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WP 1 **Deliverable 1.3**

D1.3: State of the knowledge on marine habitat restoration and literature review on the economic costs and benefits of ecosystem service restoration

Marine Ecosystem Restoration in Changing European Seas MERCES

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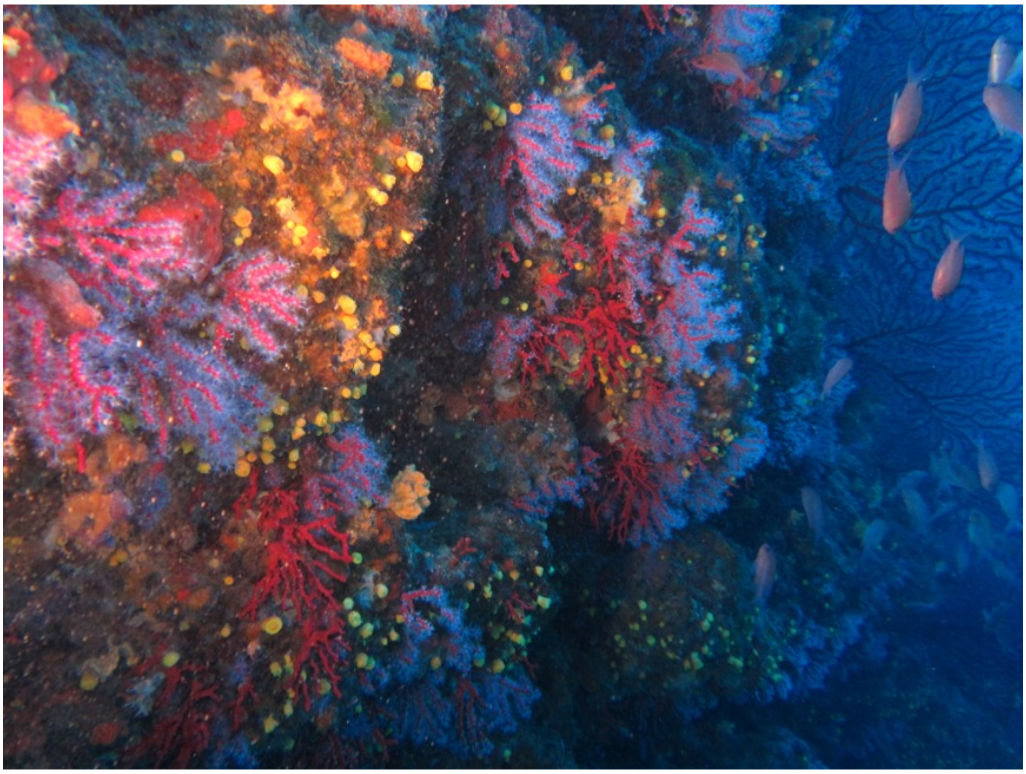

Highly diverse Mediterranean coralligenous habitat, Photo by © Carlo Cerrano, UNIVPM.

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Summary

Anthropogenic activities have had an accelerating impact on the marine world over the last 100 years. Increasing industrialisation, increasing use of resources, plus a build-up of populations on the coastline has put a high degree of pressure on the marine environment causing widespread habitat change, particularly coastal or near coastal, exacerbated by climate change. The continuing loss of biodiversity linked to habitat degradation, may lead to the unprecedented erosion of natural capital. It has been widely recognised that a range of different restoration actions are essential to halt further habitat decline, and reverse the current trends of degradation and species loss. This has led to the recognition of the needs for conservation and protection in several key habitat types, and latterly, within the Convention on Biological Diversity and EU Biodiversity Strategy for 2020, for 15% restoration targets of degraded ecosystems.

The scope of the MERCES D1.3 Report is to review the state of the knowledge of habitat restoration to support the work surrounding restoration-related activities in the MERCES project. In order to carry out the general objective, several reviews have been carried out, including: a review of unassisted restoration (spontaneous regeneration), a structured review/synthesis of peer-reviewed publications on active restoration, a state-of-the-art summary of the MERCES key habitats/species/ecosystems with respect to restoration, a review of recent European and iconic world-wide projects concerning marine restoration, and a structured review of the costs and benefits in marine restoration. In addition, several relevant issues concerning restoration are addressed, including; artificial reefs, restoring key structural species, removal of threats, No Net Loss within a mitigation hierarchy, nature-based solutions, technologies and innovation, and restoration feasibility.

There had been little consensus among scientists and practitioners as to what restoration is, with many terms used interchangeably, including restoration, remediation, reparation, recuperation, reconstruction, rehabilitation, and even re-creation. The Society of Ecological Restoration (SER) defines ecological restoration as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed". A major dichotomy is also often described between unassisted restoration or spontaneous regeneration (also called passive restoration) and so-called active restoration involving direct human intervention. Recent approaches to deal with degraded ecosystems call for the application of a family of restorative activities that can be carried out simultaneously or sequentially. The first step to allow for natural recovery is the removal of threats. This can be perceived as preventing harmful activities through regulatory management

(from controlling/banning specific activities to creating Marine Protected Areas), or removing/adding barriers in an intervention to protect an ecosystem from further harm.

Active Restoration

A review was undertaken of global peer-reviewed studies from the last 25 years with the aim to assess and summarize the methodological trends (both spatial and temporal patterns), and to provide a framework on where and how restoration has been carried out and with what outcomes, towards identifying drivers for success. In total, 498 publications were screened from 4066 publications. All world eco-regions were represented except for the Southern Ocean, with most of the efforts (58%) in the northern hemisphere. In the Temperate North Atlantic, most studies were in the USA (62%), Mediterranean (17%) and Northern European Seas (14%). Most studies were in the estuaries/wetlands (42%), followed by rocky reefs (30%), and soft-sediment environments (28%). From 2008, strictly marine studies started to exceed estuarine/wetland studies. The most targeted species in specific ecosystems include *Acropora* (coral reefs), *Spartina* (saltmarshes), *Zostera* (seagrasses), *Crassostrea* (oyster reefs), *Rhizophora* (mangroves) and *Cystoseira* (macroalgal forests). Generally, restoration efforts covered just a few years and few studies covered longer time scales. The majority of studies used transplantation or planting techniques and the most commonly used response variable was survival. Success was noted for 50–70% of the studies, whilst failure was linked to methodological aspects and overlooking important site characteristics and local threats.

Restoration of MERCES Key Habitats/Species

A review of key European habitats/species covers most current restoration methods and approaches used, timescales to success, bottlenecks/deal-breakers and means/potential for up-scaling restoration to the level of degradation. This covers 12 MERCES cases studies involving kelps, *Cystoseira*, seagrass, the bivalve *Pinna nobilis*, coralligenous habitats, red coral, sponges, deep-sea corals and seamounts. Techniques ranged from regeneration to transplantation and facilitation (e.g. by mussels) using different life-history stages or methods. Restoration is still in its infancy for some species and new protocols are being developed for deep waters. Time scales to restoration vary widely between ecosystems from months/years (kelp, sponges, some seagrasses), to decades (some seagrasses and corals), to multi-decades or centuries (deep-sea corals). Deal-breakers commonly depend on target species characteristics, the methods and techniques used, site parameters, but also the continued absence of threats. Up-scaling presents a number of challenges but will need an approach using a family of restorative activities (e.g. threat removal, unassisted regeneration, remediation, good management) combined with technological innovations, science-industry solutions and citizen science/volunteering support.

Recent Restoration Projects

In reviewing recent European research projects from the past decade, 42 projects were identified. The most prolific countries involved were Spain (24%), France (21%) and Italy (12%). The average timespan of the studied projects was 5 years. Most projects (45%) focused on a single location, with only 17% involving transnational cooperation. Funding was most commonly (45%) through European Union projects (mostly LIFE) and the average budget was 3.5 million Euros. Seagrasses were the main restoration target (36%) followed by saltmarshes (14%) and hard substrates/reefs (12%). The most common type of action was research-methodological based (33%) followed by restoration (recovery of a degraded habitat) (24%), and then enhancement (14%) (increasing value or goods and services). The most common method was species translocation (21%), seeding and planting (14%), artificial substrates (14%), then hydrological modifications, removal of contaminants and litter (10% each) – removal of invasives represented 5% of the projects.

Costs and Benefits in Restoration

The methodology and metrics on the economics of restoration are reviewed. Costing direct restoration activity may seem straightforward, but there are many not-so-obvious and hidden costs including, regulation and long term managing and monitoring. Restoration benefits come from changes in biodiversity, processes and functions resulting in changes to ecosystem goods and services. The ecosystem benefits will have societal value. Some values can be derived directly from market-values or proxies that have market-values, whilst others involve people stating values, preferences or willingness to pay, which, although commonly used methods, must be used with great care.

A review is presented on economic costs and benefits of marine restoration covering 103 published and grey-literature documents. For Costs, 72% regard monetary observations. Studies concerned rocky habitats (41%) soft-bottom habitats (42%) estuarine/wetlands (13%), and deep-sea (2%), primarily concerning restoration of degraded marine environments (88%) and mostly transplanting studies. For Benefits, most regard ecological benefits and 45% were opinions on economic benefits (e.g. increase in commercial species stocks, upgrade in an area's aesthetics, increase in tourism, increase in local income through the restoration project). Only three recent studies provided benefit data based on economic valuation methods. Most associated benefits refer to restoration of rocky subtidal (coral reefs), and soft-bottom habitats (mangroves and seagrasses). Half of the restoration benefits observations concern experiments, and 34% concern restoration projects. The majority of studies reporting benefits were successful or partially successful. Few of the studies concerned social/economic issues.

Restoration Issues

Several issues are discussed that were either not part of the targeted reviews, cross different boundaries or are important enough to warrant further development:

- Artificial reefs are a contentious issue in restoration; they have been used for various purposes and with carious success in the last 40 years, mostly introducing a new ecosystem, and are not considered as part of the original ecosystem, although their use has been justified as part of physical protection, mitigation or enhancement.
- Although the targets of restoration are recovered ecosystems, most interventions are targeted at restoring particular species. We discuss habitat forming species, keystone species or talismatic/emblematic/characteristic/habitat defining species.
- One driver of restoration is the response to a large-scale disaster, whether natural (e.g. tsunami) or anthropogenic (e.g. oil spill). Such single events may require a very large-scale response, covering multiple ecosystems and concerted management efforts; such has been the response to the Gulf of Mexico Deep Horizon oil spill.
- In addition to controlling threatening activities prior to restoration, some specific threats must be removed and we discuss litter, invasive species, keystone species (grazer or predators) and the issues of physical barriers.
- The mitigation hierarchy is a set of prioritised steps to alleviate environmental harm as far as possible through avoidance, minimisation and rehabilitation/restoration. Offset measures can be taken to compensate for any residual adverse impacts. Companies in natural resource sectors use offset management strategies (e.g. No Net Loss) to mitigate environmental changes related to extraction, pollution, biodiversity loss and climate change. Biodiversity accounting is required for selecting potential offset options but is still in its infancy from firstly being a relatively new strategy in application, and secondly because of the difficulties in accounting; for example, losses from intangibles, such as potential cumulative impacts,

and also gains from predicting how biodiversity values will change following the implementation and success of mitigation/restoration strategies.

- Another strategy for restoration is through ‘soft engineering’ (‘Nature-based Solutions’, ‘Building with Nature’ and ‘Ecological Engineering’). The primary examples of this is in coastal management, whereby hard engineering options are replaced with less extreme engineering works integrated with natural solutions for example, not building a large seawall but incorporating saltmarsh building coupled with a lower seawall. A comparison of the costs of nature-based solutions to hard engineering structures has shown that salt-marshes and mangroves could be two to five times cheaper than a submerged breakwater in certain conditions. Nature-based solutions are also important in the sequestration of carbon; whilst this may be a primary goal in planting saltmarsh, mangroves and seagrasses, other benefits include sequestering nutrients, coastal protection, increasing biodiversity and development of nursery grounds for commercial species.
- Technology and Innovation: underwater restoration can be tremendously challenging compared to any terrestrial analogue, because of the working environment, particularly in deeper or offshore areas. New technologies are becoming available or adaptable, with access to underwater vehicles or new materials for underwater work. Mechanical planters are already available for very shallow work. However large area coverage is still a major issue, as transplanting on hard bottoms is still labour intensive as for example corals need careful placement and orientation. In shallow, more accessible waters, volunteer engagement through citizen-science initiatives may be a significant way towards up-scaling restoration over wider areas, either from collection of fisheries bycatch (e.g. coral fragments for on-growing), volunteering equipment (small vessels), space (for nursery grounds), or time (for labour at any stage in the process). Also the use of social media can enhance any kind of campaign organisation reaching wider distributions than has previously been possible.

Restoration Feasibility

For restoration, the key question is whether or not to undertake action, and the decision should involve socio-economic, ecological and technological parameters. The ecological parameters pertaining to the MERCES key habitats are discussed and compared, including ecosystem features such as dynamics, connectivity, spatial distribution, vulnerability/fragility, structural complexity and diversity. Restoration timescales related to target species growth and reproduction are investigated from shallow water macroalgae and seagrasses on the scales of years to decades, to deep-water corals on the scales of multi-decades to centuries. Spatial scales are discussed in relation to the large amount of small scale (10–100s square metres) experimental work undertaken, with little in the way of larger projects (hectare scale) and how up-scaling to reach international conservation targets, that may be in the order of thousands of square kilometres, will require multiple approaches (mitigation hierarchy, rehabilitation and restoration). Finally, we discuss the economic cost and financing of restoration as well as the need to understand, value and communicate ecosystem benefits.

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Acronyms Used

ABNJ: Areas Beyond National Jurisdiction
AF: Adaptation Fund
AIC: Akaike information criterion
ALDFG: Abandoned, lost or discarded fishing gear
AUV: Autonomous underwater vehicle
BBNJ: Biodiversity Beyond National Jurisdiction
BP: British Petroleum
BWD: Bathing Water Directive
BWM: Ballast Water Management Convention
CAP: Common Agricultural Policy
CBD: Convention on Biological Diversity
C/B: Cost benefit
CDPA: Container-disperser of algal propagules
CFP: Common Fisheries Policy
CM: Choice modelling method
CORDIS: Community Research and Developments Information System of the European Commission
COTS: Crown-of-thorns starfish
CRC: Coral Restoration Consortium
CV: Contingency valuation method
EC: European Commission
EEZ: Exclusive Economic Zone
EIA: Environmental Impact Assessment Directive
EUNIS: European Nature Information System (for habitat classification)
EU: European Union
EPA: Environmental Protection Agency (USA)
FAO: Food and Agriculture Organization
FRA: Fisheries Restricted Areas
FRMD: Flood Risk Management Directive
FRMP: Flood Risk Management Plan
GBP: Great Britain Pounds
GES: Good environmental status
GFCM: General Fisheries Council of the Mediterranean
GPRA: Government Performance and Results Act (USA)
HD: Habitats Directive
ICES: International Council for the Exploration of the Sea
IEEP: Institute for European Environmental Policy
IMAP: Integrated Monitoring and Assessment Program
IMO: International Maritime Organization
ISA: International Seabed Authority
IUCN: International Union for Conservation of Nature
LMPA: Large Marine Protected Area
MARPOL: International Convention for the Prevention of Pollution from Ships
MERCES: Marine Ecosystem Restoration in Changing European Seas
MIDAS: Managing Impacts of Deep Sea Resource Exploitation
MPA: Marine Protected Area
MSFD: Marine Strategy Framework Directive
MSP: Marine Spatial Planning

NATURA:	Coordinated network of European protected areas
NBS:	Nature-based solutions
NGO:	Non-governmental organisation
Nitrates Dir:	Nitrates Directive
NNL:	No Net Loss
NOAA:	National Oceanographic and Atmospheric Administration
NPI:	Net Positive Impact (also referred to as net gain)
OPA:	Oil pollution act (USA)
OSPAR:	Convention for the Protection of the Marine Environment of the North-East Atlantic (Oslo-Paris Convention)
PAH:	Polycyclic aromatic hydrocarbon
PCB:	Polychlorinated biphenyl
PES:	Payment for ecosystem services
PPA:	Partially Protected Area
PVC:	Polyvinyl chloride
RADseq:	Restriction-site Associated DNA Sequencing
RBMP:	River Basin Management Plan
ROV:	Remotely operated vehicle
SAC:	Special Area of Conservation
SAV:	Submerged aquatic vegetation
SCUBA:	Self-contained underwater breathing apparatus
SEA Dir:	Strategic Environmental Assessment Directive
SER:	Society for Ecological Restoration
SPA:	Special Protection Area
SPAMI:	Specially Protected Areas of Mediterranean Importance
TBEP:	Tampa Bay estuary program
TEEB:	The Economics of Ecosystems and Biodiveristy
TNC:	The Nature Conservancy
UK:	United Kingdom
UNCLOS:	United Nations Convention on the Law of the Sea
UNEP/MAP:	United Nations Environment Programme / Mediterranean Action Plan
UNFCCC:	United Nations Framework Convention on Climate Change
UNISDR:	United Nations International Strategy for Disaster Risk Reduction
USA:	United States of America
VIMS:	Virginia Institute of Marine Science
VLMPA:	Very Large Marine Protected Area
VME:	Vulnerable Marine Ecosystems
WBD:	Birds Directive
WFD:	Water Framework Directive
WP:	Workpackage
WWT:	Wildfowl and Wetlands Trust

1. Introduction

1.1. Scope of the Deliverable

The overall scope of MERCES Deliverable 1.3 is to review the state of the knowledge of marine habitat restoration towards, and in support of, the MERCES project. The state of the knowledge encompasses both assisted and un-assisted restoration, reviews of existing restoration projects, restoration methods, technologies and tools, and economic costs and benefits and metrics.

In order to carry out the general objective, several individual studies (targeted reviews) are reported with a general discussion that brings together some over-riding issues for marine restoration. The individual studies include:

- A review of unassisted restoration
- An extensive structured review/synthesis of peer-reviewed publications on restoration looking at major coastal and marine ecosystems
- A review of recent European and iconic world-wide projects concerning marine restoration
- Summary state-of-the-art restoration on MERCES key habitat/species/ecosystems
- A structured review/synthesis of available literature on cost-benefits in marine restoration

This multi-faceted review is complementary to the two other reviews from MERCES WP1 which have investigated available mapping resources for habitats and degraded habitats (Bekkby et al., 2017) and marine activities and pressures and consequences for restoration (Smith et al., 2017):

1.2. Towards Marine Restoration

Anthropogenic activities have had an accelerating impact on the marine world over the last 100 years. Increasing industrialisation, increasing use of resources, plus a build up of populations on the coastline has put a high degree of pressure on the marine environment. Marine activities directly linked to marine uses such as shipping, fishing, aquaculture and coastal infrastructure, as well as land based activities such as farming, or effluents from coastal/inland industry cause a multitude of individual pressures on the marine environment (Smith et al., 2016). This has in turn caused widespread habitat change (Claudet & Fraschetti, 2010) particularly in coastal or near coastal waters (Airoldi & Beck, 2007).

Whilst the distribution and extent of anthropogenic impacts have been assessed at the regional sea level (Coll et al., 2011, Micheli et al., 2013; Korpinen et al., 2013) or even the global level (Halpern et al., 2008), there is still a high level of uncertainty regarding interactions between different pressures and their cumulative impacts (Stelzenmuller et al., 2018).

Alongside current anthropogenic impacts, there is also growing awareness that climate change is transforming our seas (Jones et al., 2014; Sweetman et al., 2017). Direct and indirect human pressures on marine ecosystems are expected to increase considerably in the next few decades, leading to an alarming loss of marine biodiversity and severe degradation of ecosystem functioning in many areas. Since the functioning of marine ecosystems within an historical range of variability is underpinned by high-levels of biodiversity (Danovaro et al., 2008), the continuing loss of biodiversity may lead to the unprecedented erosion of natural capital – healthy ecosystems and native biodiversity – in all marine ecosystems and all of the ecosystem services they supply (Worm et al., 2006; Thurber et al., 2014).

As anthropogenic activities have developed, it has slowly been recognised that the marine ecosystem is a limited resource, that can be harmed and degraded. This has led to various levels of control and regulation on activities that have produced impacting pressures, from the local, to national and international (convention/treaty) level. The degree of regulation has risen with development of the activities, to most recently ensuring sustainable development with regard to halting environmental degradation and enhancing environmental protection through conservation programmes. Conservation goals can be achieved through protection of intact habitat, through the restoration of damaged habitats or through a combination of both (Possingham et al., 2015). A well-managed restoration programme should not only halt decline, but also assist ecosystems to return or recover the functionality and habitat services to their pre-disturbance status, within a known or suspected historical range of variability. The most important causes of species decline and extinction in marine ecosystem have been attributed to habitat loss and degradation, and it has been widely recognised that a range of different restoration actions are essential to halt further decline, and reverse the current trends of degradation and species loss.

In recent decades several EU directives have come into force with increasingly stronger language for the protection and conservation of species/habitats (Habitats Directive, Birds Directive, Water Framework Directive, Marine Strategy Framework Directive, Marine Spatial Planning (MSP) Directive), culminating in the EU Biodiversity Strategy for 2020 (COM, 2011) with Target 2 that states "by 2020, ecosystems and their services are maintained and enhanced by establishing green infrastructure and restoring at least 15% of degraded ecosystems". The progression of EU targeting mirrors that from international conventions such as OSPAR or the Convention on Biological Diversity (Aichi Target 15 – 15% restoration of degraded ecosystems by 2020; see review of details in Aronson & Alexander, 2013a).

Ecological restoration in the marine ecosystem has lagged behind terrestrial restoration, partially due to underwater ecosystems being largely 'out-of-view', lack of understanding of the needs and the degree of degradation, but also from the difficulty in working in the marine environment. Restoration is well developed on the shoreline, for example for saltmarshes and mangroves (Morrison et al., 2011), less so for coastal ecosystems such as seagrasses, or corals (e.g. Rinkevich, 2008) and much less so for deeper waters (Mengerink et al., 2014; Van Dover et al., 2014). The principles and values, and the attributes selected for monitoring and evaluation largely formulated for terrestrial ecosystems must now be adapted and applied to marine ecosystems and habitats (Van Dover et al., 2014). This process will be hampered by the large questions that concern costs and spatio-temporal scales of marine ecosystem restoration work as well as the need for expanding scientific understanding, development of new tools and last but not least new governance structures related to oceans.

1.3. The Restoration Family

There has been little consensus among scientists (e.g. terrestrial and marine), practitioners, planners, and funders as to what actions can be considered as restoration, or what ecological restoration is, and to what purpose. Even distinguishing between conservation and restoration is not always easy, as conservation too can include some kind of action (e.g. breeding) (Geist & Hawkins, 2016) and many terms, including restoration, remediation, rehabilitation, and even re-creation, have been used interchangeably as synonyms (Elliott et al., 2007). Following work by the Society of Ecological Restoration (SER), ecological restoration has now been clearly defined as *"the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed"* (SER, 2004; Clewell & Aronson, 2013). The framing of this definition draws attention to a number of concepts; recovery is put across a range of degraded-to-destroyed ecosystems, then the concept of process is introduced showing that restoration is not static and that time is of key importance. Time can be integral at the socio-ecological level, for example, in terms of designing and planning a restoration project (Bayraktarov et al., 2016; Kirsch et al. 2005), building consensus by stakeholders to initiate a project (SER, 2004; Gleason et al., 2010) and monitoring short-term success and progress towards goals (Bayraktarov et al., 2016). Time is also integral at an ecosystem level in terms of biological life cycles, return/rebuilding of abiotic and biotic functions, replacement/introduction of structure, for example, replanting key structural species or providing alternative structures (Gianni et al., 2013). Most importantly the 'process' implies that a range of approaches to restoration is required at different levels of intervention. The process of restoration can be presented along a 'restorative continuum' (MacDonald et al., 2016), from reducing the causes of decline to full ecosystem restoration. Figure 1 illustrates the restoration process, indicating where some of the subsets or activities of restoration and intervention may lie.

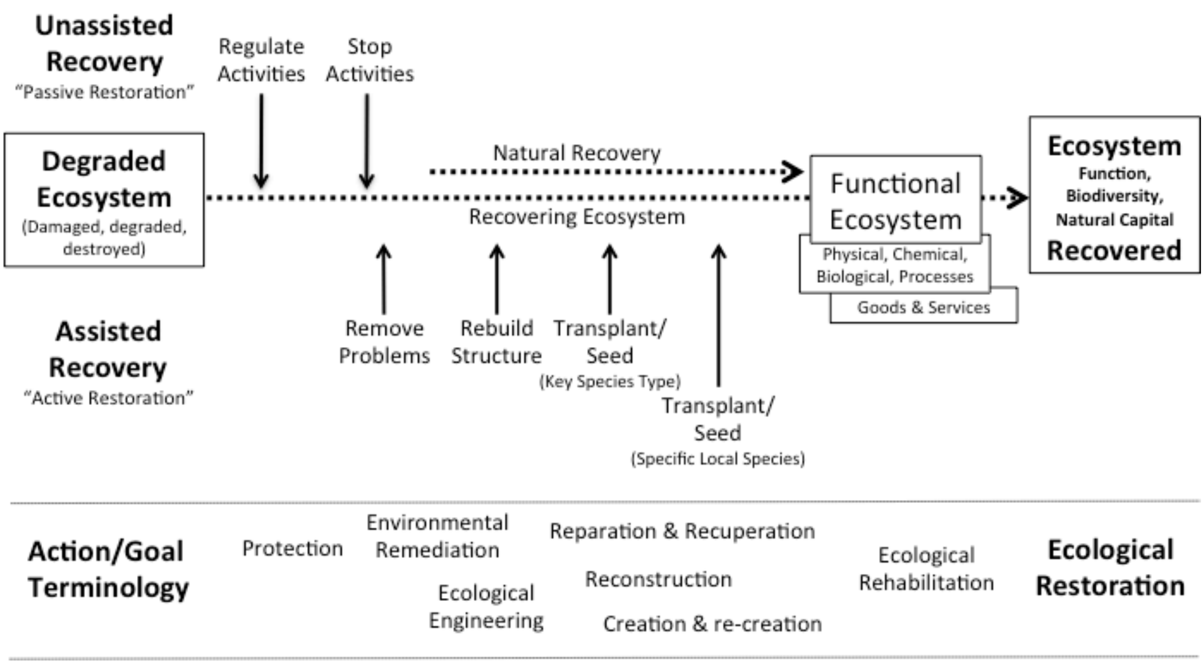


Figure 1. Concepts/terminology and relationships used in the restoration process (adapted from Ounanian et al., 2017).

A major dichotomy is often described between unassisted or spontaneous regeneration and so-called active restoration, although there are many actions or combinations of actions that may be considered as intermediate. Unassisted regeneration mostly involves modifying management approaches to better regulate and/or stop human activities that cause degrading pressures (see detailed review in Section 3). This may range from local legislation, or changes in one landowner's practices, to EU Directives and International Conventions. Management or the absence of threats (e.g. invasive species) is essential/central to both conservation and restoration efforts (MacDonald et al., 2016; Bekkby et al., 2017) and while choosing a restoration site; overlooking threats and manmade stressors such as eutrophication, altered hydrology and physical damage that impede natural regeneration often results in failures (Bayraktarov et al., 2016).

A recent approach, promoting the ecosystem services framework and the restoration of the natural capital (Blignaut et al., 2014a; Aronson et al., 2017), calls for a family of restorative activities (which may involve some of the actions, for example, in Figure 1) that can be carried out simultaneously or sequentially, to scale up restoration to lager spatial scales (Aronson et al., 2017). The terminology or inter-use of these activities is not always clear and uptake by researchers and practitioners will certainly differ, but we include as examples the following restoration options and terms:

Ecosystem **Recovery**, which is a fundamental term in many definitions, is the ability of a habitat, community or individual (or individual colony) of species to redress damage sustained as a result of an external factor (Elliott et al., 2007). Implicitly, ecosystem recovery implies returning a distressed ecosystem to a healthy condition (Duarte et al., 2015).

Protection sits between unassisted recovery where activities are regulated, and assisted recovery, where an intervention may reduce the causes of decline, for example, the removal of sea urchins causing barrens.

Remediation, defined as the action 'to rectify or make good' by Bradshaw (2002) has emphasis on the process rather than the end point (i.e. not necessarily complete restoration) and can encompass a range of approaches to restore or enhance a site's ecological value, from non-intervention through to habitat **enhancement** (Elliott et al., 2007). The term remediation is often used for the environmental clean-up of polluted areas, and the term enhancement for management actions that improve a habitat or increase a site's goods and services (e.g. increased numbers of over-wintering wading birds on an estuary (Elliott et al., 2007).

Ecological engineering is defined by Mitsch (2012) as "the design of sustainable ecosystems that integrate human society with its natural environment for the benefit of both". It concerns the manipulation of natural materials, living organisms and the physico-chemical environment to achieve specific human goals and solve technical problems (SER, 2004) and includes the restoration of human-disturbed ecosystems and the development of new sustainable ecosystems that have both human and ecological values.

Recuperation is the partial recovery of ecosystem-based productivity and services, including in a terrestrial example reforestation and soil health, where its goal is to bring a degraded site, land, or ecosystem back to a state where sustainable use is once again possible (Aronson et al., 2017).

Reconstruction is a restoration approach where the appropriate biota need to be entirely, or almost entirely, reintroduced as they cannot regenerate or recolonise within feasible time frames, even after expert assisted regeneration interventions (MacDonald et al., 2016) and often natural regeneration, assisted regeneration and reconstruction would be needed to achieve the restoration goals (Bekkby et al., 2017, deep-sea coral garden example from the Azores).

Creation is an anthropogenic intervention which produces a habitat not previously there; for example, artificial reefs placed on an otherwise sandy sea bottom should be regarded as creating new habitat aiming to increase the biodiversity of an area rather than replacing lost habitat (Elliott et al., 2007). Creation, in other words, is the intentional fabrication of an ecosystem (different from the one previously occurring on a site) for a useful purpose without a focus on achieving a reference ecosystem (MacDonald et al., 2016). Habitat creation can be the outcome of compensatory actions arising from legal obligations in cases that habitat loss cannot be avoided. Habitat **re-creation** is about re-creating a habitat that was present within historical records. As part of re-creation, **re-introduction** is the replacement of an ecosystem's structural component, i.e. re-introducing a structuring species in sufficient quantities to allow it to regain its ecological functioning. A species may be re-introduced into an area from where it disappeared until its population becomes re-established and to hopefully sustainable and self-maintaining sizes.

Rehabilitation comprises of direct or indirect actions with the aim of reinstating a level of ecosystem functionality where ecological restoration is not sought, but rather the renewed and on-going provision of ecosystem services and goods (MacDonald et al., 2016). Rehabilitation, according to Elliott et al. (2007), is the activity of partially or fully replacing structural or functional characteristics of an ecosystem that have been lost.

Ecological restoration defined as the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (SER, 2004; Clewell & Aronson, 2013), is the process of re-establishing, following degradation by human activities, a sustainable habitat or ecosystem with a natural (healthy) structure and functioning, including the re-establishment of the pre-existing biotic integrity in terms of species composition and community structure (SER, 2004).

In contrast to the restoration activities noted above, the intentional creation of biotic assemblages whose species have been selected in the design process to serve a specific purpose, are called **designer ecosystems** (Clewell & Aronson, 2013). These terms highlight the extent of human interventions from do-nothing, hands-off, i.e. let continued human induced degradation and nature fight-it-off/find a new state, to all-hands-on and skills-on-board to creating novel purpose-built ecosystems.

Human intervention is almost always required to facilitate restoration. Perhaps the only exception to this is in the abandonment of activities that have caused ecosystem degradation. Beyond that the intervention may lie as pointed out above in management activities or in some hand-on intervention from removing problems, to rebuilding structures or providing physical environments (creating the correct abiotic and biotic conditions) to seeding and transplanting (Figure 1). Recently, Bayraktarov et al. (2016) have reviewed restoration methodologies pertaining to the most common targets for restoration, including coral-reefs, seagrasses, mangroves, salt marshes and oyster reefs. The most successful restoration projects methodologies used collection of base stock (for example, corals for fragmenting and on-growing, macrophytes for seeds, seedlings, sprigs shoots or rhizomes), ex-situ and in-situ nursery culture conditions followed by out-planting and transplanting. Mixed approaches that include facilitation, biological engineering and introduction of structures have been tried out with high success rates and/or promising results; examples of successful mangrove restoration projects included facilitation by clearing out invasive plants, hydrological restoration, site contouring and excavations along with planting smooth cord grass to trap mangrove seeds; while saltmarsh projects emphasize the linking of ecological aspects and engineering and bioengineering solutions (Elliott et al., 2007; Bayraktarov et al., 2016). Humans are not the only facilitators, as other species may also facilitate marine restoration, with perhaps the largest scale example of this being mussels facilitating seagrass restoration in the Wadden Sea, by improving water quality and increasing the likelihood of seed settlement and shoot survival, and consequently bed recovery (e.g. van Katwijk et al., 2009).

1.4. Starting and end-points of restoration

The target of ecological restoration should be degraded ecosystems (SER, 2004; McDonald et al., 2016) but, as the relevant literature reveals, the target is quite often taken to be habitats, communities or even individual species. The EU Habitats Directive refers to restoration of habitats (Article 6(1)) and defines natural habitats as "terrestrial or aquatic areas distinguished by geographic, abiotic and biotic features, whether entirely natural or semi-natural". The Marine Strategy Framework Directive and Marine Spatial Planning Directive both utilise the term 'marine environment' (Recital 43 in the former for preservation, protection and restoration, and in Article 5 of the latter concerning preservation, protection and improvement). Both the CBD (Aichi targets) and the EU Biodiversity Strategy refer to restoring 'degraded ecosystems'. The European habitat mapping system, EUNIS, defines habitat as "plant and animal communities as the characterising elements of the biotic environment, together with abiotic factors (soil, climate, water availability and quality, and others), operating together at a particular scale" (Davies et al., 2004). Environmental status assessments usually require integration of multiple ecosystem components such as species and broad scale habitats as well as spatially defined outputs (Borja et al., 2016).

This plurality of definitions often leads to an overly broad use of the term "habitat", which may create confusion with the notion of ecosystem – defined in a restoration context by Clewell and Aronson (2013) as "the complex of living organisms and the abiotic environment with which they interact at a specified location". Within this project we use all three terms as and when other authors have used the terms, but remain with the definition of McDonald et al. (2016) defined above. Structural or key species and habitats are often stated as the focus of various restoration actions and projects, as key components of the ecosystems to be restored. In many cases this is the starting point of restoration but discussions and new approaches are trialled in order, for example, to maintain overall biodiversity including genetic diversity (Schopmeyer et al., 2012; Ort et al., 2014).

2. Unassisted Restoration or Spontaneous Regeneration

Unassisted regeneration of a damaged ecosystem encompasses two aspects, the removal of degrading mechanisms and the natural recovery of the ecosystem. Within the Society of Ecological Restoration key concepts underpinning best practices for ecological restoration (MacDonald et al., 2016), one of the first key attributes to be identified and addressed prior to and during restoration work is elimination or the 'Absence of Threats', such as overutilization of resources by people, biological invasions, and contamination. The causes of degradation need to be removed or at least controlled. In some cases the removal of threats allows natural (spontaneous) regeneration of the target ecosystem and there is need for further human intervention to accelerate recovery. Sometimes optimistically called 'passive restoration' or Monitored Natural Recovery (Rohr et al., 2015), this approach nonetheless calls for a full process of site investigation, long-term monitoring and ability to carry out interventions if recovery is not seen to be proceeding as predicted (Rohr et al., 2015; Fuchsman et al., 2014). A clear and useful definition of this approach is as follows:

Unassisted (or Natural or Spontaneous) regeneration *– Germination, birth or other recruitment of biota including plants, animals and microbiota, whether arising from colonization or in situ processes. A 'natural regeneration' approach to restoration relies on increases in individuals, without direct planting or seeding, after the removal of causal factors alone, as distinct from an 'assisted natural regeneration' approach that depends upon active intervention (Prach & Hobbs, 2008; Clewell & McDonald 2009).*

As threat removal is clearly one of the key steps in the restoration process, the threats need to be clearly identified and analysed. These threats can be identified as coming from individual pressures on marine ecosystem, caused by a variety of activities. The removal of threats could be considered as the creation/removal of barriers whether they are regulative or physical. In the marine environment this is in almost all cases regulative, although physical barriers may be removed or added in the coastal zone (e.g. breaching seawalls for saltmarsh recovery (French, 2006), adding semi-permeable dune fencing to enhance natural recovery (Nordstrom & Jackson, 2013)), because the creation of physical barriers is mostly impractical, although concrete barrier fields or artificial reefs have been installed to limit trawling activities (Iannibelli & Musmarra, 2008; Fabi et al., 2011). Anthropogenic activities can be regulated, thus reducing pressure impacts and consequently ecosystem threats.

Pressures are considered to be the mechanism through which an activity has an actual or potential effect on any part of the ecosystem, for example, for demersal trawling activity, one individual pressure would be abrasion of the seabed (Robinson et al., 2008) but multiple pressures arise from each activity (Smith et al., 2016). Pressures can be divided into two classes, endogenous and exogenous pressures (Elliott, 2011): endogenous pressures are those emanating from within the system that we can control (manageable – we can respond to causes and consequences), e.g. abrasion on the seabed caused by trawling activities; exogenous pressures on the other hand are those emanating from outside the system that we cannot primarily control (unmanageable – we can only respond to the consequences) and can be seen to be natural events and mostly on wide-area scales, e.g. change in seabed morphology from tectonic events. Pressures and activities relating to restoration have been detailed and reviewed in a separate deliverable (Smith et al., 2017).

2.1. Limiting or Preventing Impacts

Limiting the amount or extent of exogenous pressures is primarily achieved by controlling the activity causing the pressure. For instance, in the case of abrasion to the seabed this would be limiting those activities causing abrasion, which include fishing, dredging and anchoring. Limitations may be enabled by regulation, with partial limitation by constraining the extent in time or space, or degree of impact, or full limitation by banning/removal of the activity in total in a specific area.

Limitation may be sectoral-specific to one activity (for example, a fishing closure) or multi-sectoral, where a range of activities are regulated (for example, establishment of a protected area with no activities allowed). Regulation can be enacted at many different levels mostly through legislation although at some low levels this may be voluntary (e.g. voluntary no spear-fishing in a known diving site). There is great complexity to marine activity regulation and it can be enabled at many different levels covering different marine areas (see Figure 2 indicative regulative coverage of marine waters). Individual activities may fall under different legislation depending where they are taking place. Boyes et al. (2016) showed the complexity of geographical scope of European Policies/Directives along within Member State waters (see Figure 3).

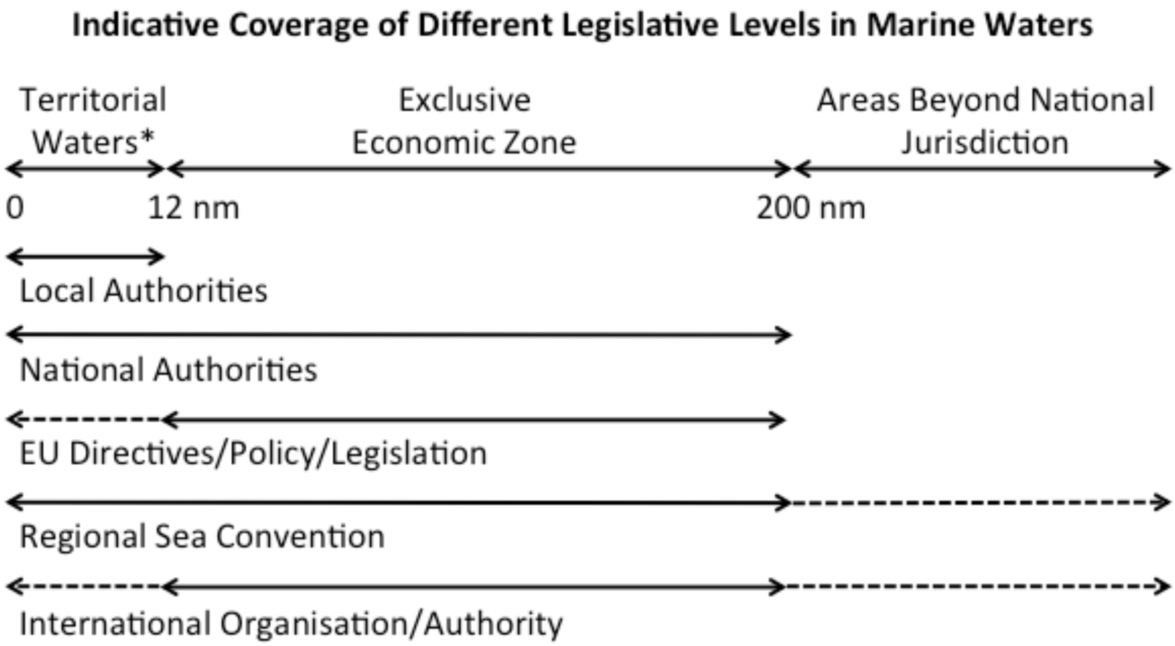


Figure 2. General indicative coverage of different legislative levels in marine waters. *Territorial Waters: may be subdivided further for various legislations. 12 nm is the maximum extent of sovereign territory and 200 nm for exclusive economic zones under UNLCOS.

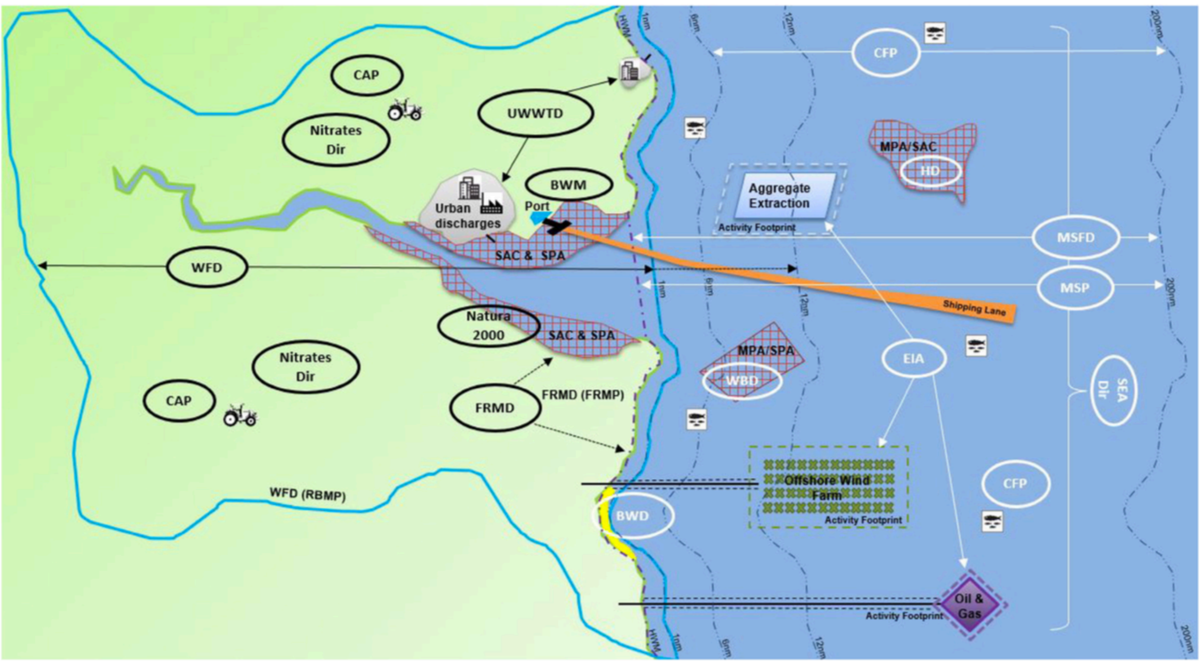


Figure 3. Geographical coverage of European Directives/Policies from Boyes et al. (2016). Abbreviations:

BWD = Bathing Water Directive; BWM = Ballast Water Management Convention; CAP = Common Agricultural Policy; CFP = Common Fisheries Policy; EIA = Environmental Impact Assessment Directive; FRMD = Flood Risk Management Directive; FRMD (FRMP) = Flood Risk Management Directive (Flood Risk Management Plan); HD = Habitats Directive; MSP = Maritime Spatial Planning Directive; MSFD = Marine Strategy Framework Directive; Natura 2000 = Habitats and Wild Birds directives; Nitrates Dir = Nitrates Directive; SAC = Special Area of Conservation; SEA Dir = Strategic Environmental Assessment Directive; SPA = Special Protection Area; UWWTD = Urban Wastewater Treatment Directive; WBD = Birds Directive; WFD = Water Framework Directive (with extension out to 12 nm for chemical status); WFD (RBMP) = Water Framework Directive (River Basin Management Plan).

In Europe, marine activities and their impacts were initially tackled by the regional sea conventions or individual sectoral policies (Boyes and Elliott, 2014). The Member State's original pre-accession sectoral polices were on the whole replaced by European policies, frameworks or directives (see Box 1), legislated either directly in European law or integrated/modified in the Member State law. Figure 4 shows the relationship between different levels of legislation control in the United Kingdom covering any form of marine environmental protection. For example, for the fisheries sector (12 o-clock axis in Figure 4), regional sea conventions (primarily OSPAR for the UK) are in the centre, with ICES feeding advice to the CFP in the EU, which has a set of basic fishery regulations which are also transposed into law in various Sea Fish and Fisheries Acts which may contain UK-specific local restrictions. Fishing vessels are also controlled through UNCLOS (UNCLOS, 1982) and the IMO at high level, and MARPOL through a variety of other directives and laws (Figure 4; 8 o'clock axis). Environmental protection which includes protection from fishing activities runs through (Figure 4; 3-5 o'clock axis) UN CBD, various other conventions, through the EU in the Integrated Maritime Policy, MSP directive, MSFD, Biodiversity Strategy, Birds and Habitats Directives and then into a various UK Acts and Regulations.

Box 1. European Legislative Terminology

Regulations

A "regulation" is a binding legislative act. It must be applied in its entirety across all EU member states. For example, when the EU wanted to make sure that there are common safeguards on goods imported from outside the EU, the Council adopted a regulation. A shipping example is the EU regulation on rules and standards for ship inspection and survey organisations (Regulation (EC) No 391/2009) and a marine resources example is the COUNCIL REGULATION (EC) No 1967/2006 of 21 December 2006 on prohibitions of fishing gear that is too harmful to the marine environment or leads to the depletion of certain stocks.

Directives

A "directive" is a legislative act that sets out a goal that all EU countries must achieve (binding on the objective). However, it is up to the individual countries to devise their own laws on how to reach these goals. Directives may establish a Framework or Policy. The Marine Strategy Framework Directive sets out the framework around which the marine environment is to be protected. The individual Member States decide exactly how to do this and what the permissible targets are.

Decisions

A "decision" is binding EU law on those to whom it is addressed (e.g. an EU country or an individual company) and is directly applicable. A recent example is the 2010/477/EU: Commission Decision of 1 September 2010 on criteria and methodological standards on good environmental status of marine waters (notified under document C(2010) 5956) Text with EEA relevance (http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32010D0477(01).

Recommendations

A "recommendation" is not binding. A recommendation allows the institutions to make their views known and to suggest a line of action without imposing any legal obligation on those to whom it is addressed. The 2002 Recommendation on Integrated Coastal Zone Management (http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32002H0413) defines the principles of sound coastal planning and management without this having any legal consequences.

Policies

A policy is a guideline used to regulate organisational affairs. EU policy is designed to support national action and help address common challenges. The FAO defines policy as a set of decisions that are oriented towards a long-term purpose or to a particular problem. Examples of EU marine policies include the Common Fisheries Policy or the EU Integrated Maritime Policy.

