestration, and nutrient cycling suggest that restored kelp forest should result in \$59–194000 USD 2020/ha/year economic benefit (Eger *et al.*, 2021). These benefits would potentially offset the costs of restoration within 1–12 years, depending on the methods used. Although the costs are currently high, if prices decrease with improved techniques and larger scales, the business case for restoring kelp populations should become stronger.

Further, carbon, nitrogen, and phosphorus credits are already being traded on local and global markets and groups that restore kelp populations could be awarded the respective number of credits, which they could then sell to offset and potentially even profit from kelp restoration projects (Rutherford & Cox, 2009; Herr *et al.*, 2017). Because the fate of kelp biomass is often unclear, the values for carbon and nutrient sequestration are still poorly understood in most kelp genera and regions. Early estimates suggested that 5–20% of a species' yearly net primary production acts as a long-term sink (Krause-Jensen & Duarte, 2016; Gouvêa *et al.*, 2020), which while smaller than for other marine macrophytes, suggests potential for the use of kelp restoration in such trading schemes. If verified trading schemes are developed for kelp restoration, then projects could contribute towards meeting a country's commitments to reduce greenhouse gas emissions under the Paris Agreement, which would provide a very strong incentive for national governments to invest in kelp restoration. Restoring kelp forests is also expected to increase fishery yields of not only the kelp itself but kelp-associated species (Bertocci *et al.*, 2015). Because many fisheries have closed due to kelp collapse, investing in restoration would help revitalise these lost industries and should also help governments justify the costs of restoration. For example, the now closed abalone fishery in northern California was valued at \$24–44 million USD dollars in 2013 (Reid *et al.*, 2016; Rogers-Bennett & Catton, 2019) and the lobster fishery in Australia was assessed at \$700 million AUD (\$520 million USD) in 2018 (ABARES, 2020).

Although large-scale restoration requires significant financial inputs, there can be potential economic and societal benefits. In the past, governments have attempted to revitalise their economies following a disaster or recession by increasing spending, often funding large infrastructure projects (Restore Act, 2012; Mannakkara & Wilkinson, 2013). Kelp restoration could be viewed as a similar investment, as financing kelp restoration would lead to substantial positive economic and social benefits. This approach was already taken in the USA in 2009, when the US administration included \$178 million USD for oyster reef restoration as part of an economic stimulus package (Smaal *et al.*, 2018). Similarly, the Australian government is investing tens of millions of dollars into coastal restoration and blue carbon as a part of its COVID-19 response spending (Prime Minister of Australia, 2021), while the EU's European Green Deal invests in nature and other technologies to achieve carbon neutrality by 2050 (European Green Deal, 2021). Other countries could look to stimulate growth by using similar approaches. The FMDP project in Japan (see Section III.2) is another model for how government groups can work together to set aside funding for restoration, provide access to experts, and facilitate collaboration across different sectors of society (Sekine, 2015; Fujita, 2019). Collaborative funding and support structures are promising ways to implement restoration at a national scale.

Finally, another potential source of funding may come from private enterprises. Business interests are increasingly looking to build social capital by 'giving back' while remaining profitable (Sneirson, 2008). For kelp restoration, companies such as Urchinomics (https://www.urchinomics.com/) and the not-for-profit Greenwave (https://www.greenwave.org/) are exploring pathways not only to restore kelp forests but also to generate sustainable revenues and operate outside the not-for-profit space. These alternate pathways could be vital to addressing the high costs of restoration (Eger *et al.*, 2020c). For example, government and fisheries groups in Korea are working with budgets of hundreds of millions of USD to restore kelp (Eger *et al.*, 2020c) and a proposed kelp restoration project by the US Army Corp of Engineers in Los Angeles, California, USA, has a budget of ~\$150 million USD (United States Army Corp of Engineers, 2019). These high-cost budgets are unattainable for many conservation groups, and green businesses may present opportunities to reduce costs and possibly create profits from kelp restoration projects.

(2) Legal frameworks for restoration

Marine management policy has often lagged behind the rapid environmental changes occurring in the oceans (Rilov *et al.*, 2019). As a result, laws initially intended to protect marine resources could now be hindering restoration efforts. Current environmental laws focus on either prohibiting the removal of resources from the oceans (e.g. fishes) or the addition of unwanted materials into the ocean (e.g. waste dumping; Lumsdaine, 1975). Restoration of kelp forests can require either or both actions. To address a hyperabundance of grazers, removal or reduction in the number of herbivorous species can be necessary. Conversely, to re-establish kelp populations the addition of biogenic materials, such as transplants or propagules, is sometimes needed, or an input of artificial substrates for kelp attachment or settlement.