Based on: https://europa.eu/european-union/eu-law/legal-acts_en#directives

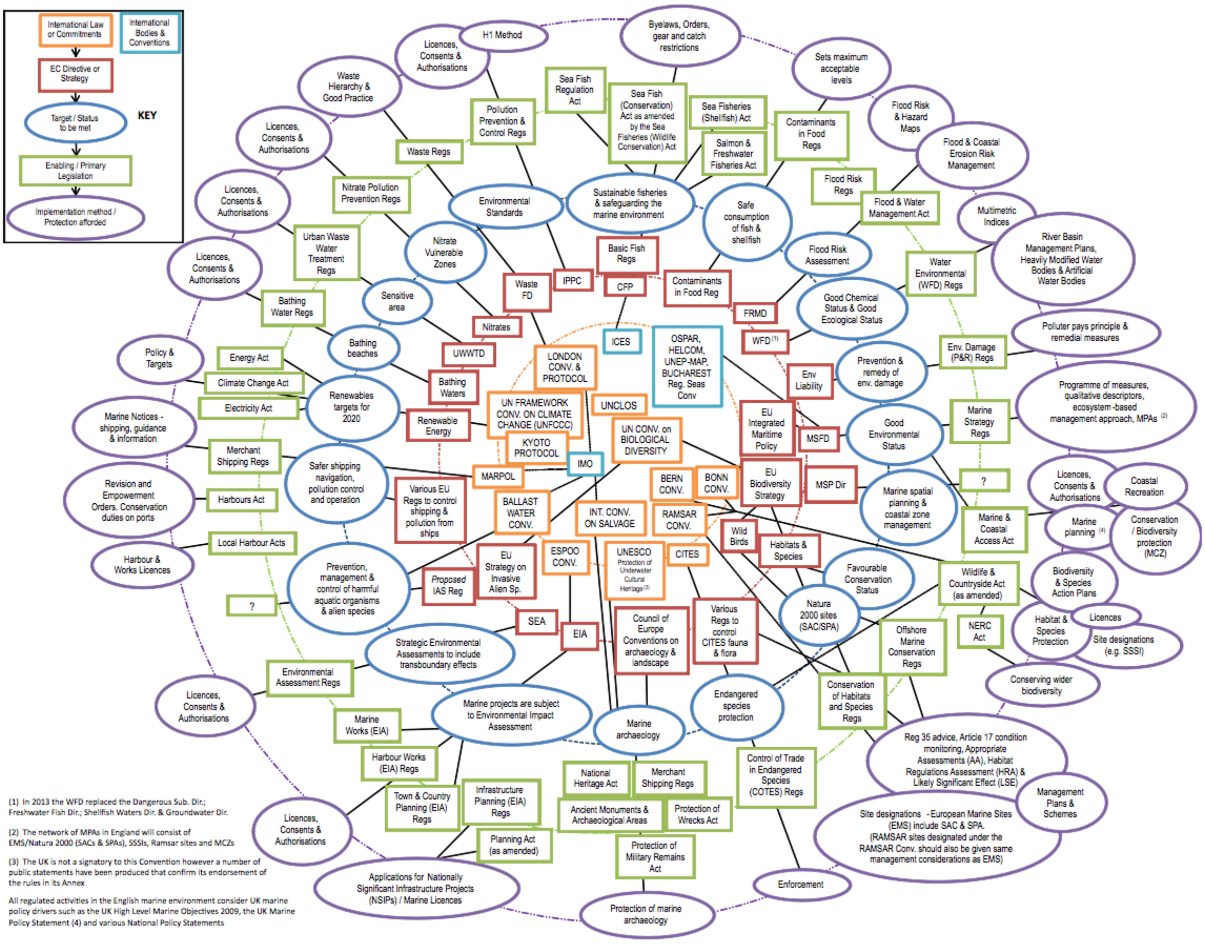


Figure 4. International, European and English legislation giving protection to the marine environment from Boyes & Elliott (2014). Central Blue Boxes, International Bodies or Conventions; Orange Boxes, International Laws or Commitments; Red Boxes, EU Directive or Strategy; Blue Circles, Target/Status to be met; Green Boxes, UK Enabling/Primary Legislation; Purple circles, Implementation method/protection afforded.

Whilst UNCLOS regulates general marine activities (principally shipping related) in Areas Beyond National Jurisdiction (ABNJ), the International Seabed Authority (ISA) regulates the seafloor in these areas. The ISA plays the major organisational role in the emerging industry of deep-sea mining (e.g. polymetallic nodules, polymetallic sulphides, cobalt rich ferromanganese crusts). It reviews applications, draft regulations and endure that mining companies comply with environmental rules, the latter of which may involve mitigation, remediation and restoration measures required by the companies. Recently, the issue of the guiding principles and objectives that should govern the activities and decisions of the ISA and, by extension, other regulatory bodies and actors in deep-sea mining was addressed within the framework of MIDAS project (Tinch & Van den Hove, 2016).

From the top down, international conventions or directives may drive restoration with generalised requirements to restore degraded ecosystems or more specific targets to restore fixed percentages of degraded ecosystems. Most restoration projects will be within Exclusive Economic Zones (EEZ's) or territorial waters, at localised sites, and enabling legislation to limit degrading activities would be at the member state local level, for example, a fishing closure area. There may be some exceptions to this, where multi-national fishing fleets may be controlled through common regulation, for example, the ban on trawling deeper than 1000 m in the Mediterranean through the General Fisheries Council of the Mediterranean (GFCM) and 800 m in waters controlled by the EU and the Eastern Central Atlantic. Another example is the recent adoption by the GFCM (October 2017) of the EU proposal for the establishment of a bilateral Fisheries Restricted Area (FRA) in the Jabuka/Pomo Pit, between Italy and Croatia, banning demersal fisheries and protecting important shared hake and scampi nursery grounds.

The creation of MPAs may also be used to limit degrading activities both in territorial waters and EEZ's under national laws as well as international waters under international convention/regulation (see Box 2). Large or very large MPAs (LMPAs, VLMPAs) are being used to reach international convention objectives, especially in deep waters (see Figure 5) and ABNJ (see Bastari et al., 2016; UNEP/MAP, 2015 for the Mediterranean). ABNJ have in general little governance, although UNCLOS establishes a general regime. Some regional seas or States have managed to apply some conservation policies to particular areas (Bastari et al., 2016) for MPAs, but also OSPAR for the Convention for the Protection of the Marine Environment in the North East Atlantic and the Commission for the Conservation of Antarctic Marine Living Resources. A new instrument is in development by the United Nations for Biodiversity Beyond National Jurisdiction (BBNJ) concerning conservation and sustainable use of biodiversity in ABNJ. These areas cover some 64% of the world's oceans (see Figure 6).

Marine Spatial Planning (MSP) may also be used to apportion marine areas for no- or low-impact activities and conservation areas. For example, offshore renewable energy areas may have small actual footprints of activity within large zones where other activities that are degrading may not be allowed, thus allowing large areas of natural recovery.

Box 2: Marine Protected Areas (MPAs)

The IUCN has defined an **MPA** as "any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment" (Kelleher, 1999). The definition goes on to note that there are two ways of establishing MPA systems: either as many relatively small sites, each strictly protected, or as a few large multiple-use areas which contain strictly protected areas within them.

MPAs may be enacted in many different ways and under a variety of different designations and laws. The different designations may either apply to the type of law that they are enacted or the degree of protection. Some may have a total ban on extractive and destructive activities (fully protected or no-take areas or Marine Reserves), others only on selected activities (partially protected areas, PPAs), whilst others may have multiple zones (multiple-use areas) with different activities banned or allowed in different zones (PISCO and UNS, 2016). In addition to established National Parks and Marine Reserves, other examples of MPA types in the European seas, designated under different legislative frameworks, may include:

- FRA: Fisheries restricted areas. Areas with some form of restrictions on fisheries (typically trawling, seining, lining or netting).
- NATURA Sites: NATURA 2000 sites are European networked designated sites of importance (core breeding or resting sites for threatened species covering 18% of EU land area and 6% of marine territory). However, not all NATURA sites are protected by legislation and most of these lack any management plan or systematic surveillance.
- SPAMI: Specially Protected Areas of Mediterranean Importance: 34 marine and coastal sites.
- VME: Vulnerable Marine Ecosystems: FAO designation for groups of species, communities or habitats that may be vulnerable to impacts from fishing activities in areas beyond national jurisdiction. Regional Fisheries Management organizations implement a 'move on rule' which requires that fishers stop fishing and move away at least 2 nautical miles from VMEs if certain indicator species are caught. The revised EC Deep-sea Fisheries Access Regime requires implementation of the 'move on rule' in the North East Atlantic if VMEs are encountered between 400 and 800 m.

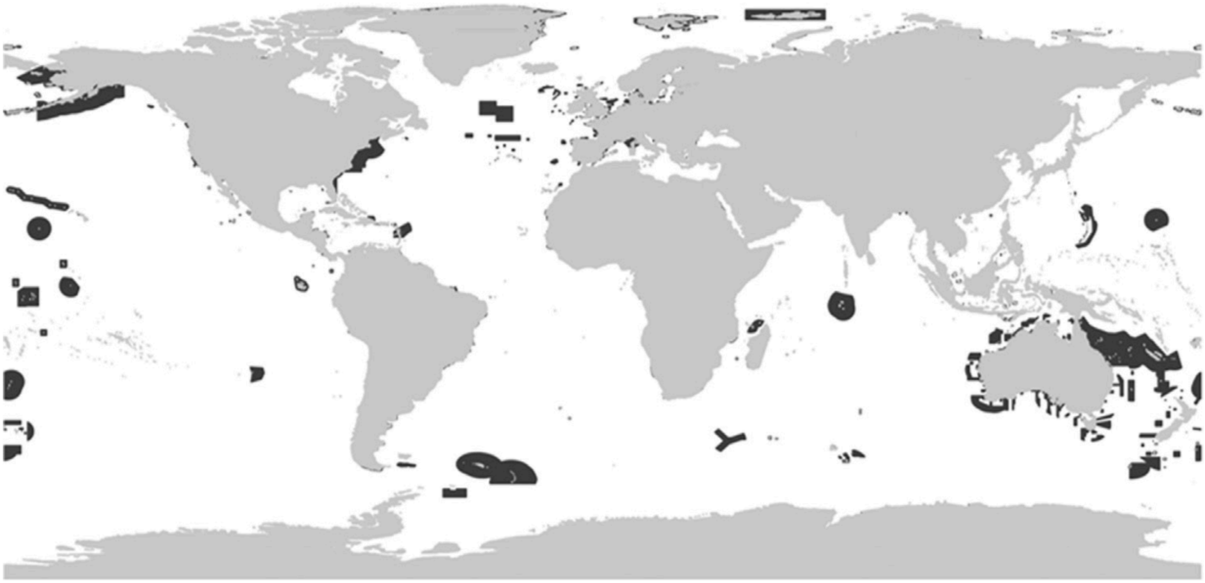


Figure 5. World-scale existing MPAs and LMPAs (black polygons) from Bastari et al. (2016).

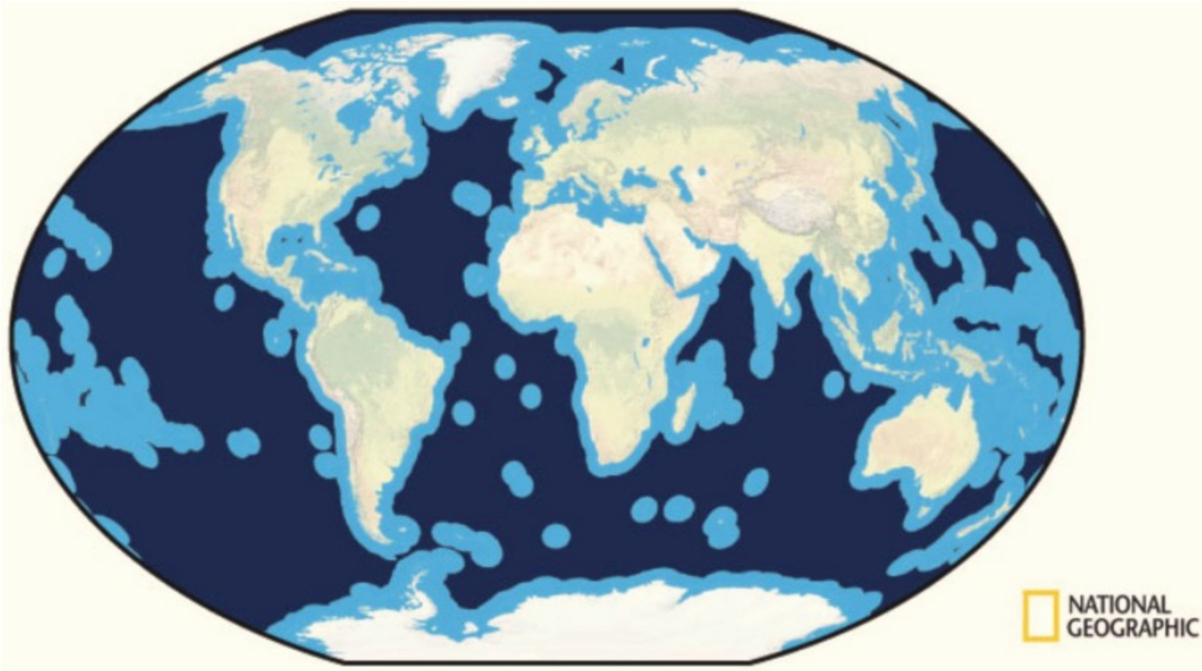


Figure 6. Areas Beyond National Jurisdiction – dark blue colour, covering 64% or the worlds' oceans. Areas of national jurisdiction are shown in light blue (from National Geographic).

A recent meta-analysis revealed that partially protected MPAs, where some activities are restricted (e.g. ban of trawling), are not as effective as no-take marine reserves, which are by far the most effective type of MPA for restoring biomass and structure of fish assemblages and ecosystems to a more complex and resilient state (Sala & Giakoumi, 2017). However, in several cases (e.g. the vast majority of Mediterranean MPAs), MPAs are weakly enforced, also called "paper parks", because they are only designated on paper and lack any formal management (Abdulla et al., 2008; PISCO and UNS, 2016). In any case, a strong part of regulating an activity within a given area, is to ensure that the regulation is enforced, otherwise ecosystem degradation may continue preventing natural recovery. This is crucial for the recovery of species and assemblages characterized by low resilience and/or slow recovery potential (e.g. deep-sea cold-water corals) (Huvenne et al., 2016).

The cost of natural recovery as a restoration process is not necessarily low because of the cost of threat removal, which as well as the cost of legislation and enforcement, may also need some level of compensation to the activity. Threat removal and law enforcement for example in the form of marine patrols to enforce blast fishing bans has been found to be more cost-effective than rehabilitation in Komodo National Park, Indonesia (Haisfield et al., 2010). An obvious benefit is that more fish recruit near living coral assemblages than dead coral, while policing healthy reefs offers additional biodiversity protection from other forms of illegal fishing. In most cases however a combination of management actions would be necessary to assist natural recovery. Monitoring programmes should also be in place to ensure that natural restoration proceeds well and no further threats appear.

3. Active Restoration

The removal of a disturbing factor or a set of pressures will allow, eventually, the natural recovery of an ecosystem, yet this depend on intrinsic properties, such as recoverability, resilience, adaptation and carrying capacity (Elliott et al., 2007), and may be a slow process, particularly slow in the case of very degraded environments (Bradshaw, 2002). To shorten the time-scale of recovery, humans intervene with management actions and techniques that aim to enhance recovery, generally termed active restoration (Elliott et al., 2007; Bayraktarov et al., 2016; Geist & Hawkins, 2016).

Human-mediated restoration actions are grouped by Elliott et al. (2007) in two major categories: (a) those used to redress a degraded environment and (b) those in response to the effects of a single stressor. The first group involves restorative activities falling within restoration, rehabilitation, remediation, re-creation, re-introduction, re-establishment, reclamation and replacement actions; the second category involves mitigation, compensation, habitat enhancement and creation actions. These terms are derived from terrestrial restoration ecology and are most often used interchangeably. There are numerous definitions (SER, 2004; Clewell & Aronson 2013; Elliott et al., 2007; McDonanld et al., 2016; Aronson et al., 2017; Section 1.3. this report) and while some experts put a lot of emphasis in clear definitions of terms, others use the term restoration in its very broad sense without bothering too much about semantics noting however, that restoration should be to a defined state (Miller & Hobbs, 2007; Geist & Hawkins, 2016). Habitat restoration should include assessing the current status and moving the "habitometer needle" from the current situation (some degradation) to a high(er) habitat value and the final restoration target (Miller & Hobbs 2007).

Having decided on the restoration goal based on ecological parameters and economic and financial constraints of target species, most often a key structural species, then the choice of method(s) of active intervention(s) are equally important. Restoration techniques for aquatic ecosystems may be static and structural, such as the introduction of biogenic reefs assisted by artificial structures, or process-related and as with the marine environment may involve interventions at the physical, hydrological and biological setting of a site (Elliott et al., 2007; Geist & Hawkins, 2016).

A wide range of methods is used as means for restoring degraded marine ecosystems (an overview of active restoration methods for five coastal and marine ecosystems is provided by Bayraktarov et al. (2016), see also this report Section 3, and Section 4 for new approaches).

Depending on the site history, and problems at hand multiple interventions might be required from structural, physical and chemical improvement, to biological solutions, and approaches from biological stressor removal (e.g. invasives, grazers), to *in situ* and *ex situ* nurseries, to transplanting with the aid of artificial support and/or facilitation (Maxwell et al., 2017 and this report Section 4). Active restoration approaches have been applied to both open marine and coastal systems, especially with regards to biogenic structure loss and to restoring keystone and engineer species, but are still lacking for several marine ecosystems, the most prominent one being the deep sea (Van Dover et al., 2014).

3.1. Active restoration peer-review publications synthesis - Introduction

The conservation of nature and the management of human activities are considered effective approaches to limit the degradation of marine ecosystems and the services they provide. Current practices are clearly inadequate to reverse present trajectories of change. Ecological restoration has the potential to reverse land degradation, increase the resilience of biodiversity and deliver important ecosystem services (Perring et al., 2015) but marine restoration is still at its infancy, due to the many gaps among current implementation methods and a substantial inconsistency in the evaluation of restoration strategies.

To investigate further, a review of studies has been undertaken, dealing with active restoration published in the last 25 years at the global scale across coastal habitats spanning from semi-terrestrial to strictly marine environments. In particular, the general aim of the review was to assess and summarize the methodological trends (both spatial and temporal pattern) in the field of active restoration, providing a framework on where and how restoration was carried out and with what outcomes. In contrast to previous quantitative assessments of restoration actions, the adopted approach allows the identification of potential drivers of success and, thus, the provisioning of specific recommendations aimed to improve the strategies addressing the urgent conservation issues faced by countries at the global scale.

Whilst restoration is being widely incorporated into natural resource strategies from the local to global level, ecological restoration is among the most expensive conservation actions worldwide (Holl et al., 2003). Consequently, understanding where and why restoration activities are more likely to be successful is critical for the development of the practice and to improve its efficiency.

The success, or failure, of restoration is dependent upon a wide range of factors, including the method(s) used (Perring et al., 2015), the species or habitat being restored (Chang et al., 2016), as well as more general contextual factors such as location (Darwiche-Criado et al., 2017), the degree of protection within the area (Keenleyside, 2012) and the duration of the project. However, to date, much of the information from past and on-going restoration studies remains untapped. As a result there is an opportunity to build upon and learn from the failures and successes of the past in order to identify the key determinates of restoration success. Therefore in addition to conducting a world review of restoration looking at where the emphasis is in terms of countries, habitats, species etc., we also performed modelling to specifically look at determinants of restoration success.

3.2. Methods

3.2.1. Review Methods

A systematic literature review was conducted consisting of three steps: (1) article identification using two search engines, (2) abstract screening and (3) review of pertinent articles (Figure 7). The aim of this activity was to provide a representative framework of active restoration practices (*sensu* Elliott et al., 2007) carried out at the global scale across coastal habitats. Besides strictly marine environments, in order to make the overview as exhaustive as possible, information was also collected dealing with the semi-terrestrial realm, such as transitional water ecosystems (i.e. estuaries and wetlands), and typical interface ecosystems, such as mangrove systems. To gain a representative sample of the literature, two databases (the ISI Web of Science [WOS] and Scopus) were searched in the time frame 1985–2016 (cut-off date 27 September 2016). The bibliographic catalogue was built searching within the title, abstract, or keyword (but also "keyword plus", which is only available for WOS) the term [Restor* OR Rehab*] combined with the term [habitat/ecosystem 1* OR habitat/ecosystem 2*, … OR habitat/ecosystem *n*]. The first section of the search terms ensured that the bulk of the literature obtained related to restoration in the broadest terminological conception (see Introduction Section 1.3 and Elliott et al. (2007) for a comprehensive discussion on the issue related to terminology). Following the approach recently adopted by Bayraktarov et al. (2016), the second set of search criteria was designed to capture likely alternative terms for the same habitat/ecosystem on which the review is focused (e.g. coral or coral reef, mangrove or mangal, saltmarsh or salt marsh, shellfish or oyster reef). The following terms (n = 31) have been identified for this purpose: eelgrass, coral, coral reef, mangrove, mangal, saltmarsh, salt marsh, oyster reef, shellfish, mussel bed, rocky bottom, hard bottom, rocky shore, rocky intertidal, rocky subtidal, seafan, sandy shore, soft intertidal, soft subtidal, sandy bottom, soft bottom, mesophotic, seamount. In other cases (i.e. deep sea, canyons, rocky substrates, kelps, barrens, macroalgal forests, sponges, mudflats) the term “marine” was added to avoid misunderstandings.

From a total of 4066 collected publications including all kinds of document (e.g. peer-reviewed articles and/or reviews, meeting and congress proceedings), 3829 records were selected for further examination based on eligibility criteria (see details in Figure 7). Of the 3829 records, 498 studies were selected for full review based on the relevance of the study objectives with those of the present review as inferred from the abstract (i.e. eligibility). Articles were excluded if they: 1) were related to passive restoration interventions (e.g. passive recovery); 2) did not include a specific case study for which the restoration technique was developed (e.g. the study was not carried out into the field); 3) mentioned the term “restoration” only for justification or discussion of results but did not provide any actual intervention of active restoration (i.e. theoretical review, modelling, or entirely laboratory experiment/study); 4) the complete article was not available or was in a non-English language.

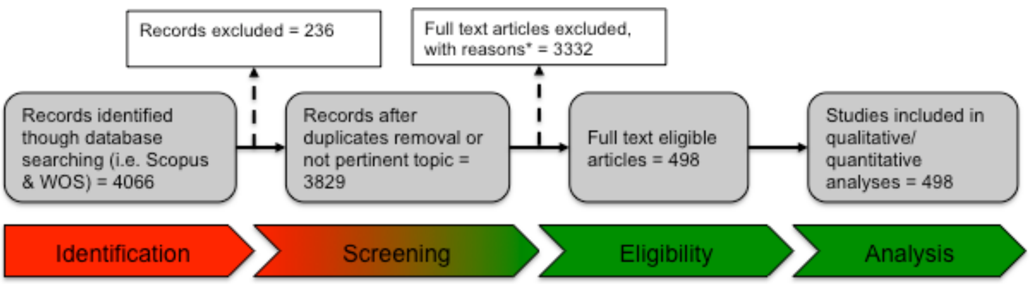


Figure 7. MERCES review flow diagram. * Reasons of exclusion are reported in the text.

From each eligible article, the following information was extracted:

- year of publication;
- geographical location of the case study, and the relevant marine biogeographic region (“realm” and “province” according to Spalding et al. (2003));
- the occurrence of some protection measures (e.g. MPAs, National Park, Sanctuary, Reserve, SPA, etc.);
- the ecosystem/habitat targeted by the restoration action;

- the target of Restoration action (i.e. degraded environment or single stressor oriented) and type of active (i.e. human mediated) restoration action (according to the conceptual model provided by Elliott et al. (2007), Figure 8);
- the temporal and spatial extent (i.e. respectively duration and scale) covered by each intervention;
- species or assemblages that were targeted (e.g. manipulated) by the study;
- the method/technology used and response variables considered to assess the outcomes of the specific intervention considered.

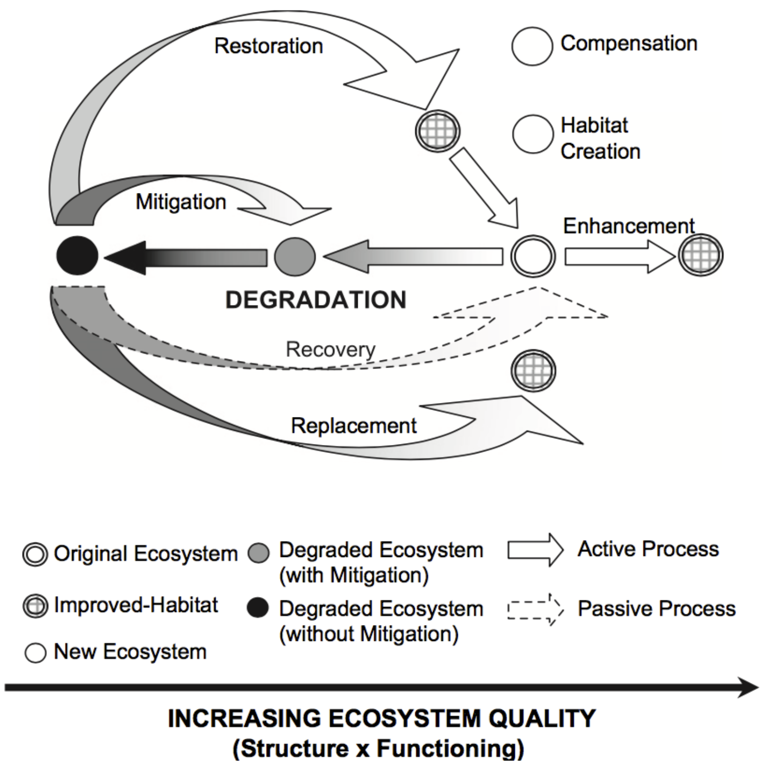


Figure 8. Conceptual model illustrating the processes of natural recovery and human-mediated restoration (= active restoration) of a degraded ecosystem through which ecosystem quality is increased to an improved or original state (from Elliott et al., 2007).

Moreover, additional proxies were identified in order to depict a cost/benefit framework of the specific approach developed in each study case. The following proxies were considered:

- the general context in which the study was carried out (e.g. field exercise, project);
- the synthetic outcome of the implemented restoration action (i.e. failure, success or a combination between the two);
- cost of the intervention (if reported);
- type of benefits coming from the restoration intervention (e.g. ecological, economical or both, and methodological).

A schematic representation of the database structure and rationale is reported below (Figure 9).

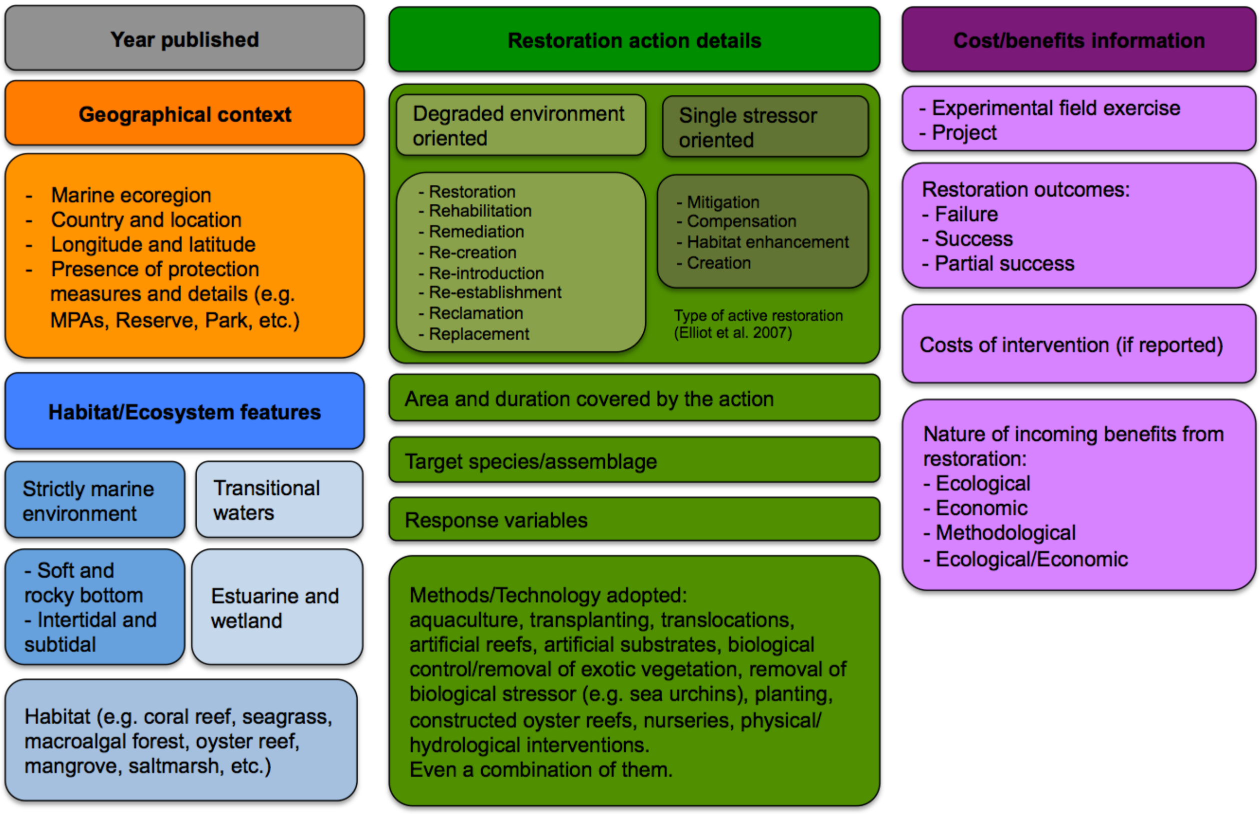


Figure 9. Conceptual model of the catalogue.

3.2.2. Modelling Methods

In order to provide robust, quantitative evidence-based recommendations for improving the application of restoration two broad questions were explored:

- What variables improve the probability of restoration being successful?
- What is their relative importance?

Data preparation

In order to aid the modelling exercise the explanatory variables were standardised and grouped into 9 categories relating to methodological approaches and contextual information (Table 1).

The outcome of the restoration activity (success, partial success and failure) and the potential explanatory variables were inferred directly from the papers considered. Furthermore, in order to provide more contextual information, a global map of cumulative human impacts was also used to determine the degree of human impact within the restoration site and the distance calculated from the site to the nearest protected area. As this was not available or applicable to all habitat types (and not applicable was not always discernible from not available), further refinements (data checking, fine tuning and data exploration) will be made towards a more in-depth version of this analysis in MERCES WP1 and WP5.

Table 1. Contextual and methodological variables considered in relation to restoration outcome.

Outcome		
Failure	**Partial success**	**Success**

Contextual variables				
Realm	**Ecosystem**	**Habitat**	**Degree of human impact**	**Distance to nearest protected area**
Arctic	Estuaries/ Wetlands	Corals	0-1	NA
Central Indo-Pacific	Rocky intertidal	Macroalgal forests		
Eastern Indo-Pacific	Rocky Subtidal	Mangroves		
Temperate Australasia	Soft-Bottom Intertidal	Other		
Temperate Northern Atlantic	Soft-Bottom Subtidal	Oyster reefs		
Temperate Northern Pacific		Saltmarshes		
Temperate South America		Seagrass		
Tropical Atlantic				
Tropical Eastern Pacific				
Western Indo-Pacific				

Methodological variables			
Type	**Duration**	**Method**	**Number of methods**
Creation Creation of habitat	< 1 month	Artificial habitat Construction of an artificial habitat	1
Mitigation Mitigating a threat	1-12 months	Growing Rearing and introducing/transplanting biota into the field	2
Re-establishment Re-establishing a habitat in an area it once existed	1-10 years	Habitat modification Physical modification of the habitat (e.g. hydrological modification)	3
Rehabilitation Reparation of ecosystem processes, services, and productivity	10-20 years	Introduction Introduction of biota to a new area	
Restoration Bringing a degraded ecosystem back into, as nearly as possible, its original condition.	20-30 years	Protection Improving water quality, creating sanctuaries etc.	
	> 50 years	Removal Clearing of exotic vegetation, pests etc.	
		Transplantation Movement of established biota into a degraded area	

Modelling approach

Using the body of evidence accumulated as part of this study, an extensive model-fitting approach was used in order to evaluate the relationship between restoration success and a range of predictor variables.

Since the outcome we were trying to predict is categorical in nature and has multiple levels (failure, partial success and success) "Multinomial Logistic Regression" statistical models (using the multinom function of the R-library nnet) was used. This is a form of linear regression analysis which is suitable in situations in which the dependent variable is nominal with more than two levels. The model works by calculating "log-odds", the ratio of the probability of success and the probability of failure, for all categories relative to a baseline. In this case the baseline was set as "failure" and determined the key variables that led to a positive change (either partial or full success) in relation to this level.

The number of factors was varied as well as their entry order, and all possible interactions and higher order terms considered. Each model was then ranked according to the Akaike information criterion (AIC). AIC is a likelihood-based measure of model fit that accounts for the number of parameters estimated in a model (i.e. models with large numbers of parameters are penalized more heavily than those with smaller numbers of parameters), such that the model with the lowest AIC has the 'best' relative fit, given the number of parameters included.

Models were ranked by their difference in AIC values; the lower the AIC value the more likely the model approximates the data. A 95% confidence set of models was then constructed beginning with the model with the highest weight and then continuously adding the model with the next highest weight, until the cumulative sum of weights exceeded 0.95. The predictors were then averaged across the 95% candidate set of models in order to derive model averaged results which take into account the uncertainty in the modelled estimates. Finally, the relative importance of each was determined variable by summing the Akaike weights for all models containing the variable. Predictor variables with a summed wi less than 0.5 were considered relatively unimportant.

3.3. Results

3.3.1. Review Results

Since 2000, a steady increase in the number of publications dealing with active restoration can be observed, resulting in a total of 498 records. All ecoregional realms (*sensu* Spalding et al., 2003) are represented in the catalogue, with the exception of the "Southern ocean" (Figure 10). In the last seventeen years, most of the efforts have been concentrated in the northern part of the globe with 58% of studies recorded (i.e. the Temperate Northern Atlantic and Pacific: 41% and 17% of studies respectively). The Tropical Atlantic and the Central Indo-Pacific also show relevant increases in restoration effort.

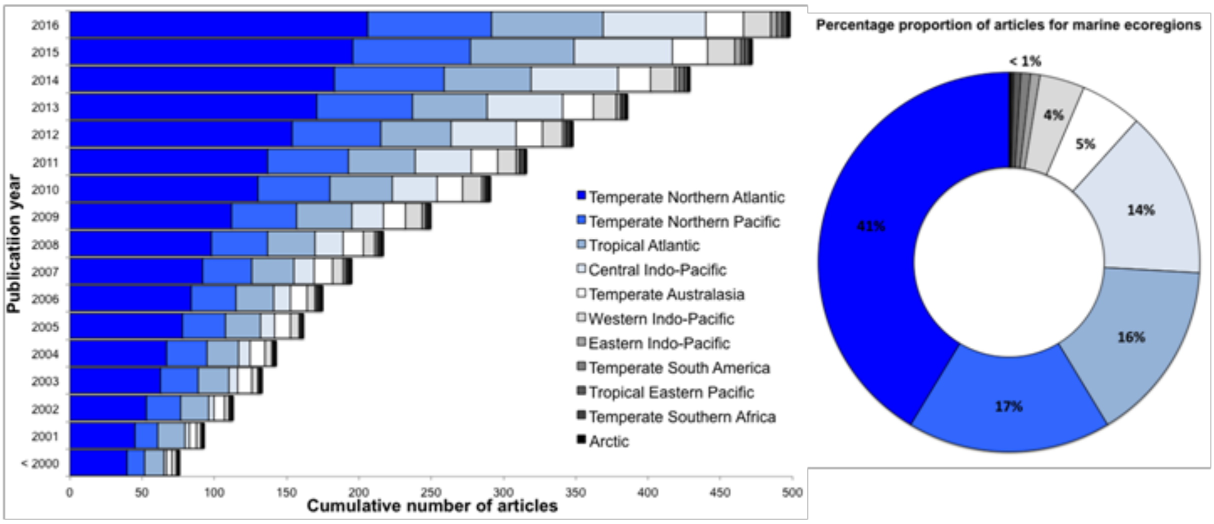


Figure 10. On the left side, the temporal and geographic cumulative number of articles is reported. On the right (donut graph), the total amount across ecoregions is shown. Colours represent the different marine "realms" according to Spalding et al. (2003).

The global map representing the geographic distribution of the studies in active restoration practices (Figure 11) shows that these efforts are not homogeneously distributed across provinces in each biogeographic realm (for details see Table 2). For instance, for the Temperate Northern Atlantic (i.e. the area showing the highest number of records) most of the studies have been carried out along the east coast of the USA (total 129 records). On the other side of this biogeographic region, much effort has been paid to the Mediterranean (35 records) as well as in Northern European Seas (29). Not all provinces in each biogeographic realm are covered by study cases (e.g. the Black Sea province).

Table 2. Total amount of studies recorded for each province (38) and realm (11).

Ecoregions	Total amount of studies
Arctic	1
1-Arctic	1
Temperate Northern Atlantic	206
2-Northern European Seas	29
3-Lusitanian	13
4-Mediterranean Sea	35
5-Cold Temperate Northwest Atlantic	75
6-Warm Temperate Northwest Atlantic	54
Temperate Northern Pacific	86
8-Cold Temperate Northwest Pacific	20
9-Warm Temperate Northwest Pacific	15
10-Cold Temperate Northeast Pacific	22
11-Warm Temperate Northeast Pacific	29
Tropical Atlantic	77
12-Tropical Northwestern Atlantic	61
14-Tropical Southwestern Atlantic	3
17-Gulf of Guinea	1
18-Red Sea and Gulf of Aden	12
Western Indo-Pacific	19
19-Somali/Arabian	2
20-Western Indian Ocean	6
21-West and South Indian Shelf	4
22-Central Indian Ocean Islands	3
23-Bay of Bengal	3
24-Andaman	1
Central Indo-Pacific	71
25-South China Sea	18
26-Sunda Shelf	15
28-South Kuroshio	5
29-Tropical Northwestern Pacific	2
30-Western Coral Triangle	28
33-Northeast Australian Shelf	3
Eastern Indo-Pacific	4
37-Hawaii	2

Ecoregions	Total amount of studies
39-Central Polynesia	1
40-Southeast Polynesia	1
Tropical Eastern Pacific	3
43-Tropical East Pacific	3
Temperate South America	4
45-Warm Temperate Southeastern Pacific	2
47-Warm Temperate Southwestern Atlantic	1
48-Magellanic	1
Temperate Southern Africa	1
51-Agulhas	1
Temperate Australasia	26
54-Southern New Zealand	1
55-East Central Australian Shelf	8
56-Southeast Australian Shelf	3
57-Southwest Australian Shelf	14

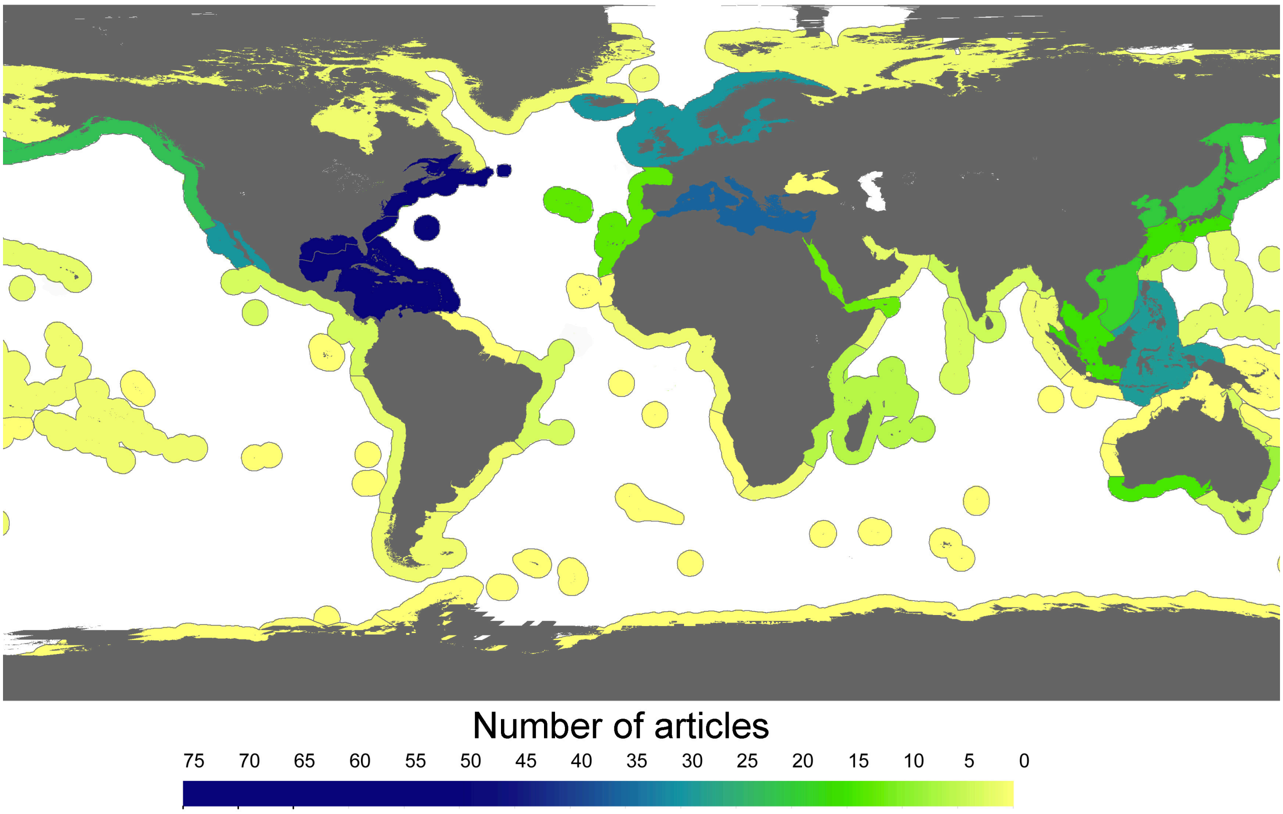


Figure 11. The geographic partitioning of the research effort expended in active restoration practices. Colour scale represents the total amount of articles across marine "provinces" from Spalding et al. (2003).

As far as the restoration effort partitioning across habitats/ecosystems (Figure 12 A,B), it is possible to observe a substantially consistent increase both in strictly marine and estuarine/wetland systems (Figure 12A). The highest number of studies has been recorded in estuarine/wetlands system (42%), followed by rocky reefs (30% considering both intertidal and subtidal), and finally soft-bottoms environments (28%). In particular, studies for strictly marine environments started to exceed studies on estuarine/wetlands system from 2008, as a result of an increasing number of studies carried out in the rocky subtidal (mainly focused on coral reef habitat, Figure 12B) and in soft-bottom systems (mainly seagrass and mangrove habitats, Figure 12B). The most targeted habitat of restoration actions is represented by saltmarshes (in total 131 documents, Figure 12B) followed by seagrasses and coral reefs.

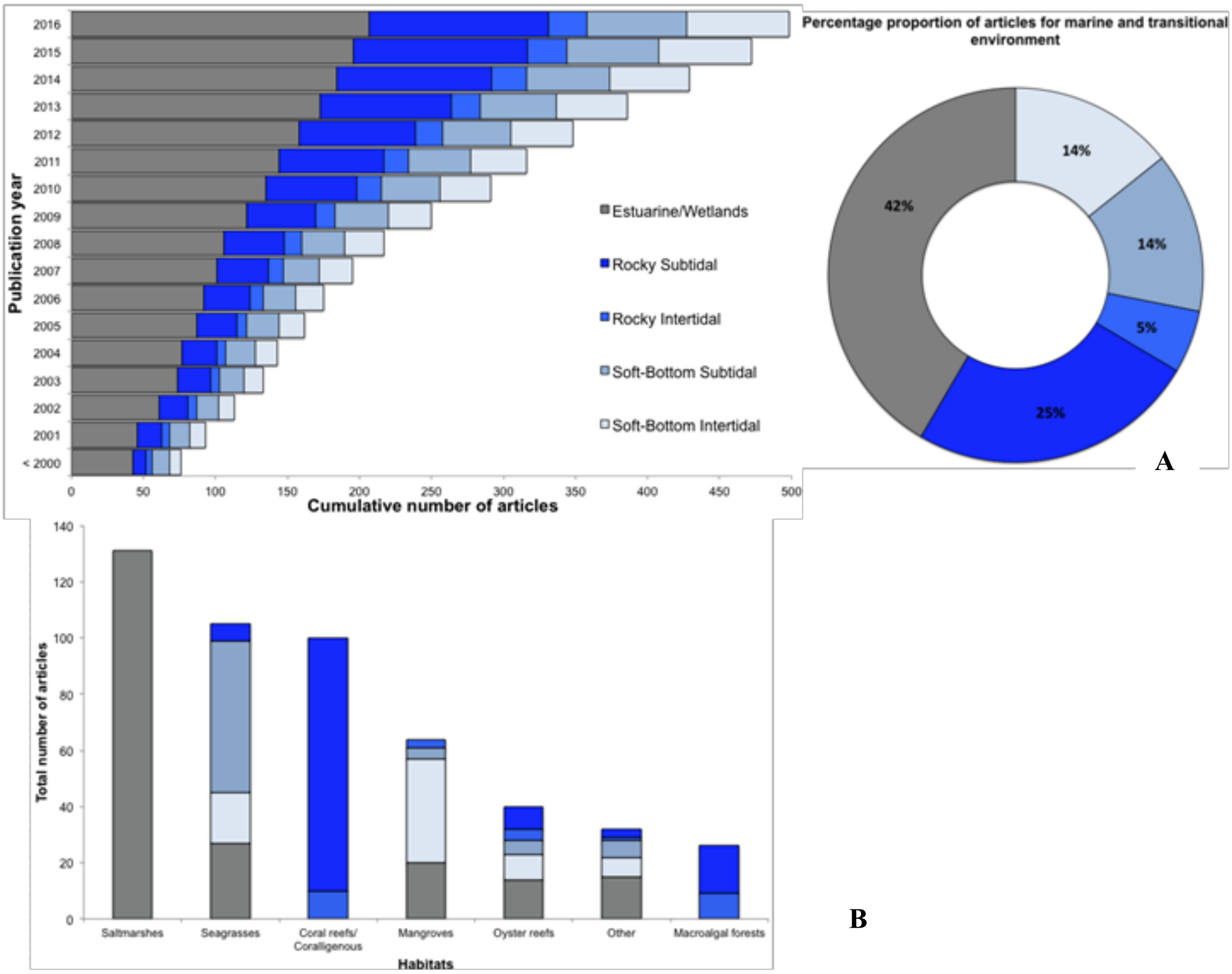


Figure 12. A) On the left side, the temporal cumulative number of articles across systems (i.e. strictly marine and transitional waters) is reported. In the donut graph, on the right side, the total amount % is shown. B) Total number of articles for each habitat across systems. Colours represent the different systems: grey = transitional waters; from dark blue to light blue = strictly marine environments. Habitats showing a sporadic occurrence (i.e. < 3%) in the database (e.g. mudflats, mussel beds, scallops) were grouped in "Other".

Independently from the habitat targeted by the restoration action, and/or the presence of any kind of protection measures characterizing the sites of interventions (e.g. reserves, parks, MPAs, etc.), studies documented at least the 50% success in terms of restoration outcome (Figure 13).

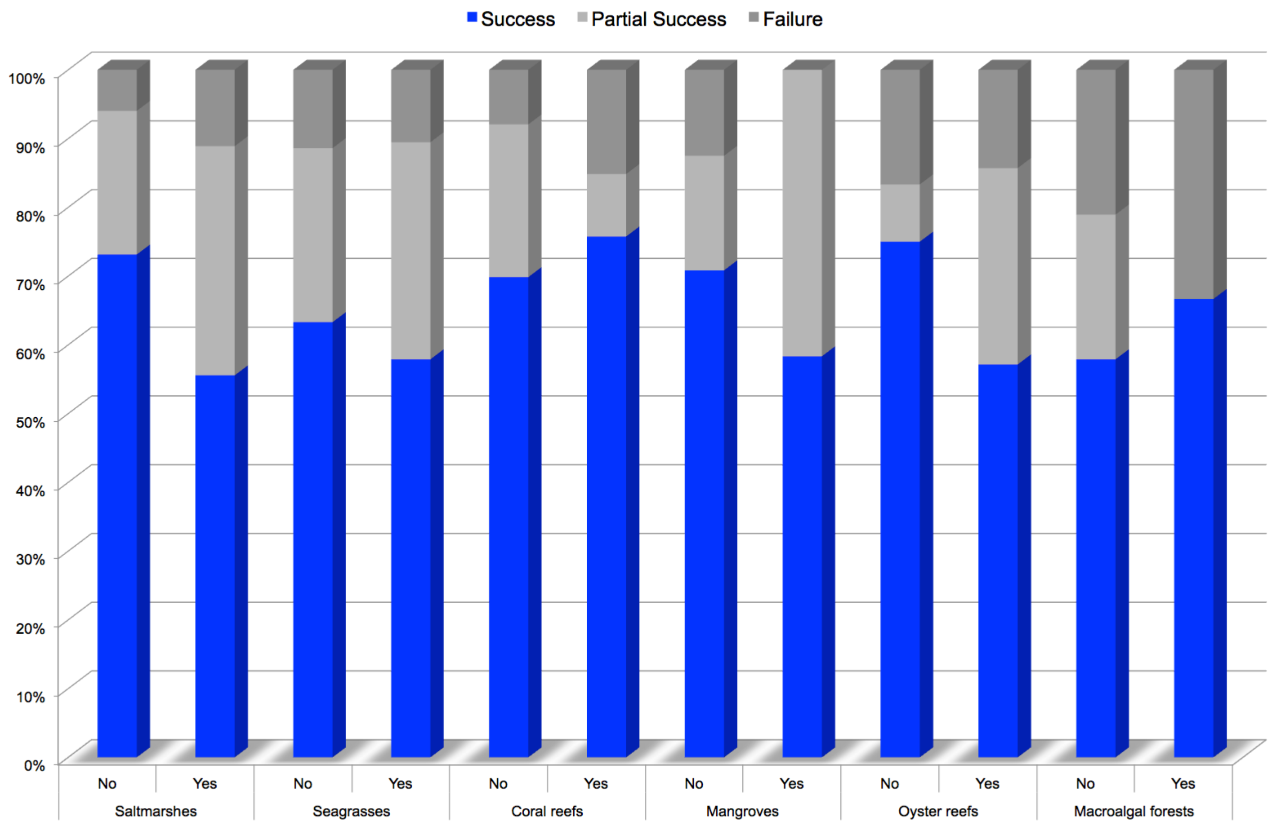


Figure 13. Partitioning (i.e. percentage representativeness) of the restoration outcomes (i.e. from success to failure) across habitats. The presence of some form of protection for each habitat is also reported separately. No = absence; Yes = presence of protection measures.