Current discussions regarding reforming environmental laws have focused on identifying appropriate baselines and target species (McCormack, 2019); additional discussions are also needed to revisit the rules regarding exploitation of 'unwanted' or hyperabundant species and the addition of desirable materials. For example, marine reserves often prohibit the removal of sea urchins which can prevent kelp from returning, as in Hong Kong for example (Leung *et al*., 2014). While no-take marine reserves remain the gold standard in marine conservation (Sala & Giakoumi, 2018), shifting these paradigms to allow for limited removal of endemic grazer species (such as for the project in Gwaii Haanas, BC, Canada; Lee *et al*., 2021) and invasive grazing species and potential addition of habitat into reserves may be needed to address specific issues. As an example of changing legislation, in September 2021, the state of California passed Bill AB-63 to facilitate restoration and monitoring activities within marine conservation areas. The challenges presented by modern restoration projects will therefore require adaptive legislative frameworks that allow for the trialling of environmental interventions, scaling them up when successful, and the reconsideration of previously held tenets.

Other laws or directives will also be useful in motivating restoration efforts – specifically, laws that require the offset of habitat destruction. For instance, offsetting regulations were responsible for a 172 ha project in southern California which is working to ensure no net loss of kelp (Bull & Strange, 2018) from that project (Schroeter *et al*., 2018). The USA, Canada, Australia, the EU, Korea, and New Zealand have offsetting regulations and policies (Niner *et al*., 2017) which are useful examples for how to create such policies. Interestingly, we only recorded four offsetting projects in our database, potentially because these project reports are not easily accessible or because offsetting for kelp is uncommon. Regardless, future offsetting projects should be reported in public repositories to allow for open consideration of their success. Notably, Norway, Japan, and Chile, do not have offsetting directives. Although offsetting policies are important, they can only ensure no net loss of kelp and are not necessarily effective for increasing kelp area. Governments can look to increase kelp populations by setting directives such as Law N°20.925 in Chile which legally sets aside funds for restoration.

VII. CONCLUSIONS

(1) Kelp forest restoration has a long history, spanning 16 countries and over 300 years of practice. The field is diverse with representation in many sectors of society, including academia, governments, communities, indigenous groups, and businesses. The field is accelerating with more projects in the 10 years between 2009 and 2019 than ever before. While a global field, more restoration projects have occurred in Japan than the rest of the world combined, but access to the results of those projects remains limited.

(2) To date, most restoration projects have been small in size, short in duration, and focused on a few genera (*Macrocystis*, *Ecklonia*, *Cystoseira*, and *Sargassum*).

(3) Six recorded projects have achieved large-scale success (100 and 1000 s of ha) in restoring kelp forests. This success shows that large-scale restoration is currently possible and a reasonable goal to strive for.

(4) The most successful restoration projects are those that are near existing kelp forests. Preventing kelp forest decline aids kelp recovery, therefore actions to ensure that kelp is not lost from a system are critical.

(5) Urchin grazing is the most frequent reason that kelp restoration is needed and also the most common cause of project failure. Projects should work to mitigate this stress prior to restoration and maintain low grazer densities to achieve success. Although not necessarily acceptable due to potential ecological risks, quicklime maybe a technically viable solution to remove large numbers of sea urchins at low financial cost. Urchin fisheries and/or urchin ranching are other options which may profitably remove urchins.

(6) Transplanting kelps should work to establish significant population sizes for the best chance of success, particularly if they are adjacent to existing kelp beds.

(7) Artificial reefs are a common but expensive and contentious tool for afforestation and restoration. Projects need to carefully consider the economic and environmental costs and benefits before deploying artificial reefs.

(8) Further work is needed to investigate seeding methods for restoration. If successful, this method could help scale up kelp restoration projects to larger sizes at reasonable costs.

(9) Projects have been very expensive to date, but costs are reducing, and the social and economic benefits of kelp restoration are high.

(10) Future methods for restoration (genetic manipulation, kelp aquaculture, autonomous technology) have the potential to address barriers to restoration (warming oceans, low abundance of existing kelp, high urchin populations), but risks and benefits must be weighted, and considered in context of holistic ocean management.

(11) Legal frameworks are often inappropriate for kelp restoration and may need to be reconsidered to allow for careful manipulation of ocean spaces for restoration where needed (e.g. transplanting, seeding, herbivore removal).

(12) Kelp restoration initiatives present opportunities for rich collaborations among individuals, organisations, and countries, to reforest the ocean, achieve benefits

for multiple user groups, and link into the UN Sustainable Development Goals. Global efforts to consolidate and share experiences and learning, such as the Kelp Forest Alliance (kelpforestalliance.com), represent concrete steps towards advancing future efforts.

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X. Supporting information

Additional supporting information may be found online in the Supporting Information section at the end of the article.

Appendix S1. Search terms and results for published literature search.
Appendix S2. Data collection template.
Appendix S3. Data identifiers used in plotting the different descriptive results.
Appendix S4. Full data set used in analyses.
Appendix S5. Groups involved in kelp restoration projects.
Appendix S6. Motivations for initiating restoration projects.
Appendix S7. Location of restoration projects in Japan 1970–2014.

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