The most targeted species of restoration actions belong to the genera *Acropora* for coral reefs, *Spartina* for saltmarshes, *Zostera* for seagrasses, *Crassostrea* for oyster reefs, *Rhizophora* for mangroves and *Cystoseira* for macroalgal forests (Figure 14).

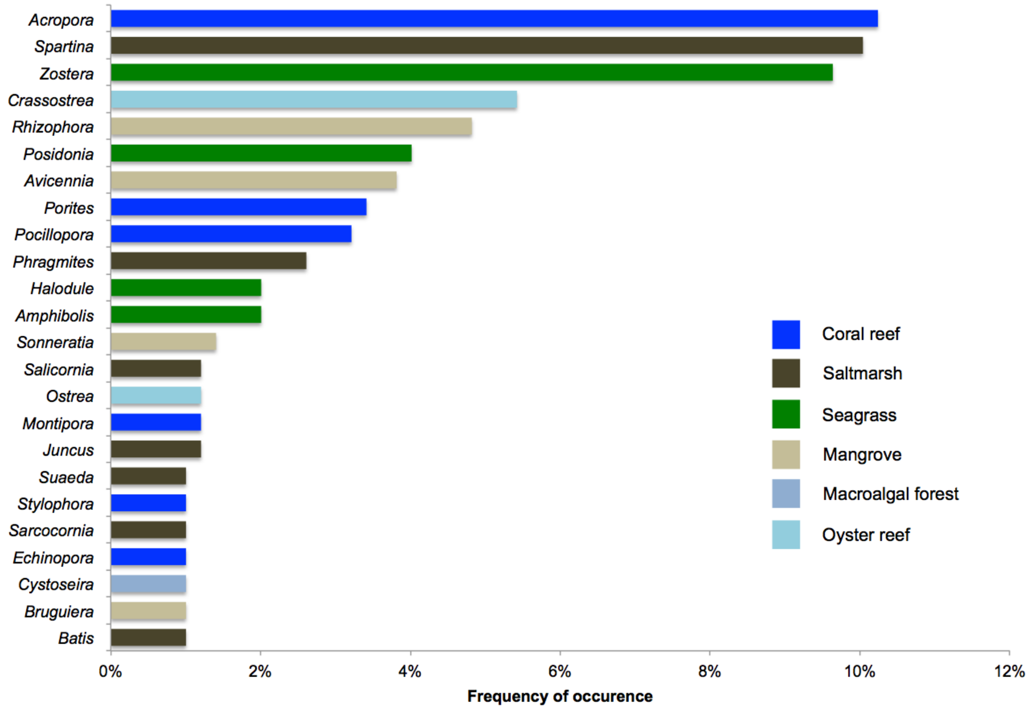


Figure 14. Summary of the most targeted organisms (expressed in genera). Different colours represent the different hosting habitats. Genera showing occurrence < 1% were not reported.

Differences can be observed across habitats in terms of duration and area covered by each study (Figure 15 A,B). Generally, most of the effort in the implementation of active restoration actions covered a temporal interval of few years (Figure 15A). In particular, an increasing number of studies across habitats adopt an experimental duration of up to 1–2 years, with the exception of macroalgal forests (< 12 months). Few studies cover longer time scales. An exception is represented by saltmarshes, seagrasses and mangroves habitats, showing examples of long-term experiments. A number of restoration efforts in saltmarshes in particular, exceed a length of 16 years.

Most active interventions have been carried out at a spatial scale lower than one hectare. An exception is represented by saltmarshes showing the highest number of studies with a spatial extent more than one hectare (Figure 15B).

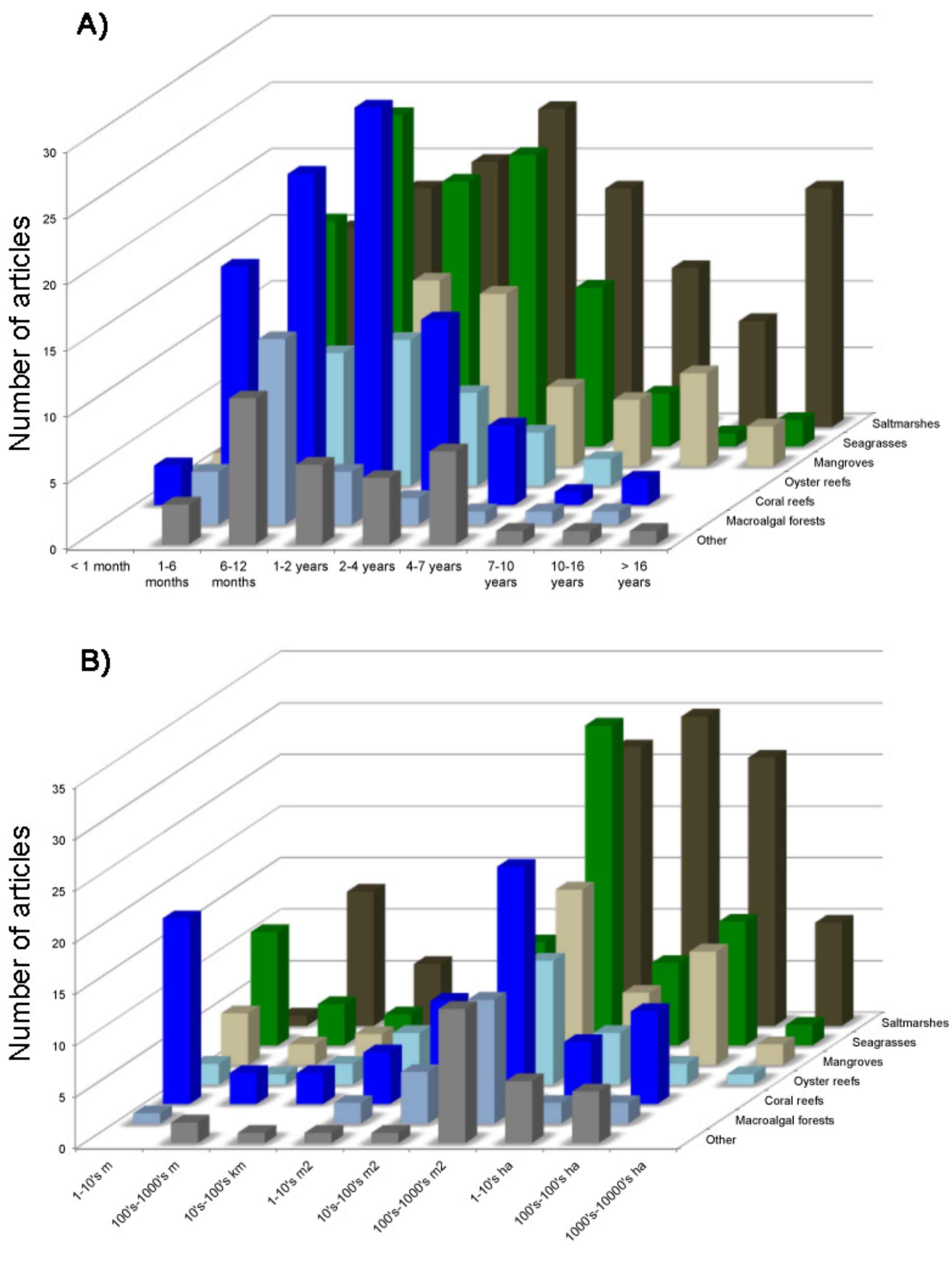


Figure 15. A) Time span (i.e. duration) and B) area covered (i.e. extent) by each publication across habitats. Habitats showing a sporadic occurrence (i.e. < 3%) in the database (e.g. mudflats, mussel beds, scallops) were grouped in "Other".

Different approaches have been adopted in terms of restoration. As reported in Figure 16A most of the articles (57%) referred to restoration and synonyms (i.e. Rehabilitation, Remediation, Re-creation) or are based on a replacement (20%) of an ecosystem's structural component (i.e. Re-introduction, Re-establishment, Reclamation, Replacement). According to the classification provided by Elliott et al. (2007) these actions are degraded-environment oriented. The remaining articles (23%) are single stressors oriented. They refer to mitigation or compensation measures if respectively aimed to *in situ* or *ex situ* creation of habitat (6%), or if they refer to habitat enhancement or creation if aimed to the production of new habitat (not necessarily *in situ*). Three large categories relative to methodology and techniques (i.e. biological, physical, or a combination) were grouped according to the specific environmental features targeted in the specific restoration actions. In this framework, it was observed that generally the highest number of articles involved the manipulation of a biological component (usually a habitat forming species, Figure 16B) independently of the objectives of the restoration action (i.e. I, II, III, IV).

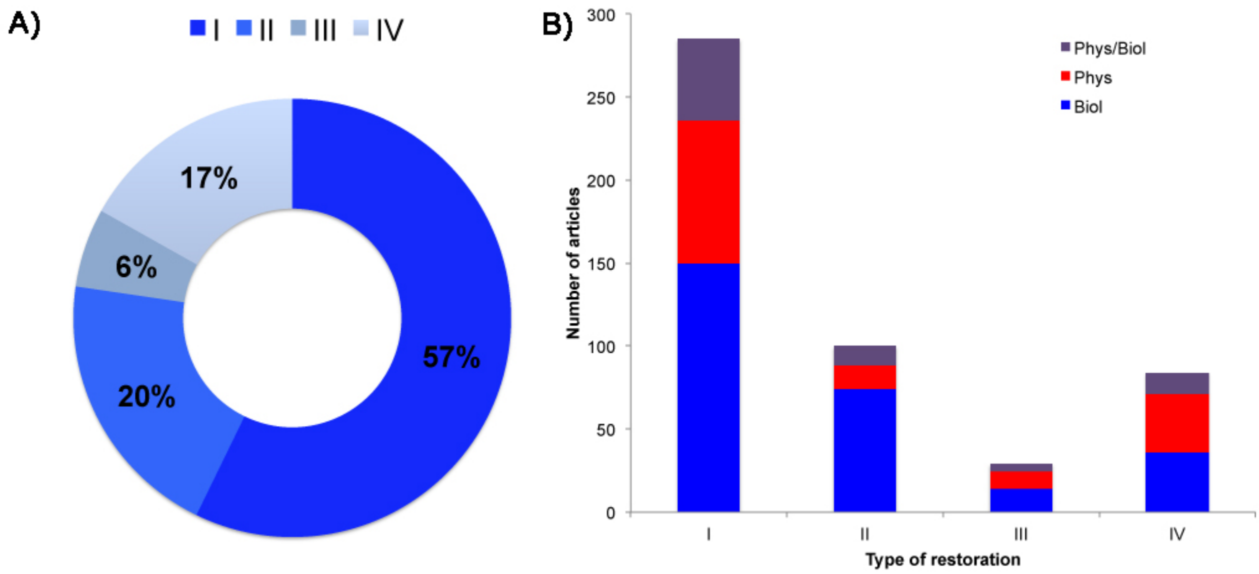


Figure 16. A) Percentage partitioning of articles on the basis of the type of restoration according to Elliott et al. (2007). B) Partitioning of articles, across different type of restoration, on the basis of the manipulated environmental features (Biol = biological; Phys = physical). I = Synonyms of restoration actions referred to habitats (all actions tend, as much as possible, to the structural and functional features of the original system; II = Replacement of an ecosystem's structural component (single structuring species); III = *in situ* (i.e. mitigation) or *ex situ* (i.e. compensation) creation of habitat; IV = producing new habitat (not necessarily *in situ*).

Generally (Figure 17), the highest number of studies (i.e. 33%) used transplantation techniques independently from the aim of restoration action. Among the other methods/techniques, planting and those involving modifications of environmental features (i.e. physical and/or hydrological - which are usually targeted on transitional water systems, e.g. saltmarshes), were also well ranked (respectively 15 and 21%).

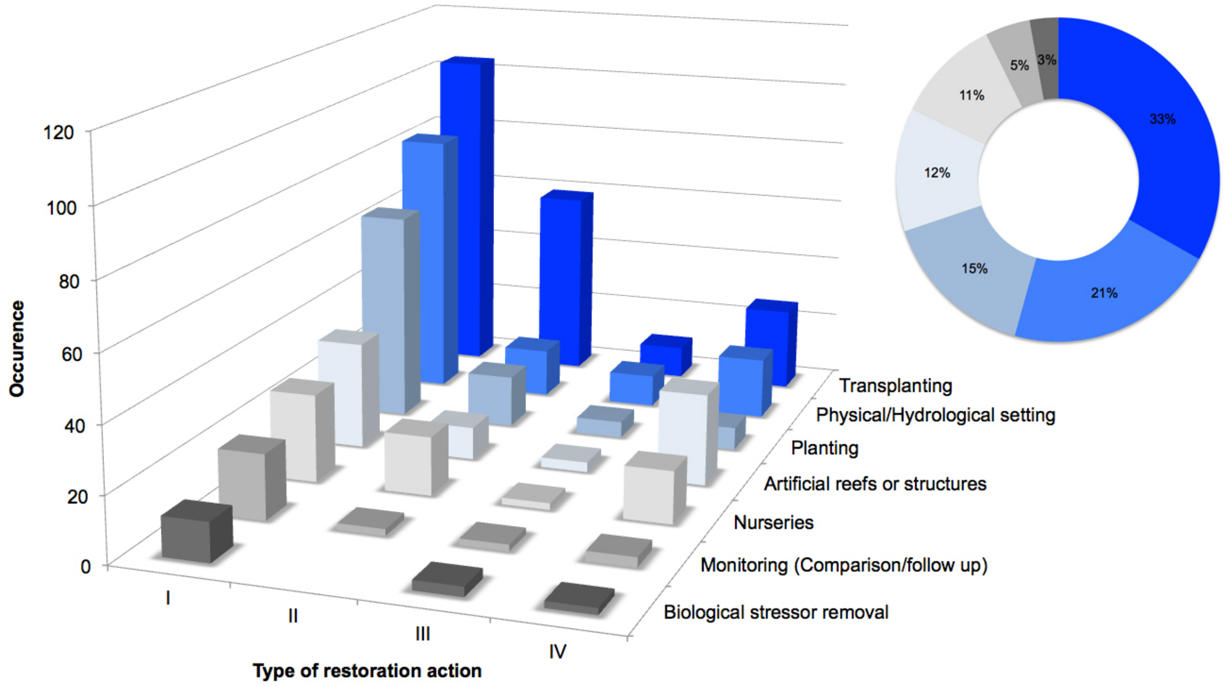


Figure 17. Detailed partitioning of methods/techniques adopted in the studies across type of restoration actions (I,II,III,IV = see details in the caption above). In the donut graph the total percentage occurrence of each method/technique is reported. Different colours represent the different methods/techniques reported in the z-axis of the 3d histograms graph. Methods/Techniques were sorted on the basis of their overall occurrence.

Usually, the response variable mostly used to assess the outcomes of the restoration action is the survival (Figure 18).

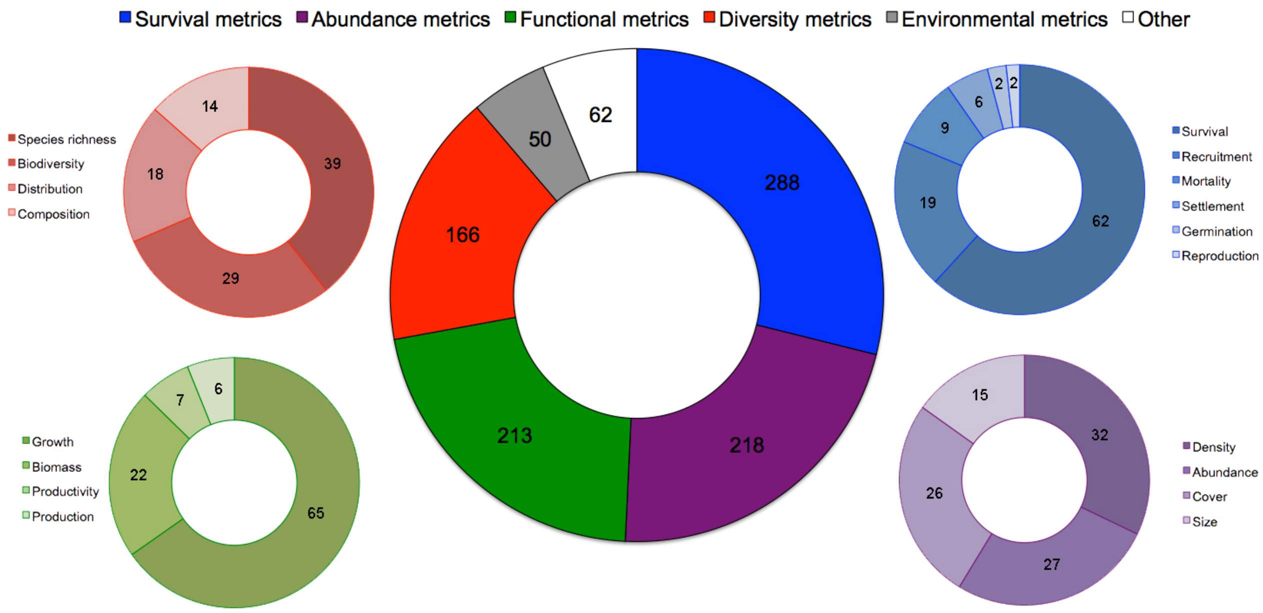


Figure 18. Summary of the response variables, grouped for categories, which have been used to assess the restoration outcomes across articles. For each category of metrics (excepting for environmental and other) details of each specific metric (reported as percentage) is also provided.

3.3.2. Modelling Results

The models performed relatively well, explaining 67% of the observed variation in restoration success.

Achieving partial success

Of the factors considered, the ecosystem type, the habitat, the degree of human impact, the method and the type of restoration were found to be most important in achieving "partial success" in restoration activities. Specifically, restoration activities in macroalgal forests, saltmarshes, seagrasses and "Other" tended to be partially successful, as did activities in areas with higher degrees of human impact and those where re-establishment was the goal. In contrast, restoration in soft-bottom Intertidal and soft-bottom intertidal ecosystems, and restoration using transplantation methods tended to not be partially successful (Table 3).

Table 3. The variables that contributed to making restoration activities partially successful, along with the model averaged estimates and p-values. Variables in light blue relate to contextual factors whilst variables in white relate to methodical.

Factot	Variables	Estimate		
Ecosystem type	Soft-Bottom Intertidal	-1.04	0.01	*
Ecosystem type	Soft-Bottom Subtidal	-1.38	0.01	**
Habitat	Macroalgal forests	4.00	0.00	***
Habitat	Other	3.04	0.00	**
Habitat	Saltmarshes	4.69	0.00	***
Habitat	Seagrasses	5.54	< 2e-16	***
Degree of human impact	NA	0.33	0.00	***
Method	Transplantation	-11.64	0.08	.
Restoration type	Re-establishment	2.08	0.01	*

Achieving success

Of the factors considered, the habitat, the degree of human impact, the realm and the method of restoration were found to be most important in achieving "success" in restoration activities. Specifically, restoration activities in macroalgal forests, saltmarshes, seagrasses tended to be successful, as did activities in areas with higher degrees of human impact and those in the Western Indo-Pacific, which had the highest bearing impact. In contrast, restoration in soft-bottom Intertidal ecosystems tended to not be successful (Table 4).

Table 4. The variables that contributed to making restoration activities successful, along with the model averaged estimates and p-values. Variables in light blue relate to contextual factors whilst variables in white relate to methodical.

Factor	Variable	Estimate	Pr(>|z|)	
Ecosystem type	Soft-Bottom Intertidal	-0.79	0.03	*
Habitat	Macroalgal forests	1.45	0.04	*
Habitat	Saltmarshes	2.24	0.01	*
Habitat	Seagrasses	2.81	0.00	***
Degree of human impact	NA	0.19	0.00	**
Realm	Western Indo-Pacific	25.25	0.06	.
Method	Mitigation	2.42	0.01	*
Method	Re-establishment	3.45	0.00	***
Method	Rehabilitation	1.32	0.01	*
Method	Restoration	3.42	0.00	***

Relative importance of variables

Of the factors considered, contextual variables (e.g. the type of the ecosystem being restored, the relative degree of human impact in the area) rather than methodological variables (e.g. how the restoration activity was conducted), tended to have high relative importance in determining the outcome (Table 5). However, as seen in Tables 3 and 4, the method and type of restoration are still of significance.

Table 5. The relative importance of the variables that had a bearing on whether restoration failed or not. Variables in blue relate to contextual factors whilst variables in white relate to methodical.

Factor	Relative Importance
Duration of restoration	1.00
Ecosystem type	1.00
Habitat	1.00
Degree of human impact	1.00
Method of restoration	1.00
Realm	1.00
Type of restoration	1.00
Distance from protected area	0.61
Method: Removal of species	0.51
Method: Introduction	0.50
Number of different techniques	0.50
Method: Artificial habitat	0.49
Method: *In situ* growing	0.49
Method: Protection	0.49
Method: Transplantation	0.49

3.4. Discussion

Over the past two decades large efforts have been documented at the global scale to develop and implement active restoration actions across different habitats. However, even if a consistent positive trend in the publication rates was detected across ecoregions (on both developed and developing countries), most of coastal restoration studies/projects were conducted in the USA and Europe, probably as a consequence that these areas also rank among the most threatened in the planet. For these countries, ecological restoration represents a big deal but also a big opportunity. In particular, for the USA, the "restoration economy" contributes annually over $24 billion and 221,000 jobs (BenDor et al., 2015).

In spite of the substantial and consistent increase in restoration efforts across habitats/ecosystems, to date studies/projects on restoration have been mainly focused on semi-terrestrial (substantially saltmarsh habitats), rather than on strictly marine environments. The latter with a significant delay compared to the former, and in many cases we are still in the phase of filling the gaps (e.g. seagrass and macroalgal forests). This gap has been mitigated in the last decade (i.e. starting from 2008), as a growing number of studies/projects carried out within strictly marine environment has been observed (mainly focussed on seagrass and coral reef habitats). As indicated by recent studies, costs of restoration actions, which are much higher in the marine environment (Bayraktarov et al., 2016), coupled with the intrinsic features of this system (i.e. much less stable/manageable with respect to confined environments) are the main drivers affecting the observed patterns.

Independently of the habitat targeted by the restoration actions, at least 50% of the studies documented the success of restoration outcomes. However, despite the high recognition that active restoration can have a critical role in the recovery of disturbed systems, results highlight a general heterogeneity of targets, implementation methods, approaches and standards across habitats. For instance, with the exception of wetlands and estuarine systems (i.e. mostly saltmarshes and mangroves), most restoration projects cover too small areas (< 1 ha) to match the scale of human disturbance. In addition, most restoration studies/projects in the database showed a limited duration (i.e. no more than one to two years) corresponding to the lifetime of development projects, research grants, or academic theses. Ten years ago, a study reported that restoration research on coral reefs had been focused on the development of techniques rather than on assessing the application of established methodologies in large-scale restoration projects

(Zimmer, 2006). The present review indicates that still little has changed and many studies are still experimental covering small scale sand very limited durations. In particular, the short project duration of marine coastal restoration projects has been criticized as being unsuitable to assess recovery of ecosystem function and that the outcome of restoration (success or failure) is directly related to the period of observation (Bayraktarov et al., 2016 and references therein). Thus, the limited spatial and temporal scale covered by the studies/project, combined with a frequent lack of consideration of control areas and knowledge of baselines, largely impair the potential to actually show robust success stories.

Moreover, response variables used to assess the outcomes of restoration action are rather heterogeneous and too often vaguely reported. The majority of studies reported item-based success in terms of survival and lacked clearly defined and measurable success. Where criteria for success were stated explicitly, they typically aimed for simple metrics, such as a particular level of biomass or coverage, and rarely focused on the recovery of ecosystem function or services. This makes it very difficult to provide objective and actual information concerning the success rates of restoration interventions.

Modelling results suggest that in the marine environment the key factors responsible for the restoration outcome are related to habitat type and the degree of human impacts, which may translate to the level of habitat degradation, although methodological aspects related to the restoration approach (e.g. re-establishment), or techniques used (e.g. transplantation) are also of importance. Suding (2011) reports that incomplete recovery is often attributed to a mixture of local and landscape constraints, including shifts in species distributions and legacies of past land use, which seems to be confirmed by the results of our modelling analysis for the marine environment as well. On a similar note, Bayraktarov et al. (2016) found that in the marine environment, restoration success, approximated through survival parameters, varied for the different ecosystems, being lower for seagrasses and higher for coral reefs and saltmarshes. In contrast, failure, which is believed to be under-reported (Hobbs, 2009; Knight, 2009; Suding, 2011), is most likely related to inadequate site selection, stochastic events, or human disturbance (Bayraktarov et al., 2016). Overall, the varying performance outcomes of restoration projects and practices underpin the various challenges restoration practitioners need to tackle for the successful implementation of ecological restoration, which are ecological, technical, and socio-economic in nature (Maron et al., 2012).

Our results further show the importance of the MERCES project to set the stage (e.g. protocol availability, monitoring of the effects, reasons for failure) in marine restoration for the development of best practices to apply at spatial and temporal scales so as to answer to present disturbance regimes. In particular, research and monitoring of restored sites over longer time periods is a much more challenging task, but where possible should be the ultimate aim of ecological restoration. The need to assess the long-term success of restoration projects should be embraced by policies aimed to the effective restoration of marine coastal habitats.

4. Restoration and MERCES Key Habitats/Species

4.1. Introduction

MERCES WP1 has so far reviewed various aspects of information on the status of European habitats by regional sea and habitat type, looking, for example, at causes of decline and deterioration. Smith et al. (2017) (D.1.2 Deliverable) reviewed the most common activities and pressures acting on shallow-soft, shallow-hard and deep-sea habitats, their impacts (e.g. algal blooms, overgrowth of turf algae, removal of predation pressure, loss of density and cover, increased patchiness and decreased patch size, decreased connectivity), and the consequences for recovery and mitigation/restoration potential of these habitats). In addition, Bekkby et al. (2017) (D.1.1 Deliverable) reviewed 6 key MERCES habitats (including kelp and *Cystoseira* forests, seagrass meadows, coralligenous assemblages and deep-sea coral gardens and soft-bottom communities), with respect to 6 major features, such as their dynamics, connectivity, structural complexity and vulnerability, deemed critical to restoration and likelihood of restoration success.

Summarizing Bekkby et al. (2017), questions on dynamics, for example, how fast a habitat-building species grows, and how old it gets, are essential for the restoration time period. Slow-growing species with long life spans usually become fertile at a high age, which will extend the time needed for full restoration. Coral reefs are such a habitat, while kelps grow fast and reproduce early. When it comes to connectivity and spatial distribution, seagrass seeds can for instance disperse between two seagrass meadows if they are sufficiently close to each other, thereby increasing the genetic diversity and possibility of survival of the species when there are changing environmental conditions.

Habitats with high structural complexity provide substrate, food and shelter for numerous species, and are the foundation for a high diversity of genes, species, communities and functions in the ecosystem. For most habitat types, high diversity is naturally obtained if the habitat's physical structure is intact. For ecosystems without habitat-building species, efficient restoration actions probably involve restoration of groups of species with different functions, rather than single species.

Vulnerability is another feature of high relevance. Bottom trawling may destroy coral reefs, while eutrophication and dredging threatens kelp forests, macroalgal forests and seagrass meadows. CO_2-emissions contribute to the rise in temperatures and ocean acidification, the latter being harmful for corals and their production of calcium carbonate for skeletal structure. Human activities constitute the largest threat to marine habitats both in Europe and globally, and harmful activities must be stopped or strongly reduced if the degraded habitats and ecosystems shall be repaired.

Section 3 of this report reviews the extent of global restoration effort by region and ecosystem/habitat, types of restoration approaches applied, response variables, etc. Here, we present a mini-review of specific examples of MERCES Case Studies on the European key habitats/species going a step further. The information presented here includes notes on most current methods and approaches used, on timescales to success, and on bottlenecks/deal breakers and means/potential for up-scaling restoration to the level of degradation. The information is presented in a standardised descriptive table for each of the following 12 cases:

- Kelp forests in Norway
- *Cystoseira* forests in Spain
- Seagrass meadows overview, in Norway, Baltic Sea and Mediterranean
- *Pinna nobilis* in Croatia
- Coralligenous habitat in Spain
- Red corals in Italy
- Sponges in Italy
- Deep-sea corals in Azores
- Deep-sea seamounts in Italy

4.2. MERCES Key Habitats

4.2.1. Key Habitat 1: Kelp forests

Key Habitat 1: Kelp forests
1. Habitat Description
Along the Norwegian coast (Skagerrak, North Norwegian and Barents Seas), kelp forests are found on rocky seabed (including soft sediments with different sizes of stones), on sheltered areas (sugar kelp *Saccharina latissima*), and on rocky moderately exposed and exposed areas (*Laminaria hyperborea*), see Figure 19. Kelp forests are found below the seaweed belt down to about 20–25 m maximum depth (Bekkby et al., 2009). The kelp forest is a productive system (e.g. Abdullah & Fredriksen, 2004), with a diverse association of flora and fauna (e.g. Christie et al., 2003), providing habitat for several fish species of commercial interest (Norderhaug et al., 2005). Grazing by sea urchins is a threat to kelp forests in the north, and the combined influence of nutrients and warmer water decimate kelp in southern Norway.
2. Main, most recent, or successful techniques and methods used in restoration actions
Restoration actions of kelp forests in Norway involve removal of sea urchins (destructive by lime or

Key Habitat 1: Kelp forests
by harvesting for food), deployment of artificial reefs (sometimes combined with lime treatment in sea urchin barrens), and transplantation of kelp plants. In MERCES, transplantation of adult kelp plants (sugar kelp as well as *L. hyperborea*) in sea urchin barrens, are tested.
3. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions
The transplantation techniques applied seem to work well at small scale, but are too costly on larger scale (because of hard physical work by diving). Kelps recover if sea urchins are removed (Leinaas & Christie, 1996). Establishment of a profitable sea urchin harvesting industry that removes sea urchins in the restoration site is likely to enhance the restoration success. Further harvesting of sea urchins outside the restoration sites will enhance natural kelp recovery. Increased abundance of different species of crab, seem to control/reduce sea urchin density (Norway unpublished NIVA data, also shown by Steneck et al., 2013). Hence managing the crab fishery to allow a sufficient high crab population will contribute to the natural kelp recovery in the north.
4. Major barrier/deal breaker and new means/solution/hope to up-scale restoration to the level of degradation
Restoration of kelp forests in Norway is a new discipline, looking at why kelp plants have disappeared in the north and in the south. The decimation of kelp forests involves grazing by sea urchins in the north and a combination of warmer water, eutrophication, overfishing, and increased run-off from land in the south. The large-scale grazing by sea urchins in the north started in the 1970s, and has not been directly related to any anthropogenic pressure that can be easily removed. It also occurs at a large spatial scale that makes it very costly to restore. But failing to remove urchins from sea urchin barrens is a major barrier for the success of restoration in the north. Hence restoration of sea urchin barrens at a large spatial scale has so far not been considered feasible. In the southern Norway where there is the possibility of regime shifts from kelp to filamentous algae due to warm water and high nutrient loadings, such shifts can be harder or even impossible to recover.
5. Relevant time scales for restoration
Survival of kelp transplants can be measured at both short- (months) and longer-term period (1–several years). However, the restoration success and the time it takes will largely depend on the scale of the experiment/the restoration actions relative to the size of the area that you aim to restore, as well as the effort needed to control the sea urchin abundance. If the area is cleaned for sea urchins, the natural recovery of kelp plants will be fast (often sugar kelp will be established first, then *L. hyperborea*, if the site is wave exposed). The survival of transplanted kelp plants does not necessarily mean recovery of total biodiversity and ecological functions. However, the high mobility of the associated fauna and using old kelp plants with established epiphytes on stipes (just relevant for *L. hyperborea*) indicate that full recovery on small spatial scale can be achieved in a few years, in particular in areas close to natural kelp forests.

Figure 19. Restoration of the sugar kelp *Saccharina latissima* (left) and *Laminaria hyperborea* (right) conducted in Norway in the framework of MERCES project (see Key Habitat 1), photo left: sugar kelps (attractive food for sea urchins) are set up on lines to avoid grazing by urchins, photo on the right: removing sea urchins around the transplanted kelps. Photos by Camilla With Fagerli, NIVA.

4.2.2. Key Habitat 2: *Cystoseira* forests

Key Habitat 2: *Cystoseira* forests
1. Habitat Description
Marine forests of species of the genus *Cystoseira* (Fucales) are some of the most important ecosystem-engineers on photophilic Mediterranean rocky bottoms (Giaccone, 1973; Ballesteros, 1988, 1990; Zabala & Ballesteros, 1989). *Cystoseira* stands are structurally and functionally very similar to terrestrial forests (Figure 20). Like trees, *Cystoseira* are foundation species that provide habitat for many other associated species (algae, invertebrates and fish). Although miniaturized, they can be considered as submerged Mediterranean forests. *Cystoseira* is a genus of canopy-forming brown algae, amongst the largest and most long-lived in the Mediterranean. Several species are endemic and many of them are highly vulnerable to anthropogenic activities and thus, have severely declined over the past decades (Benedetti-Cecchi et al., 2001; Thibaut et al., 2005; Serio et al., 2006).
2. Main, most recent, or successful techniques and methods used in restoration actions
Two different restoration techniques have been experimentally tested to promote *C. barbata* recovery (Verdura et al., *submitted*). The first one, *in situ* seeding, consists in collecting fertile apical branches from the donor population and placing them in Container-Disperser of Algal Propagules (CDPAs). The second one, *ex situ* seeding, consists in obtaining and culturing recruits of *C. barbata* in the laboratory, in tanks with filtered seawater that contain flat stones without biotic covering. *C. barbata* fertile apical branches are placed on CDAPs floating over the water of each tank for three days during which the hydrodynamic tank conditions have to be as stable as possible, in order to facilitate zygote settlement on the stones. After a period of approximately two months the recruits can be transplanted in the field with the help of volunteers.
3. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions
The techniques used are time-consuming and the only way to up-scale the restoration actions is by involving volunteers.
4. Major barrier/deal breaker and new means/solution/hope to up-scale restoration to the level of degradation
Of course, restoration only makes sense once the impact that caused degradation is reduced or

Key Habitat 2: *Cystoseira* forests
eliminated. Natural sensibility of the first life stages of *Cystoseira* species, may compromise restoration success, since many factors such as temperature, sedimentation, irradiance or overgrazing strongly influence juvenile survival. Therefore, deal breaker is to avoid both local and global stressors.
5. Relevant time scales for restoration
Just on the second year after the restoration actions, density of individuals obtained by the two applied techniques was similar and comparable with the natural populations. However, it took more than 5 years to reach comparable size structure distribution between restored and natural populations (Verdura et al., *submitted*).
6. Other relevant point/key message relevant to the habitat
For *Cystoseira*, as well as for other macroalgal engineering species, size structure distribution may determine population dynamics, conservation state and maturity, therefore, besides density, size-structure distribution is also an essential indicator to reflect the overall habitat restoration (Verdura et al., *submitted*). However, the period required to reach similar size structure to natural populations will vary among species and may tend to decades given the slow growth rates of most of the species (Montero-Serra et al., 2017).

Figure 20. *Cystoseira* stands (top photo right) are structurally and functionally very similar to terrestrial forests (top photo left) (see Key Habitat 2). Collection of fertile apical branches from a donor population for *ex situ* seeding. Transplantation of recruits in the field is then implemented with the help of volunteers. Photos by CSIC.

4.2.3. Key Habitat 3: Seagrass meadows

Key Habitat 3: Seagrass meadows
1. Habitat Description
Seagrasses are common in soft-bottom, shallow coastal areas as they need high light availability and stable sediments. The depth limit depends on the species and on light availability. There are approximately 60 species in 12 genera and 4 families. In Europe, there are four native species: *Zostera marina* (subtidal and intertidal; global distribution in northern hemisphere), *Zostera noltii* (intertidal, distribution throughout NE Atlantic and European Seas), *Cymodocea nodosa* and *Posidonia oceanica* (subtidal, found in Mediterranean Sea). Seagrass meadows provide numerous ecosystem services, including food and shelter for numerous species (mammals, reptiles, fish, and invertebrates), stabilise the seabed, filter water, and store carbon (Campagne et al., 2015; Cole & Moksnes, 2016; Nordlund et al., 2016). Seagrasses are vulnerable to environmental change, and 30% of seagrass meadows have been lost over the last 50 years (Waycott et al., 2009; Lefcheck et al., 2017a). Seagrass restoration leads to the restoration of ecosystem services (Greiner et al., 2013; Reynolds et al., 2016; Lefcheck et al., 2017b).
2. Main, most recent, or successful techniques and methods used in restoration actions
Many methods have been tried in different areas and for different species (Orth et al., 2006; Busch et al., 2008; van Katwijk et al., 2016). These include (a) transplanting of seeds (Marion & Orth, 2010; Orth et al., 2012; Infantes et al., 2016), seedlings (Infantes et al., 2011), (b) rhizome fragments (Davis & Short, 1997), (c) adult shoots (Meehan & West, 2002; Eriander et al., 2016), or (d) entire sods (including sediment; Uhrin et al., 2009; Matheson et al., 2017). Different anchoring methods have also been tried using (a) rods and pegs of different material (metal, bamboo, wood), or by (b) attaching plants to substrate such as biodegradable mesh (Kidder et al., 2015), hessian bags (Tanner, 2015), or shells (Lee & Park, 2008). Planting of adult plants with intact rhizomes and sods seems to have the highest success rate (van Katwijk et al., 2016). Mechanical planting has also been tried (Fishman et al., 2004) as well as sediment fertilization which can be effective in areas that are nutrient-depleted (Balestri & Lardicci, 2014), but inhibits plant growth at high nutrient levels (Peralta et al., 2003).
3. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions
Large scale efforts to restore appropriate sediment through sand-capping could enable restoration in areas sediments are no longer suitable, i.e. sediments resuspended due to exposure of glacial clay making the water permanently murky, or sediments are too coarse, hindering natural re-establishment of seagrass. In such conditions all restoration efforts have failed (Eriander et al., 2016). This will soon be tested in the North Sea area.
4. Major barrier/deal breaker and new means/solution/hope to up-scale restoration to the level of degradation
Overall, restoration success has been low (37%), the main cause being unsuitable environmental conditions for seagrass re-establishment (van Katwijk et al., 2016). This includes high turbidity (Thorhaug, 1985; Eriander et al., 2016), filamentous algal loads and drifting algal mats (Gustafsson & Boström, 2014), wave exposure (Infantes et al., 2011), and changes in sediment characteristics (Suykerbyk et al., 2016a), which can lead to reduced growth, or transplants being buried or washed away. In addition, changes in top-down pressure due to overfishing can reduce herbivore densities and lead to higher biomasses of filamentous drifting algae or epiphytic loads on leaf surfaces (Baden et al., 2012). Importantly, anthropogenic pressures must be fully ceased prior to restoration. Other factors which could increase restoration success include ensuring high genetic diversity in the transplanted population (Jahnke et al., 2015; Evans et al., 2017) and planting on a large scale (van Katwijk et al., 2016). In areas with mixed meadows of different species, plant diversity can also

Key Habitat 3: Seagrass meadows
increase short-term success (Salo et al., 2009; Gustafsson & Boström, 2011). Recently, more attention has been paid to the importance of feedbacks and species interactions between seagrasses and other co-existing species such as bivalves (van der Heide, 2007; van Katwijk et al., 2009; Suykerbuyk et al., 2016b; Maxwell et al., 2017.). Another potentially important, but unexplored factor, is the role of the microbial community surrounding seagrass roots (Garcias-Bonet et al., 2012; Fahimipour et al., 2017; Cucio et al., 2016). If microbes and/or macrofauna are important, then they must be transplanted together to promote positive feedbacks and care must be taken to reduce negative feedbacks (Suykerbuyk et al., 2012).
5. Relevant time scales for restoration
Time scales differ depending on the species in question. For example, *Zostera marina* is very fast growing, and restoration success can occur within several years (e.g. Orth et al., 2012). However, in other species, it may take many more years to fully return to the "natural state" (e.g. Bell et al., 2008). *Posidonia oceanica* grows very slowly, so restoration and recovery time scales may be on the scale of several decades (Meehan & West, 2000).

4.2.4. Key Habitat 3.1: Seagrass (*Zostera marina*) meadows - Norway

Key Habitat 3.1: Seagrass *(Zostera marina)* meadows – Norway
1. Habitat Description
Along the Norwegian coast, seagrass meadows (mainly *Zostera marina*) are found from muddy to sandy seabeds, from the north (the Norwegian Sea and Barents Sea), along the western coast (the North Sea), to the south (Skagerrak). Seagrass meadows are also found from the intertidal zone down to a maximum of 12 m (on the West coast, Bekkby et al., 2008), but most commonly down to 4–8 m. The habitat has an important function as a three-dimensional habitat for many species (e.g. Fredriksen et al., 2005; Christie et al., 2009).
2. Main, most recent, or successful techniques and methods used in restoration actions
The Project "Indre Viksfjord" is the only seagrass restoration project in Norway. The main focus of the project is to improve the conditions for *Z. marina* by removing mats of floating green algae (Figure 21). However, other approaches such as aeration to improve oxygen conditions in the sediment are tested. The effect of tidal ports on water exchange is tested by modelling, and implantation in practice is on the projects priority list. In MERCES we test survival and expansion of transplanted adult plants, with and without addition of blue mussels (*Mytilus edulis*), in one exposed and one sheltered sandy site.
3. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions
To scale up restoration of seagrass beds by transplanting adults, we must use degradable nets to attach the plants (not plastic as in the experiments). Adding seeds (a much easier and rapid technique) to exposed areas will probably not work. Involving volunteers to fasten plants to the nets and to deploy the nets would enhance the process, and it will be a means to enhance public ocean literacy.
4. Major barrier/deal breaker and new means/solution/hope to up-scale restoration to the level of degradation
In Viksfjorden, the most important approach to save the seagrass beds will be to reduce the nutrient loading to the area. The environmental authorities have a long-term plan to achieve this goal. Meanwhile, the removal of algae mats hopefully will enhance the seagrass beds changes of surviving. A cut in the funding to inner Viksfjord's projects removal of the algal mats would be a deal

Key Habitat 3.1: Seagrass *(Zostera marina)* meadows – Norway
breaker. Techniques on seeding and growing seagrass plants on degradable nets, which can be put into the sediment by means of boats instead of diving, would be an efficient way to scale up restoration of seagrass beds. Severe storms will have the potential to destroy newly planted seagrass beds.
5. Relevant time scales for restoration
Zostera marina plants grow and spread quickly, so it is possible to estimate the success on both short (a couple of months) and longer term (1–more years). However, the restoration success and the time it takes will largely depend on the scale of the experiment/the restoration actions relative to the size of the area to be restored. A few transplants in a large area will probably need many years to occupy the whole habitat. But if the area is heavily populated by transplants, this can take a few years. The recovery of the seagrass itself does not necessarily mean recovery of total biodiversity and ecological functions. However, the mobility of the associated seagrass fauna, and the short life of seagrass leaves, indicate that this can be achieved in a few years, if not too far from other seagrass beds.

Figure 21. Removal of mats of floating green algae to improve local environmental conditions for *Zostera marina* in the framework of "Indre Viksfjord" seagrass restoration project in Norway (see Key Habitat 3.1). Photos by the Viksfjord-project. Photos by NIVA

4.2.5. Key Habitat 3.2: Seagrass (*Zostera marina*) meadows – Baltic Sea

Key Habitat 3.2: Seagrass (Zostera marina) meadows – Baltic Sea
1. Habitat Description
In the Baltic Sea the distribution area of the eelgrass *Zostera marina* was estimated at 1200 km^2 at minimum, which is about four times greater than the estimates given for the Western Europe in 2003 (340 km^2) (Spalding et al., 2003; Boström et al., 2014). Due to reduced salinity and short evolutionary time of the Baltic Sea, eelgrass is largely the only phanerogam on sandy habitats where it significantly promotes floral and faunal richness on otherwise species-poor sandy substrates. Despite of this relatively large distribution, large-scale losses of eelgrass have been documented including the northeastern parts of the Baltic Sea. Eelgrass is found exclusively subtidally, growing at depths of 2–6 m. Eutrophication leading to decreased water clarity and increased filamentous algal blooms threatens to reduce the depth limit and growth of seagrass meadows in the Baltic Sea. The failure of eelgrass to re-establish despite reduction in background nutrient levels signals complex recovery trajectories and calls for much greater conservation effort to protect existing seagrass meadows. To prevent further loss of eelgrass, region-specific management and monitoring actions

Key Habitat 3.2: Seagrass (Zostera marina) meadows – Baltic Sea
are needed to identify and control the local loss drivers. Restoration is considered as one of rewarding measures to promote seagrass recovery in such habitats. In the Gulf of Riga study area, *Z. marina* reproduces only vegetatively and therefore the restoration actions can be done via transplanting only.
2. Main, most recent, or successful techniques and methods used in restoration actions
Several techniques are currently tested in the Baltic Sea. One of them regards planting single shoots one by one and evenly spaced, without using sediment or anchoring. Another experimental approach carried out in parallel in Finland and Estonia concerns the testing, both in situ and ex situ, of eelgrass *Zostera marina* planted with two types of bivalves; epifaunal blue mussels *Mytilus edulis* (Figure 22), and infaunal Baltic clam *Macoma balthica*. The seagrass shoots used in the experiments are attached to underground meshes and then the bivalves are added to the experimental plots. A third technique currently under investigation in the Gulf of Riga, Estonia, is replanting using *Zostera* shoots with the rhizomes attached to ropes which are afterwards buried.
3. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions
Seagrass restoration is a new approach in the Baltic Sea and these MERCES cases are the first tests of such kind in the study areas.
4. Major barrier/deal breaker and new means/solution/hope to up-scale restoration to the level of degradation
Poor water quality and high filamentous algal loads as result of eutrophication are the major threat to seagrasses in the Baltic Sea. Successful restoration efforts need action at a larger scale to reduce nutrient input into the Baltic Sea and improve water conditions. At the first experimental site in Estonia, the transplanted *Z. marina* individuals were buried under the sand. Prior to restoration action, it is very challenging to predict the sand movement and accumulation, especially considering extreme weather patterns in the recent years. Thus, it is recommended that initial recovery trials are carried out at multiple locations and at small scales to avoid large-scale failures. Moreover, in fine sand habitats with high sand mobility, the restoration attempt was successful in case negatively buoyant ropes were used instead of plastic frame to fix the transplanted seagrass individuals.
5. Relevant time scales for restoration
Small-scale field experiments have shown that *Zostera* shoots can establish within one growing season, so restoration could be quick if environmental conditions are improved. In Estonia, the success of *Z. marina* restoration was highly site-specific. At semi-exposed site experiencing an intense fine sand accumulation, the majority of transplanted plants were lost. However, in similar environment, the restoration attempt was successful in case negatively buoyant ropes were used instead of plastic frame to fix the transplanted seagrass individuals. Interestingly, in the exposed site, the restoration was quite successful. Some plants were lost due to strong wave action; however, those plants that succeeded to root, showed a good growth condition. In another semi-exposed site characterized by coarse sand environment, the restoration rate was the highest. Within a few months *Z. marina* had a number of new shoots in the most of experimental plots. As the experiment is still on-going a better knowledge of the restoration success will be obtained by summer 2018.

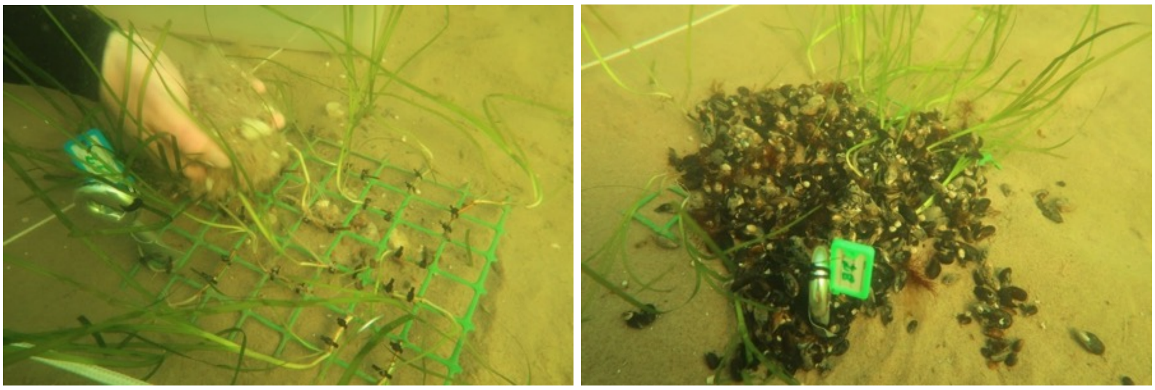


Figure 22. Establishment of a restoration plot of the seagrass *Zostera marina* in Estonia (left). The seagrass shoots are attached to the plastic frame using cable ties and then metal pegs are used to anchor the plot into a sand soil. The diver adds the sand to the plot in order to ensure a stability of the plot. Suspension feeding mussels (*Mytilus trossulus*) are added to the plot at later stages (right) to facilitate the establishment of seagrasses. The right photo shows a fully restored *Z. marina* plot with added mussels. Photos by UTARTU.

4.2.6. Key Habitat 3.3: Seagrass (*Posidonia oceanica*) meadows - Mediterranean

Key Habitat 3.3: Seagrass *(Posidonia oceanica)* meadows – Mediterranean
1. Habitat Description
Seagrass species in the Mediterranean cover the soft bottom habitats of the sea floor from 1 m to approximately 40 m depth, depending on light penetration. *Posidonia* beds are present in all Mediterranean countries from 31˚N to 45˚N (Green & Short, 2003) and cover 2.5–4.5 million ha of sea floor (Pasqualini et al., 1998). The seagrass canopy, rhizomes and roots harbour highly diverse communities for attachment, spawning and protection from predators (Diaz-Almela & Duarte, 2008). The epiphytic community of *P. oceanica* leaves consists of micro- (mainly cyanobacteria and diatoms) and macro-algae, various sessile animals (e.g. *Sertularia perpusilla* and *Plumularia obliqua posidoniae*), bryozoa (e.g. *Electra posidoniae*) and microscopic foraminifera (e.g. *Conorboides posidonicola*) (Colom, 1974 The rhizome network of *P. oceanica* meadows harbours, foraminifer *Miniacina miniacea* and fan mussel *Pinna nobilis*, sessile species such as the worm *Sabella spallanzanii*, red algae (e.g. *Peyssonnelia squamaria* and *Udotea petiolata*), mollusc and crustacean species, various echinoids and sea stars (e.g. *Echinocardium spp.*, *Spatangus spp.*, *Asterina pancerii*) and crinoids (e.g. *Antedon mediterranea*) (Diaz-Almela & Duarte, 2008).
2. Main, most recent, or successful techniques and methods used in restoration actions
The common method used for *P. oceanica* meadow restoration is transplanting. There are few techniques that are used in transplantations, such as using different donors to improve the resilience of the population (Procaccini & Piazzi, 2001) as well as stimulation of root formation (Balestri & Lardicci, 2006). There are two types of transplanting: one is using laboratory grown seedlings and the second is using adult plants from a meadow. Some experiments with *P. oceanica* and other seagrass transplantation have been conducted in France, Italy and Spain. Although *Zostera* species showed success for transplantation, *P. oceanica* have generally failed due to the slow growth rate and the lack of knowledge (Diaz-Almela & Duarte, 2008).
3. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions
A new approach to transplantation is to use storm-generated fragments as source for transplantation of damaged meadows (Balestri et al., 2011). Utilizing fragments, either as a result of storms or of human activities, will increase the awareness for coastal areas and will help restore meadows without damaging others. A new approach is currently tested in Gogova Bay, Aegean Sea, Turkey, within and outside an MPA. Half of the *Posidonia* transplants are covered with cages to

Key Habitat 3.3: Seagrass *(Posidonia oceanica)* meadows – Mediterranean
avoid the effect of the herbivore *Siganus* fish species, and in the remaining experimental plots *Posidonia* transplants are placed with and without *Pinna nobilis* to estimate their effect on seagrass growth and survival (Figure 23).
4. Major barrier/deal breaker and new means/solution/hope to up-scale restoration to the level of degradation
Posidonia is a very slow growing species and currently there are no quick or large-scale solutions for up-scaling restoration efforts.
5. Relevant time scales for restoration
Seagrass matte (1m^2 surface, 0.40 m high) transplantation results in 15% shoot survival rate 7 months after planting (González-Correa et al., 2007). The first year of the transplantation is crucial because the plant starts to adapt to the new environment and develops roots and it is the period when the largest transplant losses usually take place (Diaz-Almela & Duarte, 2008). Average number of leaves produced per shoot in a year is 5.7–8.9, rhizome growth is 1–6 cm and shoot recruitment rate is 0.02 to 0.5 in units yr-1 (Marba et al., 1996). The success of the transplantation will be assessed at the end of 1st year through the number of the survived shoots. The recovery of the habitat/meadow however will take longer because of the slow growth rate of the species (at least 4 years).
6. Other relevant point/key message relevant to the habitat
Survival of the transplants will be the most important indicator of the restoration success.

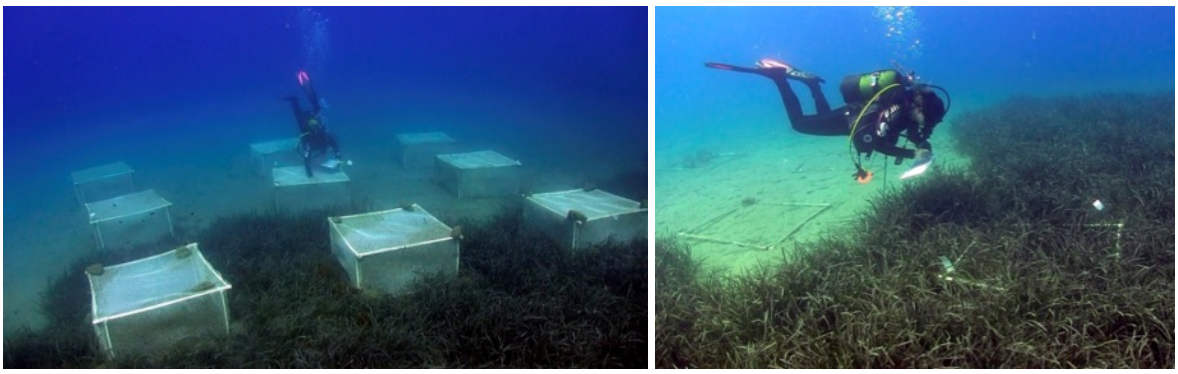


Figure 23. *Posidonia oceanica* transplantation experiment without and with cages to protect transplants from herbivore fish (*Siganus luridus*) in Turkey (see Key Habitat 3.3). Photo by MCS.

4.2.7. Key Habitat 4: The noble pen shell *Pinna nobilis*

Key Habitat 4: The noble pen shell *Pinna nobilis*
1. Habitat Description
Strictly protected species *Pinna nobilis* lives mostly in shallow coastal areas, in seagrass meadows as well as on bare sediment, but can be found in areas as deep as 60 m. This long-lived species is endemic to Mediterranean and it is there considered as the biggest bivalve. Anthropogenic and environmental threats have contributed to the decline of populations of this species in the Mediterranean.
2. Main, most recent, or successful techniques and methods used in restoration actions
Transplantation of *Pinna* as a conservation action to protect the species was successfully tested

Key Habitat 4: The noble pen shell *Pinna nobilis*
(Katsanevakis, 2016; Bottari et al., 2017).
3. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions
The most critical point is careful digging out of bivalves with special attention not to damage the byssus. In a pilot project of transplanting *Pinna* from Pula harbour (where a nautical centre will be build) to nearby Brijuni National park almost 200 specimens were transplanted in both *Cymodocea* meadow and bare sediment with the help of divers from local diving clubs (Figure 24).
4. Major barrier/deal breaker and new means/solution/hope to up-scale restoration to the level of degradation
Transplantation of *Pinna* needs to be done carefully, one bivalve at the time; so a well-organized team of local divers can help in this respect. It is important that they are educated about the major points in the process that may influence the success of transplantation.
5. Relevant time scales for restoration
Time scale with this species is not short as *Pinna*'s life span is over 20 years. The most critical point in transplantation is the immediate survival of the transplants (assessed in one to two months time after the transplantation). Critical points in transplantation/translocation are 1) not to damage byssus and 2) to do the transplantation during good weather conditions since transplanted *Pinna*'s are sensitive to stronger hydrodynamism, until their byssus is again firmly connected to sediment (around two weeks). Further monitoring of transplanted *Pinna* should be performed for at least the next two subsequent years. As *Pinna* reproduces in late spring/early summer for the monitoring purposes it would be necessary to monitor them during that period to see if they are reproducing - in case that they are and no additional higher mortalities are recorded the transplantation is successful and transplanted *Pinna* population could be considered viable and fully functional.
6. Other relevant point/key message relevant to the habitat
Due to recent catastrophic mass mortality recorded in the Western Mediterranean caused by a parasite (Vázquez-Luis et al., 2017), which is spreading, it would be important to urgently establish aquarium cultivation of a number of *Pinna nobilis* populations in closed systems. Only in that way there will be stocks of healthy animals with which repopulation of areas affected by disease could be done.

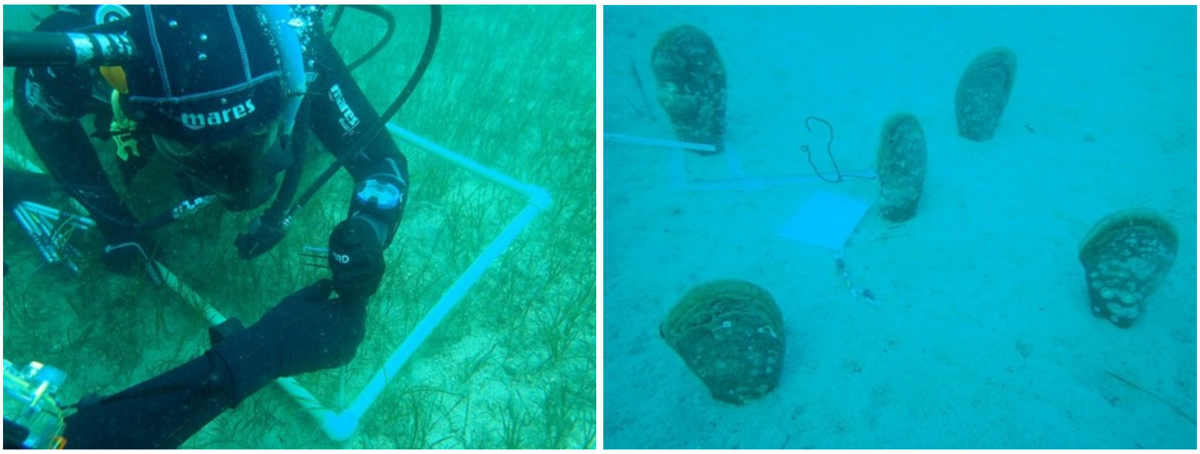


Figure 24. Transplantation of 200 *Pinna nobilis* specimens from Pula harbour to Brijuni National Park (Croatia) in a *Cymodocea* meadow (down left) and bare sediment (down right) with help of local diving clubs (see Key Habitat 4). Photos by PMF – Zagreb.

4.2.8. Key Habitat 5: Coralligenous assemblages

Key Habitat 5: Coralligenous assemblages
1. Habitat Description
Coralligenous assemblages are hard bottoms of biogenic origin that are mainly produced by the accumulation of calcareous encrusting algae growing at low irradiance levels. This habitat is extended around all the Mediterranean coasts with a bathymetrical distribution ranging from 20 to 120 m depth depending on the local environmental variables. Coralligenous assemblages are important biodiversity hotspots harbouring approximately 10% of marine Mediterranean species, most of them long-lived algae and sessile invertebrates (gorgonians, sponges, bryozoans, etc.), which exhibit low dynamics and are very vulnerable to pressures.
2. Main, most recent, or successful techniques and methods used in restoration actions
Restoration protocols on the targeted species for the mesophotic coralligenous habitats are based on transplants of small-medium individuals collected from donor specimens (Linares et al., 2008; Baldaccioni et al., 2010; Fava et al., 2010; Montero-Serra et al., 2017), see Figure 25.
3. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions
The techniques used are time-consuming and the only way to up-scale the restoration actions is by involving volunteers. In a pilot action carried out within MERCES, instructors from different diving centres were involved to transplant more than 400 gorgonians fragments.
4. Major barrier/deal breaker and new means/solution/hope to up-scale restoration to the level of degradation
The deal breaker is both local and global stressors that should be avoided. Regarding timing, summer periods should be avoided because this species may suffer mass mortalities due to exceptional warming. Likewise, for demonstration restoration actions, working in Marine Protected Areas where mainly fishing is limited or absent can enhance the success of transplanting.
5. Relevant time scales for restoration
Transplants can exhibit similar growth and reproductive rates to donor colonies after few years (< 4 yrs). However, the period required to fully recover habitat complexity will tend to be decades given the slow growth rates of most of the species (Montero-Serra et al., 2017).
6. Other relevant point/key message relevant to the habitat

Key Habitat 5: Coralligenous assemblages
Survival and growth of transplants and recruitment rates in the restored location would be the most suitable indicators of the success of the restoration actions.

Figure 25. Transplantation of invertebrate key species of Mediterranean coralligenous assemblages (see Key Habitat 5); apical fragments of red gorgonians ready to be transplanted (left) and monitoring of red coral transplants (right). Photos by Quim Garrabou (left) and Cristina Linares (right) CSIC and University of Barcelona.

4.2.9. Key Habitat 6: Red coral coralligenous assemblages

Key Habitat 6: Red coral coralligenous assemblages
1. Habitat Description
The extremely slow-growing and threatened octocoral *Corallium rubrum* is typical of the Mediterranean area, occurring from 25 to 120 m depth. This precious red coral has a long history of exploitation in the Mediterranean Sea with harvesting following the "boom and bust" cycles with newly discovered beds being overexploited to depletion (Jaziri et al., 2017).
2. Main, most recent, or successful techniques and methods used in restoration actions
Regarding red coral, the most common method is transplantation of fragments using epoxy glue, either directly on the hard bottom or using plastic nets with colonies inserted in (Montero-Serra et al., 2017).
3. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions
To be sure of the success of the restoration we need to know if we are restoring a population with the optimal sex-ratio, which is difficult. The possibility to easily recognize underwater male and female colonies could increase the possibility to restore a nucleus of breeding populations. An important disequilibrium between sexes could compromise the final results. The utilization of plastic panels (Figure 26) to recruit red coral larvae in donor populations could be a good complement to the traditional transplant of adult colonies. This approach is under test in the mesophotic coralligenous habitat.
4. Major barrier/deal breaker and new means/solution/hope to up-scale restoration to the level of degradation
Thermal anomalies and fishing activities, both artisanal and recreational, are the main threats when erect species are transplanted. Red coral colonies, anyway, should be transplanted upside-down under small overhangs, sheltering colonies from mechanical impacts. The larval behaviour is crucial

Key Habitat 6: Red coral coralligenous assemblages
to forecast time scale required by transplanted species to develop a breeding population. If colonies are not placed in the right habitat, larvae can be developed but without any possibility to settle.
5. Relevant time scales for restoration
If the transplants are adequately positioned, red coral could need very few years (2–4) to contribute to the population density with new larvae. A transplantation experiment for 300 *C. rubrum* colonies (intercepted from illegal harvesting in Catalonia, Spain) was highly successful over a relatively short term due to high survival and reproductive potential of the transplanted colonies. However, demographic projections predict that 3 to 4 decades may be needed for fully functional *C. rubrum* populations to develop (Montero-Serra et al., 2017). At the moment no data are available on this issue.
6. Other relevant point/key message relevant to the habitat
The facilitation processes triggered by the presence of red coral colonies are unknown. The species can be easily handled and its reappearance in a diving spot can incredibly increase the attractiveness of the area. A sound knowledge of the demography and life history traits of the target species for restoration actions is imperative for anticipating the dynamics and timescales of restored populations. Furthermore, long-term monitoring of restoration actions using appropriate metrics, for example, survival, but also growth and reproduction, is the only way to assess the effectiveness of these actions.

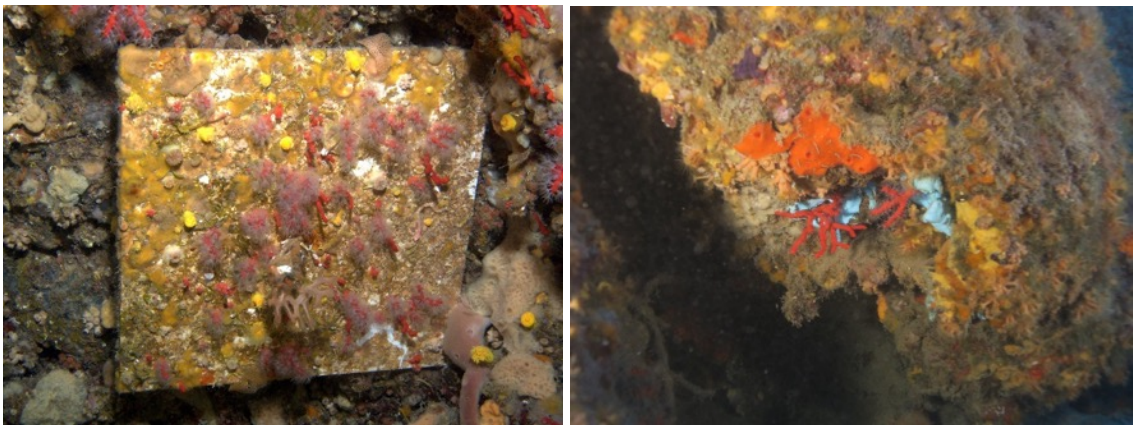


Figure 26. Use of plastic panels to aid recruitment of red coral larvae (*Corallium rubrum*) (left) and transplantation of coral fragments using epoxy glue on hard bottom (right) (see Key Habitat 6). Photos by UNIVPM/GAIA.

4.2.10. Key Habitat 7: Sponges - Coralligenous assemblages

Key Habitat 7: Sponges – Coralligenous assemblages
1. Habitat Description
The actual structural and functional role of Porifera in coralligenous assemblages is yet to be adequately understood but it is evident they play a fundamental role in the homeostasis of the system, keeping under control dynamics such as the growth of bioconcretions, microorganisms fluctuations and nutrient cycles. During the last twenty years the Mediterranean Sea is facing a dramatic change in its assemblages with a fast decrease of many filter feeders. The loss of sponges is scarcely perceived owing to the presence of a skeleton that rapidly dissolves after the sponge death, limiting our understanding on the responses of coralligenous assemblages to climate change.

Key Habitat 7: Sponges – Coralligenous assemblages
The transplants of sponges where their disappearance has been documented is an important opportunity not only for restoration per se but also to increase our knowledge on their key ecological role in the coralligenous habitat.
2. Main, most recent, or successful techniques and methods used in restoration actions
There has been considerable experimental work aiming at farming sponges, with the goal to produce biomass to be used for pharmaceutical or cosmetic purposes (e.g. Duckworth, 2009; Schippers et al., 2012). However, very little work has been aligned with the idea of restoring habitats harbouring sponges (e.g. the various methods described to rear *Spongia officinalis* for the production of biomass – like the one described in Corriero et al. (2004) may not fully apply to restoration). The only published work specifically related to sponge restoration refers to the Caribbean species *Xestospongia muta* (McMurray & Pawlik, 2009), but exclusively considers re-attachment of broken fragments, rather than a comprehensive response to the loss of sponges (e.g. following mass mortality events). Moreover, it appears that different sponge species are more suitable for certain methods than others – this depends on each species morphology/growth strategy, biology, habitat, etc. Said this, the general principle that applies in the majority of approaches at propagating sponges (for the production of biomass or for restoration) is currently based on the use of transplants from donor sponges. Transplants are then "seeded" in different ways, depending on the scope of the work, on the characteristics of the species and on the conditions at the working site (Figure 27).
3. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions
Currently, sponge restoration (or habitat restoration with a considerable sponge component) is in its infancy – the concept itself is innovative. MERCES work is focusing on the identification of successful protocols for a limited number of sponge species, bearing in mind that other species may require very different methods.
4. Major barrier/deal breaker and new means/solution/hope to up-scale restoration to the level of degradation
Local and global stressors should be avoided. Summer periods should be avoided as well, because species may suffer mass mortalities due to exceptional warming. The success of transplants is enhanced where fishing is limited, e.g. in MPAs. Two bottlenecks in the scaling up of such successful protocols stand in: 1) the limited amount of available donor organisms; 2) the needed underwater working time.
5. Relevant time scales for restoration
Initial indicator of success is survival in the few days or weeks after the transplants – manipulation is a primary source of stress. Once the transplantation process is technically successful (i.e., no or very limited immediate mortality), success shall be measured in terms of new sponges generated from transplants. Indeed, most sponges have very slow net growth, especially in a competitive habitat and for massive species, and transplants' growth may be a misleading indicator. However, for this same reason, it may take years to observe a conspicuous appearance of new sponges. To our best knowledge there is not much information supporting forecasts for the recovery of functions and services. Speculations and the scant literature on coral reef restoration and *X. muta* would point in the range of tens of years (McMurray & Pawlik, 2009; Schmahl et al., 2006). A major source of uncertainty lies in environmental changes or in acute stressful events that may trigger mass mortalities, interrupting the recovery process or causing shifts in community compositions (as for sponges, this may turn into different species becoming predominant at a given site before and after the event – examples under study in MERCES in Portofino, Italy).
6. Other relevant point/key message relevant to the habitat

Key Habitat 7: Sponges – Coralligenous assemblages
Being able to work with recruits (and ideally selecting those best adapted to current conditions) might one day facilitate recovery, also addressing points regarding up-scale of restoration and time scales.

Figure 27. Transplantation experiment of *Spongia officinalis* and *S. lamella* (bottom right) in Portofino MPA, Italy, in the framework of MERCES project (various types of plastic and metal screws shown, blue epoxy glue and sponge fragments, see Key Habitat 7). Photos by UNIVPM/GAIA.

4.2.11. Key Habitat 8: Coral gardens

Key Habitat 8: Coral gardens
1. Habitat Description
Coral gardens are defined as dense single- or multi-species aggregations of cold-water corals, where Alcyonacea (gorgonians and soft corals), Pennatulacea (seapens) Antipatharia (black corals) and Stylasteridae (hydrocorals) are the most conspicuous components. Coral gardens of all sorts are distributed in the deep sea of all worlds' oceans. In the Azores, coral gardens are found in areas of high current flow on seamounts and island slopes, typically below 200 m depth, where they form complex three-dimensional habitats that support high levels of biodiversity by providing feeding, spawning and nursery grounds for a wide range of organisms, including commercially important fish species. Cold-water corals' life history characteristics, such as slow growth rates, long lifespan, low fecundity, and larvae with potentially low dispersal capabilities makes them, and the habitats they form, particularly vulnerable to impacts from human activities, which has resulted in coral gardens' listing as vulnerable marine ecosystems (VME´s) of utmost conservation importance.
2. Main, most recent, or successful techniques and methods used in restoration actions
The first time that restoration is attempted for deep-sea coral gardens is within MERCES.

Key Habitat 8: Coral gardens
Restoration protocols are based on techniques developed for tropical coral reefs and Mediterranean red coral populations, whereby transplants of small to medium size coral fragments from adult donor specimens are transplanted to impacted areas (Rinkevich, 1995; Linares et al., 2008).
3. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions
Pilot restoration actions in the Azores involve fisherman and fisheries observers to recover corals accidentally caught during fishing operations and test the feasibility of replanting them back at sea. This approach aims to increase the survival of accidentally caught corals (bycatch) that would otherwise be thrown overboard (discarded) with no likelihood of survival, and restore impacted coral gardens at a reduced cost. Currently, coral fragments are being deployed in landers back to the deep sea in areas known to have suitable conditions for coral survival (Figure 28). For up-scaling this restoration action new transplantation techniques will have to be developed. The use of coral transplants attached to cobbles, as attempted for shallow-water gorgonian populations in the Mediterranean (A. Gori, ShelfReCover project), may represent a potential low cost solution for coral restoration of large spatial areas. Cobbles can be easily thrown from a boat over a large extent of the target restoration sites.
4. Major barrier/deal breaker and new means/solution/hope to up-scale restoration to the level of degradation
Corals are highly vulnerable to human pressures. Therefore, any restoration actions should act in concert with protection measures that remove as much pressures as possible from the area to be restored (e.g. closures to fishing activities), until a certain threshold of size/biomass of coral colonies or area covered by coral colonies is attained. Moreover, because of the patchy or fragmented nature of deep-sea coral gardens, a combination of restoration approaches will likely be necessary, with natural spontaneous regeneration (through fisheries closures, MPAs) at large scales, and assisted regeneration and reconstruction at smaller scales.
5. Relevant time scales for restoration
The life history characteristics of cold water corals suggest that assisted regeneration of this habitat encompass long time scales (from decades to centuries). This aspect may create increased uncertainty in the restoration success of these habitats under future climate conditions.
6. Other relevant point/key message relevant to the habitat
Due to the complexity of restoring coral gardens in the deep sea, along with current limitations for scaling up restoration actions and the extremely long time scales required for restoration success, conservation will better be achieved through both the protection of intact habitats and the restoration of key, crucial, or essential degraded habitats. Survival and growth of transplants and recruitment rates in the restored location are the most suitable indicators of the success of the restoration actions.

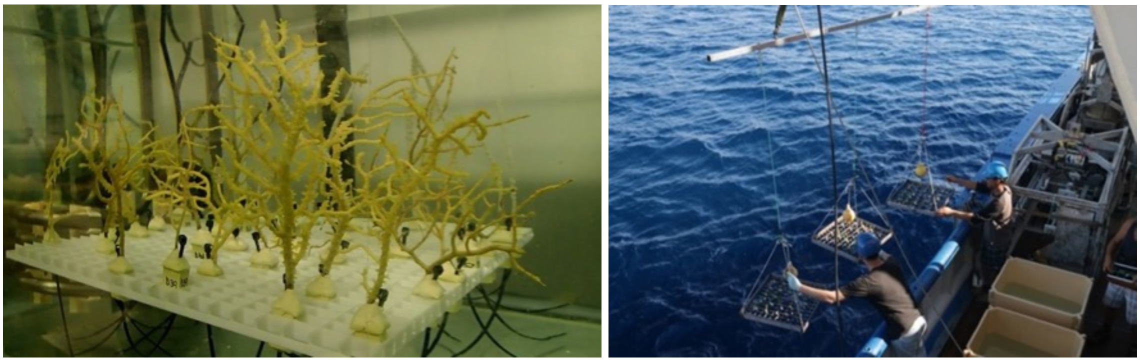


Figure 28. Restoration of deep-sea coral gardens though transplantation of accidentally caught corals (fisheries bycatch) deployed on landers (see Key Habitat 8). Photos by IMAR-UAZ.

4.2.12. Key Habitat 9: Deep-sea communities: seamount soft-bottom communities

Key Habitat 9: Deep-sea communities: seamount soft-bottom communities
1. Habitat Description
Seamounts are prominent features of the world's underwater topography (Wessel et al., 2010). It is estimated that there are potentially up to 100000, seamounts over 1 km high and many more of smaller elevation. Seamounts are often highly productive ecosystems and may play an important role in patterns of marine biogeography. Seamounts are indeed characterized by the presence of large standing stocks of economically valuable fish species and are known to support special biological communities, with high levels of endemic species. Because of their unique characteristics, it has been hypothesized that seamounts may play important roles in ocean biodiversity including acting as centres of speciation, refuge for relict populations, and stepping stones for trans-oceanic dispersal. Seamounts are important to both benthic and pelagic realms (Taranto et al., 2012; Morato et al., 2010). Seamounts are threatened by fishing and bottom trawling in particular, rock-drilling and dredging and potentially in the future by deep sea mining (Morato et al., 2006; Petersen et al., 2014, EU MIDAS project: www.eu-midas.net).
2. Main, most recent, or successful techniques and methods used in restoration actions
At present, we have no evidence of restoration projects on soft bottom communities inhabiting seamounts. We have examples of the restoration approaches referred to the ‘regeneration’ of the ecosystem based on its resilience after the end of disturbance. These approaches can be collectively referred to as ‘passive’ restoration (McDonald et al., 2016). Some investigations studied the faunal recovery after the end of the fishing activities (Althaus et al., 2009). The recovery is however very different when different benthic components (soft bottom communities: meio-, macro- and megafauna *vs* hard bottom communities: corals and sessile fauna) are considered.
3. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions
Extending the area where impacts (fishing or rock-drilling) are ceased and setting up a monitoring plan to assess the success and the efficacy of the ecosystems regeneration using a holistic approach including benthic abundance, diversity (number of species and species community composition) and ecosystem functioning.
4. Major barrier/deal breaker and new means/solution/hope to up-scale restoration to the level of degradation
Major limit for deep-sea ecosystems restoration are the high costs, in particular with regards to ship-time and the equipment requested to work in the deep-sea ecosystems. There is increasing awareness to optimize the findings combining different activities during oceanographic cruises.

Key Habitat 9: Deep-sea communities: seamount soft-bottom communities
5. Relevant time scales for restoration
Regeneration can occur in years-scale for certain groups of soft-bottom communities. For instance, megafauna recovery can occur in 5–10 years for soft-bottom communities after the end of fishing activities (Williams et al., 2010). Similar patterns are observed for prokaryotes, meiofauna and macrofauna in case of impacts induced by drilling activities.
6. Other relevant point/key message relevant to the habitat
The regeneration is strictly related to type of disturbance, the spatial and temporal scale of the impacts, the target organisms (soft vs hard bottoms and associated communities), and local trophic and environmental conditions. Moreover, we have to set up monitoring activities to assess the success of restoration projects.

4.3. Discussion - Key messages from the MERCES cases

4.3.1. Techniques

In general, techniques for restoration projects on coastal and shallow-water habitats are well developed, however, effective evidence-based restoration methods for deeper habitats such as sponge grounds, coral gardens and deep soft bottoms are still lacking. The MERCES project is a pioneer in developing restoration protocols for a number of species where restoration is still in its infancy. Lessons learnt from shallow habitats or from biomass production techniques are being trialled for deep-sea species. The most common restoration technique employed is transplantation of the target restoration species using different life-history stages or methods. This includes, for example, the translocation of adult *Pinna* bivalves (from a harbour site to a MPA as part of compensatory restoration obligation), to transplanting various coral fragments (caught as fisheries bycatch), to hand-picking fertile apical branches to use for *in-situ* and *ex-situ* seeding for *Cystoseira*. Various tools and anchoring supports are used in the field, from simple natural stones (to attach seedlings and fragments), plastic ties, nets, mesh, bags, lines and ropes, screws, glues to various artificial supports, frames, plates etc. (see also Section 7.1 on artificial reefs). A variety of materials are used, including plastic, metal, shells and stones and numerous biodegradable materials (see also Section Litter how restoration contributes or not to adding litter in the marine environment). Going beyond single structural or keystone species approaches (see also Section 7.2 on restoring species), recent efforts have concentrated on positive interactions between species (Suyberbuyk et al., 2016b), such as feedback between seagrass and other ecosystem engineers (one topic of research in MERCES) or considering transplanting the associated microbial community within sediments to increase the chances of success.

4.3.2. Scales

Time scales to restoration and transplantation success vary between the species and key habitats examined. Even time scales to initial success (survival of transplants) vary from a matter of days/weeks (e.g. kelp and sponges) to some months/year for others. Seagrass transplantation success depends on the species, with *Zostera* being able to establish in a couple of months and *Posidonia* growing so slowly that by the end of the first year, success is still uncertain. Comparable growth and reproductive outputs to donor natural populations can be attained within few to several years for some habitats/species (e.g., kelp and macroalgal forests, Verdura et al., submitted). Recovery of different functions and additional species within the same ecosystem will require different times, for example, recovery of associated mobile megafauna and macrofauna and meiobenthic infauna might be different and take considerable time (e.g. Hiddink et al., 2017). Full recovery of functions can take from a few years to several decades (e.g. red corals) to multi-decades or centuries (e.g., deep-sea coral gardens) (Orth et al., 2012; Montero-Serra et al., 2017). The restoration success and time scales to recovery will also depend on the effort invested and the spatial scale of the restoration actions (e.g., spread of transplants, thinly transplanting across large areas, etc.).

4.3.3. Deal breakers

Restoration success will depend on a number of key issues and these comprise the target species characteristics, the methods and techniques used, and parameters, such as site location and site history (degradation causes and impacts), hydrological setting and sediment suitability. There are a number of deal breakers and success rates differ among habitats (see also Section 3), as does the restoration potential (Bekkby et al., 2017) with seagrasses exhibiting mixed potential, low success rates or even complete failures.

Several experiments have been conducted, for example, transplanting *P. oceanica* and most of them have failed. Even in some success stories, it is hard to define the success rate in a short time frame. Restoration of *P. oceanica* meadows requires long time periods (decades) and better knowledge on a number of issues. This includes the essential environmental conditions of *P. oceanica* in the restoration area (sediment type, sedimentation rate, water flow, nutrition, water quality, dissolved organic and inorganic matters of the water column, depth, light penetration etc.); sediment choice is a key factor. Seedlings have highest survival rates on dead matte with no germination on pebbles or gravel (Balestri et al., 1998). Other important considerations include the donor meadow health (or the storm generated material), the location of the restoration area (for example, protected areas are relatively less affected by human activities such as trawling and anchoring), the transplanting season, and the genetic diversity of the donor meadow (Meinesz at al., 1992; Procaccini & Piazzi, 2001).

Similarly, even for faster growing *Zostera* species, the science and practice of restoration have a long way to go and capitalizing on lessons learnt from major failures is important. Most restoration efforts focussing on *Zostera* bed seem to fail because the environmental conditions (water clarity, wave exposure or sediment) are no longer appropriate for seagrass growth. Location/environmental setting and sediment suitability is a major deal breaker; even large scale well planned restoration projects end up failing with transplants being washed away since the sediments can no longer support them (Suyberbuyk et al., 2016a, recent BESE experiments in the Netherlands in the Wadden sea, http://www.merces-project.eu/?q=content/restoration-marine-ecosystems-using-natural-biodegradable-materials). Sand accumulation leading to transplant burial is also an important issue impacting success rate. Depending on intensity of the phenomenon and site location characteristics (e.g. exposure), solutions are available; for example in Estonia (MERCES case study) negatively buoyant ropes were used instead of plastic frame to fix the transplanted seagrass individuals. Sand capping is a new technique to be tested in the North Sea as a way to restore the sediment suitability for restoration. Sediment fertilization has been tested for *Cymodocea nodosa* in nutrient-depleted areas providing rapid colonization of restoration sites but restoration practitioners should carefully evaluate the site setting before applying fertilizers to avoid failure (Balestri & Lardicci, 2014). In the Norwegian seagrass case study other approaches are trialled (under the Indre Viksfjord and MERCES projects) including aeration to improve sediment oxygenation.

4.3.4. Absence of threats, removal of pressures and undesirables: a part of deal breakers.

Absence of threats (*sensu* McDonald et al., 2016, e.g. decontamination, removal of invasives, cessation of over-utilization of resources) and removal of local pressures (e.g. abrasion, changes in siltation and light, smothering, etc.) from any restoration area is a must, either as a prerequisite or as part of the restoration (Bekkby et al., 2017; Smith et al., 2017). The priority for most cases is to first improve the environmental conditions, e.g. reduce eutrophication and nutrient input, and reduce coastal disturbances), reduce degradation sources and/or manage activities and impacts. Our cases demonstrate the effects of eutrophication, for example with the build-up of floating algal mats posing a serious threat to Baltic Sea seagrass meadows (Gustafsson & Bostrom, 2014). Removal of the mats is deemed critical to the survival of seagrass and in that respect not securing funding for their continued removal is a deal breaker, rendering the restoration efforts futile. Restricting certain human activities or banning certain practices is also a prerequisite to restoration; trawling and anchoring in particular causes (for example in coral areas) can cause habitat loss, fragmentation, damage and mechanical impacts. In two of our cases trawl caught material is either used for transplantation or experiments are impacted by illegal trawling activities. Choosing MPAs as restoration areas is convenient because of the greater control of human activities and less influence of certain pressures (see invasive species Section 7.4.2.). In a number of our MERCES cases we deal with undesirable species, from the invasive herbivorous fish in the Aegean case to the grazing sea urchins in the kelp case in Norway and the *Cymodocea* case in Spain. In experimental set-ups this is dealt with exclusion from the experiments (by using cages) but in Norway other solutions are suggested. These include the use of chemicals, creating the conditions for commercially profitable harvesting of the sea urchins and/or by managing the crab fisheries to allow crabs to control the urchins.

4.3.5. Up-scaling - innovations and hope for solutions

In the last 5 years, after the 2012 ratification of the CBD Aichi Biodiversity targets, the mainstreaming and up-scaling of restoration and rehabilitation have become widely recognized among international decision makers, including the EU and many governments worldwide (Aronson et al., 2017). To turn these commitments and aspirations into restoration actions, reversing loss and matching the level of degradation caused by human activities, needs mature science and social acceptance in many forms (from policy to funding to social licence and citizen science). Most importantly, to up-scale and mainstream ecosystem restoration, we need a family of restorative activities acting at different levels while restoring natural capital and ecosystem services (Aronson et al., 2007). These activities range, for example, from threat removal, unassisted regeneration to remediation and depend on management and input-output solutions, while technological innovations, science-industry solutions and citizen science and volunteering are of paramount importance for assisted regeneration by restoration.

A few technological innovations are demonstrated by MERCES experimental cases (e.g., specialized equipment for skimming the water surface to collect floating mats, underwater landers employed to place metal wire plates with coral fragments on the deep-sea bed) but more are in use in support of restoration both in the laboratory and in the field (see Section 7.7. on technologies). Low-tech approaches are also used and this includes throwing stones (with fragments attached) into the sea from boats in an analogy with bombing the land with trees by plane in reforestation projects. Additional approaches will have to be developed to support up-scaling.

Finally, support from citizen scientists and volunteers is the common strength of many success stories in restoration (see Section 5 on restoration projects and Section 7.7.3 on citizen science). This is also evident in several of the highlighted MERCES cases here, for example, with the support of local diving communities being instrumental in the restoration cases of gorgonians in Spain and *Pinna* shells in Croatia. A drastic shift in social thinking and awareness will undoubtedly help up-scale restoration and reverse degradation along with employing all possible restoration and management approaches (Aronson et al., 2017).

5. Review of Recent Restoration Projects

5.1. Introduction

This section aims to acquire knowledge and evaluate the current state of marine restoration initiatives, emphasizing in the European Seas but also gaining a vision to current world trends. This knowledge was considered essential in order to understand how commonly restoration approaches are applied to marine ecosystems in Europe, as well as their main targets, requirements, and outcomes. Interpretation of the extracted data aims to help towards the identification of potential strengths, as well as implementation weaknesses regarding existing applied marine restoration schemes. Liu et al. (2016) in their systematic review on China's coastal restoration projects spanning from the 1950's to date pointed out the paramount importance of such synthetic understanding as it provides valuable lessons that will help in the future success of restoration activities. Considering that marine restoration is a relatively new concept and in order to restrict to the current trends and practices, the past decade (i.e. 2007 to present) was selected as the appropriate timespan for the queries. The subject of the performed searches was research or applied projects that either exclusively or partially involve marine restoration approaches.

5.2. Methods

For the collection of restoration projects at the European scale that either exclusively or partially involve marine restoration approaches, web searches were performed with a standard search engine (Google), using keyword combinations. Keywords included "restoration", "marine" or "coastal", "Europe" and "project". For all searches, the first 100 search results were reviewed. Apart from the open web search, specific searches for relevant projects were performed in European project inventories, such as the 'keep' database for EU Interreg projects (https://www.keep.eu/keep/search), the EU Life projects database (http://ec.europa.eu/environment/life/project/Projects/index.cfm) and the Community Research and Development Information Service of the European Commission (CORDIS: http://cordis.europa.eu/projects/home_en.html), as well as national research project lists, usually maintained by academic departments or development and innovation clusters and networks (e.g. https://www.uantwerpen.be/en/rg/ecobe/research/research-projects/ from University of Antwerp, Belgium; http://en.polemermediterranee.com/DAS-Projects of the Pôle Mer Méditerranée). At the global scale, the searchable Global Restoration Network database (http://www.globalrestorationnetwork.org/database/), maintained by the Society for Ecological Restoration was queried, as well as the Reef Resilience initiative (http://www.reefresilience.org/case-studies/), the latter restricted to tropical coral reefs. Besides web searches, expert suggestions were asked and evaluated. Furthermore, the project's dataset was enriched with projects targeting on the MERCES key habitats which were examined in the web review synthesis presented in Section 6.3 of the present report.

The results were catalogued in Microsoft Excel, and information was extracted and assigned to 17 data fields:

1. Country: the country where the project or the restoration action is implemented.
2. Region: the MSFD Region where the country of the restoration project/action belongs. Options to this field are: (a) Baltic Sea, (b) North-East Atlantic, (c) Mediterranean Sea, (d) Black Sea, (e) Outside MSFD, (f) Multiple. The latter category refers to actions/projects implemented in multiple regions.
3. Subregion: the MSFD sub-region where the country of the restoration project/action belongs (applying only for the North-East Atlantic and the Mediterranean Sea). Options for the North-East Atlantic are: (a) Greater North Sea, including the Kattegat, and the English Channel, (b) Celtic Seas, (c) Bay of Biscay and the Iberian Coast, (d) Macaronesian biogeographic region (Azores, Madeira, Canary Islands). Options for the Mediterranean Sea are (a) Western Mediterranean Sea, (b) Adriatic Sea, (c) Ionian Sea and the Central Mediterranean Sea, (d) Aegean-Levantine Sea.
4. Launch year: the year the restoration project/action started.
5. End year: the year the restoration project/action ended.
6. Timespan of the restoration project/action: if not stated, derived from the launch and end years.
7. Current status of the project, described as either completed or on-going.
8. Scope of restoration, with two options i.e. (a) experimental or (b) operational); the first refers to projects/actions of research or demonstration nature, while the latter refers to projects/actions attempting actual restoration of degraded marine habitats.
9. Number of restoration application sites, with two options: (a) single or (b) multiple.
10. Budget of the project, as officially reported. All international currencies were transformed (September 2017) to euros for comparative purposes.
11. Funding source, falling into three categories, (a) EU, (b) national, and (c) private sector.

12. Coordinating authority, classified as (a) university or research body, (b) government, or (c) private.
13. Target of restoration action, with 3 options available: (a) habitats, (b) species, or (c) pressures.
14. Type of restoration, according to the definitions of Elliott et al., 2007. Options to this field are: (a) methodological, i.e. targeted to the development and evaluation of methodologies, pilot studies, and demonstration of restoration techniques, (b) restoration, i.e. actions taken in an effort to return a degraded habitat back to its original (undisturbed) state, (c) enhancement, i.e. actions in an effort to increase the ecological value, goods and services of existing habitats, (d) compensation, i.e. actions applied outside the degraded area in an effort to compensate for losses caused, (e) habitat creation, i.e. anthropogenic intervention which produces a habitat not previously there and may be within or outside the degraded area, (f) mitigation, i.e. actions in an effort to reduce pressures causing the degradation of a habitat, and (g) replacement, i.e. actions in an effort to turn a degraded habitat to a different, improved state than the original state.
15. Methods implemented (loosely based on the definitions in Bayraktarov et al., 2016). Options to this field are: (a) translocation, (b) seed harvesting and planting, (c) artificial substrates, (d) development of tools and methods, (e) hydrological restoration, (f) removal of contaminants and litter, (g) removal of invasive species, (h) protection measures, (i) translocation and juvenile cultivation, (j) multiple.
16. Inclusion of protected sites (e.g. MPAs or NATURA sites); with three options (a) Yes (b) No, or (c) No data.
17. Reference Link, providing a web link to the project.

The data included in the catalogue were graphically summarised, either individually or in combination, and used to describe the current state of European marine restoration initiatives.

Apart from the comprehensive inventory of recent European projects, several restoration projects or initiatives at the global level were selected according to criteria of broad implementation scale, effective integration of sectors and approaches, or extended timespan. These projects were individually showcased and the main aspects of their scope, approach, and implementation practices were presented.

5.3. Results

In total, 42 European projects that either exclusively or partially involve marine restoration approaches and were implemented in the past decade, were included in the inventory.

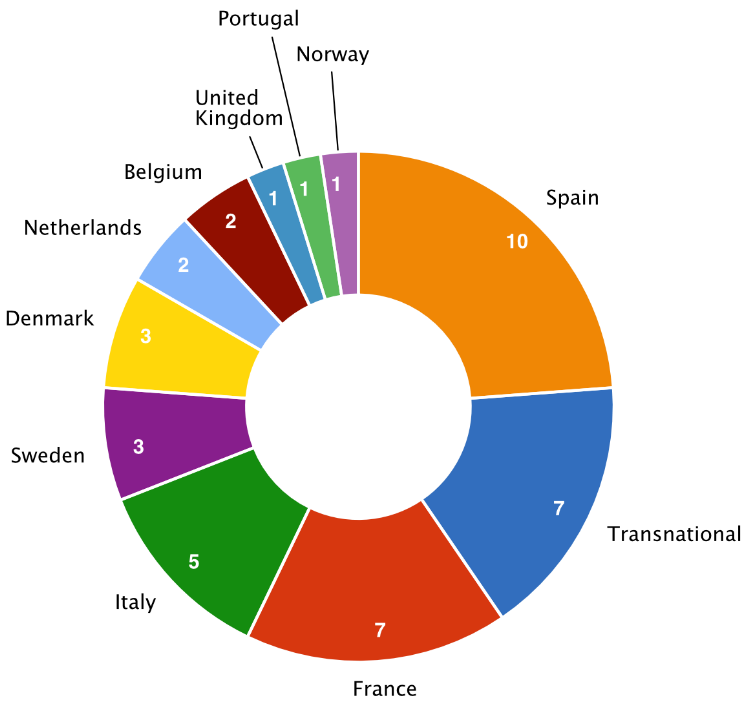


Figure 29. Distribution of the 42 recent European restoration projects examined across countries.

Ten European countries (9 belonging to the EU and Norway) have implemented marine restoration projects at the national level in the past decade (Figure 29). Out of them, the most prolific are Spain (10 projects), France (9 projects) and Italy (5 projects). Seven out of the 42 projects (17%) involve transnational cooperation. Regarding the region of implementation (Figure 30), the Mediterranean Sea is the one that hosts the most marine restoration projects, reaching 52% of the examined recent initiatives. The western basin is by far (92%) the main field for marine restoration applications in the Mediterranean, with only one project implemented in the Ionian and the Adriatic Seas (Figure 30B). The Mediterranean is followed by the North-East Atlantic (26%), with the Greater North Sea emerging as the main implementation field (Figure 30C). Four restoration projects are implemented in multiple regions and one outside the MSFD regions, in the Norwegian Sea (Norway).

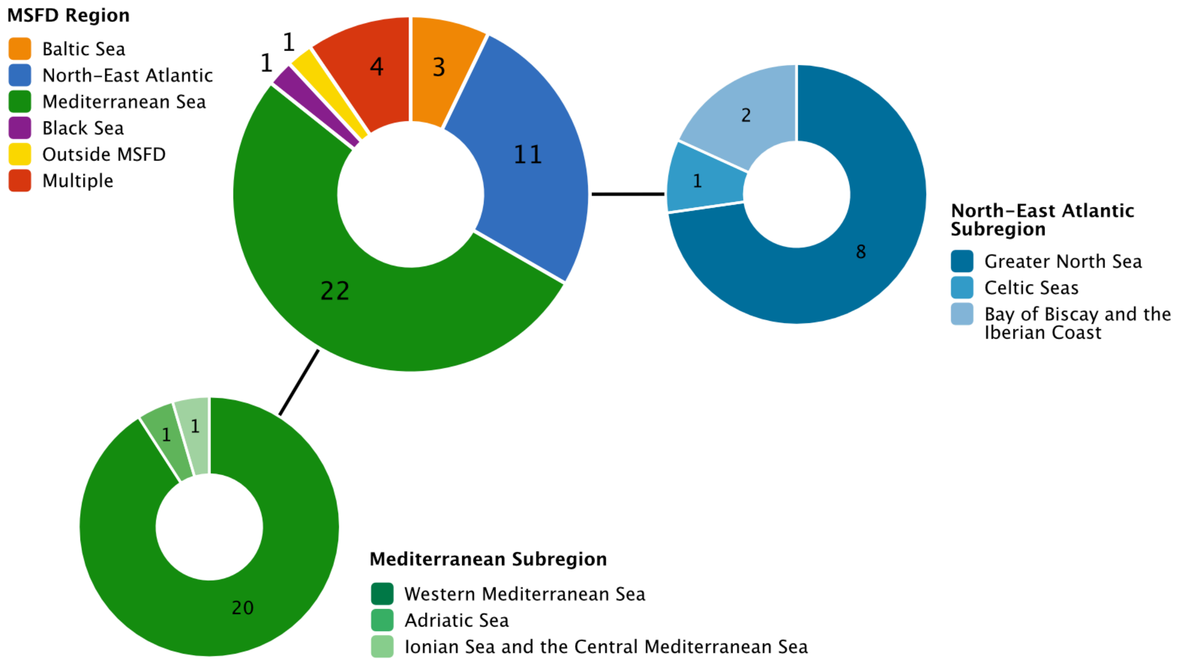


Figure 30. (A) European MSFD region where the project or the restoration action is implemented for the 42 recent European restoration projects examined, and subdivision into sub-regions for (B) the Mediterranean Sea and (C) the North-East Atlantic.

The average timespan of the studied recent European marine restoration projects is 5 years, with duration extending from less than one year (small-scale applied restoration initiatives) to 8 years for long-term implementations (Figure 31). However, for a considerable number of sources (33%), we were unable to retrieve information regarding the duration of the project. A good portion of the projects (38%) are still on-going during the preparation of this report, while 40% of them were completed by 2017. For the remaining 22% no information is given.

The scope of restoration actions within the examined projects are divided equally between operational (implying specific actions to restore a degraded marine area) and experimental (in the sense of restoration actions designed to support research or demonstration activities) (Figure 32). Ten percent of the examined projects had both operational and experimental scope, while for another 10% no data were retrieved.

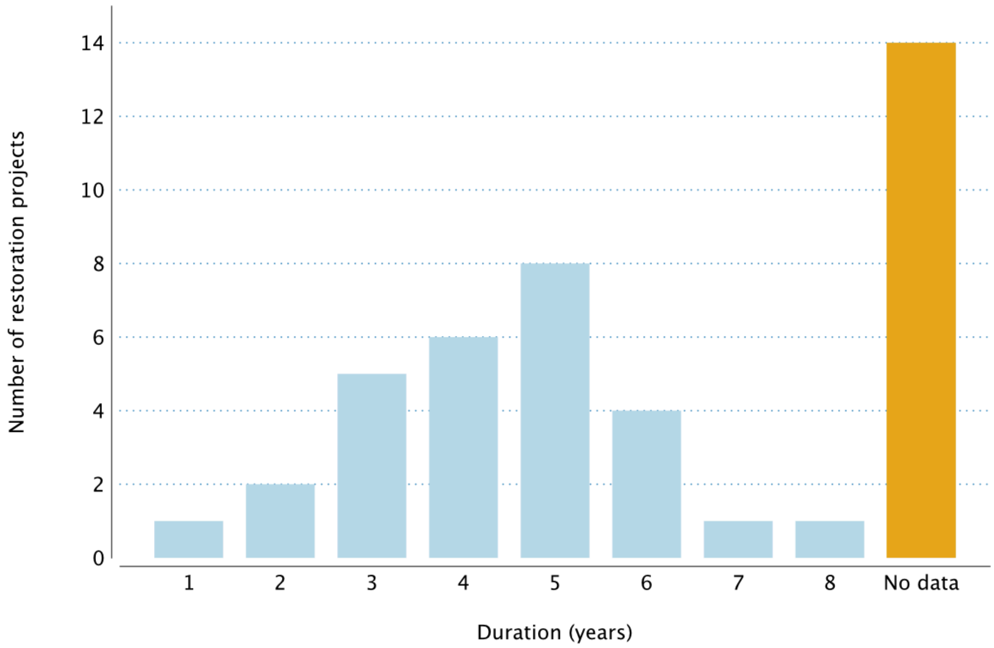


Figure 31. Duration in terms of years for the 42 recent European restoration projects examined.

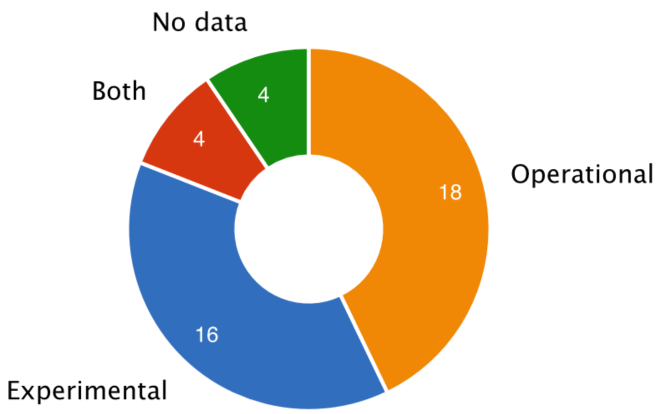


Figure 32. Number of restoration projects with regard to the scope of the initiative for the 42 recent European restoration projects examined.

Regarding the number of application sites for restoration, most projects (45%) focused on a single location, while 30% involved multiple sites of implementation (Figure 33). In most cases (40%), at least one Marine Protected Area (MPA) was included among the restored sites (Figure 34). This was most commonly a Natura 2000 site, but also designated Marine Parks and other protection schemes were represented.

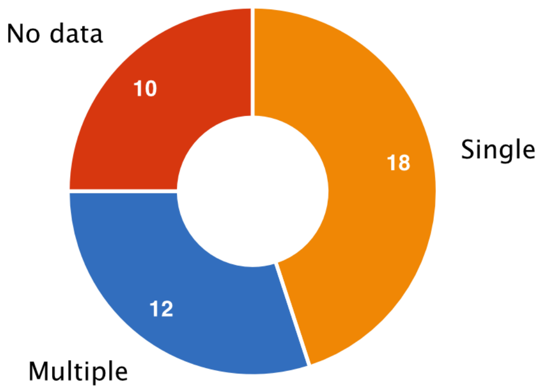


Figure 33. Number of restoration projects with regard to the number of application sites involved for the 42 recent European restoration projects examined.

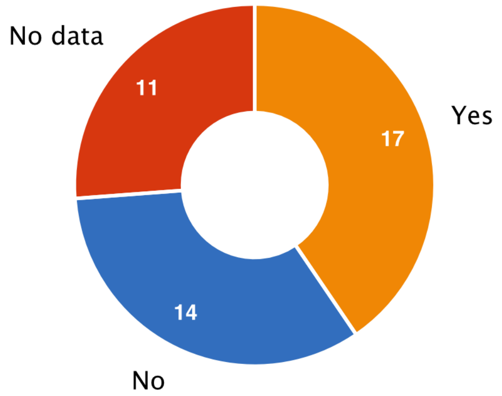


Figure 34. Number of restoration projects including MPAs among the restoration sites for the 42 recent European restoration projects examined.

The funding source for the studied recent European marine restoration projects was most commonly (45%) the European Union (Figure 35). However, half of those EU-funded projects are Life projects, which often require a substantial partial national co-financing. 31% of the total projects were fully nationally funded, while 7% are funded by the private sector, occasionally with some amount of national co-financing. The budget of the examined projects ranges from 100.000 Euros to more than 20 million Euros. The average budget is 3.5 million Euros; however, the median is at approximately 1.5 million Euros. A fair proportion of the projects (14%) are low-budget (100–500 KEuros), while the same proportion applies to projects with individual budgets of over 3 million Euros (Figure 36).

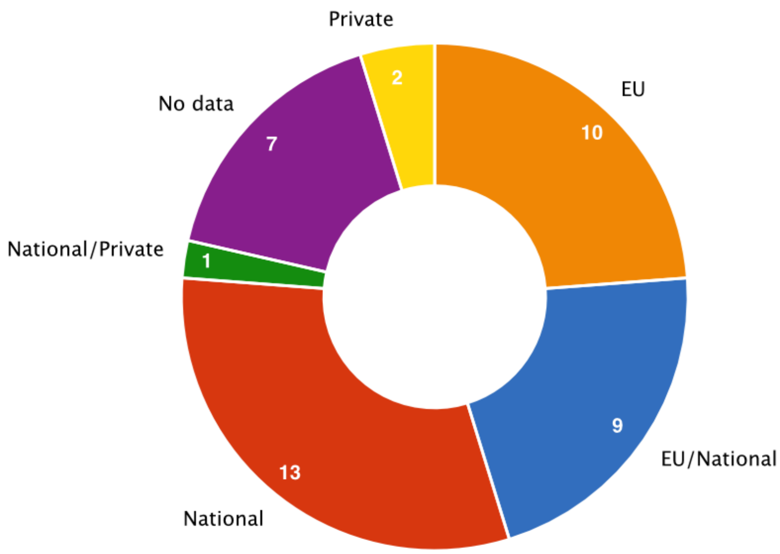


Figure 35. Number of restoration projects by source of funding for the 42 recent European restoration projects examined.

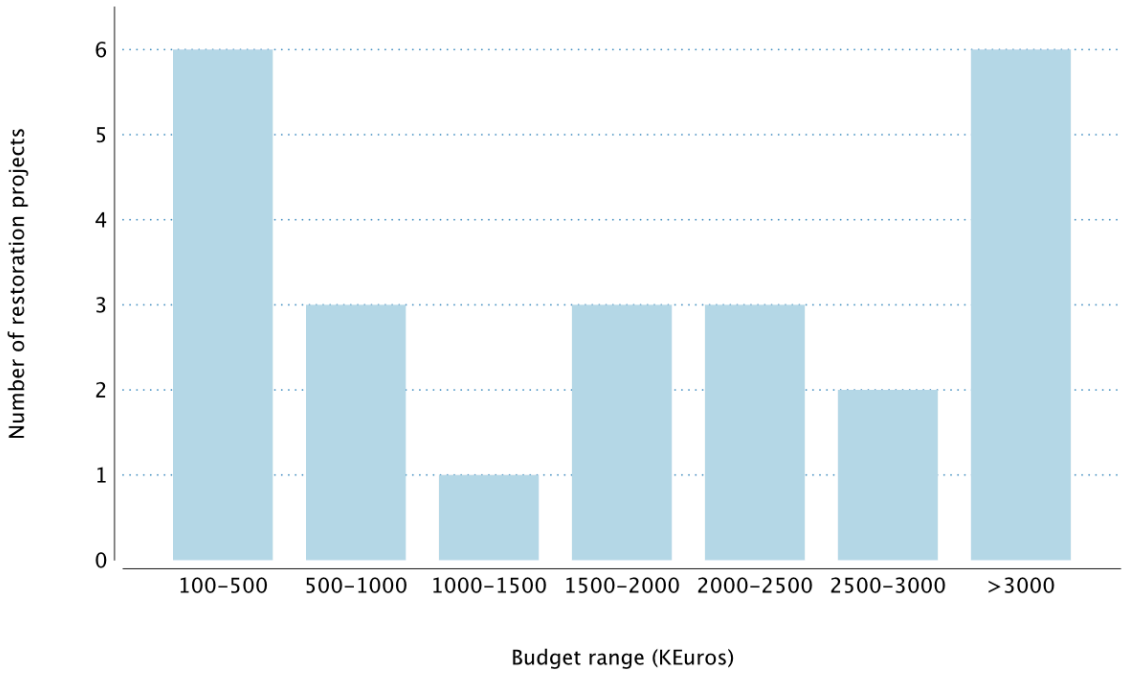


Figure 36. Distribution of budget (KEuros) across the 42 recent European restoration projects examined.

The coordinating body of the studied restoration projects was most often (52%) academic, i.e. a university or research institute (Figure 37). Government, in the form of local authorities, management bodies, and environmental agencies, was the coordinator for 26% of the projects. The private sector, mainly via NGOs, was coordinating a smaller part (14%) of the projects.

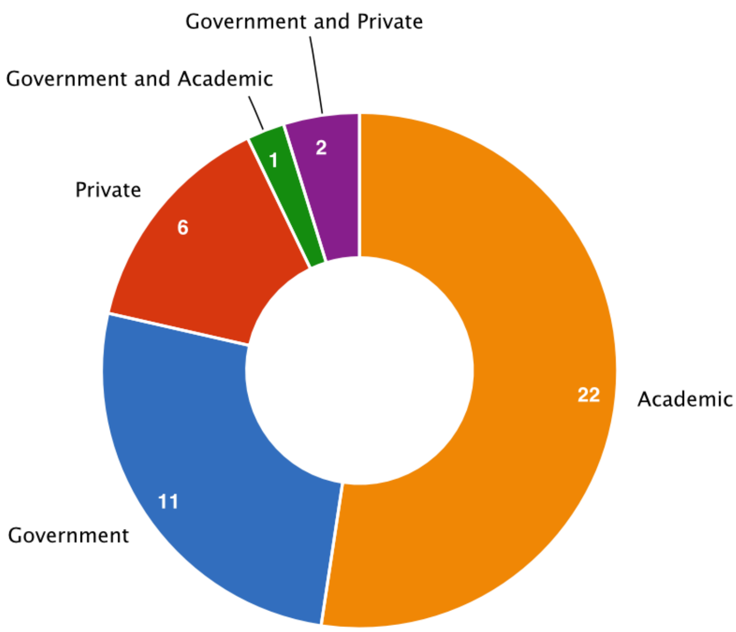


Figure 37. Number of restoration projects by coordinating body for the 42 recent European restoration projects examined.

Seagrasses (mainly *Posidonia oceanica* and *Zostera marina* and *Z. noltii*) emerge as the main target (36%) of the restoration action for the studied recent European marine restoration projects (Figure 38). Saltmarshes follow with 14% and include projects applying hydrological restoration and other enhancements to intertidal habitats. Hard substrates and reefs come third (12%), concerning mainly the creation of new hard-substrate habitats for the settlement (induced, facilitated or natural) of sessile organisms. Marine algae and pollution mitigation share a percentage of 10% for each. The first category includes mainly *Cystoseira spp*. translocation and regeneration applications; although algae, kelps are treated as a separate category, represented by only one project (2%) at the European scale. The final category concerns the active removal of marine litter (lost fishing gear, artificial structures, and general litter), as well as other contaminants. Coral restoration and invasive species removal are represented by two projects (5%) each. It is interesting to note that three of the examined projects (7%) incorporate multiple restoration targets, employing integrated or interdisciplinary approaches. These projects are, namely, MERCES (transnational collaboration), GIREL (France) and DRIVER (France).

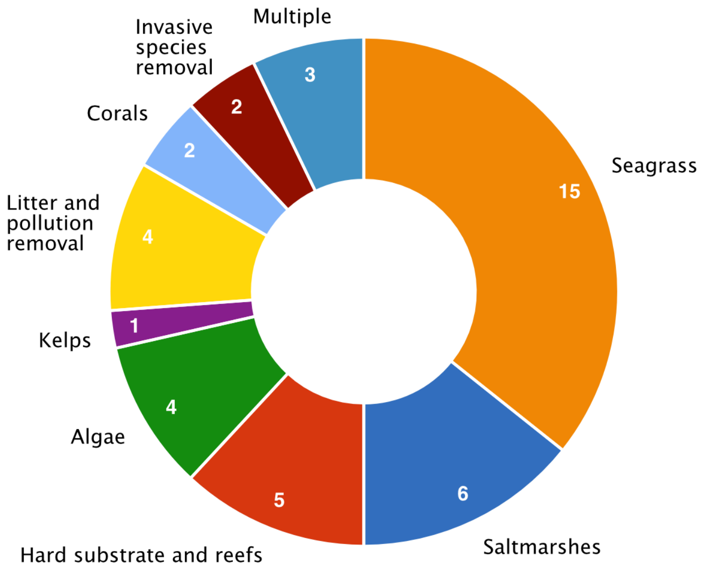


Figure 38. Number of restoration projects by target of restoration action (species, habitat, or pressure) for the 42 recent European restoration projects examined.

Regarding the type of the restoration action employed, the bulk amount (33%) of the studied recent European marine restoration projects are research-oriented, in the sense that they are targeted to the development and evaluation of methodologies, pilot studies, and demonstration of restoration techniques (Figure 39). Apart from the methodological projects, the most common type of action (24%) is restoration, i.e. actions taken in an effort to return a degraded habitat back to its original (undisturbed) state. Enhancement (i.e. actions in an effort to increase the ecological value, goods and services of existing habitats) follow with 14%; these projects mainly employ the introduction of artificial habitats in marine ecosystems, or the manipulation and engineering of the hydrological characteristics of intertidal habitats. Two categories, compensation and habitat creation, share a 7% each. Compensation refers to actions applied outside the degraded area in an effort to compensate for losses caused; this category applies to three projects from our list, dealing with translocation of deep-sea corals and seagrasses in new, not necessarily degraded, habitats. Habitat creation concerns anthropogenic intervention which produces a habitat not previously there and may be within or outside the degraded area; in our catalogue, this category includes diverse initiatives employing the creation of structures for the settlement of kelps (RESTORE project, Norway), artificial hard substrates (Bluereef project, Denmark), as well as newly-created marshes (Steart Marshes, UK). Among the recent European marine restoration projects there are single cases for both mitigation (i.e. actions in an effort to reduce pressures causing the degradation of a habitat) and replacement (i.e. actions in an effort to turn a degraded habitat to a different, improved state than the original state). The first (POSEIDONE project, Italy) concerns the deployment of protective structures to restrict seafloor fishing and its impacts to degraded seagrass meadows, while the latter (Port of Den Helder, Netherlands) employs actions for the improvement of the state of the habitat within and around a marine port.

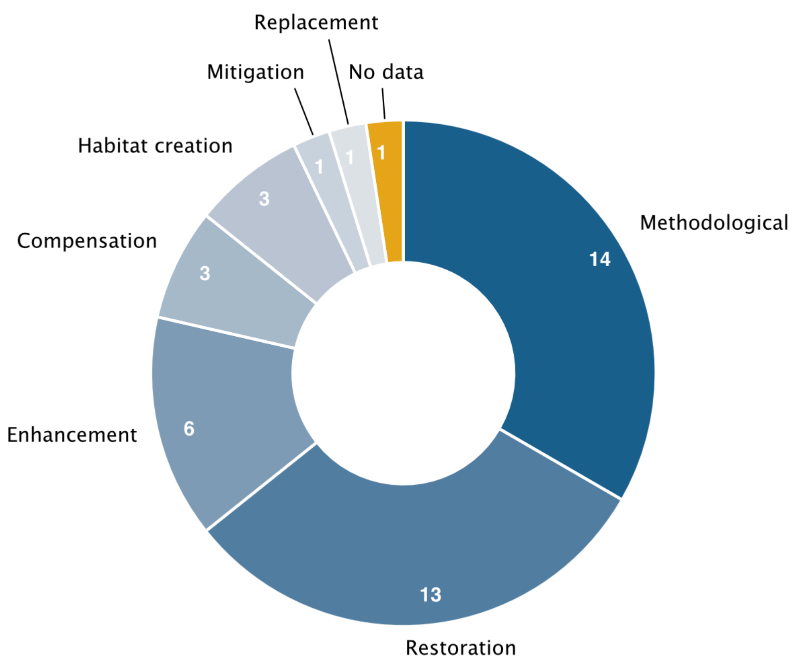


Figure 39. Type of the restoration actions (based on definitions by Elliott et al., 2007) for the 42 recent European restoration projects examined.

A variety of restoration methods were employed in the 42 studied recent European marine restoration projects, as visible in Figure 40. The most common (21%) is species translocation, i.e. collection of developed plants (seagrasses or algae) or sessile animals (e.g. corals, sponges) from one location and restocking in another (for example SeResto, Italy; ShelfReCover, Spain). Seed harvesting and planting (14%) is relevant to translocation but, instead of utilizing adult specimens it relies on collection or production of seeds for the enrichment or reestablishment in degraded habitats (e.g. see Novagrass, Denmark); among the presently catalogued projects, this method exclusively concerns seagrasses. The second rank is also shared by artificial substrates (14%), which may refer to the introduction of artificial structures for either the enhancement of existing hard substrates, or the creation of new habitats (e.g. see projects Bluereef, Denmark and REEFS, transnational collaboration). Development of tools and methods (fourth, 12%) is relevant to the creation and evaluation of methodological approaches towards restoration, e.g. development of mathematical models for hydrological restoration, optimization of restoration practices, development of novel restoration methodologies. Hydrological modifications and active removal of contaminants and litter rank also high at 10% each, while active removal of invasive species stands somewhat lower at 5% of the total examined projects. Single cases of initiatives are present for applied protection measures for the mitigation of pressures (POSEIDONE, Italy), as well as for algal (*Cystoseira*) juvenile cultivation for restocking purposes (Marine Forest, Spain).

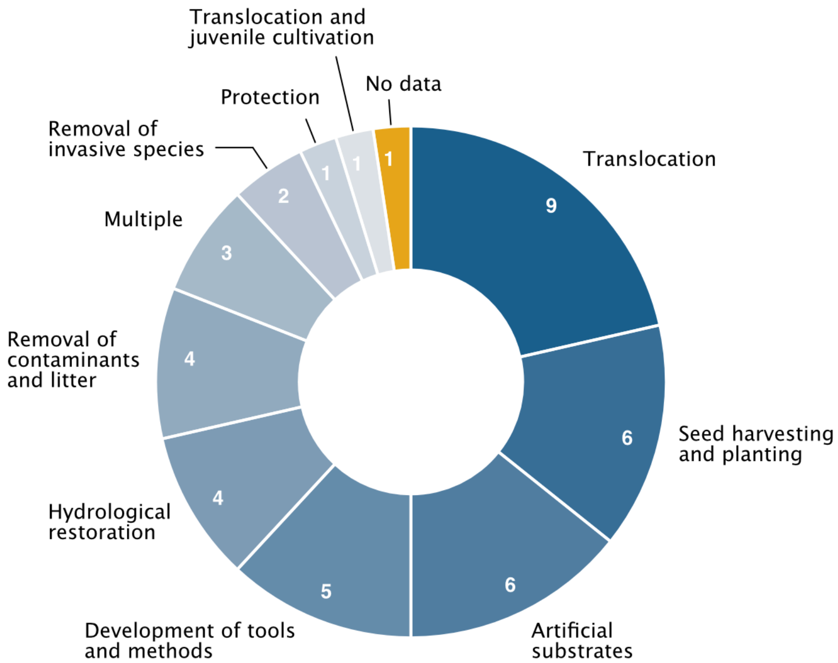


Figure 40. Type of the restoration actions (based on definitions by Elliott et al., 2007) for the 42 recent European restoration projects examined.

5.4. Discussion

Marine restoration initiatives are not evenly spread across Europe. The Mediterranean Sea emerges as a major field for the implementation of restoration projects in the past decade; approximately half of the restoration initiatives examined herein are conducted in Mediterranean waters. Some Mediterranean countries (Spain, France and Italy) are quite active and have taken over several restoration initiatives during the past decade; together they account for 53% of the total examined projects. However, there is a prominent geographical accumulation to the western Mediterranean basin, since only single restoration initiatives are implemented in the Adriatic and the Central Mediterranean (Life4MarPiccolo and SeResto, respectively), while no restoration initiatives are visible in the Eastern Mediterranean and the Levantine. Common goals and stronger collaboration across EU has been suggested by Hermoso et al. (2016) as the ultimate strategical approach to face environmental problems. In this direction, implementation gaps in ecosystem service restoration may be overcome by the operation of large-scale multinational restoration initiatives, such as MERCES, which involve Central and Eastern Mediterranean countries (Greece, Turkey, Croatia). As EU currently evaluates the relevance of its environmental policies (Hermoso, 2015) and the efficiency of the conservation efforts implemented over the last two decades (Hermoso et al., 2016), it is timely for thoughtful consideration of restoration needs, giving prominence to the marine environment, pointing out the gaps in funding at EU scale and the need for more strategic funding guidelines.

The European Union appears as a major financial promoter for marine restoration initiatives, funding (fully or partially) almost half of past decade's initiatives in the European continent. National financing is somewhat equally important, followed by a small percentage of privately-funded projects. Since nationally funded projects usually imply local, small-scale implementation, it can be presumed that their respective budgets are also low-level. Nevertheless, this could not be confirmed by the collected information, since for most cases of nationally funded initiatives the budget was not visible in the data. However, it is interesting to note that the costliest project out of those examined (Steart Marshes, UK; see also the global restoration projects showcase, this section) is jointly financed by national and private funds.

Generally, there is no prominent trend visible regarding the amount of financing for restoration-related projects. Initiatives are evenly spread across the low- (less than one million Euros), medium- (between 1 and 3 million Euros), and high-budget (over 3 million Euros). However, since many of the projects examined are not fully dedicated to marine restoration but have a restricted component responsible for a restoration application or testing, it is not always possible to determine which amount of the total budget is actually channelled to the restoration effort. The lack of detailed restoration cost information and how they are distributed among different categories of expenses is however the *status quo* in ecosystem service restoration studies (de Groot et al., 2013) and more prominent with regards to the marine environment (Bayraktarov et al., 2016).

The duration for all projects examined is 8 years or less, with an average of 5 years. Considering the slow processes involved to the progress of marine restoration practices (Bell et al., 2014), this appears short and is probably justified by the experimental nature of most of the examined projects (see below). This has also been found by Bayraktarov et al., (2016), which report that most marine restoration projects in their study had also a short duration of about 1-2 years, corresponding to the duration of development projects, research grants, or academic theses), with implications for assessing the recovery of ecosystem services and functions.

The examined recent European marine restoration projects are equally shared between experimental (i.e. those investigating and evaluating the application of marine restoration) and operational scope (i.e. those targeting to resolve actual habitat degradation problems). It is interesting that the latter are those with the most increased budgets (11 out of 12 operational-scale projects with financial data available have total budgets of more than one million Euros). It is also interesting to see that a fair amount of the projects with experimental scope (9 out of 16) are nationally funded, implying that marine restoration research and demonstration is interesting for national financing bodies. This is also highlighted by the fact that for a substantial proportion of the examined projects (14 out of 42) government bodies (e.g. local authorities, environmental agencies) are coordinating, or participating to the project's coordination scheme. Another notable point is that there is an absence of funding mechanisms commonly present in global restoration initiatives (e.g. the World Bank or the Global Environmental Facility) among the examined European projects, which may highlight the major role of the EU as an overarching funding mechanism for research and development in the European continent.

It is evident from the analysis of the restoration target for the examined recent European marine restoration projects that some types of marine habitats or organisms are more commonly addressed by applied restoration actions. These are the seagrasses and algae, along with the intertidal systems (saltmarshes) and hard substrate and reefs. The reason for this bias may be mostly related to the effectiveness of currently present restoration methods for those ecosystem components, or their accessibility, since as is apparent in the MERCES D1.1 report (Bekkby et al., 2017), degradation –and, hence, need for restoration– can apply to a varied array of marine habitats and species assemblages. Also, since most of the examined projects target a single component of the marine ecosystem, there is an apparent need for multi-dimensional approaches, or initiatives that would combine the knowledge from previous efforts towards integrated restoration of degraded marine habitats.

5.5. Iconic Projects

Whilst undertaking the review of the recent European restoration projects, a number of iconic projects were identified including some beyond Europe. These projects are iconic with respect to either their scope/approach, subject, methodology or extension and include 7 projects, from Sweden, USA, the UK and different sea areas, covering seagrasses, saltmarshes, mangroves, estuarine habitats and shallow and deep corals. A short description of the projects are given in the following sections with images from the individual projects in Figure 41 sourced from their websites.

5.5.1. Interdisciplinary research for seagrass restoration – Sweden's Zorro project

http://havochsamhalle.gu.se/english/ocean-science/zorro---eelgras

Zorro is an interdisciplinary research program started in 2011 at the University of Gothenburg. It has the goal to improve the management of coastal ecosystems with focus on eelgrass meadows along the Swedish west coast. The program constitutes the base for research collaboration between marine ecologists, environmental legal scholars, and environmental economists about management and restoration of eelgrass ecosystems, and is carried out in close collaboration with local authorities and the Swedish Agency for Marine and Water management. The research within Zorro, apart from developing cost-effective methods and guidelines for large-scale restoration of eelgrass in Sweden (Infantes et al., 2016), has equally focused on the investigation of ecological causes of eelgrass loss, economic valuation of eelgrass ecosystem services, legal aspects of eelgrass exploitation and management, as well as methods and policies for mitigation of eelgrass net-losses.

The research has resulted in both scientific publications as well as reports, guidelines and suggestion of policies for management and restoration of eelgrass in Sweden (e.g. Moksnes et al., 2016) and has also developed video guidelines for eelgrass restoration. The research within the program is financed by the Swedish Environmental Protection Agency, Swedish Agency for

Marine and Water Management, the Swedish Research Council, and local administration resources.

Zorro integrates a number of research actions investigating different aspects of eelgrass restoration:

- No-net-loss and restoration of marine habitats: Legal and ecological constraints and solutions (2012–2017)
- Developing management and restoration of eelgrass ecosystems (2014–2017)
- Integrating seascape ecology and ecosystem services of eelgrass meadows for marine spatial management (2016–2018)
- Towards science-based coastal management: Tipping points for seagrass conservation and restoration (2015–2019)
- Long-term carbon and nitrogen storage in Swedish eelgrass sediments (2015–2017)

5.5.2. Volunteering for large-scale seagrass restoration – VIMS, USA

http://web.vims.edu/bio/sav/index.html

Scientists from the Virginia Institute of Marine Science (VIMS), collaborating with The Nature Conservancy (TNC) volunteers began planting eelgrass seeds and shoots into Virginia's coastal bays in 1997. From 1999 through 2010, VIMS staff and TNC volunteers had collected and broadcast 37.8 million eelgrass seeds across 309 acres in 4 coastal bays. Those plantings have now expanded through natural re-seeding into 4,200 acres of lush eelgrass meadow, denoting VIMS' Submerged Aquatic Vegetation (SAV) program as the largest and most successful seagrass restoration project in the world (Orth & McGlathery, 2012). Along the project's duration, restoration efforts have taken place with both seeds and transplanted shoots, while the techniques needed to successfully harvest, keep, and plant eelgrass seeds have been gradually perfected. Moreover, the seagrass restoration effort is part of an even broader program that also aims to restore bay scallops and oysters. VIMS's broader 18-year effort to restore eelgrass to the seaside bays has been funded by grants from numerous agencies, notably the Coastal Programs of the Virginia Department of Environmental Quality (administered by NOAA's Office of Ocean and Coastal Resource Management), the Virginia Recreational Fishing License Fund, the American Recovery and Reinvestment Act, The Nature Conservancy, the U.S. Army Corps of Engineers, the Virginia Department of Transportation, complemented by private grants. Maps of the seagrass beds in these coastal bays can be viewed on the VIMS SAV interactive map, while information on real-time water quality data such as water temperature, oxygen, turbidity, and salinity can be viewed via the website of the Virginia Estuarine and Coastal Observing System (VECOS).

Recently, a link was established with the European initiative NOVAGRASS (http://www.novagrass.dk), a collaborative international effort funded by the Danish government to refine and scale-up methods for seagrass restoration along the shallow costal bays of Denmark's Jutland peninsula.

5.5.3. Multifaceted coral restoration initiatives through the Reef Resilience Network

http://www.reefresilience.org

The Reef Resilience Network is networking formation for the facilitation of management towards conservation and restoration of coral reefs, led by The Nature Conservancy (TNC, https://www.nature.org), a global-scale non-profit organisation. The Network includes a Coral Reef Restoration Module that compiles the latest scientific guidance and tools to help managers, researchers and practitioners ensure the maximum success of a coral reef restoration project and the most efficient use of limited resources. Through a partnership with experts from the Coral Restoration Consortium (CRC) this component covers the following coral restoration topics: (a) key considerations to be made before starting a restoration program; (b) methods for propagating branching corals and massive corals; (c) using artificial structures in restoration; (d) Promoting ecological processes that enhance coral populations; and (e) guidance for enhancing and sustaining a restoration program.

Case studies for the Reef Resilience coral restoration component includes coral restoration initiatives in the tropical Atlantic (Florida and the Bahamas) and Indian Ocean (Fiji and Seychelles). In the Bahamas, TNC has founded the Atlantis Blue Project Foundation (http://blueprojectatlantis.org) since 2007 to promote coral conservation and restoration. To date, more than 4,000 *Acropora* fragments nurseries have been established in Southwest New Providence, Paradise Island, and Andros Island. In Florida, TNC launched a $3.3 million, 3-year coral restoration project funded from NOAA. The project is a regional effort designed to aid the recovery of populations of *Acropora* corals and to provide social and economic benefits for local communities in addition to long-term ecological habitat improvements. In Fiji Islands, over 14,000 corals consisting of more than 25 species have been propagated and transplanted back to the reef in village MPAs since 2006 and village youth have received basic training in cost-effective coral propagation techniques, reef ecology and fauna, and integrating this work into guiding snorkelling tours. In the Seychelles, the first-ever large scale active reef restoration project in the region by growing small pieces of healthy coral in underwater nurseries prior to transplanting them to degraded sites that have been affected by coral bleaching; between 2011 and 2014, a total of 24,431 nursery-grown coral colonies were transplanted to 5,225 m^2 of degraded reef.

Moreover, the Reef Resilience Networks incorporates invasive species removal projects in Hawaii, Honduras, and Bonaire. For these case studies, TNC incorporates community involvement and technical means to remove or control the populations of allochthonous or native species, such as invasive algae, sea-urchins, and the lionfish. The consortium has published several best-practice and management guides on tropical coral restoration (e.g. Edwards & Gomez, 2007; Johnson et al., 2011) and relevant video tutorials, as well as hosted webinars and in-person and online training courses aiming at conservation managers and practitioners.

5.5.4. Creating new intertidal habitats: Steart Marshes, UK

http://www.wwt.org.uk/wetland-centres/steart-marshes/

The Steart Marshes initiative regards the assisted formation of new saltmarsh habitats through mechanical engineering of the coastline. These newly created saltmarshes go some way to replacing those lost to the sea. It is a joint effort of the Wildfowl & Wetlands Trust (WWT), a conservation non-governmental organisation focusing on wetlands, and UK's Environment Agency, with a budget of 21 million GBP. Its design and pre-implementation phase started in 2009, while actual land transformation works lasted from 2011 to 2013, creating 300 hectares of tidal marshlands and making Steart Marshes the biggest new coastal wetland in Britain. The tidal area is nearly 3km long and over 1km wide and its formation is expected to protect homes and businesses from flooding due to climate change and rising sea levels. The marshes lie between the mouth of the River Parrett and the Bristol Channel on the Somerset coast. This position attracts migrating birds, as well as a diverse wetland fauna including otters, egrets, owls, waders and wildfowl. The new habitat is owned by the Environment Agency and is managed by the WWT and was opened to the public in 2014.

5.5.5. Mangrove enhancement and replacement in Port Everglades, USA

http://www.porteverglades.net

Port Everglades (Boward County, Florida) is a major urban industrial seaport struggling to accommodate today's growing number of larger cargo ships that are bringing goods to South

Florida's growing population of 6 million consumers and 110 million visitors state-wide. Through a creative green initiative, the berthing capacity shortfall is addressed by developing new wetlands that support wildlife and ecological quality within the seaport area. The project cost $15.8 million and was managed by the port administration while implementation was assigned to private companies. In 2016, the initiative had concluded cultivation of 16.5 acres of nursery-grown mangrove and native plants on property that was originally dry land intended for other uses. This action allowed releasing 8.7 acres of an existing mangrove conservation easement adjacent to the harbour docks, effectively doubling the amount of mangrove conservation area in a more environmentally advantageous location within port property. The released acres will be excavated and the Southport Turning Notch will be expanded to make way for up to five new cargo ship berths. The new Upland Mangrove Enhancement area is adjacent to an existing 40+ acre conservation easement and contains approximately 70,000 Florida-native, nursery-grown mangrove and wetland plants along with transition buffer plants. Surplus mangrove seedlings that were not used for maintenance plantings were planted within nearby restoration areas with the help of more than 40 volunteers. The enhancement restores and creates mangrove wetlands, which gives rise to habitat for aquatic species of fish and invertebrates, as well as nesting habitat for birds. The project recently won IHS Maritime and Trade magazine's Dredging and Port Construction Innovation Award in the "Working/Engineering/Building with Nature Award" category.

5.5.6. Diverse habitat restoration through community involvement in Tampa Bay, USA

http://www.tbep.org/index.html

Spanning 400 square miles, with a drainage area nearly six times as large, Tampa Bay is Florida's largest open-water estuary, harbouring a rich and diverse assemblage of plants and animals, along with a rapidly growing human population that has made the region the second largest metropolitan area in the state. It is also an area comprising a diverse array of human activities: three major commercial harbours, as well as a thriving tourism industry attracting 5 million visitors per year. Tampa Bay was designated an "estuary of national significance" by US Congress in 1990, paving the way for development of a long-term blueprint for bay restoration through the Tampa Bay Estuary Program (TBEP) through a partnership of local authorities and the US Environmental Protection Agency (US E.P.A.). The restoration scheme is operated by the TBPE management with a broad involvement of local community and stakeholders, as well as private partners. Through annual actions dating since 1996, TBEP managed to restore and enhance existing key mangrove/salt marsh and seagrass habitats to pre-1950 levels. TBEP reports habitat restoration data of areas protected or restored on an annual basis in the Tampa Bay watershed to US E.P.A. per the requirement of the Government Performance and Results Act (GPRA). TBEP works with its partners to ensure that all restoration data is counted. From 2008 to 2011, a total of 78 distinct restoration actions had been successfully completed, collectively restoring approximately 4,000 acres of land, with a pursued goal of preserving and enhancing the whole extent of the bay's 18,800 acres of mangrove and salt marsh habitat.

5.5.7. Deep-sea coral restoration from oil spill in the Gulf of Mexico

https://restoreactscienceprogram.noaa.gov/funded-projects/deepwater-corals

An extended area in the Gulf of Mexico was damaged by the 2010 Deepwater Horizon accident which resulted in the largest marine oil spill in history. The spill had a catastrophic impact on the Gulf's vast, interconnected ecosystems. It harmed natural resources as diverse as fish and shellfish, productive wetland habitats, sand beaches, birds, endangered and threatened sea turtles, protected marine mammals – as well as deep-water coral communities (Demopoulos et al., 2016). The US NOAA funded a $1.3 million research project through the RESTORE Act Science Program to address crucial gaps in the understanding of the processes that shape population connectivity patterns in habitat-forming deep-water corals living between 150 and 7,500 feet deep in the Gulf of Mexico, including species directly impacted by the Deepwater Horizon oil spill. Through three on-site expeditions with remotely operated vehicles (ROVs), the research team will seek to determine the diversity and genetic structure of key coral populations. They will also identify the direction and rate of genetic exchange among coral populations to establish which ones are the source of the most successful larvae that may form new colonies. Using state-of-the-art population genomic approaches (Restriction-site Associated DNA Sequencing – RADseq) and predictive models of larval dispersal the research group will be able to estimate how far particular coral "families" spread and determine connectivity patterns among coral populations.

Volunteering for seagrass restoration – VIMS, USA

(image source: http://www.vims.edu/newsandevents/topstories/seagrass_denmark.php)

Coral restoration through the Reef Resilience Network

(image source: http://www.reefresilience.org/restoration/)

Creating new intertidal habitats: Steart Marshes, UK

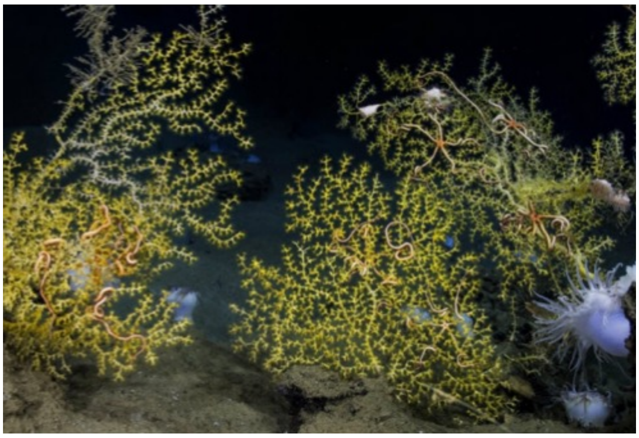
(image source: http://prolandscapermagazine.com/chew-valley-contribute-to-uk's-largest-wetland-project/)

Mangrove enhancement in Port Everglades, USA

(image source: http://www.porteverglades.net/articles/post/port-everglades-upland-mangrove-enhancement-trends-for-success/)

Diverse habitat restoration through community involvement in Tampa Bay, USA

(image source: http://www.tbep.org/about_the_tampa_bay_estuary_program-state_of_the_bay.html)



Deep-sea coral restoration from oil spill in the Gulf of Mexico

(image source: https://www1.lehigh.edu/news/unlocking-mysteries-deepwater-corals-restore-gulf-mexico)

Figure 41. Representative images from iconic world restoration projects.

6. The Costs and Benefits in Restoration

6.1. An overview of the methodology and metrics on the economic effects of restoration

The restoration of degraded ecosystems has implicit costs related to restoration actions, but the restoration action may provide us with a wide variety of benefits that are not always obvious, but generally difficult to monetize. An early step in any restoration issue may concern the balance of costs and benefits and whether it is worth undertaking a restoration programme and to what extent. In terms of economic assessment, the full range of ecosystem service benefits have to be understood and a system to cost the benefits is required, in order to estimate the net economic societal benefit. Pendleton (2010) has provided a major methodological outline towards the process-understanding behind the economic effects of marine habitat restoration, which we have summarised in Figure 42 and we examine further in the following short sections.

6.1.1. Costing Restoration

Though economic aspects of restoration costs is a recent addition to ecological restoration (de Groot et al., 2013; Blignaut et al., 2014a; Bayraktarov et al., 2016; Iftekhar et al., 2017), the costs of an active project could be arrived at in a straight-forward economic analysis if we considered the whole spectra of costs involved in the restoration process. This may, for example, involve assessing the cost of making structural changes, collecting local fauna as genitors/donors, nursery growing for seeds/larvae/clones/juveniles, transplanting/seeding, land acquisition, labour or monitoring costs. In the marine environment the cost of underwater work may increase with depth, as costs of accessibility and required technologies become a larger issue. The longer lived species, such as corals and sponges, may also increase restoration costs as they take longer to grow in a nursery and they also require longer term monitoring after restoration actions. Simple solutions, for example regulation of an area for reducing degrading impacts, may also have costs from desktop studies concerning the issue, production, implementation and enforcement of regulation. At a higher level if this involves the implementation of a MPA there may be considerable costs in setting up, managing, enforcing and monitoring the MPA.

Iftekhar et al. (2017) describe 4 main categories in restoration costs, namely acquisition (e.g. for acquiring property rights for the area to be restored), establishment (e.g. site preparation, planting), maintenance (e.g. administration, monitoring) and transaction (e.g. searching for suitable sites, organizing programs). The economic tools used in valuing restoration costs differ with regard to type of costs, the easiest being establishment and maintenance costs, for which market prices are usually available and used. For acquisition costs, capitalized gross revenue or gross margin of the productive use of land are used, or methods based on property prices, depending on the nature of acquired property. Transaction costs can be estimated by conducting surveys among the participating landholders or agencies and reviewing documents (Iftekhar et al., 2017 and references therein).

6.1.2. Restoration Benefits – Ecosystem Goods and Services

Benefits coming from basic changes in biodiversity, processes and functions result in changes to goods and services provided by the ecosystem, where ecosystem services are defined as the direct and indirect contributions of ecosystems to human well-being (de Groot et al., 2010). Goods and services have been categorised by many authors (e.g. de Groot, 1992; Costanza et al., 1997; Beaumont et al., 2007), with one of the definitive works given by Böhnke-Henrichs et al. (2013), where they are categorised into 4 typologies (Figure 42) including; Provisioning Services (e.g. seafood, genetic resources, medicinal resources, etc.), Regulating Services (e.g. air purification, climate regulation, coastal erosion prevention, etc.), Habitat Services (e.g. life-cycle maintenance, gene pool protection), and Cultural and Amenity Services (e.g. recreation and leisure, inspiration, cultural heritage and identify, etc.).

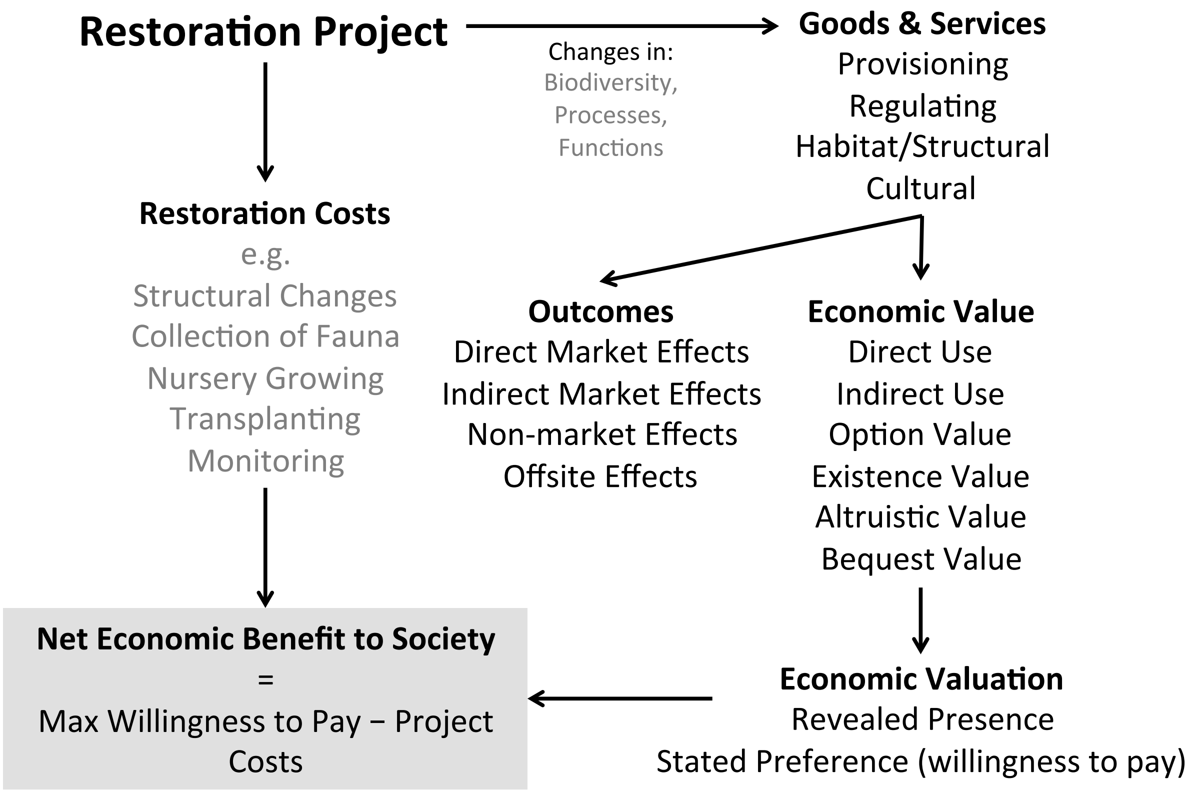


Figure 42. Cost-Benefit Factors in Restoration (based on Pendleton, 2007)

The benefits or change in benefits provided by an ecosystem will have some societal value and habitat restoration can create economic value if it produces new ecosystem services, can increase the value of existing goods and services, or increase the value of other economic activities that depend on ecosystem conditions (Pendleton, 2010). Ecosystem service valuation is the process of assessing the values of these benefits and gives the opportunity to also consider ecosystem benefits and costs that might be overlooked in management and planning within a marketplace framework alone (Börger et al., 2014). The challenge in measuring benefits is in being able to identify the whole range of ecosystem services and the benefits sourced to the societies, and select appropriate parameters reflecting ecosystem services that can translate into monetary values (Adame et al., 2014; Hanley et al., 2015; Hattam et al., 2015).

The outcome of a restoration project may have a number of different effects (Figure 42), which Pendleton (2010) lists as direct market effects (e.g. willingness to pay to visit a restored area), indirect market effects (e.g. restoration provides an increase in fish nursery grounds that lead to increased commercial catches), non-market effects (e.g. increase in cultural benefits such as recreational values or inspirational activities), and offsite effects (e.g. water quality is increased downstream, leading to increased amenity activities).

Whilst Pendleton (2010) categorised the basic components of economic value associated with an ecosystem to be restored simply into Use Value and Non-use (passive) values, this may be more complex with a fuller typology given by Pascual et al. (2010) shown with explanations in Table 6 and Figure 43. Use values include direct use (e.g. commercial fish sales), indirect use (e.g. regulation services – oxygen produced or carbon sequestered by seagrass meadows), and option values (the price given for the future availability of an ecosystem service). Non-use values include bequest value (value attached to the fact that future generations will benefit), altruist value (value attached to the fact that in future other people will benefit) and existence value (value deriving from the knowledge that something continues to exist).

Table 6. Typology of values (Pascual et al., 2010)

Value type	Value sub-type	Meaning
Use values	Direct use value	Results from direct human use of biodiversity (consumptive or non-consumptive)
	Indirect use value	Derived from the regulation services provided by species and ecosystems
	Option value	Relates to the importance that people give to the future availability of ecosystem services for personal benefit (option value in a strict sense)
Non-use values	Bequest value	Value attached by individuals to the fact that future generations will also have access to the benefits from species and ecosystems (intergenerational equity concerns)
	Altruist value	Value attached by individuals to the fact that other people of the present generation have access to the benefits provided by species and ecosystems (intragenerational equity concerns)
	Existence value	Value related to the satisfaction that individuals derive from the mere knowledge that species and ecosystems continue to exist

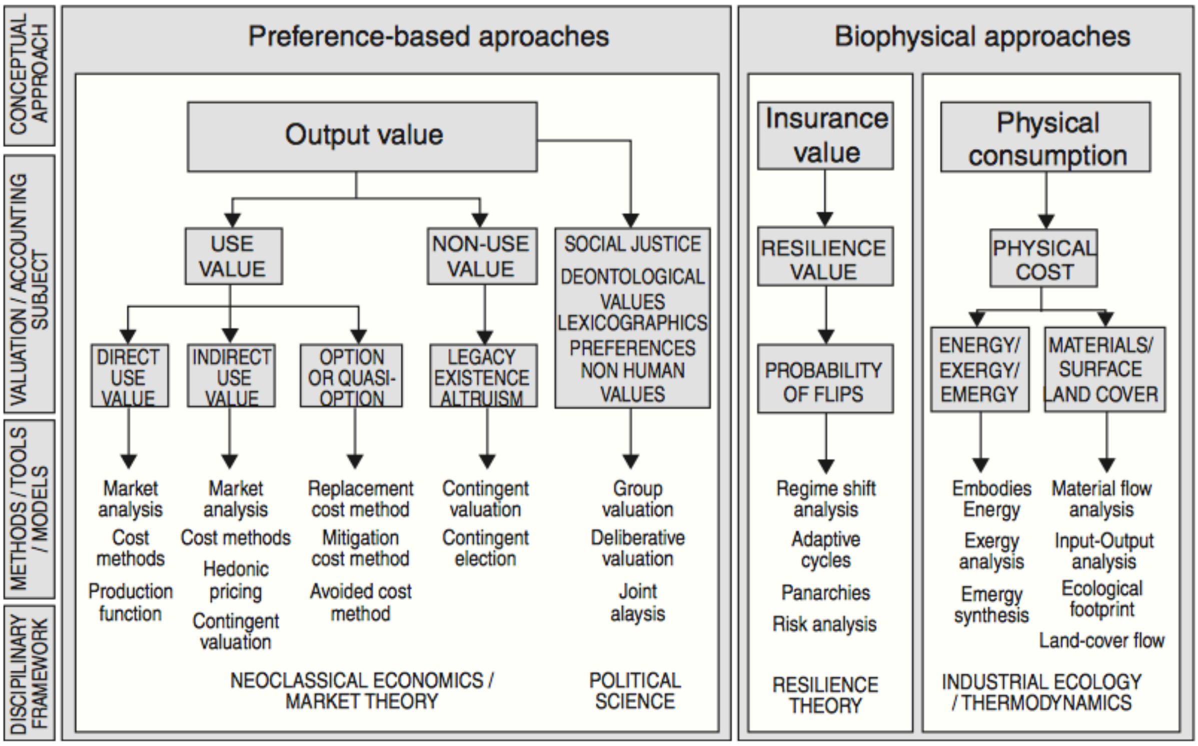


Figure 43. Approaches for the estimation of nature's values (from Pascual et al. (2010), based on Gómez-Baggethun & de Groot (2010)).

Pendleton (2010) notes that to show restoration has created value, it is necessary to isolate the effects of restoration from other factors, show that restoration does change value, and show that the ecological and environmental outcomes of restoration are the cause of these changes in values.

Economic valuation varies in terms of techniques used, goods and services assessed and assumptions made (Brander et al., 2007). The methodology selected for the assessment depends on the type of value expected to be generated by the project, which in most cases is a non-market value (Iftekhar et al., 2017). The measurement of these non-market values is generally covered by two methodologies: the revealed preference method, which is applied for measuring use values (e.g. recreation) and the stated preference method applied in cases of non-use values (e.g. preservation of threatened species for future generations) (Iftekhar et al., 2017). In his seminal paper on measuring the economic effects of marine habitat restoration, Pendleton (2010) explains that revealed preference methods are price-based and estimate the potential economic value of restoration by how values differ between sites with or without restoration, while stated preference methods involve people stating values, rather than inferring values from actual choices.

The stated preference method has been criticised, its major weakness being its hypothetical nature (Northern Economics, 2009). Moreover, most values are based on passive uses and the quality of data is inferior to observing revealed preferences (FAO, 2000). However stated preference is the approach mostly used in marine studies (Hanley et al., 2015), and for marine restoration in particular, it is perhaps the only way to assign monetary values to largely intangible non-use values. The choice modelling method (CM) and contingent valuation method (CV) are the most common ones within this category. CV relies on people's preferences and their willingness to pay for improved ecological conditions or avoided loss in the value of goods or services predicted to result from restoration, and is normally carried out through questionnaires or interviews. In choice experiments, respondents are requested to choose between specific options, each of which describes a specific set of attributes of the project and the amount of money one would have to pay to achieve that option. The choices made by the respondents are subsequently used to assess an individual's willingness to pay and to estimate the value of the non-market goods/service. Results based on willingness to pay not only depend on the subjects' preference to pay but also their ability to pay. Having an analysis based on how people respond to hypothetical questions means that great care must be taken on the specific survey context, what the questions concern, how they are phrased and the selection/representation of the people asked. In some surveys, respondents may also be placed in unfamiliar situations in which complete information may not be available (Northern Economics, 2009).

Rather recently, when time and/or funding is limited and does not allow data collection, the benefit transfer approach is used. In this case, the results from existing valuation studies based on revealed or stated preference methods are spatially and/or temporally transferred to a new area. The use of this methodology depends greatly on the availability of similar valuation data and whether stakeholders require accurate or approximate valuation data for their site (Holland et al., 2010), and fits better in situations where the projected goods and benefits can be measured in fairly homogeneous, divisible units (Ready and Navrud, 2005). The TEEB team used this approach as the basis to estimate potential benefits of restored ecosystems for biomes for which benefit values, based on solid data, were not available (TEEB, 2009). These estimations were based on a rather large available data set, i.e. 104 studies with 507 values from 22 different ecosystem services for 9 major biomes. Nevertheless, the TEEB team explicitly states that careful site-specific analysis of costs and benefits is needed before investment decisions are taken, and thus such an approach should only be seen as indicative of the scope of potential benefits. This scepticism with respect to the use of the benefit transfer method is based on the fact that the benefit values derived from literature cannot be imported elsewhere without making various context specific adjustments to reflect the local realities such as population numbers, income, and benefits derived from an ecosystem vis-à-vis that of the system from which the data is imported (Blignaut, personal communication). Blignaut makes this remark based on, among others, the fact that ecosystem services do not have a value, in and by themselves, outside of human use and such human interaction with ecosystems is very much context specific. Extreme caution should therefore be taken when using "global" figures, or values derived within a different context (Blignaut, personal communication).

6.2. Literature Review on the Economic Cost and Benefits of Restoration

6.2.1. Introduction/Scope

Marine restoration targets and aims at the recovery of biodiversity and ecosystem functions, thus it needs to be based on ecological knowledge. Yet finances and social aspects are among the decision parameters that drive restoration goals, addressing important issues related to the potential for social benefits or the likely imposed costs to the communities/stakeholders, therefore determining whether restoration projects are realistic (Miller & Hobbs, 2007). Funds for ecological restoration are usually limited (Adame et al., 2014) and therefore need to be carefully allocated by setting priorities for restoration areas, scale, habitats, services and benefits (both ecological and economic) that need to be achieved. This is of particular importance when it comes to the restoration of marine ecosystems, as it is more expensive than for any other ecosystem (Bayraktarov et al., 2016). Despite costs and gains being in the core of ecological restoration prioritization and decision making, these types of information are still scarce in relevant marine studies and provided in a rather incomplete form (Bayraktarov et al., 2016).

Our review on economic costs and benefits of marine restoration aims at summarizing and listing all the available but sparsely found cost- and benefit- related information in relation to a range of coastal and marine habitat types, areas, restoration techniques, target species, outcome, etc. This will help (a) elucidate the financial setting of marine ecological restoration, (b) identify gaps and economic constraints for restoration success, and (c) support decision making and restoration planning. Basic statistics on the type of available sources and cost-related information are illustrated as a first step to identify existing gaps in cost and benefit data reporting in marine restoration studies.

6.2.2. Approach

A catalogue on the available information on the economic cost and benefits of marine and coastal ecosystem service restoration was compiled based on four main sources:

(1) the peer-review publications synthesis on the knowledge on marine habitat active restoration methods, technologies and tools (details presented in Section 3.1). Among the peer-reviewed papers considered in the active restoration review chapter (see section 3), 83 included some kind of information on restoration costs, of which 73 considered also aspects of restoration benefits

(2) a Google Scholar search of published articles covering the most recent period 2016-2017 that was not considered in the previous source. The combination of keywords used included “marine”, “restoration”, “cost” or “benefit” and the first 100 search results were reviewed for content relevance

(3) the mini search for grey literature sources (project reports and online sources) carried out for the six European key habitats studied within MERCES WP1 (see following Section 6.3)

(4) grey literature sources based on expert knowledge.

The catalogue on the economic cost and benefits of marine restoration is a simple Excel workbook with a single row per observation and a series of columns corresponding to the desired information. In most cases, an observation corresponds to a unique source entry (article, review, report, thesis); nevertheless, there were studies presenting costs for more than one restoration techniques, or habitats, or areas, in which case they were incorporated in the catalogue with multiple entries.

The catalogue consisted of all those columns/information described in details for the peer-reviewed publications synthesis on the knowledge on marine habitat active restoration methods, technologies and tools (Section 3.1). Because studies did not provide cost data or estimations of restoration costs in a comprehensive manner, we added also two extra, descriptive columns for the overall categorization of information:

- Type of cost estimation: classified as (a) Monetary, when costs were specified in numbers (independent of the currency used); (b) Qualitative/Comparative, when reference to costs was in a descriptive way (e.g., inexpensive, cost and labour efficient, etc) or in relation to other techniques/tools/approaches (e.g., “by rough estimation, the shell method reduced the cost for the eelgrass transplantation by 50–70% compared to the traditional staple method"); (c) Opinion, in cases where the authors expressed their personal view on the expected cost of a restoration technique/tool/method (e.g., "Transplantation did not appear a cost-effective option to aid reef rehabilitation, there being significant costs but no clear benefits over a 5−10 year time scale")

- Type of restoration cost: classified as either (a) Total, in cases where the cited restoration costs concern all categories of costs involved in a restoration project, including capital and operating costs, (b) Partial, when a subset of the total costs is given or the description/opinion concerns a subset of all the probable costs (usually refers to costs of specific techniques/tools).

An extra, generic column on restoration benefits was also inserted in the catalogue in order to assign the relevant information derived from the reported literature sources in four main categories: (a) Ecological benefits, for information related to ecological aspects, such as biodiversity, stocks, habitat quality, ecosystem functioning/services, etc.; (b) Economic benefits, when the source specifically includes information on economic benefits through the use of valuation techniques, (c) Methodological benefits, for those cases discussing methodological issues and the advantages of the studied approach as compared to other approaches/techniques (either investigated within the same study or in other investigations), (d) Ecological & Economic benefits, when both ecological and economic benefits are mentioned in the study, although in most of these cases the economic part of the information is without any value, and (e) a No Benefit category, when no apparent benefits result from a specific restoration approach/technique.

The catalogue analysis focused exclusively on the costs and benefits aspects of the reported sources with a view to highlight their type of information and their range of distribution over different types of sources, habitat types and categories, target of restoration, restoration techniques, and restoration outcome.

6.2.3. Results

6.2.3.1. Marine restoration costs

Overall, the cost- and benefit-related data catalogue consists of 118 entries sourced from a total of 103 individual documents. Despite the increasing concern and interest on marine restoration activities and the multitude of peer-reviewed articles addressing aspects of restoration in the marine realm (for details see Section 3.1), less than 20% (98 articles) provided cost-related data information. Cost data were further derived from 4 reports and 1 master thesis.

Among the 118 entries of our cost and benefit data catalogue, 72% (85 observations) regard monetary estimations (Figure 44), even if these may only refer to a part of restoration costs (Table 7, 52 observations). The rest concerns comparative cost estimations, with 14% (17 observations) being cost estimations in relation to other restoration techniques, types or approaches, and 14% (16 observations) presenting an opinion on possible costs (with indications such as low, inexpensive, etc.).

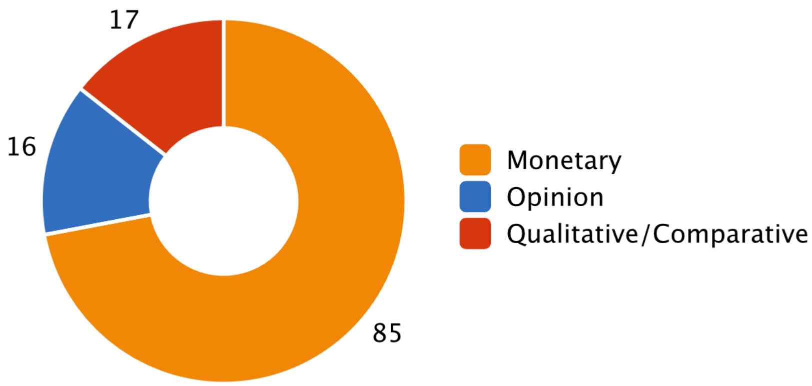


Figure 44. Number of catalogue entries by type of cost estimation.

Table 7. Number of entries with restoration costs by type of cost estimation and restoration category costs.

Type of cost	Partial	Total	Unknown	Total
Monetary	52	31	2	**85**
Opinion	15	1		**16**
Qualitative/Comparative	13	4		**15**
Total	80	36	2	**118**

A high number of restoration cost data (94%) are published in peer-reviewed journals, with 104 observations derived from 93 articles and 7 from 5 review articles, and only a small number of 7 observations were sourced from grey literature (6 from reports and 1 from master thesis) (Table 8).

Table 8. Number of entries with restoration costs by type of cost estimation and source type.

Type of source	Monetary	Qualitative/Comparative	Opinion	Total
Article	74	15	13	**104**
Review articles	4		3	**7**
Master thesis	1			**1**
Report	4			**4**

Type of source	Monetary	Qualitative/Comparative	Opinion	Total
Review reports	2			**2**
Total	85	15	16	**118**

Most information on restoration activities costs refer to rocky and soft-bottom habitats (Figure 45). Among them, 41 (35%) and 9 (8%) observations relate to rocky subtidal and intertidal respectively, of which the most concern restoration aspects of subtidal coral reefs (Table 9, 33 observations) spanning from the Tropical Northwestern Atlantic to the Indian Ocean; a quarter of the observations (29) corresponded to soft-bottom intertidal, primarily mangroves, seagrasses and saltmarshes (15, 5 and 4 observations respectively) from Cold Temperate areas of the Atlantic and the Pacific to Tropical Atlantic areas, Australia and the Mediterranean; and 16% (19 observations) to restoration activities in soft-bottom subtidal habitats, mainly targeted to the restoration of seagrasses (Table 9, 16 observations). Saltmarshes are the restoration target in most of the studies providing cost data within the generic habitat category Estuarine/Wetlands (8 observations out of 17), which also include restoration costs for mangroves, seagrasses and oyster reefs (2 observations for each habitat type). Deep sea appears in the cost catalogue with two observations included in one paper (Van Dover et al., 2014), which provide costs for two hypothetical restoration scenarios: the Darwin Mounds located off the coast of Scotland, where bottom trawling has damaged mounds of stony coral; and Solwara 1 hydrothermal vent site located off the coast of Papua New Guinea, where commercial mineral extraction for recovering a copper-, gold-, and silver-rich seafloor massive sulfide deposit will remove some actively venting and inactive substrata and their associated organisms.

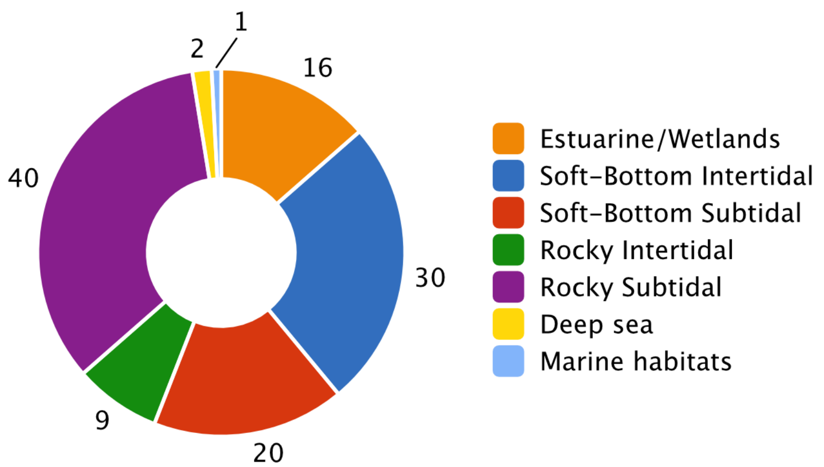


Figure 45. Number of restoration cost entries by habitat category.

Table 9. Number of entries with restoration costs by type of cost estimation and habitat category/type.

Habitat category/type	Monetary	Qualitative/Comparative	Opinion	Total
Coral reefs	28	6	4	38
Seagrasses	11	8	4	23
Mangroves	18			18
Saltmarshes	9	1	3	13
Oyster reefs	6		1	7
Macroalgal forests	2			2
Other	11	2	4	17
Total	85	17	16	118

Total costs of restoration activities (Table 7) are only available at 31% of the catalogue cases (36 observations), yet most of them are in monetary terms (Table 7, 31 observations) for soft-bottom intertidal mangroves, coral reefs, saltmarshes and seagrasses (11, 9, 3 and 2 observations respectively). Similarly, partial restoration costs are given in most studies in monetary values (Table 7, 52 observations) and also concern mangroves, coral reefs and seagrasses (6, 19 and 9 observations respectively) focusing mainly on costs of suggested transplantation/gardening techniques.

Reported costs concern primarily activities for restoring degraded marine environments (104 observations, 88%), mainly at rocky subtidal and soft-bottom habitats (37 and 40 observations respectively), whereas for single stressor restoration actions the number of cases with reported costs are rather low (14 observations, 12%), but they also focus at the same types of habitats (8 and 4 observations for soft-bottom habitats and rocky subtidal respectively). For both categories of restoration target, 72% of the reported cost information is provided in monetary terms (75 and 10 observations for degraded environment and single stressor respectively), while for the rest they are equally divided in the other two cost estimations types, i.e. qualitative/comparative and opinion (Figure 46).

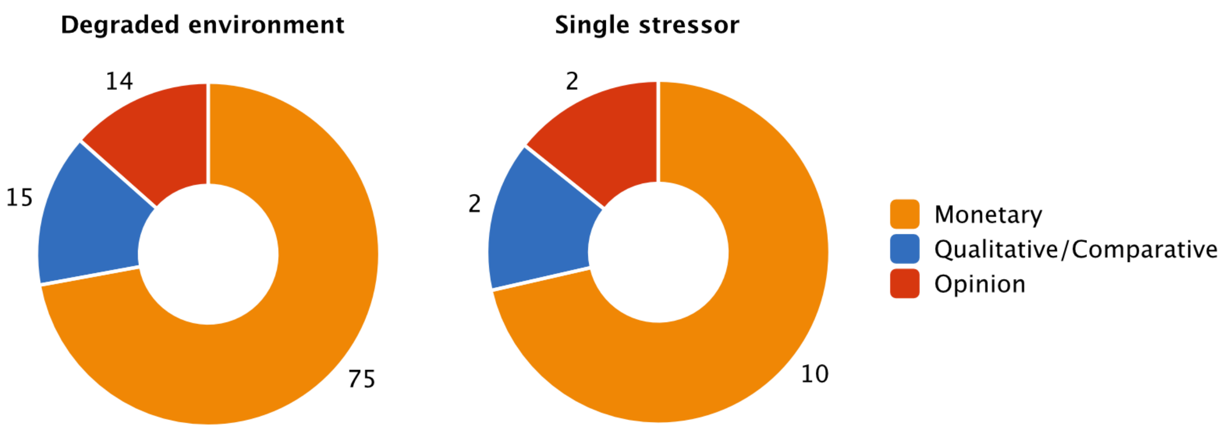


Figure 46. Number of catalogue entries by type of cost estimation for each of the two different restoration targets (Degraded environment, Single stressor) considered in the restoration review catalogues

Cost data are available for 41 observations regarding implementation of restoration projects (Table 10), of which 33 are provided in monetary terms. Almost half of the reported observations though (58) regard field or laboratory exercises and experiments that test suggestions of new techniques or tools, with 37 of these given in monetary values, while 10 of them are compared to other techniques or restoration approaches and 11 are simply estimations of costs by the researchers themselves (Table 10). For 16% of the observations (19 out of 118 entries) we could not extract such kind of information (Table 10).

Table 10. Number of entries with restoration costs by type of cost estimation and restoration activity.

	Monetary	Opinion	Qualitative/Comparative	Total
Project	33	5	3	**41**
Experimental	37	11	10	**58**
Other	15		4	**19**
Total	85	16	17	**118**

More than half (51%) of the restoration cost entries included in the catalogue concern primarily biological restoration techniques (60 observations; 40 monetary, 7 opinion, 13 comparative observations), mostly transplanting, either as a standalone technique (26 observations) or complemented by supportive activities such as the use of nurseries (Nurseries & Transplanting 11 observations), or aquaculture (Aquaculture & Transplanting 5 observations) (Table 11). Cost data were also estimated for restoration with physical means (24%, 29 observations), related mainly to habitat constructions or reformations, and excavations, while 17% of the entries (20 observations) provided cost estimations with regard to techniques using a combination of both biological and physical approaches, of which 12 observations concerned transplantation of organisms supported by some type of artificial structure (Table 11). For most of the described restoration techniques/approaches (68%, 80 observations) only partial cost estimations were provided relating mostly to the costs of the suggested/studied technique.

Table 11. Number of entries with restoration costs by type of cost estimation and restoration category/technique.

Restoration techniques	Monetary	Opinion	Qualitative/Comparative	Total
Biological	**40**	**7**	**13**	**60**
Planting	8	1		**9**
Transplanting	14	4	8	**26**
Nurseries	3	2	2	**7**
Nurseries & Transplanting	15		1	**16**
Biological stressor removal (clearing exotic vegetation)			1	**1**
Other			1	**1**
Physical	**24**	**4**	**1**	**29**
Physical/Hydrological	18	2	1	**21**
Physical/Hydrological, biological stressor removal (clearing exotic vegetation)	1	2		**3**
Artificial reefs/structures	5			**5**
Hydrological		**3**		**3**
Hydrological		3		**3**
Physical/Biological	**17**	**2**	**1**	**20**
Planting & Physical/Hydrological	7			**7**
Transplanting & Artificial reefs/structures	9	2	1	**12**
Nurseries & Artificial substrates	1			**1**
Passive	**1**			**1**
Policing vs Artificial reef structures	1			**1**
Other	**3**		**2**	**5**
Total	85	16	17	**118**

Among the observations of the restoration cost and benefit catalogue a rather high number (63%, 74 entries) concerns cost estimations of successful restoration activities/techniques, while in 23 of the entries there was no indication of the restoration outcome (Table 12).

Table 12. Number of entries with restoration costs by type of cost estimation and restoration outcome.

Restoration outcome	Monetary	Opinion	Qualitative/Comparative	Total
Success	50	12	12	**74**
Partial Success	7	2	1	**10**
Failure	9	2		**11**
NA	19		4	**23**
Total	*85*	*16*	*17*	***118***

6.2.4.1. Marine restoration benefits

Overall, 100 observations from 91 literature sources with restoration cost data also provided information on benefits resulted, or presumed to result, from restoration activities or proposed/studied restoration techniques. In most of these entries (82), the benefits related to ecological aspects, concerning either a direct increase/growth/augmentation of the investigated species/habitat, or a better approach for the protection of the studied habitat (e.g. shore protection). Among the 82 observations with ecological benefits, 37 also included an opinion on potential economic benefits (Figure 47), emphasizing mainly on the reduction of associated restoration costs through the use of the studied restoration approach, while some foresee an economic benefit through the increase of commercial species stocks, an upgrade in the area's aesthetics, with a subsequent increase in tourism activities, or simply by an increase in local income through the implementation of a restoration project (provision of new jobs, increase of visitors resulting in markets turnover increase, etc). Only three, recent studies (Caffey et al., 2014; Blignaut et al., 2016; Chang et al., 2017) provided exclusively economic benefit data as a result of marine restoration activities based on economic valuation methods. A few observations (13) were grouped under the category "Methodological", indicating benefits and advantages, in comparison to other methods, if their techniques/approaches were used (Figure 47), while two studies concluded that there were no benefits related to the studied restoration techniques (for coral reef growth and seagrass transplantation).

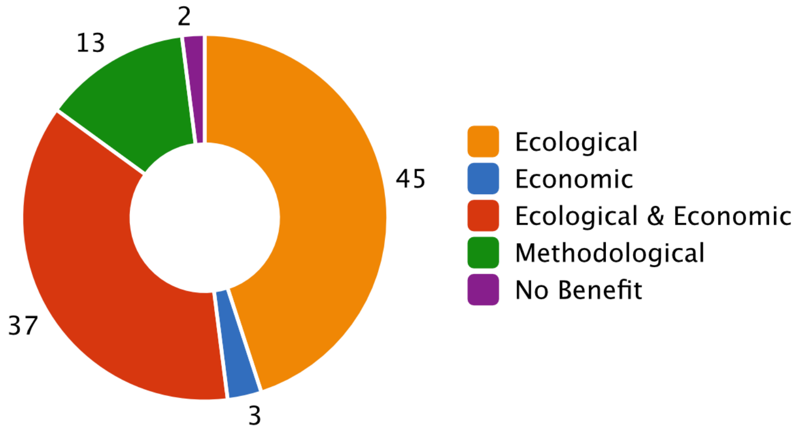


Figure 47. Number of catalogue entries by type of benefit information.

The information on the benefits from restoration studies with cost estimations was mainly sourced from peer-reviewed papers (96%), either in the form of reviews or research articles (4 and 92 observations respectively), with only 4 observations extracted from grey literature (reports and master thesis) (Figure 48).

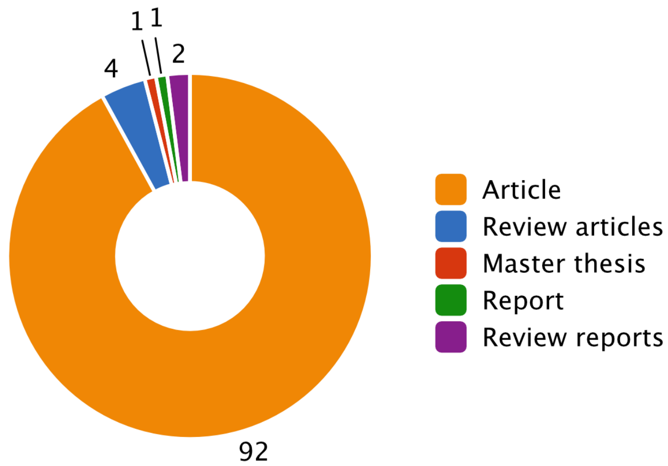


Figure 48. Number of catalogue entries with benefit information by source type.

Similar to the information on restoration activities costs, most associated benefits refer to the rocky subtidal (37%), mainly to coral reefs restoration, and soft-bottom intertidal and subtidal habitats (26% and 14% respectively), mainly mangroves and seagrasses respectively, for which either exclusively ecological or ecological and economic benefits are discussed (Table 13). The three studies on economic benefits from marine restoration concern two intertidal habitats (in Korea and Saudi Arabia) and a marsh wetland in the USA, whereas most methodological benefits are derived from studies on subtidal coral reefs. The two deep-sea observations including restoration benefits are also coming from the only deep-sea study of the restoration costs and benefits catalogue, i.e. Van Dover et al. (2014), in which, along with the two hypothetical restoration scenarios costs (more details in the previous section), ecological benefits are also discussed.

Table 13. Number of entries with restoration costs and benefits by type of benefit information and habitat category/type.

Habitat category/type	Ecological	Economic	Methodol ogical	Ecological & Economic	No Benefit	Total
Coral reefs	14	0	7	13	1	35
Seagrasses	8		3	4	1	16
Mangroves	7		2	8		17
Saltmarshes	7	1		3		11
Oyster reefs	3			3		6
Macroalgal forests	2					2
Other	4	2	1	6	0	13

The reported benefits of the specific catalogue were found in studies that targeted degraded marine habitats (Figure 49, 88 observations), primarily involving "Restoration" as a type of active restoration action (59 observations) mainly for coral reefs, seagrasses and mangroves (23, 10 and 9 observations respectively). As expected, the reported benefits were also either exclusively ecological (41 entries) or also included some economic aspects (33 entries), while 11 observations related to degraded environments studies had a methodological focus when reporting the expected benefits of their approach. Benefits from activities targeting single stressors were included in the catalogue with only 12 entries (Figure 49), relating mostly to habitat enhancement activities.

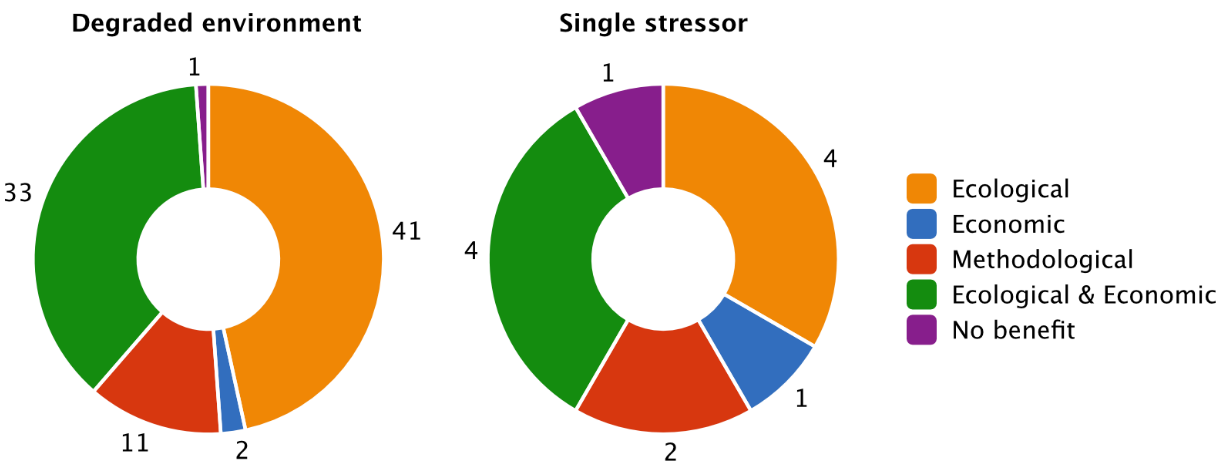


Figure 49. Number of catalogue entries by type of benefit information for each of the two different restoration targets (Degraded environment, Single stressor) considered in the restoration review catalogues.

Half of the restoration benefits observations from sources with restoration cost information concern field or laboratory exercises/experiments of restoration techniques and tools, whereas 34 are observations on benefits from the implementation of restoration projects (Figure 50). Once more, mainly ecological benefits are reported in both project and field exercise type of entries (16 and 27 entries respectively), followed by observations that include information on both ecological and economic benefits (12 and 14 entries respectively).

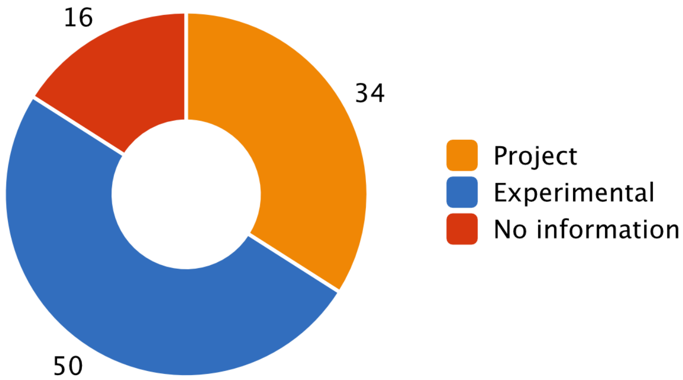


Figure 50. Number of entries with benefit information by type of restoration activity.

Restoration benefits concern primarily biological restoration techniques (53 observations), mostly as a result of transplanting, or nurseries and transplanting (20 and 11 observations respectively), with the benefit categories "Ecological" or "Ecological and Economic" prevailing (Table 14). Benefits were also discussed in literature sources for restoration with physical means (26 observations), and similar to the restoration costs results, they related to physical or hydrological habitat alteration, while a few studies (4 observations) dealt with benefits from using oyster reef constructions (Table 14). Restoration benefits were also discussed in a number of sources investigating marine restoration using a combination of both biological and physical approaches (in total 18 entries), among which 12 observations concerned transplantation of organisms supported by some type of artificial structure that are foreseen to have ecological (7 entries) or both ecological and economic (4 entries) benefits (Table 14).

Table 14. Number of entries by type of benefit information and restoration category/technique.

Restoration techniques	Ecological	Economic	Methodological	Ecological & Economic	No Benefit	Total
Biological	**23**		**8**	**20**	**2**	**53**
Planting	3			6		**9**
Transplanting	11		6	3		**20**
Nurseries	1		1	3	1	**6**
Nurseries & Transplanting	6		1	8	1	**16**

Restoration techniques	Ecological	Economic	Methodological	Ecological & Economic	No Benefit	Total
Biological stressor removal (clearing exotic vegetation)	1					**1**
Other	1					**1**
Physical	**10**	**1**	**3**	**12**		**26**
Physical/Hydrological	5	1	1	11		**18**
Physical/Hydrological, biological stressor removal (clearing exotic vegetation & urchin removal)	3					**3**
Artificial reefs/structures	2		2	1		**5**
Physical/Biological	**12**		**2**	**4**		**18**
Planting & Physical/Hydrological	4		1			**5**
Transplanting & Artificial reefs/structures	7		1	4		**12**
Nurseries & Artificial substrates	1					**1**
Other		**2**		**1**		**3**
Total	*45*	*3*	*13*	*37*	*2*	***100***

For 64 of the observations regarding information on restoration benefits the restoration approach or technique used was considered successful, with ecological (31 entries), ecological and economic (24 entries), methodological (7 entries) or economic (2 entries) benefits (Table 15).

Table 15. Number of entries by type of benefit information and restoration outcome.

Restoration outcome	Ecological	Economic	Methodological	Ecological & Economic	No Benefit	Total
Success	31	2	7	24		**64**
Partial Success	7		4	3	1	**15**
Failure					1	**1**
NA	7	1	2	10		**20**
Total	*45*	*3*	*13*	*37*	*2*	***100***

6.2.4. Discussion

Over the last decades, terrestrial and marine restoration has emerged as a global priority and an urgent task scientists and practitioners need to tackle in an effort to compensate for the extended habitat and biodiversity loss caused by human activities and coastal development (Halpern et al., 2008; Aronson & Alexander, 2013 a, b; Tobon et al., 2017; Possingham et al., 2015).

Nevertheless, aspirations and commitments would only appear as a paper exercise unless they are complemented by cost, benefit and feasibility estimations. Our synthesis on available sources with information on the costs and benefits of marine restoration reveals that though several attempts have already been made with respect to this field, considerable efforts need still to be placed on the economics of marine restoration.

6.2.4.1. How fragmented are the reported restoration costs? (Total vs Partial costs)

Bayraktarov et al. (2016) have pointed out that there is not a comprehensive way in restoration cost data reporting for five distinct marine habitat types (coral reefs, seagrass, mangroves, saltmarshes, and oyster reefs), which is also verified by our review. As Iftekhar et al. (2017) summarize in their study "restoration costs are rarely reported by ecological restoration studies, published cost data are often collected using different approaches, making them hard to compare (Bullock et al., 2011), and sometimes not all types of costs are considered during planning (Pastorok et al., 1997; de Groot et al., 2013)". Though ecosystem restoration costs are made up by (and can be broken down to) several components, such as planning, construction, labour, maintenance and monitoring costs, only in few cases the source entries in our catalogue included an overall assessment of aggregated costs (30%) and similar to the findings of Bullock et al. (2011) only a few studies include detailed information on different categories of costs. In fact, most literature sources in the catalogue provide cost information related to a suggested/studied technique. This is most likely related to the fact that, until now, most studies on marine restoration focus on research purposes, and specifically on the improvement of technical, technological or methodological aspects, and rarely concern the study and implementation of integrated approaches. Further to this, it seems quite possible that when all other factors of a restoration approach are similar for a specific degraded environment, cost estimations for the diversified aspect of the approach may appear sufficient, making the extra effort, as well as expenses, for an economic assessment perhaps perceived as redundant.

6.2.4.2. Monetary costs vs estimations of costs and consistency in cost data reporting

An additional issue that has been already noted by Bullock et al. (2011) and de Groot et al. (2013), which appeared in our review as well, is the lack of meaningful cost data and of a consistent manner when reporting economic aspects of restoration. In our synthesis catalogue we could only extract some kind of restoration cost information from approximately a hundred literature sources, despite the multitude of available studies on marine restoration (Section 3.1).

Nevertheless, in most of these sources monetary costs were presented, even if these concerned partial costs (e.g. monetary cost for a suggested restoration technique), revealing the current trend in understanding and considering the socio-economic effects of marine restoration. The substantial percentage of studies (almost 30%), though, restricted to a qualitative/comparative type of cost information or to forming an opinion on restoration costs, stresses the fact that there is still a lot of way to cover before full information is disclosed and, perhaps even more importantly, in a standardised format, which is rather a prerequisite for making robust environmental management decisions (Bayraktarov et al., 2017).

6.2.4.3. The aspect of restoration performance when reporting restoration costs

In her comprehensive review Suding (2011) states that comprehensive evaluations and available information on restoration outcomes are rare. This also applies to marine ecosystems as Bayraktarov et al. (2016) found that only 11% of the observations of their study provided combined data on cost, area and survival of organisms as a measure of feasibility. Verdura et al. (submitted) in their recent study on cost-effective methods for the restoration of marine algal forests state that biological traits are important for selecting appropriate restoration techniques and further suggest that costs should be taken into account only when the compared techniques are equally efficient. Our study demonstrated that there is a rather high number of sources with restoration cost information from which the restoration outcome can be inferred (95 entries), yet the rather unstructured and incomplete presentation of the cost and feasibility data point to the need of more comprehensive assessments in a standardised way.

6.2.4.4. The gaps and needs in restoration costs and benefits reporting

Our review on restoration literature sources revealed that most efforts on cost and benefits estimations concentrate on the coastal environment, where human activities but also marine restoration mainly take place, especially in the intertidal and shallow subtidal, living the biggest part of the marine environment, namely the open sea, the deep-sea and the pelagic habitats, largely unexplored. The sources of the catalogue also indicate that academics are still focusing on developing techniques for maximising the efficiency and outcome of restoration actions and thus costs and benefits are mostly associated with this type of restoration costs, lacking to a great extent information on other categories or overall costs. As a consequence of this, restoration costs and benefits are indicative for activities at small scale, primarily as a result of testing suggested biological techniques, such as organism growth in hatcheries or transplanting.

Apparently, as the focus in restoration studies need to shift in order to encompass more habitats, larger scales in both space and time, and integrated approaches combining various organisms and techniques, so does the focus of studies on the costs and benefits of marine restoration. A further demand in addressing the economic aspects of ecological restoration is the assessment of all categories and activities that impose economic costs, as well as the benefits arising from the recovery of ecosystem services, both of which may differ between habitats, locations, restoration techniques, targeted species/assemblage or even the societies involved. This will ideally be carried out in a standardised way that will further enable not only the assessment of the success, feasibility and associated benefits of specific approaches and restoration plans, but also the comparison of different methods, a substantial issue for decision making, prioritisation of areas and restoration planning.

An important issue that has been recently discussed in the field of ecological restoration is the emerging policy focus on ecosystem services with potential implications for the conservation of biodiversity (Bullock et al., 2011). This shift in restoration target gives the opportunity to realise that the analysis of ecosystem service restoration benefits is in its infancy and even more so is their economic valuation. Laurans et al., (2013) indicate six main challenges that apply not only to the ecosystem service valuation in the marine realm but to most environments: (1) inaccuracies associated with valuation data, (2) inadequate valuation data availability, (3) cost of valuation studies, (4) training of policy makers to apply ecosystem service valuation, (5) regulatory frameworks not conducive to ecosystem service valuation, and (6) the potential of ecosystem service valuation to hamper political strategies. Among them, the lack of adequate valuation data stands out as a major challenge for marine applications (Börger et al., 2014). Brander et al. (2007) conducted a meta-analysis on the recreational value of coral reefs, and though the valuation literature provided value estimates for almost all economic services provided by coral reefs, they found that valuation studies collected for the purposes of their meta-analysis lacked fundamental information, such as the characteristics of the coral reef studied (e.g. area, quality, location), and the specifics of the methods used (e.g. sample size, number of non-respondents). Furthermore, Börger et al. (2014), pointed out that ecosystem service valuation studies are extremely unevenly distributed across different types of marine habitats, ecosystem services and geographic locations with studies on value estimates of near-coast provisioning, regulating and cultural (especially recreational) services prevailing. In contrast, studies concerning the open ocean and the deep sea, or the less well recognised cultural services are minimal. According to the same investigators, valuation approaches need further standardisation and development within an ecosystem service valuation context. Having this in mind, they propose the methodological development of stated preference approaches for application to marine ecosystem services and for improvement of ecological content validity, based on innovative tools.

The currently limited efforts on ecosystem service restoration benefits are coupled with great uncertainties owing to the fact that ecosystem benefits are quite often unknown or imprecise (Bullock et al., 2011), and with regard to restoration they need to be calculated for long term periods and sometimes their focus should be on social preferences basis rather than on biological traits. Blignaut et al. (2013) consider that the concept of ecosystem services, as explicitly linking services to beneficiaries and demonstrating values, has not yet been mainstreamed in the science, public policy or practice of ecological restoration. Blignaut et al. (2014a) promote the investment in the restoration of the natural capital not only as a game changer in the path to sustainability but also because it is ecologically and economic beneficial. Despite increasing evidence that restoration 'pays' long-term (Tucker et al., 2013), progress in detailed cost reporting is still needed; this will help elucidate the cost-effectiveness of restoration, provide evidence to convince society that the benefits outweigh the investment costs, with the view to propose funding schemes for covering the expenses.

Another fundamental linkage that needs to be addressed is that of restoration costs with target goals under specific desired scenarios and scales. Because restoration goals are context dependent and differ among locales and societal needs, estimation of costs simply on the basis of habitat/species type or technique may lead to unrealistic hypothesis. Especially for large spatial scale restoration to match the degradation scales, both rehabilitation and restoration actions are needed (Aronson et al., 2017). Either way, it is likely that restoration will affect large areas of the world and millions of people over the coming decades (Reid & Aronson, 2017).

6.3. Recent Cost Benefit Work in the MERCES Key Habitats

6.3.1. Introduction/Scope

Information on the economic value, cost and benefits of ecological restoration was extracted from the peer-review publications examined for the global review on restoration actions in marine ecosystems (Section 6.2). Nevertheless, acknowledging that such data was not always available in the reviewed papers, we performed an additional mini search for grey literature sources (project reports and online sources) and papers which were not addressed in the global review, with a special focus on the six European key habitats which were examined in the framework of MERCES WP1 as individual case studies.

6.3.2. Methods

The literature search was conducted in Google search engine for cost/benefit information related to restoration activities for the six key habitats was performed using keyword combinations. Keywords included “restoration”, “Europe” and “economic value” or “cost” or “benefit” and the examined types of habitats, i.e. “Kelp”, “seagrass”, “*Cystoseira*”, “coralligenous”, “deep-sea corals”, and “deep sea sediment”, respectively. For all the above cases, the first 100 search results were reviewed and catalogued in a simple Excel workbook with a single row per entry (600 entries in total) and a series of columns (7) corresponding to the desired meta-data. The columns are described in more detail below:

- ID: the unique entry number for this record (filled by the catalogue administrators)
- Key habitat: the examined key habitat
- Title or Description of the source (free text field)
- Reference Link: free text field, providing a web link to the reference
- Source type: (a) On-line resource, (b) Paper, (c) Report, (d) Conference paper, (e) Book/Chapter
- Categorization according to the provision of information concerning restoration activities and relevant cost/benefit data: (a) Restoration activity with cost/benefit (C/B) data, (b) Review paper with C/B data, (c) Restoration activity without C/B data, (d) Review paper without C/B data, (e) MERCES, i.e. sources linked to MERCES project, (f) Management/Conservation, (g) Other habitat, (h) Other, i.e. other irrelevant sources, (i) Duplicates, i.e. duplicate sources.
- Comments: free text, further details about the source or findings of the paper/report, or any other useful information, e.g. habitat type in case of other habitats.

6.3.3. Results and Discussion

A total of 600 sources were reviewed for the examined key habitats (100 per habitat). Peer review papers, accounted for 14-28% of the reviewed sources while grey literature (mostly online sources and reports) accounted for 72-86% of the reviewed sources (Figure 51).

For most key habitats, the majority of reviewed sources concerned management and/or conservation initiatives while a considerable percentage of the examined sources were irrelevant to the key habitat concerned (Other habitat, e.g. coral reefs in the results for deep sea corals). The category "Other" includes sources with minimal information on the topic (e.g. news items, publications lists and researchers' CVs) or different uses of the habitats examined (e.g. kelp aquaculture). Interestingly, sources linked to MERCES project (e.g. newsletters) appeared in the search results for all key habitats (19 sources; 1-5 per habitat).

Only 5 sources included information regarding cost and/or benefits of restoration of the examined key habitats (Figure 52); specifically: one paper with hypothetical costs for deep-sea coral restoration (Van Dover et al., 2014) which appeared multiple times in the search results; one review paper about seagrass beds, kelp forests and other marine habitats (Narayan et al., 2016); one methodological paper (Marion and Orth, 2010) and one Swedish project report (Moksnes et al., 2016) about seagrass restoration; and one paper about *Cystoseira* gardening on coastal defence structures as an enhancement action (Firth et al., 2014). No data related to cost or benefits for restoration/enhancement activities were found for deep-sea sediment and coralligenous assemblages.

In addition, our review revealed 21 projects (e.g., BIOMARES, CORGARD, DRIVER, GIREL, MMMPA, RESTORE), which were incorporated to the restoration projects' catalogue, as well as sources for cost/benefit data on restoration of other types of habitats (e.g. coral reefs, mangroves and oyster reefs) in review papers (Grabowski et al., 2012; Narayan et al., 2016) and in papers which did not properly address the restoration issue (Firth et al., 2014) which were not addressed in the global review. Review papers tended to appear several times in the search results (i.e. Duplicates) even if they did not include cost/benefit data for the given key habitat.

Although this review was not thorough, the results are indicative of the restoration effort invested in the examined key habitats. Indeed, no restoration initiatives had so far focused on deep-sea habitats – except for the hypothetical case studies by Van Dover et al. (2014) – or coralligenous assemblages, except for MERCES project which already appears in the online search results. On the other hand, relatively more data were available for seagrass beds.

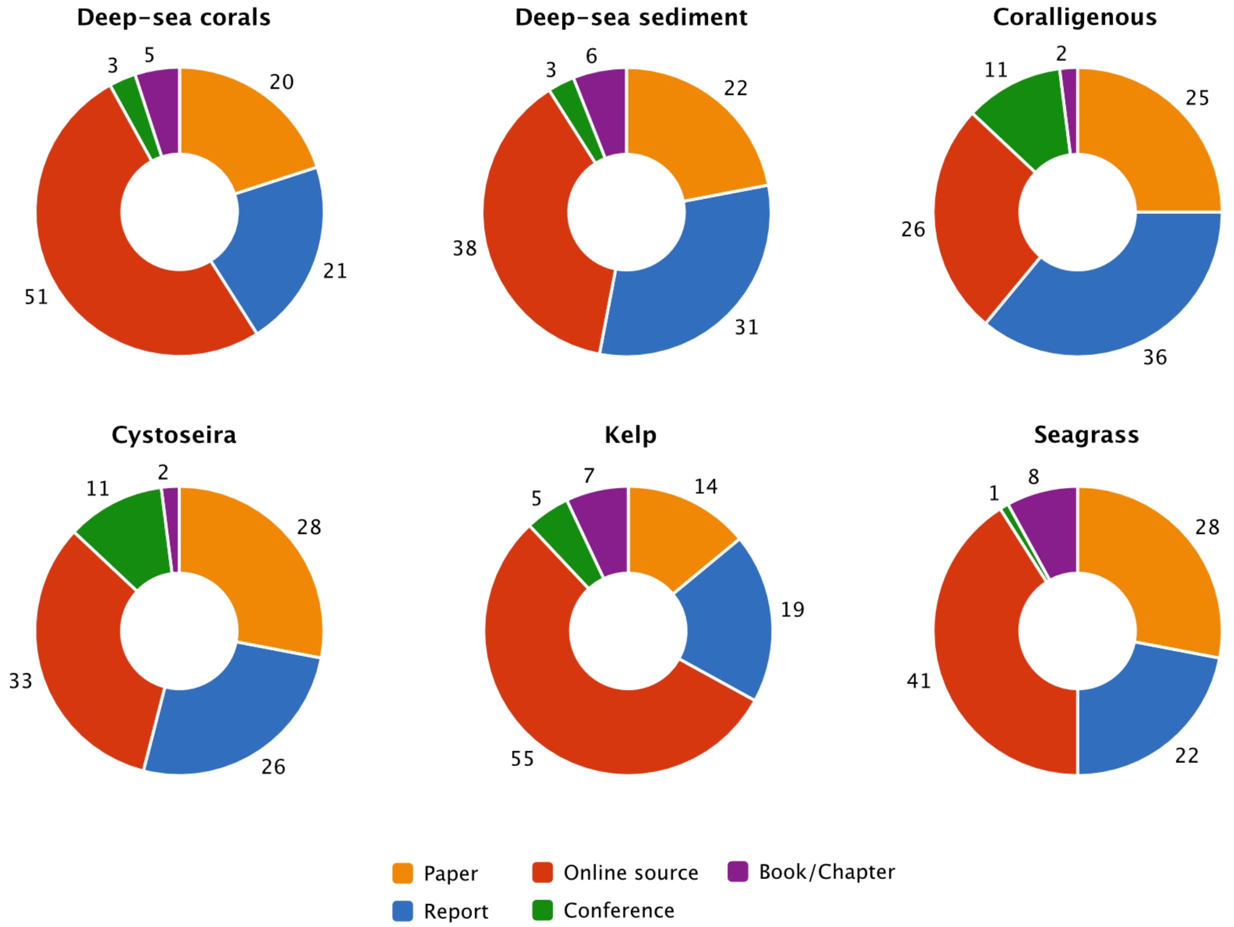


Figure. 51. Number of reviewed sources for the examined key habitats by source type.

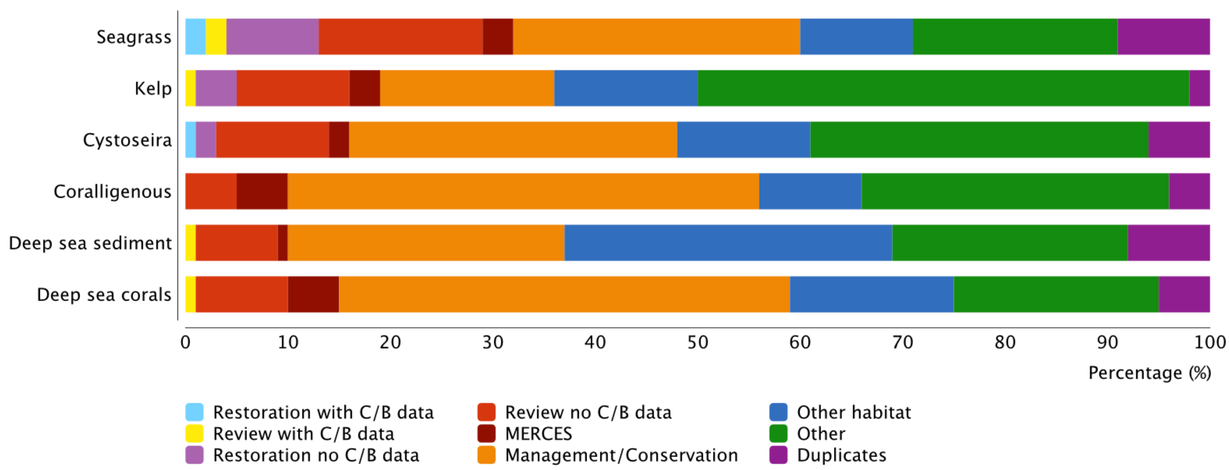


Figure. 52: Percentage of reviewed sources for the examined key habitats with respect to the available information regarding restoration activities and relevant cost/benefit (C/B) data.

7. Discussion

In addition to some discussion points with respect to individual preceding review sections, several issues concerning marine restoration were either not part of the targeted reviews, cross different boundaries or are important enough to warrant further development. In the following sections several such topics are developed including the place of artificial reefs in restoration, the removal of threats, restoring species as part of ecosystem restoration, environmental disaster affecting multiple ecosystems, restoring natural capital, nature-based solutions, technologies and innovation and the feasibility of restoration.

7.1. Artificial Reefs

Artificial reefs were introduced as a concept in the 18th century and since then several reefs have been deployed around the world (Seaman, 2002; Bortone et al., 2011). In Europe, most artificial reefs have been deployed in the Mediterranean Sea and the Bay of Biscay in the OSPAR Maritime Area, especially during the last 40 years, as a measure for the enhancement of fish stocks and the management of fisheries and to a lesser extent for the protection and enhancement of seabed habitats, mainly through the exclusion of destructive fishing practices (e.g. trawling) and other activities (e.g. dumping) (Fabi et al., 2011; Fabi & Spagnolo, 2011). Although artificial reefs are not part of the original ecosystem, they may be used as part of mitigation, enhancement and restoration initiatives (Dupont, 2008).

A broad variety of modules of various shapes and sizes, constructed by different materials, has been tested and used for the construction of artificial reefs; the most commonly followed practice involves the random or geometric (e.g. pyramids) deployment of concrete modules which bear holes (Fabi et al., 2011). Furthermore, various types of artificial structures, frames and devices have been developed to (a) enhance species recruitment by providing structurally complex substrates (e.g. Lam, 2003; Levy et al., 2010; Al-Horani & Khalaf, 2013) and (b) to facilitate transplantation of macroalgae (e.g. Deysher et al., 2002), seagrass (e.g. Park & Lee, 2007) and sessile invertebrates for restoring their populations (e.g. Van Treeck & Schuhmacher, 1997). Transplantation of keystone species on artificial reefs could also enhance regeneration by accelerating community succession (Fariñas-Franco et al., 2013). Hence, the deployment of biogenic substrates of marine origin (e.g. coral rubble) seeded with organisms that jump start succession (e.g. sponges) was found to facilitate natural recovery processes, thus improving coral reed restoration outcomes (Biggs, 2013). On the other hand, the use of other natural materials (e.g. wood or bamboo) for coral restoration purposes was found to generate low survival of transplants (Ferse, 2010). In addition to the deployment of artificial substrates for restoration purposes, ecological engineering approaches have recently investigated the enhancement of coastal defence structures (e.g. seawalls) by transplanting native habitat-forming species (e.g. gardening of canopy-forming algae) (Firth et al., 2014).

It is noteworthy that 14% of the European restoration projects which have taken place during the last decade (see Section 3 of this report) involve the deployment of artificial reefs/structures (e.g. CIRCE, RESTORE, Bluereef). Half of these projects were funded by the EU while the other half received national or private funding, in agreement to a review about artificial reefs in Europe by Fabi et al. (2011) which found that their development has been financially supported mainly from the EU as well as from local authorities and the private sector (e.g. diving clubs interested in the development of marine tourism).

7.2. Restoring Species

Whilst the overall target of a restoration programme may be a fully recovered ecosystem with full functioning and high biodiversity, actual restoration activity is often targeted at restoring particular species. This may involve the transplantation of seeds, sprigs, and shoots (macrophytes), transplantation of juveniles, fragments, adults (invertebrates, e.g. corals and sponges), or release of larvae, juveniles and adults (invertebrates and vertebrates). One of the main cases for species restoration is when the natural process of recovery of the species is not able to happen, either because there is no genitor stock of the species available or there is a lack of connectivity in the bio-geographical spread of the species, whereby larvae, juveniles or adults are not able to move in and recolonize an area. A second case may be when the natural recovery rate for a species is slow and there is a need to accelerate the natural process by introducing the species to a level where its replication may be faster or, for example, where it provides an essential structure to the habitat that other species are dependent on. An example of both cases could include deep-water corals where extensive areas might have been destroyed by trawling (for example on the North Atlantic Margin – Freiwald et al., 2004; Grehan et al., 2005), which may have also removed connectivity patches preventing recolonisation of distant areas. At the same time deep-water corals are extremely slow growing and fragments may need to be transplanted into a degraded area to provide both genitor stock for local recolonisation as well as the 3-dimensional structure that supports highly diverse hot-spot ecosystems (Roberts et al., 2006).

Different species may be the target of restoration depending on their roles in the ecosystem, this may include:

- **Habitat forming species** – structural species. These species characterise the physical structure of a habitat (e.g. seagrasses, kelps, corals, oyster reefs, vermiform reefs species, etc.). Most of these species are integral parts of ecosystems that have suffered extensive losses and degradation from numeral human activities and even extreme natural phenomena (Cambell et al., 2014; Mbije et al., 2013; Griffin et al., 2015; Sharma et al., 2016; Villamayor et al., 2016). The severity of the impacts differs between these and their status ranges from formally recognised as critically endangered (Griffin et al., 2015; Lirman et al., 2014) to vulnerable to anthropogenic disturbances (Waycott et al., 2009). They are all important for the habitat they provide, on which other species may be dependent for food, attraction, substrate, mating, nursery areas, and various direct ecosystem goods or service, for example, in the case of seagrass beds, seawater oxygenation or carbon sequestration (Drexler et al., 2014; Liquette et al., 2013; Nordlund et al., 2016). It is no surprise that these highly valuable, in so many ways, habitat forming species have been the major focus of restoration projects worldwide (Bayraktarov et al., 2016, see Section 3.).
- **Keystone species**: defined originally as predator species that have a disproportionate effect on its environment relative to its abundance (Paine, 1969; 1995), and may cause a dramatic shift in an ecosystem if is removed. The original strict definition of predator has been more broadly defined in a recent review (Menge et al., 2013) and includes *key-industry species* (abundant species supporting consumers) and *foundation species* (critical species which define much of the structure of the community, including species that create or maintain habitats). The latter as well as including the habitat species mentioned above, also includes, for example, bioturbators (engineering species), which increase the soft sedimentary living space by increasing oxygen fluxes and strongly affect biogeochemical processes (e.g. the Norwegian lobster/langoustine *Nephrops norvegicus*), which in turn maintains particular habitats or trophic structures. Keystone engineer species such as *Spartina maritima* (small cordgrass), *Corallium rubrum* (red coral) and *Pinna nobilis* (fan mussel) are also the target of restoration efforts (Castillo et al., 2009; Benedetti et al., 2011; Bottari et al., 2017, and on-going MERCES cases).

- **Talismanic, emblematic, charismatic species, characteristic species, habitat defining species** (e.g. penguins on southern Atlantic/Antarctic beach, polar bears in the Arctic, manatees in the Caribbean/tropical Atlantic, wreckfish in the deep sea): these species are neither structural species, nor keystone, i.e. they do not necessarily have a strong functional role in the ecosystem, but rather a strongly perceived importance where their presence is a sign/indicator of an ecosystem in good condition. Restoration of these types of species in the terrestrial domain would refer to a breeding programme, which has not had a marine equivalency. Captive breeding of marine mammals has been undertaken successfully (polar bear, sea-lions, seals, dolphins, killer whales), but invariably this has been mostly to provide further stock for captivity, with less examples of rehabilitation of stranded and entangled animals and release to the wild (Moore et al., 2007). Captive breeding of marine wild fish under aquaculture facilities (e.g. tuna ranching) is conducted solely for consumption and not conservation purposes.

A specific type worth noting is when restoration is applied to commercial fish and other stocks, such as scallops and oysters. This in almost all cases refers to rebuilding or recovery programmes for the stock of a particular commercial species, which would then allow it to be sustainably harvested. Rebuilding has in many cases shown to be a difficult process (Murawski, 2010). The goal is not so much protection of a species but rather conservation towards the continued sustainable use of that species. Restorative actions and recovery plans can range from technical measures and temporary banning of the fishing activity, to the creation of no-take areas and/or no-harvest sanctuaries and the restoration of essential fish habitats (e.g. *Posidonia* beds are nursery grounds for many juvenile fish) and can confer various benefits (Powers et al., 2009). Stock recovery can be a step following a successful habitat restoration effort, for example, re-introduction of scallops in restored eelgrass habitats (Schmitt et al., 2016).

7.3. Environmental Disaster and Multiple Related Restorative Actions

Many of the current drivers for restoration activities concern our understanding of how widespread degraded habitats are, that it is not acceptable to leave habitats in a degraded status, and the need to protect areas from future damage. This has resulted in the international conventions and directives requiring actions and spatial targets within reasonable timeframes (e.g. the CBD Aichi targets or the Habitats and Marine Strategy Framework Directive). Another driver is the immediate response to a single damaging impact through an environmental accident.

The response in this case may be reasonably straightforward or could be very complex at a high level involving a wide-scale environmental disaster. Most wide-scale human related accidents involve high volume oil spills, but wide-scale damage could also be natural, for example the impact of a tsunami in a region, where beyond the physical impact of the wave damage other impacts are caused from secondary wave-mediated impacts (e.g. destruction of protective structures, grounding of vessels over reefs, spillage or released of contaminants, deposits of terrestrial material in the marine environment, transport of alien species). There are contingency plans for many types of accidents at sea or on land that may affect the marine ecosystem, notably to limit the cause of the impact, to prevent further damage, remove the danger and to restore the ecosystem. Environmental disaster may involve large areas encompassing a wide range of different ecosystems, may involve many different types of organisations and authorities and may cross national borders. The response requires a high level of management and organisation. One of the most recent cases of environmental disaster was the Gulf of Mexico Deepwater Horizon incident. Both the magnitude of the accident and the extent of the response that followed were extreme and unprecedented in scale, and in the following section we provide this as a case study in disaster management requiring multiple restorative actions.

7.3.1. The Deepwater Horizon incident: an unprecedented environmental disaster

Background

On April 20, 2010, the Deepwater Horizon mobile drilling unit located about 50 miles offshore from Louisiana, exploded, caught fire, and eventually sank in the Gulf of Mexico, resulting in a massive release of oil and other substances from BP's Macondo well. The well continuously and uncontrollably discharged oil and natural gas into the northern Gulf of Mexico for 87 days after the explosion. Approximately 4 million barrels of oil were released into the Gulf of Mexico (McNutt et al., 2011), resulting in a surface oil slick which covered a cumulative area of 112,100 square kilometres, and was washed onto at least 2,100 kilometres of shoreline. Additionally, oil contamination settled on the seafloor over thousands of kilometres (Camilli et al., 2010). The impacts caused by the Deepwater Horizon spill applied to a wide range of organisms, and resulted in injuries to multiple habitats, species, and ecological functions that affected the entire ecosystem of the northern Gulf of Mexico.

Under the US Oil Pollution Act (OPA) of 1990, a council of federal and state "trustees" acting on behalf of the public was established to assess the natural resource injuries, develop a restoration plan, and acquire funding to make restoration possible. They jointly developed a baseline document (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016), for (a) providing a natural resource damage assessment and restoration plan, and (b) presenting an examination of the environmental impacts of various restoration alternatives. Public input was requested and considered important for restoration planning, both during the preparation of the programmatic document and after its completion. The responsible party (BP) was charged to pay for all actions of restoration implementation, as dictated by OPA, with a total of $8.1 billion, the largest natural resource damages settlement in history (Bradshaw, 2016).

Immediate response actions and assessment of injury

The programmatic documents of NOAA Gulf Spill Restoration program define ‘injury’ according to the Oil Pollution Act, as: *“an observable or measurable adverse change in a natural resource or impairment of a natural resource and/or service”.*

Immediate response actions were undertaken to reduce the extent of the oil spill and reduce human exposure and injuries to natural resources. However, some of these response activities also affected the environment in negative ways (e.g. burning oil produced air pollution, increased boat traffic and shoreline activity disturbed habitats). Environmental injury assessment identified direct toxicity effects from exposure to oil (e.g., death, disease, reduced growth) as well as mechanical damages to a wide range of habitats, from marshlands to the deep-sea) as well as associated organisms from the entire food web spectrum over a broad geographical scale. Impacts to marine and coastal ecosystems and species have been assessed and documented for a variety of species and habitats in an array of papers and reports following the incident. For example, severe to moderate impacts have been reported for marshland vegetation (*Spartina* and *Juncus*) in intertidal soils subjected to different levels of oil pollution (Lin & Mendelssohn, 2012). In the deep sea, impacts of the spill to coral communities was documented as partial or total necrosis and signs of stress in populations several kilometres away from the source spilling source (Fisher et al., 2014; White et al., 2012).

Restoration plan

The programmatic documents of NOAA Gulf Spill Restoration program consider ‘restoration’ as: ***“****Any action that restores, rehabilitates, replaces, or acquires the equivalent of the injured natural resources”.*

The extensive impacts of the incident to multiple habitats and species over a broad geographical scale establish the need for comprehensive restoration planning on the ecosystem scale that recognises and strengthens existing connectivity among habitats, resources, and services in the Gulf of Mexico. Since the restoration efforts are planned to be applied over extended areas and over a long timeframe (15 years), the programmatic document (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016) gives the participating bodies flexibility to accommodate changes over the lifetime of the restoration process both regarding the implementation approaches and the allocation of the budget, in order to adjust to scientific or technological progress.

The trustees' consortium evaluated programmatic restoration alternatives and developed a comprehensive, integrated ecosystem restoration plan based on five goals and 13 restoration types, along with 8 restoration areas: the 5 states affected by the spill, along with Region-wide, the Open Ocean, and an eighth "Restoration Area" which refers to additional funds reserved for currently Unknown Conditions and Adaptive Management. The main restoration approaches categorised by the planning goals, are shown in a schematic in Figure 53. They span from pressure reduction, to conservation and restoration of numerous species (from fish to mammals and vegetation) and ecosystems (e.g. wetlands, near shore habitats and the deep-sea including submerged aquatic vegetation, beaches, oyster reef habitats), restoration of ecosystem services and monitoring of adherence to the project goals and they follow five high level objectives (restore and conserve habitat, restore water quality, replenish and protect living coastal and marine resources, provide and enhance recreational opportunities and provide for monitoring, adaptive management, and administrative oversight). Emphasis is given in adaptive management which allows fine-tuning of the restoration program over time, based on monitoring data and evolving scientific understanding.

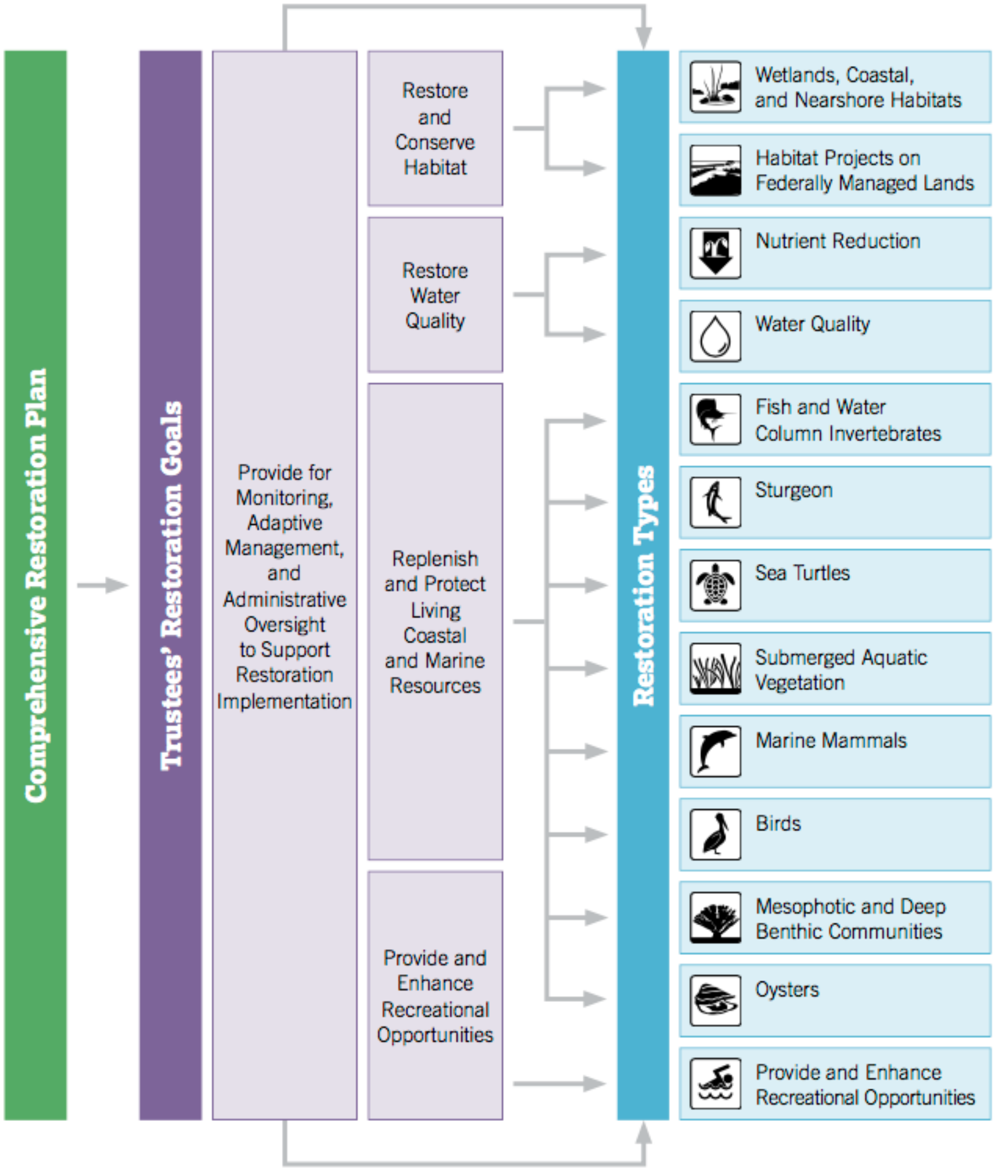


Figure 53. The comprehensive restoration plan for the Gulf of Mexico incorporates 5 goals and focuses to 13 restoration types.

The restoration plan for the Deepwater Horizon oil spill environmental injury serves as an interesting example of response to a massive degradation event and showcases the appropriate directions for any environmental disaster mitigation initiative, or – even more generally – any broad-scale restoration effort.

Key success elements include:

- Aiming restoration at the ecosystem level, attempting to combine and link all specific restoration actions to ensure the broader benefit.

- Performing an extensive and detailed assessment of the level of degradation prior to design and planning of concrete restoration actions.
- Applying the ‘polluter pays’ line, charging the responsible for the injury for the environmental impact assessment, the immediate actions, and the restoration efforts.
- Securing the involvement of all major administrative, authoritative, and scientific stakeholders in managerial structures.
- Actively engaging the general public to all phases of preparation and implementation.
- Adopting a long timeframe for preparation and restoration actions.
- Assessing the environmental impact of the restoration measures themselves in the design and implementation phase.
- Employing comprehensive monitoring plan and an adaptive management approach, fine-tuning the restoration program over time based on monitoring data and evolving scientific understanding.

7.4. Removal of Threats

Despite the increasingly wide realization that marine ecosystems are vital to human well-being through the many services they provide and of their high ecological, cultural and economic value, deterioration and damage is still widespread (TEEB, 2010; Halpern et al., 2008). Pre-existing and new pressures remain and/or are increasing to alarming levels with significant effects. Numerous management options and interventions are available to tackle some of these threats along with appropriate legislative means to support actions through policy implementation. Various directives are implemented at EU and Regional Sea levels aiming at managing the threats with the aim to achieve a good environmental status for European seas and a sustainable blue economy under the ecosystem approach to management. Non-point sources and very fast spreading pressures such as marine litter and invasive species are at the top of the conservation and restoration agenda both as prerequisite for restoration and as threat to any restoration action. Equally, the overpowering effect of keystone grazing species is a cause for widespread degradation of various marine ecosystems and a challenge to control if any restoration action is to succeed. Control and removal is the answer to these three threats and this is further developed below along with a few examples demonstrating the opposite option of blocking access to a threat by building barriers.

7.4.1. Litter

Marine litter is one of the most widespread pressures on the marine environment and one that is increasingly mapped for its extent and intensity across the European Seas (Smith et al., 2017).

Marine litter issues, sources and solutions, increasingly dominate the environmental news and social media outlets. This is not surprising as marine litter is practically everywhere, polluting our shores, the sea surface with floating plastic and other rubbish, the water column and the seabed from the shallow to very deep waters (Pham et al., 2014; Vlachogianni et al., 2017; Lopez-Lopez et al., 2017). It has entered the food chains and is being consumed and affecting sea turtles, sea mammals, sea birds and fish (Anastasopoulou et al., 2013; Hardesty et al., 2015) and has been implicated in spread of invasive species (Tutman et al., 2017). It comes in all sizes from microplastics, to small plastic fragments and fibres, to very large items. It has been attributed to various sectors of human activity, including, for example, from recreation and tourism (e.g. plastic single use items) to defence (e.g. munitions). Fisheries are a contributor to marine litter by losing nets, traps, pots and lines that entangle or smother species (Hardesty et al., 2015) as well as with direct inputs of ordinary items from fishing vessels.

There are various examples of litter restoration projects including fishing-for-litter schemes and removing abandoned, lost or discarded fishing gear (ALDFG) by divers (see for example DeFishGear an IPA Adriatic project involving all countries bordering the Adriatic Sea (http://www.defishgear.net/) and worldwide marine litter projects (http://marinelitternetwork.com/global-projects/). One iconic restoration example concerns the removal of 2500 car tyres deposited underwater in the 1980's and arranged to represent an artificial reef in the current NATURA 2000 site "Baie et cap d'Antibes - Iles de Lérins" in the Alpes Maritimes in the south of France. The project conducted in 2005 by the French MPA Agency (Poiret, 2015) had the aim to restore the integrity of the marine environment over which the tyres had scattered in recent decades, and thus avoid any alteration of the site's natural habitats of European importance. The tyres, originally set to restore fishery resources, had not only altered the underwater landscape but also mechanically damaged the seabed, threatening habitats of community interest such as *Posidonia* and coralligenous formations. Another restoration project from the Adriatic Sea investigated the recovery of rocky habitats by removing abandoned/lost/discarded ghost nets/gears and willingness to pay for different restoration options (a LIFE-GHOST EU project, Tonin & Lucaronin, 2016). Alongside other human activities, research is also responsible for contributing to the problem by losing or leaving on the sea bed, nets, traps, sampling equipment, ballast weights and frames of all kinds and sizes (e.g. loss of benthic landers, underwater remotely operated vehicles such as the NEREUS (Showstack, 2014; Cressey, 2014); ISIS ROVs and AUVs (Copley, 2014).

Reducing or removing threats is paramount to any project and litter has been identified as one of pressures to be removed before any restorative action can take place in any site (McDonald et al., 2016; Bekkby et al., 2017). Biodegradable plastic bags have been shown to represent a future threat for seagrass meadows by affecting sediments and altering above/below seagrass compartments and plant/species relationships (Balestri et al., 2017). However, restoration actions also can contribute to litter and plastic pollution. Numerous designs of *in situ* coral nurseries supported by metal and PVC frames and/or cement (varying from low relief to Christmas-tree-like-trees have been tested for coral gardening around the world and are being implemented in many reef restoration projects (Shaish et al., 2008; Young et al., 2012; Meesters et al., 2015). These structures can remain, be lost due to storms or be abandoned in the marine environment due to overcapacity and lack of resources, or bad management where orphan nursery platforms have been left unattended, collapsing and resulting in mortality of threatened/endangered corals (Lirman & Schopmeyer, 2016). Recently a joint effort of three projects: MERCES, OBN-Griend project and STW Bridging Thresholds are experimenting with an undisclosed biodegradable frame system to support landscape-scale mussel-seagrass restoration in the Netherlands (Christianen et al., 2017). The use of degradable materials such as hessian bags for seagrass, bamboo frames for corals and biodegradable ties) have been tested previously with different levels of success (Ferse, 2010; Park & Lee, 2007; Irving et al., 2014). Beyond the artificial frames, various glues, screws, lines and ties of different materials are used to anchor and support transplantation efforts for corals and sponges (McMurray & Pawlik, 2009; Griffin et al., 2015). Currently there is no substitute for several plastic essentials and ceramic and plastic tiles seem to be a good substrate for coral larvae (Okamoto et al., 2008; Carlo Cerrano, personal communication, MERCES on-going experiment). Coral and other larvae also settle on oil-gas-platforms which are the subject of the recent rigs-to-reefs debate, i.e. whether platforms acting as species connectivity points should be removed after decommissioning (Macreadie et al., 2011).

7.4.2. Invasive species

Invasive species cause significant impact on ecosystem services and biodiversity in the European seas (Katsanevakis et al., 2014). The adoption of management actions for controlling their populations and mitigating their impacts is globally acknowledged as a major challenge, though it has been overlooked in marine conservation plans (Giakoumi et al., 2016). The Aichi Target 9 of the Convention on Biological Diversity (CBD) states that by 2020, (a) invasive alien species and pathways must be identified and prioritized, (b) priority species must be controlled or eradicated, and (c) measures must be in place to manage pathways to prevent their introduction and establishment.

At the European Union level, under the Marine Strategy Framework Directive – MSFD (2008/56/EC), member states are committed to develop strategies to achieve Good Environmental Status (GES), determined on the basis of eleven qualitative descriptors; specifically, the second descriptor (D2) of the MSFD dictates that "*Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystems*". At the Mediterranean level, an indicator (EO2) focusing on "*trends in abundance, temporal occurrence, and spatial distribution of non-indigenous species, particularly invasive, non-indigenous species, notably in risk areas (in relation to the main vectors and pathways of spreading of such species)*" has been considered as a key element in the ecosystem-based Integrated Monitoring and Assessment Program – IMAP (UNEP(DEPI)/MED IG.22/28. Decision IG.22/7), which was recently adopted by the Contracting Parties of the Barcelona Convention (2016).

Listing of any non-native, and particularly invasive, species in a given site forms an essential component of ecosystem baseline inventories for the planning and design of ecological restoration projects, and the elimination or control of invasive species has been considered as a particular goal in one of the six key ecosystem attribute categories (i.e., Absence of threats) required prior to developing longer-term goals and shorter-term objectives for ecological restoration (McDonald et al., 2016). In some cases, even if full recovery has been achieved, interventions such as the removal of invasive species may be needed for ecosystem maintenance purposes (McDonald et al., 2016).

According to the CBD (1992), the general policy approach for management of invasive species involves the three following stages: 1) prevention, 2) early detection and eradication, and 3) control and long-term containment. In cases where eradication is not feasible, control and/or containment measures should be implemented. Eradication methods include: (a) mechanical and physical control (e.g. cutting and uprooting plants or culling fish); (b) chemical control (e.g. use of pesticides, such as piscicides and molluscicides); (c) biological control (i.e. introduction of biological control agents) which is considered as a "green" alternative to chemical control; as well as (d) novel management approaches which may arise as serendipitous by-products and technologies of entirely different research fields and directions (e.g., endocrinology and molecular genetics) (Simberloff, 2014).

Prevention of introduction is considered the most cost-efficient and preferred control method (Ray, 2005; COM, 2008) since the elimination of existing populations of a given species from an area does not prevent re-invasion or further introductions, especially in areas which are susceptible to such invasions while in most cases invasions are detected only after the introduced species have spread considerably (Simberloff, 2014).

Till the last decade, very few eradications programs had successfully taken place in Europe, basically for conservation purposes (e.g. LIFE projects), but none of these focused on invertebrates or marine species (Genovesi, 2005). However, according to the latter review, this may be partly due to the fact that several small-scale removals are often reported only in grey literature sources which are difficult to access. The main reasons for the small number of eradication programs, compared to other parts of the world (e.g. New Zealand) include the inadequate and unclear legal and authorization framework, the lack of awareness and often public opposition (especially for vertebrates), the lack of funding, and the failure to detect new invasions at their early stage of introduction (Genovesi, 2005 and references therein).

Historically, there are very few cases of successful eradication of established marine invasive species in the scientific literature, such as that of the polychaete *Terebrasabella heterouncinata* from California (Kuris & Culver, 1999) and the fouling mussel *Mytilopsis* sp. from the Darwin Harbour estuary, Australia (Bax et al., 2002). In both cases, the invading species were introduced into a small spatial scale, thus resulting to a relatively lower eradication cost, they were detected at an early colonization phase, while the lines of authority were clear enough to allow individuals or agencies to take all necessary actions (Myers et al., 2000; Ray, 2005). However, these conditions are rarely met and thus, the adoption of an effective early-warning, rapid-response system, for example through the development of collaborative thematic networks (Zenetos et al., 2015) and citizen science initiatives (i.e. trained citizen volunteers), could greatly assist in the reporting and eventual eradication of invasive species (COM, 2008; Sambrook et al., 2014; Simberloff, 2014). One of the most well-known successful examples of eradication of marine species is that of the green macroalgae *Caulerpa taxifolia* from California, where a group of recreational divers recognized the species and contacted the authorities at an early stage of introduction (Anderson, 2005). On the other hand, the potential to eradicate the same species from north-western Mediterranean areas was largely lost as eradication efforts were initiated more than a decade after its first discovery at Monaco in 1984 (Myers et al., 2000 and references therein). Another notable example of active involvement of volunteers in the eradication of marine invasive species was that of the host-specific polychaete *T. heterouncinata* from an infected area in California, where an army of volunteers removed 1.6 million gastropods which were susceptible hosts to this polychaete species (Kuris & Culver, 1999). Volunteers have also supported efforts to control populations of the highly invasive alien lionfish (*Pterois volitans* and *P. miles*), and even reverse declines in their native prey fish, in the Caribbean Sea by using selective fishing gear (de León et al., 2013; Côté et al., 2014; Anderson et al., 2017). A relevant project was recently initiated in Cyprus (RELIONMED; LIFE+ Nature and Biodiversity 2016) and will last for 4 years (2017–2021).

The aforementioned eradication examples involved mechanical or physical control (e.g. lionfish culling) or chemical eradication methods (e.g. eradication of *Caulerpa taxifolia* and *Mytilopsis* sp.). Chemical eradication methods (e.g. herbicides) are also commonly used for the removal of invasive plants with the aim to restore tidal marsh habitats (e.g. Turner & Warren, 2003; Kimball et al., 2010; Kerr et al., 2016). Examples of biological control of marine invasive species include the use of other invasives, for instance the predation by the blue crabs *Caliinectes sapidus* on the gastropod *Rapana venossa* in USA (Harding, 2003) and the use of exotic mangrove species to control *Spartina alterniflora* invasion in coastal China, as a model to promote native community restoration during the control of exotic invasion (Zhou et al., 2015).

7.4.3. Keystone species (echinoid barrens)

The loss of macroalgal forests due to overgrazing by sea urchins has been documented in several areas of the world, including the European seas (Bekkby et al., 2017 and references therein). During the last decades several studies have suggested that the dramatic shift of macroalgal forests into barrens is indirectly linked to overfishing and the related trophic cascade effects (Sala et al., 1998 and references therein). In a simplified scheme, the depletion of predator fish species may result in significant increase of sea urchins which by turn overgraze and deplete canopy-forming algae (e.g. kelps and *Cystoseira* spp.). Comparative studies between protected and unprotected Mediterranean areas showed that fish predation impact on sea urchins was higher at protected areas, thus suggesting that fishing restrictions (e.g. establishment of no-take marine reserves) could re-establish lost interactions (Guidetti, 2006).

The control of urchin populations in overgrazed areas has been commonly adopted as a practice which could assist in the restoration of degraded macroalgal beds (Bekkby et al., 2017; Fraschetti et al., 2017 and references therein). The main methods used include the manual removal of sea urchins by divers, including citizen volunteers (Watanuki et al., 2010; Guarnieri et al., 2016) and their exclusion – along with other herbivore species – when applying restoration/enhancement methods (e.g. transplantation of macroalgae on artificial structures) by using anti-grazing nets or cages (e.g. Falace & Bressan, 2002; Perkol-Finkel et al., 2012). Nevertheless, colonization capacities of target species should be considered when designing removal actions in order to define the appropriate spatial scale (Fraschetti et al., 2017). In addition to the above methods of physical control, the use of biological control such as the increase of predatory pressure (e.g. introduction of crabs) has been also considered in certain cases in an effort to increase the probability of restoration success (Fagerli et al., 2014; Bekkby et al., 2017).

Similar physical (e.g. manual removal) and biological control methods (e.g. inclusion of predators such as tritons), as well as chemical ones (i.e. injection of bile salts, household vinegar, copper sulphate or sodium bisulphate by SCUBA divers) have been applied for managing the outbreaks of the crown-of-thorns starfish (COTS) *Acanthaster planci*, which is one of the largest causes of coral cover loss in Indo-Pacific reefs (Johnson et al., 1990; Morello et al., 2014; GBRMPA, 2017).

7.4.4. Fencing

In terrestrial ecosystems, some threats to an area to be restored can be easily managed by fencing. Pest-exclusion fences have been typically erected to prevent access of certain types of animal pests (e.g. rodents, rabbits, sheep, deer or cattle) to horticulture and grasslands as well as protected areas, allowing local wild animals or vegetation to develop and flourish. A variety of fence types has been developed, from simple or electrified wire fences, which may include subsurface fencing elements to multispecies fence designs and exclusion barrier systems (Day & MacGibbon, 2007). One of the most striking examples is the 5,614-km long Dingo Fence for the protection of sheep flocks from predation in Australia, which is one of the longest man-made structures in the world and is the world's longest fence.

In the marine world fences have been used on very small scales, particularly in exclusion experiments (e.g. Sala et al., 2011). But in many cases, these are anti-grazing nets or cages that prevent an animal from climbing or swimming over the fence (e.g. Falace & Bressan, 2002; Perkol-Finkel et al., 2012). Fences have been used on the shoreline to protect, for example swimmers from sharks (shark nets and barriers), box jellyfish (stinger nets) or shorelines/embayments from floating oil slicks. In contrast to terrestrial analogues, underwater continuous fencing for large areas is impractical as the medium of water allows fauna easily to climb or swim over and expenses of caging would be impractical. Furthermore, while in terrestrial forestry individual seedlings can be planted in grazer protection tubing, in the marine environment, the medium of water allows grazers to easily enter such tubing and damage the planted individuals.

In large areas, barrier schemes also known as anti-trawling protection reefs have been used to restrict trawling activities from entering a protected area or areas where artificial reefs have been deployed. These modules are usually concrete units heavy enough to hamper illegal trawling and bear iron beams to entangle the nets (Fabi et al., 2011). Mixed modules, combining the above characteristics typical artificial reefs are also available. In Europe, anti-trawling protection reefs have been deployed in France, Italy, Spain and Portugal (Fabi & Spagnolo, 2011). Underwater barrier schemes do not have to be continuous but spaced to catch a trawl. If the area is very large the barrier must be in-depth so that a trawl is not deployed and recovered from within the barrier-defined area.

7.5. No Net Loss (NNL) and Net Positive Impact (NPI)

Many companies in the primary natural resource sectors recognise that they need to manage their operational and reputational risks to mitigate environmental changes related to extraction, pollution, biodiversity loss and climate change (Aiama et al., 2015) to ensure they continue to operate with a 'social license'. With regards to biodiversity related risks, the concepts of 'No Net Loss' (NNL) or 'Net Positive Impact' (NPI) are taking hold (Rainey et al., 2014). The goal is to ensure that negative biodiversity impacts caused by a project are either balanced (for NNL) or outweighed (for NPI, also referred to as net gain) by biodiversity gains through compensation measures implemented by the project (Aiama et al., 2015). The 'net' in NNL and NPI acknowledges that some biodiversity losses at the development site are inevitable, and that biodiversity gains may not be perfectly balanced in regards to the time, space, or type of biodiversity impacted.

The European Commission (EC), in 2014, consulted with interested citizens, public authorities, business and NGOs about their views on a future No Net Loss Initiative at the EU level. A majority of the respondents felt that agriculture, forestry, fisheries and aquaculture were priority sectors for inclusion in a future initiative. There were also calls for the scope to cover all economic sectors in order to include all pressures on biodiversity (EC, 2014).

For NNL or NPI goals to be achieved credibly, they typically must follow a systematic biodiversity management approach commonly known as the 'mitigation hierarchy' (Aiama et al., 2015). The underpinning principle of biodiversity offsetting is NNL and NPI – i.e. the counterbalancing of biodiversity losses with biodiversity gains. There was a strong agreement among the EC's survey respondents that respecting the principles of the mitigation hierarchy was essential for achieving the objective of NNL (EC, 2014). The Institute for European Environmental Policy (IEEP) in collaboration with others, subsequently, undertook a scoping impact assessment for a future EU initiative on No Net Loss of Biodiversity and Ecosystem Services (Tucker et al., 2016).

7.5.1. The Mitigation Hierarchy

The **mitigation hierarchy** is a set of prioritised steps to alleviate environmental harm as far as possible through avoidance, minimisation (or reduction) and restoration of detrimental impacts to biodiversity and ecosystem services. It is illustrated in relation to net impacts in Figure 54. It is not a standard or a goal, but a 'best practice' approach to mitigation planning. It can be defined as: '*the sequence of actions to anticipate and avoid impacts on biodiversity and ecosystem services; and where avoidance is not possible, minimize; and, when impacts occur, rehabilitate or restore; and where significant residual impacts remain, offset* (CSBI, 2015).

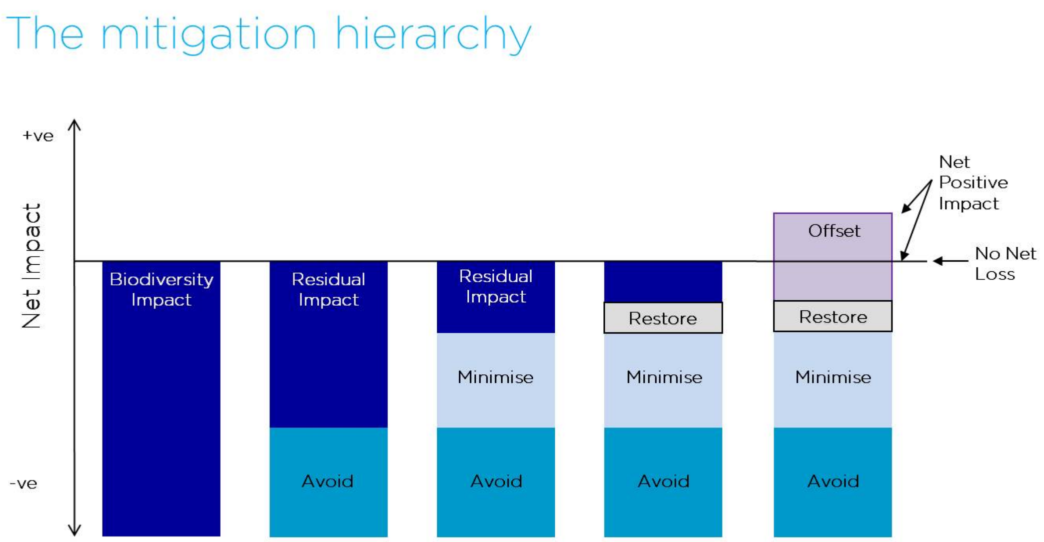


Figure 54. The mitigation hierarchy for managing biodiversity risk (from CSBI, 2015).

The steps of the mitigation hierarchy are as follows (from CSBI, 2015):

1. Avoidance: the first step of the mitigation hierarchy comprises measures taken to avoid creating impacts from the outset, such as careful spatial or temporal placement of infrastructure or disturbance. For example, placement of roads outside of rare habitats or key species' breeding grounds, or timing of seismic operations when aggregations of whales are not present. Avoidance is often the easiest, cheapest and most effective way of reducing potential negative impacts, but it requires biodiversity to be considered in the early stages of a project.

2. Minimisation: measures taken to reduce the duration, intensity and/or extent of impacts that cannot be completely avoided. Effective minimisation can eliminate some negative impacts. Examples include such measures as reducing noise and pollution, designing powerlines to reduce the likelihood of bird electrocutions, or building wildlife crossings on roads.

3. Rehabilitation/restoration: measures taken to improve degraded or removed ecosystems following exposure to impacts that cannot be completely avoided or minimised. Restoration tries to return an area to the original ecosystem that occurred before impacts, whereas rehabilitation only aims to restore basic ecological functions and/or ecosystem services (e.g. through planting trees to stabilise bare soil). Rehabilitation and restoration are frequently needed towards the end of a project's life-cycle, but may be possible in some areas during operation (e.g. after temporary borrow pits have fulfilled their use).

Collectively **avoidance**, **minimisation** and **rehabilitation/restoration** serve to reduce, as far as possible, the residual impacts that a project has on biodiversity. Typically, however, even after their effective application, additional steps will be required to achieve no overall negative impact or a net gain for biodiversity.

4. Offset: measures taken to compensate for any residual, adverse impacts after full implementation of the previous three steps of the mitigation hierarchy. Biodiversity offsets are of two main types: '**restoration offsets'** which aim to rehabilitate or restore degraded habitat, and '**averted loss offsets'** which aim to reduce or stop biodiversity loss (e.g. future habitat degradation) in areas where this is predicted. Offsets are often complex and expensive, so attention to earlier steps in the mitigation hierarchy is usually preferable.

The mitigation hierarchy is useful as a framework because it can:

- Promote performance measurement
- Reduce scheduling delays and instigate cost-effective approaches

- Function as a risk assessment and management tool

Figure 55 illustrates the iterative process of *avoiding* and *minimizing* until remaining risks and impacts can be managed through the remediative measures of *restoration* and *offsetting*.

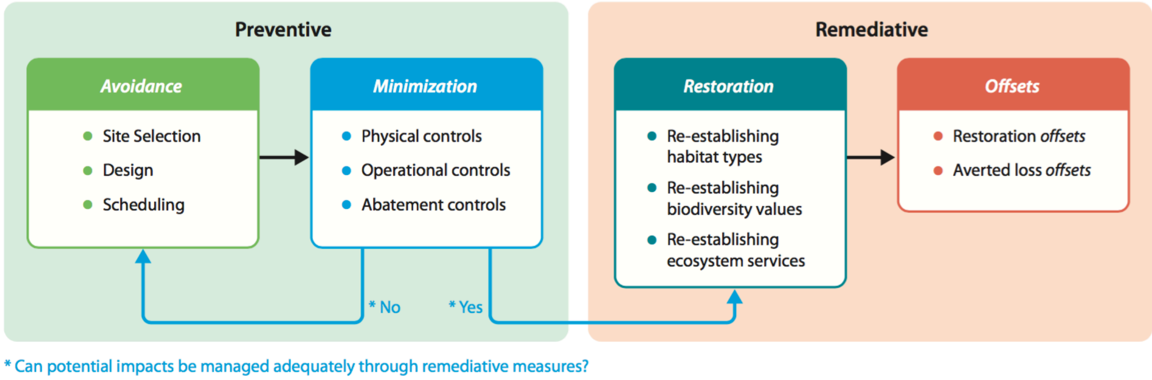


Figure 55. Schematic diagram showing the implementation of the mitigation hierarchy (from CSBI, 2015).

The mitigation hierarchy can be viewed as a set of prioritized, sequential components that are applied to reduce the potential negative impacts of project activities on the natural environment. It is not a one-way linear process but usually involves iteration of its steps. It can be applied to both biodiversity and related ecosystem services. There are two preventive components, *avoid* and *minimize*, and two remediative components, *restore* (or rehabilitate) and *offset* (see Figure 56). As a rule, preventive measures are always preferable to remediative measures, from ecological, social and financial perspectives.

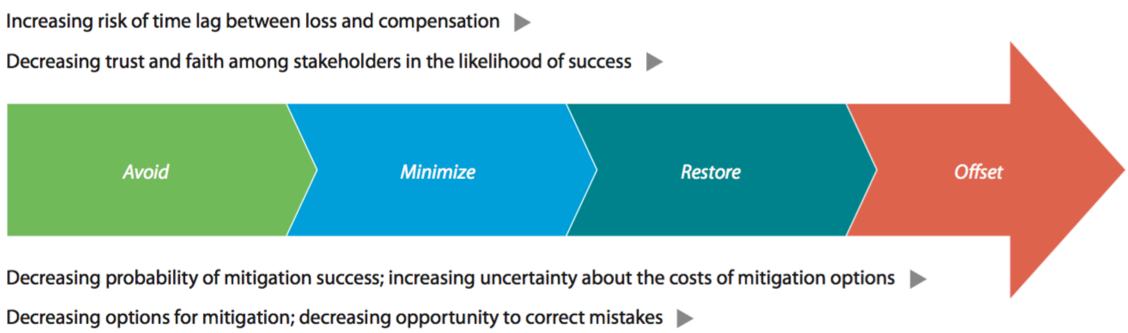


Figure 56. Avoid, minimize, restore, offset (from CSBI, 2015).

Both industry and financial institutions apply the mitigation hierarchy across the different stages of the project cycle, but for slightly different purposes. For industry, the mitigation hierarchy is mainly a tool for planning and adaptive management; for financial institutions it provides a framework to guide clients, and a means to audit performance.

The mitigation hierarchy is not a one-way linear process, and entails both feedback and adaptive management to optimize investments. The question of 'How much *avoidance* is enough?' depends on the mitigation options remaining for the biodiversity features of concern. Iteration may therefore be necessary (Figure 57).

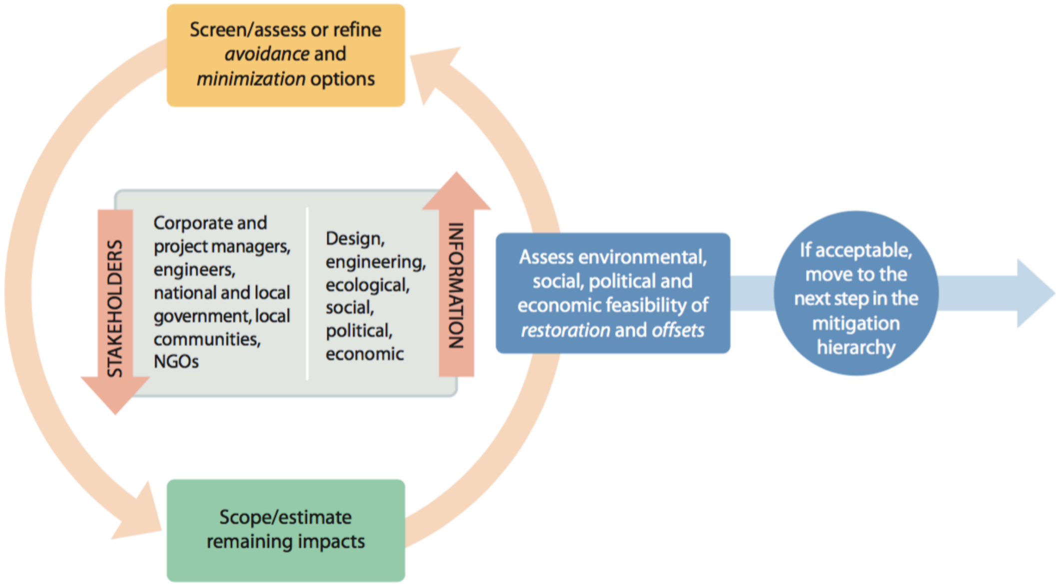


Figure 57. The iterative stages in the assessment of options and impacts, to optimize investment in components of the mitigation hierarchy (from CSBI, 2015).

The following feedback and adaptive management steps can be taken;

- Apply *avoidance* and *minimization* measures to potential impacts on biodiversity and ecosystem services using a risk-based approach.
- Characterize and estimate the magnitude of the potential remaining impacts to be addressed by *restoration* and, if necessary by *offsetting*.
- Assess the environmental, social, political and economic feasibility of *restoring* or *offsetting* this type and magnitude of impact on biodiversity and ecosystem services values.

7.5.2. Biodiversity Accounting

Biodiversity is intrinsically difficult to measure and compare quantitatively; no single metric can describe biodiversity as a whole therefore adding up losses and gains, a type of 'biodiversity accounting' is difficult. Nevertheless, methods have been developed (and continue to be refined) for calculating loss and gain of particular biodiversity values. These may focus on habitat as a useful proxy for biodiversity as a whole (e.g. 'quality hectares', a measure of habitat area x condition), or on a small set of key species (e.g. 'units of distribution', a measure of the proportion of the population of a particular species in a defined area).

Any *offset* approach that aims to demonstrate how far residual impacts have been addressed (including a NNL or net gain approach) calls for calculation of project-attributable losses and gains for the specific biodiversity features of concern. For calculating losses, residual impacts on biodiversity features need to be considered at a landscape scale relevant to the ecology of the biodiversity features of concern (including 'indirect' impacts such as induced/facilitated access to an area with noteworthy biodiversity). In identifying the biodiversity features of concern, and selecting potential *offset* options (if warranted), cumulative impacts of multiple projects across the landscape may also need to be taken into account. The baseline for loss calculation is normally the situation prevailing before project implementation begins—one reason why it is important to undertake baseline biodiversity surveys for impact assessment.

Calculating gains involves a number of predictions of how biodiversity values will change following the implementation of an *offset* compared to what would have happened without the *offset* (the 'counterfactual' scenario). This is a complex technical task as biodiversity features show natural variation over time, and can be affected either positively (e.g. through other expected conservation investments) or negatively (e.g. through on-going habitat loss). Gain calculations thus call for a knowledge of baseline conditions pre-*offset* (gained, ideally, by targeted baseline surveys) and an estimate of trends in pressures and conservation responses (based on expert knowledge). Biodiversity accounting and offset design methods are illustrated in Figure 58.

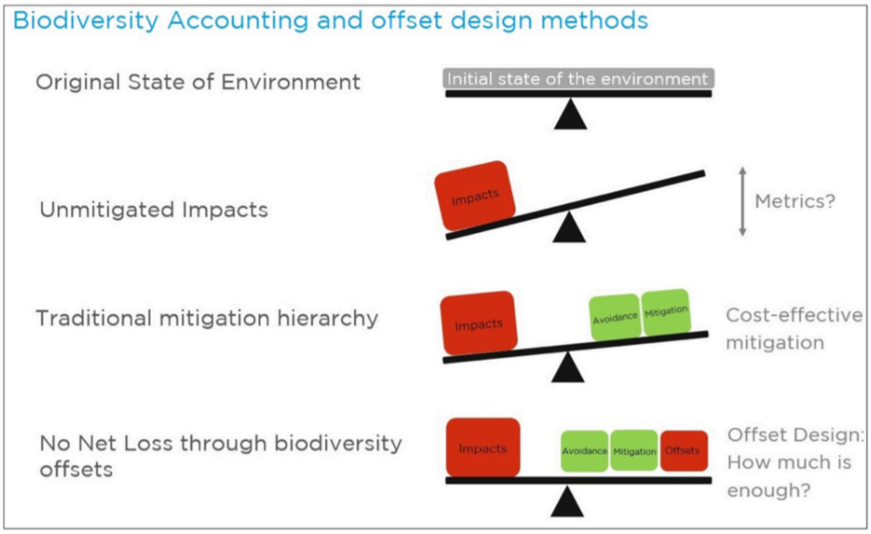


Figure 58. A summary of biodiversity accounting and offset design methods (from CSBI, 2015).

7.5.3. Feasibility assessments

Feasibility assessments consider technical, social, political and economic issues. To answer the question, 'Is it possible to achieve a target?' (such as NNL), the burden of proof goes through the stages of theoretical feasibility, technical feasibility (including cost considerations) and socio-political feasibility (including sustainability considerations) (Figure 59). As greater certainty is achieved, the project mitigation and *offset* options are narrowed down, as in any project design process.

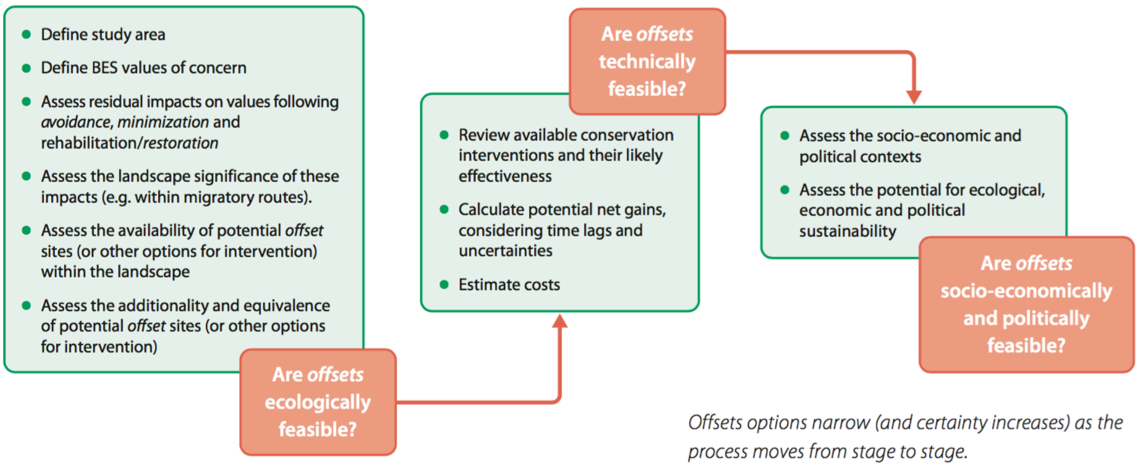


Figure 59. Steps in assessing the technical and political/business feasibility of a biodiversity conservation target (e.g. no net loss) (from CSBI, 2015).

7.5.4. Marine Biodiversity Offsetting

Biodiversity offsetting policies applicable to marine environments exist in six countries (US, Canada, Australia, France, Germany and Colombia) and have been actively considered in at least

27 others (Niner et al., 2017). Outside of these, a wide range of other approaches ranging from preliminary studies to identify potential compensatory habitat, to nascent biodiversity markets, and project-level application of corporate standards of NNL are also being considered (Niner et al., 2017a).

The EC includes Compensation under Article 6 of the Habitats Directive and is considering how application of NNL might extend beyond the key habitats and species listed in the Directive (McGillivray, 2012). Evidence suggests that where offsetting policy is developed for a specific marine application, the preferred approach is to pool financial contributions from developers into funds for strategic action for biodiversity benefit (Niner et al., 2017a).

Conceptually, the implementation of biodiversity offsets can take one of three forms: (1) *ad-hoc* projects delivered directly by the proponent of development causing biodiversity loss; (2) third party habitat banks (also referred to as species, conservation or mitigation banks) where 'biodiversity credits' equivalent to meeting offsetting requirements can be purchased or otherwise exchanged; and, (3) in-lieu fees where financial compensation for biodiversity impacts is pooled for strategic level conservation projects (Niner et al., 2017a). To guide the appropriate application of biodiversity offsets a set of key principles have been widely accepted as necessary for the success of the approach (Table 16).

Table 16. Key principles for biodiversity offsetting success (adapted from Niner et al., 2017a)

Principle	Detail
Mitigation hierarchy	Biodiversity offsets should be considered only as a last resort for residual impacts after avoidance and mitigation has been explored.
Equivalence	Demonstration of the balance between biodiversity losses and gains is required].
Additionality	Biodiversity offsets should not displace existing commitments or activity; they should deliver benefits beyond those that would occur in the absence of the offset project.
Continuity	Supply of biodiversity through offset projects requires consideration from a temporal and financial perspective.
Compliance success	Non-compliance with biodiversity offset requirements is a significant risk to achieving the aim of NNL.

According to Niner et al. (2017a), discussion and use of biodiversity offsetting has rapidly increased over the last decade for a number of reasons. Political agendas to promote the use of market-based instruments for conservation purposes have been identified as one of the main drivers for uptake of the approach. This political push has outpaced the development of ecological foundations for the approach, which are yet to be clearly defined. Given the knowledge gaps in the underpinning ecological science, the outcomes of biodiversity offsetting in terms of environmental protection are unclear. The challenges of this approach include those concerning our fundamental ability to restore ecology, inappropriate implementation and design of offsets, the need to seek equivalence across ecological components and ineffectual compliance regimes.

The guiding principles for the success of biodiversity offsets in marine environments are considered by Niner et al. (2017a) to be almost identical to those required in terrestrial environments. However even in terrestrial environments, success of the approach to counter biodiversity losses and the application of these principles has proved to be challenging and there are concerns that it's misuse may be contributing to declining trends of biodiversity. The difficulties faced in the terrestrial environment include; the accounting of biodiversity (often across biodiversity types) to ensure that the aim of NNL is met; our ability to restore ecological components and habitats; those relating to compliance, such as the appropriate application of the mitigation hierarchy and post-consent monitoring; and the avoidance of the perverse application of the approach. These challenges all apply to the marine application of biodiversity offsetting but are further exacerbated by three key factors; (1) the high level of uncertainty within marine impact assessment owing to the highly variable and connected nature of the environment; (2) the limited evidence of ecological restoration success in a marine context; (3) the diffuse, complicated and at times remote governance arrangements managing the resource (Niner et al., 2017a).

Scientific knowledge gaps and other practical challenges have necessitated flexibility concerning the manner in which key offsetting principles are implemented in policy frameworks relevant to such environments. The potential trade-off of such flexibility is that consequent marine offsetting practice may not be compatible with the ultimate objective of no net loss of biodiversity (Niner et al., 2017b).

Australia has one of the most developed policy frameworks for biodiversity offsetting in the world, and has only recently started the process of developing its first marine-specific offsetting policy (Maron et al., 2016; Niner et al., 2017b). The Australian experience is illustrative of the challenges associated with marine application of biodiversity offsetting, in particular the challenge of reconciling the need for practical flexibility with the fundamental objective of NNL. The Australian experience is illustrative of the challenges associated with marine application of biodiversity offsetting, in particular the challenge of reconciling the need for practical flexibility with the fundamental objective of NNL.

Addressing these challenges in the context of intensifying ocean-based development is likely to require both focused efforts to address outstanding scientific and technical challenges, and the possible re-interpretation of the concept of NNL, for example by allowing 'trading up' of biodiversity losses for gains of greater conservation value (Habib et al., 2013; Niner et al., 2017b).

7.6. Marine ecosystem restoration and nature-based solutions in coastal management

In recent years there has been an increase in the number of 'soft engineering' solutions to coastal management, especially in relation to flood protection (Borsje et al., 2011; van Slobbe et al., 2013). Depending on the location 'soft engineering' options may be more applicable and more cost-effective over the long term than building breakwaters and increasing the height of seawalls. Soft engineering methods have been described under a variety of terms including 'Nature-based Solutions' (NbS or NBS), 'Building with Nature' and 'Ecological Engineering' (Pontee et al., 2016). These schemes have generally been proposed in the context of coastal protection, but the benefits from them are much broader including 1) the sequestration of carbon by salt marshes, seagrass meadows, mangrove forests and coral reefs, 2) the sequestration of nutrients (nitrogen, phosphate) from terrestrial and agricultural run-off, 3) the provision of nursery areas for inshore fish populations, 4) the protection of farmland, 5) the conservation of biodiversity, 6) the provision of recreational activities and 7) safeguarding valuable real estate. Generally Nature-based Solutions in the past have focussed on just one of these ecosystem services. However, there is a growing realisation that for good coastal management all ecosystem services should be considered equally and their relative values over different timeframes taken into account.

The use of the term nature-based solutions to a certain extent depends on different interests. Some groups, such as ecologists, may include only plant and faunal components, while engineers may focus on processes that copy natural procedures (Pontee et al., 2016). In addition, NbS activities may be undertaken to restore degraded habitats, such as the wide scale loss of seagrass meadows or mangrove forests (International Federation of Red Cross and Red Crescent Societies: IFRC, 2011), or, in more recent times, undertaken to compensate for habitats lost on the coastline nearby, such as in relation to compensate for habitats lost to the expansion of a port or the creation of tidal barrages.

Narayan et al. (2016) studied the cost effectiveness and benefits of natural and nature-based coastal defences. Sixty-nine field coastal habitats around the world were examined to assess the effectiveness of salt marshes, seagrass meadows, mangrove forests, kelp beds and coral reefs in reducing wave height and hence coastal erosion. A comparison of the costs of nature-based solutions to hard engineering structures showed that salt-marshes and mangroves could be two to five times cheaper than a submerged breakwater in certain conditions. The actual benefits depend on the specific site being considered. The authors also found that there are very few studies which integrate and synthesise engineering and ecological knowledge for effective comparisons of soft and hard engineering coastal defences.

Nature-based solutions are also important in the sequestration of carbon, both within the vegetation of salt marshes, seagrass meadows and mangrove forests and the sediments which are accreted around them. This is often termed ‘blue carbon’ (Howard et al., 2014). Carbon sequestered in coastal sediments remains trapped for centuries to millennia. Sediment carbon pools including roots, rhizomes, and leaf litter may constitute 50% to over 98% of the total ecosystem carbon stock (Fourqurean et al., 2012; Howard et al., 2014). Per unit area salt marshes, seagrasses and mangroves store considerably more carbon than tropical and boreal forests (Fourqurean et al., 2012; Pendleton et al., 2012). However, coastal blue carbon ecosystems are being destroyed at an alarming rate (Sifleet et al., 2011). If current trends continue all unprotected mangroves could be lost in the next 100 years (Pendleton et al., 2012). The restoration of salt marshes, seagrasses and mangroves, therefore, is of considerable importance in management of carbon budgets for coastal nations the world over (Macreadie et al., 2017). Potentially, better coastal management with carbon budgeting could fund coastal ecosystem restoration projects (Howard et al., 2014) especially in developing countries.

Apart from the sequestration of carbon there are also significant benefits by sediments sequestering nutrients introduced into coastal ecosystems from terrestrial and agricultural runoff.

The capture of the nutrients within accreted sediments has the potential to reduce eutrophication in the coastal zone. Cole and Moksnes (2016) estimated multiple ecosystem services provided by seagrass meadows in Sweden, notably carbon and nitrogen uptake and the provision of habitat for fish. The authors placed monetary values on the flow of future benefits associated with commercial fishing, avoiding climate change damages and reducing eutrophication. Fish production, the most commonly valued ecosystem service in seagrasses, represented only 25% of the total ecosystem services value while nitrogen regulation constituted 46%. Most ecosystem services studies therefore will be underrating the value of salt marshes, seagrass meadows and mangrove forests.

Seagrasses are likely to support the development of juvenile fish of commercial value. Bertelli and Unsworth (2014) recorded nine commercial species including Plaice, Pollock and Herring in seagrass meadows. Similarly Blandon and Zu Ermgassen (2014) found juveniles of twelve commercial fish species in seagrass areas and restoration measures of seagrasses have a potential payback time of less than five years, based in fishery enhancements alone. In addition to commercial species it is likely that there are also monetary benefits from seagrass meadow restoration from the enhancement of species targeted by recreational fishing (Jackson et al., 2015).

In more recent times, consideration of multiple ecosystem services in the restoration of coastal wetlands is taking place (Barbier, 2013). A comprehensive assessment of a wide variety of ecosystem services was made as part of the planning for a large-scale salt marsh restoration project on the Steart Peninsula, Somerset, UK (Vieira da Silva et al., 2014). Valuations were made of benefits from agricultural enhancements and of one commercial species (sea bass), as well as carbon sequestration and nutrient recycling. Notably, though, the study also quantified the benefits to the local economy through recreation and tourism and the advantages of including multi-use pathways and observation points in the design plan. It was estimated that visitors to the site would increase to 33,000 per annum bringing benefits of the order of several hundreds of thousands of pounds to the local economy. Estimates of values arising from school trips and stimulating citizen science were also made. A net annual economic benefit from the restoration of the salt marsh was estimated to be up to one million pounds.

Despite the growing evidence of the benefits from the restoration of coastal zone ecosystems through nature-based solutions (NbS) these approaches are still not being considered on an equal footing as hard engineering methods (Pontee et al., 2016). There is limited consideration of alternatives by coastal managers and of valuing the multiple benefits that might be derived from

NbS. The lack of broader thinking by coastal zone managers is constraining the development of businesses with technological solutions and specialist knowledge in providing NbS solutions. Global funding mechanisms, such as the United Nations Framework Convention on Climate Change (UNFCCC) and the green Adaptation Fund (AF), as well as the United Nations International Strategy for Disaster Risk Reduction (UNISDR) and lending from the World Bank are staring to lead to a change (Narayan et al., 2016). As for scientific research, which is now combining skills to consider multiple ecosystem services in marine ecosystem restoration, a similar revolution in thinking is required in coastal zone managers and policy makers.

7.7. Technologies and Innovation and adaptations to the marine environment

7.7.1. Working underwater

In addition to the general difficulties involved in any restoration initiative, underwater restoration can be tremendously challenging compared to any terrestrial analogue. The underwater environment is complex, characterised by a fluid medium, hostile to humans, difficult to access and corrosive – all requiring special equipment. Inevitably a considerable number of marine restoration initiatives in shallow waters involve diving. Nevertheless, as humans enter into the marine environment their ability to undertake work becomes limited as no-decompression bottom time sharply decreases with depth. In shallow waters conventional SCUBA diving using compressed air limits individual dives to work periods of hours, reducing to less than half an hour at 30 m depth without having to spend extra time in decompression. Air diving work is very limited in time beyond 30 m without lengthy decompression either in-water or with a surface recompression chamber. Using mixed gases or closed-circuit rebreather systems, bottom times can be extended, and for technical diving depths can be increased up to typically 100 m, but again workable bottom time is very limited, equipment costs and risks increase, and a high level of trained/practised expertise is required (Bozanic, 2007; Sieber & Pyle, 2010). Overall, although divers have limited depths and bottom time, they have a high degree of dexterity, response, and stereovision, but limited carrying capacity (for materials and tools).

Underwater operations and research activities in great depths are mainly conducted with Remotely Operated Vehicles (ROVs) and Autonomous Underwater Vehicles (AUVs). ROVs have become widely available in the last decades, with differing levels of size, power, payload and manipulation capacity (Smith & Rumohr, 2013; Rogers et al., 2015). Although they need support/connection to surface vessels, they can reach considerable depths (thousands of metres) and stay operational on the seabed for as long at the surface vessel can remain on station. They can also operate in harsh underwater conditions (temperature, currents, etc.). AUVs only require a surface vessel for launch and recovery (although they could also be launched from the shore), however they have lower payloads and intervention capabilities than ROVs, being pre-programmed for missions with low possibilities for re-tasking during a mission and require batteries to operate (Jamieson et al., 2013). Some remote operations can be undertaken by landers to deliver and recover instruments (cameras, trackers, sensing modules) or materials to the seabed. Hybrid AUV/lander systems have also been developed in the form of bottom crawlers, tank-like tracked vehicles that are able to act autonomously on the seabed, moving around with sensor packages (video/photos, water measurements, sediment biogeochemistry) over extended periods of months (e.g. Sherman and Smith, 2009). While ROVs are very useful for deep-water work they are generally large machines requiring large surface vessels which are expensive to operate. However, there may be ways of reducing costs in restoration actions by using stand-by time in normal commercial ROV operations. Successful collaboration between science and industry has been demonstrated with scientists having access offshore to ROV systems during stand-by time, although not for actual restoration actions, in the SERPENT project (www.serpentproject.com).

Although deep-sea restoration is still in its infancy (Van Dover et al., 2014) restoration trials in deep water coral gardens are recently undertaken in the framework of FP7 MIDAS, BBVA ShelfReCover and H2020 MERCES projects. Restoration work involves the transplantation of corals obtained from fisheries bycatch onto landers and frames (Figure 60) which are then deployed to the seabed. A sonar reflector located on each lander enables their easy relocation with sonar. Thereafter, survival rates and physiological condition of coral fragments is periodically assessed by means of ROV/AUV video and photography (Carreiro-Silva et al., 2017; Gori et al., 2017).

Figure 60. Lander deployed coral stubs for growth. Photo by © Telmo Morato, IMAR-UAz, MIDAS Project.

7.7.2. New ideas and alternative approaches

As we enter a new era of innovative marine restoration, new techniques are continuously being tested for different habitat types and their structural key species. These include the use of mineral accretion technology for increasing coral survival and growth (Borell et al., 2009), methodologies for maintaining genetic diversity in restored coral and seagrass populations (e.g. Linden & Rinkevich, 2011; Ort et al., 2014), kelp-farming and other aquaculture technologies (e.g. Vasquez et al., 2014), the use of various biodegradable structures (e.g. potato waste) for creating oyster and mussel reefs (BESE-elements®, https://www.bese-elements.com/ and Dideren 2017) or even the use of (semio)chemicals, optimized biofilms, and modified coral stocks (van Oppen et al., 2017). Alternative approaches have been also suggested, such as the introduction of biological components generating natural recovery processes (e.g. Obolski et al., 2016) and the use of 'assisted evolution' options (e.g. selective breeding, assisted gene flow and the manipulation of the associated microbiome) aiming to enhance environmental stress and the success of restoration initiatives (van Oppen et al., 2017).

Novel techniques, innovative ideas and serendipitous by-products arising from various research fields and directions, ranging from land-based agriculture to marine-based industrial applications, molecular genetics and biotechnology could greatly facilitate underwater restoration initiatives in the future. For example, similarly to terrestrial agriculture, mechanization has already influenced marine restoration. Mechanical harvesters and seed planting machines (Figure 61) which reduce seed-processing labour have already been tested for large-scale eelgrass restoration (Traber et al., 2003; Orth et al., 2009; Marion & Orth, 2010; NOVAGRASS project http://www.novagrass.dk/en/ and Kristensen & Flindt, 2017).

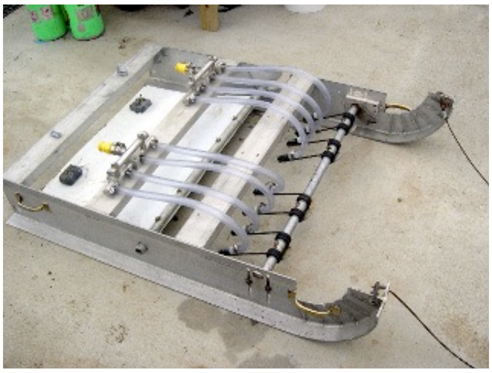



Figure 61. Automated seed harvester, NOVAGRASS project (left) (Photo by Flemming Gertz) and underwater mechanical seed planter (right) developed for eelgrass (*Zostera marina*) transplantation (from Orth et al., 2007).

Large area coverage restoration has been attempted in terrestrial ecosystems through aerial bombing. This has taken the form of either seeding using seed balls, or shoots in biodegradable penetrators released from aircraft over the intended habitats (grassland or planned forestland). The idea originated in the 1930's and there are reports of applications in Japan, South Africa, Thailand and the USA, generally in homogenous areas that are hard to access (Wikipedia: "Aerial Seeding", "Seed Bombing", https://www.theguardian.com/uk/1999/sep/02/paulbrown). There is however, little information about these activities/methodologies especially concerning their success. In the marine environment, this approach could be used, but there are more technical difficulties concerning delivery of a seedling/shoot or coral fragment to the seabed in the correct orientation, dispersal in the water column and therefore arrival in conditions conducive to survival and growth. In deep waters new technologies such as AUVs could theoretically deliver a payload (seeds, shoots, fragments or larvae) close to the seabed to target areas, negating water column dispersal. This would need much experimentation as survival conditions for settling fauna are sometimes quite specific, for example, deep water corals that might have preferences on parts of morphological features (e.g. coral/carbonate mounds, outcrops and boulders) with specific orientation related to current or food supply, or next to other settled species (e.g. De Mol et al., 2002; Vertino et al., 2010).

Technological advances, new infrastructures and novel applications developed for coastal and marine engineering, marine-based industrial activities (e.g. deep-sea mining, hydrocarbon drilling, dredging, energy generation) and transport could also be potentially used in marine restoration. Currently, the ever-increasing development of marine-based activities has led to a parallel demand for compensating ‘green’ solutions. For instance, nowadays environmental engineering options are being applied in coastal defence works (e.g. Firth et al., 2014 and Green Engineering Workgroup of the World Harbour Project) while the continuous development of deep-sea mining activities has debated the urgent need for adopting precautionary environmental management and restoration measures (Van Dover et al., 2014; Durden et al., 2017). An interesting approach that could possibly facilitate conservation and restoration of deep-sea benthos involves the on-going transformation of several offshore oil rigs due for decommissioning, into artificial reef complexes on unprecedentedly large scales, under the “rigs-to-reefs” program (Macreadie et al., 2011).

Although effectiveness of the above techniques may vary between different sites and case studies, the developed technology and approaches can be potentially modified for other habitat types, species and depths. Nevertheless, multi-disciplinary approaches and better involvement of marine-based industries are still needed.

7.7.3. Public engagement through citizen science approaches

Volunteer engagement through citizen science initiatives is greatly acknowledged as a significant way towards up-scaling data collection to higher spatial scales, especially considering marine biodiversity inventories and monitoring (Thiel et al., 2014; Edgar et al., 2016; Garcia-Soto et al., 2017). Several citizen science projects materialized in the European Seas within the last years, targeting mostly vulnerable/protected species and habitats and tracking litter (e.g. Seawatchers, http://www.observadorsdelmar.cat/; CIGESMED for divers, http://cs.cigesmed.eu/, Gerovasileiou et al., 2016, Marine LitterWatch mobile application https://www.eea.europa.eu/themes/water/europes-seas-and-coasts/marine-litterwatch).

Reporting of threats by citizen divers at an early stage of introduction has proved to be critical for the success of restoration measures, through the eradication of invasive species (see section 6.4.2. and references therein). The development of user-friendly data submission or early-warning systems could facilitate rapid and cost-effective restoration.

In addition to simple data reporting, dissemination platforms and social network-volunteer schemes could be very important tools in restoration. There are several documented cases where volunteers have been massively involved in restoration activities, such as the removal of invasive species or native overgrazing/coral-preying echinoderms (see sections 6.4.2. and 3.4.3., respectively), the construction and monitoring of oyster reefs (Hadley et al., 2010) and the transplantation of coral fragments (Forrester et al., 2014; Hesley et al., 2017). Local fishing communities have also contributed to restoration initiatives though active involvement at different stages of mangrove reforestation (Rao, 2009) or by providing various coral species caught as by-catch for restoration purposes (e.g. Bilan et al., 2017; Gori et al., 2017). So far, volunteer engagement in the above initiatives has yielded positive restoration outcomes. Within MERCES, volunteers from dive clubs, for example in Spain and Croatia have made possible the transplantation and translocation respectively of hundreds of specimens of coral fragments and noble pen shells.

Recruitment of volunteers in marine restoration activities, likewise any citizen science project, can be achieved through campaigns organized in social media platforms (e.g. Facebook, Twitter, Instagram), and through local public seminars and workshops in collaboration with local stakeholders (e.g. diving clubs, fishermen, local authorities, NGOs). A minimum training and field trials with volunteers is often needed to safeguard better restoration outcomes (Hesley et al., 2017). Development of educational material, thematic websites and smartphone applications provide valuable training tools, which could also enhance environmental awareness (Garcia-Soto et al., 2017).

In certain cases, the involvement of volunteers in restoration activities has been even introduced as a “commercial product” in the diving/eco-tourism industry. In Okinawa (Japan), coral transplantation is financially supported on a voluntary basis by tourist divers who pay to participate in special diving tours organized by local operators. These tours cost 20% more expensive than similar ones without transplantation, while divers also pay extra for the artificially produced coral transplants, yet deriving utility from the cost incurred in term of personal enjoyment (Okubo & Onuma, 2015).

Conclusively, the engagement of volunteers in marine restoration activities though specialized citizen science approaches provides a unique opportunity to maximize restoration outcomes with a parallel (a) decrease of cost, (b) up-scaling of spatial cover, and (c) enhancement of environmental awareness and of natural capital value while linking local societies with the marine research sector. Such opportunities should definitely be considered in future restoration initiatives.

7.8. Restoration Feasibility

A clear set of decision parameters that would determine whether or not to take restorative actions on a degraded ecosystem has been developed by Van Dover et al. (2014). The parameters fall into three major categories (with various sub-categories). Socio-economic parameters reflect factors that are likely to benefit or affect society/people and include; likelihood of ecosystem benefits, governance, cost, societal pressure, financial incentives and wider socio-economic impacts. Ecological parameters reflect the ecological aspects of the proposed restoration actions and include; ecological vulnerability, likelihood of wider ecological benefits, natural recovery and large relative ecological impact. Technological parameters deal with the real-world issues in restoration actions and the overall likelihood of a successful outcome and include; likelihood of success, likelihood of technical feasibility, likelihood of technological advancement. In the following sections we investigate some of these, as well as other issues that shape restoration work, including ecological features, temporal and spatial scales and the value of restoration.

7.8.1. Ecological Features Effecting Restoration

A recent review (Bekkby et al., 2017) focusing on critical ecosystem features such as dynamics (e.g. growth rate and longevity), connectivity (e.g. dispersal and gene flow), spatial distribution, vulnerability/fragility (e.g. to physical pressures), structural complexity (e.g. 3D complexity) and diversity (at the species, functional, genetic and community diversity levels) produced a traffic light approach/summary to the restoration potential of the key habitat types examined within MERCES. By looking at parameters relevant to restoration success and features influencing chances for habitat recovery, shallow-soft, shallow-hard and deep-sea habitats were assigned to a rough ranking of restoration potential mirroring known and anticipated challenges. A common challenge for suggesting restoration practices and guidelines is the lack of comprehensive knowledge on the link between a pressure and a change in ecological state or condition. Although a lot of progress is being made in defining harm and serious harm, both scientifically and legally, in relation to deep-sea mining exploration, we are still not in a position to link the extent or proportion of habitats affected with the good environmental status of our habitats and a common understanding and interpretation on how to assess degradation (and thresholds of change) across habitats is lacking (Levin et al., 2016; Bekkby et al., 2017; Smith et al., 2017).

Staying with the more biological challenges, according to the Bekkby et al. (2017) scoring (Figure 62), deep-sea coral communities (and other deep-sea bottom communities) are likely to be the most challenging when it comes to achieving acceptable restoration goals. In part this is due to the extremely slow growth rates, long lifespans (thus likely late age of first maturity), low fecundity, high vulnerability to human impacts of key indicator species and the limited information on larvae biology, dispersal and population connectivity. Shelf subtidal coralligenous communities, with slow growth rates, low connectivity, high vulnerability, fragility to human activities and extreme structural complexity, are also challenging to restore. The restoration potential of seagrass meadows depends on the target species, and as with other communities (e.g. sponge communities), facilitation by other species, and the physical setting (from presence of threats to sediment and water parameters) in the location of the restoration activity (see Section 4). Shallow-water, hard-bottom macroalgal forests are classified as "medium" in terms of their likelihood of achieving restoration goals, owing to their accessibility (for field work) and for some species, to their higher connectivity levels and growth rates but medium to high vulnerability to pressures. Of the MERCES case study habitats selected, shallow hard bottom kelp forests have the highest chances of restoration success due to their fast growth rates, high levels of connectivity and low levels of vulnerability. Having noted that, even for kelp habitats and despite their shallow depth and high accessibility, numerous challenges remain posing a serious threat to up-scaling their restoration. These challenges vary in difficulty and include, among others, widespread eutrophication effects and expansive highly degraded areas with urchin barrens. Controlling the grazing pressure of sea urchins on the kelps and that of the crabs on the urchins is not a trivial issue and, for restoration and transplantation to be successful, will require a change of management both creating and controlling extracting human activities.

In conclusion, mitigation of pressures, spatio-temporal regulation of activities, and compensation are still considered the most cost-effective strategies for managing present trajectories of change. Although prevention is better than cure (and greater efforts are needed to conserve existing biodiversity to ensure continuation of ecosystem services) even minor rehabilitation of degraded ecosystems can put back some biodiversity and key services (Geist & Hawkins, 2016). Ecological restoration approaches for most marine habitats should consider the combination of the three restoration approaches (natural regeneration, assisted regeneration and reconstruction).

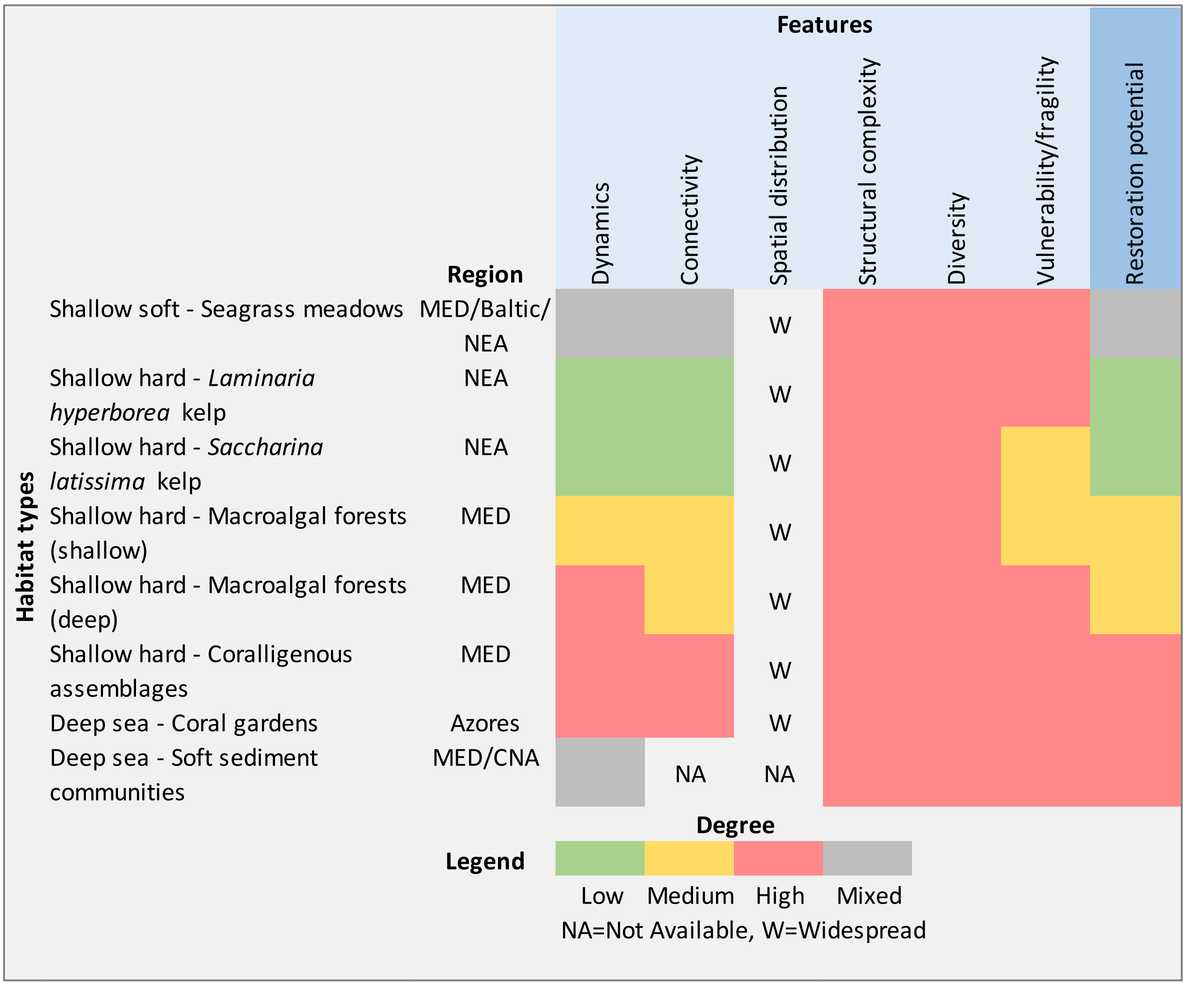


Figure 62. Restoration potential from Bekkby et al. (2017). Green shading relates to a feature that may facilitates achieving the restoration goals, orange shading represents medium and red shading denotes that the feature makes restoration relatively difficult. Grey shading represents conditions where different factors (e.g. species or location) may lead to different degrees of restoration success. NA indicates that there is scarce or no available information. NA indicates that there is scarce or not available information concerning connectivity and spatial distribution (for deep-seas sediment communities). Baltic: Baltic Sea; NEA: North-East Atlantic Ocean; CNA: Central-Northern Atlantic; MED: Mediterranean Sea.

7.8.2. Timescales in Restoration

As noted in Section 4.3, timescales of recovery in the marine environment may be extremely long compared to terrestrial ecosystems, particularly in deep waters. In terms of species recovery, deep-sea coral, slow-growing structural species, may require centennial or multi-centennial timescales for recovery to fully functioning reef structures (Roberts et al., 2006). Published growth values for solitary black coral *Leiopathes* sp. are 0.008–0.022 millimetres/year (Prouty et al., 2011), and the octocoral *Paramuricea* spp. 0.071–0.205 millimetres/year, although reef forming white corals *Lophelia pertusa* have published growth rates of 2.6–32 millimetres/year (Freiwald and Henrich, 1997; Mikkelsen et al., 1982; Mortensen and Rapp, 1998, Larcom et al., 2014) and *Madrepora oculata* 3−18 millimetres/year (reported in Roberts et al., 2009). Over even longer timescales, deep-sea manganese nodules, the focus of deep-sea mining, are parts of the substrate fabric covering large areas on which some bottom fauna are dependent and which may take millennial timescales to reform (Morgan, 2000). In light of these timescales, the question that immediately arises is how a restoration programme can be established to move towards full ecosystem restoration, when it is well beyond a foreseeable management programme/timescale. The way forward, if restoration is judged necessary, is to at the very least, to get on the trajectory of recovery by initiating a start, whether it is just to cease degrading activities, legally protect areas, or to move further with interventions to provide constructions or transplant species. Of course these are the extremes of timescales in marine restoration and in typical shallower water European marine ecosystems; timescales that may be decadal at maximum and may be much more manageable. Orth et al. (2012) reported that *Zostera marina* large-scale bed recovery took years rather than decades and for a variety of seagrasses, the timescale for bed recovery after disturbance is frequently around 3 years (reported in Bell et al., 2007). This time frame of less than decadal scales is noted also for the macroalgae kelps and *Cystoseira* reported in Section 4.2**.** Plants have reasonably high growth rates and slower growing invertebrates such as *Pinna nobilis* and shallower water corals also reported in Section 4.2. have recovery rates in decades. These shorter term recoveries imply that a restoration and management programme are certainly feasible within natural funding lifecycles.

7.8.3. Spatial Scales in Restoration

A great deal of the work undertaken to date on restoration has been experimental or on small scales and there remains the question of how to up-scale these works to a scale that can provide environmental and ecological benefits (Macdonald et al., 2016). Degraded ecosystems should be restored and at a high level, programmes need to be put in place to reach the restoration targets for degraded ecosystems set by international Directives and Conventions (see Section 1.2). Within Europe there is a lack of precise and accurate ground-truthed habitat mapping (Bekkby et al., 2017), with even lesser information concerning the geographical extent of habitat degradation (Smith et al., 2017), with basic inconsistencies in defining practically what level of density of a key habitat species may define that habitat (e.g what density of seagrass constitutes a seagrass meadow), or what level of degradation defines a habitat as degraded. These are key in being able to understand what the targets mean in terms of site location or extent for actions towards a 15% restoration figure. The scales for these restoration targets may be huge, for example bottom trawling activities affect sedimentary habitats on the order of hundreds of thousands of square kilometres in European waters and the entire area could be implied to be degraded. Eigaard et al. (2017) estimated the trawling footprint in waters less than 200 m depth, to be between 28 and 99% in the EU management areas of the Northeastern Atlantic and between 57 and 86% in the EU Mediterranean Sea (they also estimated, that 40% of the macrophyte-dominated sediments and biogenic habitats in the EU Mediterranean are trawled). Considering the EU Biodiversity Strategy target of 15% restoration of degraded ecosystems, this could require restoration scales of tens of thousands of square kilometres covering a variety of sedimentary habitats. Most key habitats considered in MERCES are patchy and on small scales in any particular area. The most extensive of these could be considered to be seagrass habitats, which are intermittent along the coasts. In the Mediterranean, *Posidonia oceanica* meadows cover just over 12,000 square kilometres with an estimated regression of 34% in the last 50 years (Telesca et al., 2015). A conservative area for restoring meadows under current targets would therefore be approximately 500 square kilometres, spread over 12,000 kilometres of coastline where *P. oceanica* occurs. Typically this scale of restoration would involve a high level of planning, coordination, financing and consensus building (Aronson et al., 2017), as well as commitment, and most likely the application of new technologies for effective area coverage (see Section 7.8.2.). Small pockets and mosaics of habitats are more typical for restoration. They may consist of a grouping of individuals or colonies, for example the bivalve *Pinna nobilis* or deep-sea corals over metre to hundred square metre patches, separated from other patches over various distances. In order to scale up, it may not be necessary to restore in a continuous way, but to set up new patches that may grow and/or seed further additional patches in the vicinity of the restoration area. The simplest approach to wide areas is to apply some form of area protection, such as MPAs (Section 2), but this should be just the start in the application of a family of restorative activities to reach a true ecological restoration.

7.8.4. The Value of Restoration

Ecological restoration has received increasing attention in the new millennium and has been recognised as an integral part of the tools society has in order to tackle major environmental concerns, such as the widespread ecosystem degradation, desertification, anthropogenic climate change, and the unprecedented loss of biodiversity due to human activities and pressures (Blignaut et al., 2014b). Nowadays, it is widely accepted that for successful restoration projects we need to set clear and achievable goals and draw on ecological, economic, technological and social knowledge and constraints (Miller and Hobbs, 2007; Adame et al., 2014; Van Dover et al., 2014; McDonald et al., 2016). Information on costs is particularly important when planning ecological restoration as it helps decide on issues such as whether or not to restore in a given area, which projects to implement, and which methods or techniques should be preferred (Iftekhar et al., 2017). Yet decisions should not be based on economic criteria alone, as ultimately ecological restoration is motivated by ecological, ethical, social, and cultural values along with economic ones. Restoration ecologists and stakeholders need to consider both economic and ecological costs, the latter in the sense of environmental impacts with possible effects on human well-being and prosper in the long run, and view restoration costs in relation to both existing and future habitat loss with associated loss of benefits.

Achieving this goal is dependent on our ability to define, assess, and valuate ecosystem services, as well as the benefits they can provide to people, and to highlight how these and their restoration will benefit society over time. To this end, economic valuation of ecosystem service benefits – that in most cases, though not fully recognised and evaluated, are multiple – arises as an essential but rather overlooked discipline in restoration ecology that could help in forming scenario plans and weigh their economic trade-offs (Börger et al., 2014). This should be coupled with cost-benefit analysis, an established analytical tool in environmental sciences, the application of which is still challenging within the marine environment (Börger et al., 2016).

Apart from the above issues on the link between environmental science and economic valuation and the possible limitations in valuation metrics, several other aspects should be considered when considering the value of restoration projects. One important question to address is "have we already destroyed too much or can we allow more destruction?" Taking the precautionary approach, especially in cases of uncertainty in current or future benefits of a degraded environment, restoration projects, though often costly, may be in fact invaluable. A recent analysis by de Groot et al. (2013) of over 200 studies concludes that the majority of the restoration projects analyzed provided net benefits and should be considered not only as profitable but also as high-yielding investments.

A further consideration relates to the targets and goals of the restoration project. Restoration costs may rise according to different scenarios and therefore decision-makers must carefully consider different levels of targets and goals in order to achieve an optimal ratio between restoration costs and economic benefits from ecosystem service restoration. Tucker et al. (2013) showed that restoration costs vary widely among EU countries and depend on the condition of the targeted ecosystem but also on the degree of restoration undertaken, and they further pointed out that minimising costs may not always prove to be cost-efficient since the economic benefits of ecosystem restoration can be greater than the costs, up to a certain point. In addition, integrating various motivations (e.g. from water quality improvement to offsetting) in the restoration planning might deliver multiple benefits (Hagger et al., 2017). The development of modelling applications that will investigate the potential increase in restoration costs in relation to an increase in restoration targets/goals, a change in geographical location, or even a shift in restoration techniques will be helpful in this direction.

We should also bear in mind that restoration costs may vary considerably in both space and time (Wätzold & Schwerdtner, 2005), as a result of differences associated with salaries, costs of materials, equipment or acquisition of land, or the phase of the restoration project. These potential sources of cost increase could be narrowed by certain actions in order to allow for benefits to outweigh economic costs. These actions may regard, for example, the involvement of local communities and volunteers when implementing the restoration project, or a focus of research on innovative tools that may limit labour costs or render expensive techniques redundant.

A major challenge with restoration projects is to cover the costs, which are usually considered as expenses that should be covered by governmental funds, companies that cause environmental degradation with their activities, or through biobanking and biodiversity initiatives (Bullock et al., 2011). Recently, an ecosystem service perspective approach has produced Payment for Ecosystem Services (PES) schemes, which have been developed to compensate individuals (e.g. farmers) or communities for actions that maintain or increase the provision of services (Bullock et al., 2011 and references therein). Such schemes require buyers and sellers of a service facilitated by a functioning institutional arrangement. PES schemes create restoration opportunities and have also been established for financing restoration. Nevertheless, they still need further development to ensure that restoration targets and the stakeholder needs are met (Bullock et al., 2011). Barbier et al (2014) moot the potential for an international finance facility, which would mobilize resources for deep-sea restoration from international capital markets by issuing long-term bonds to be repaid by donor countries over 20–30 years. When restoration is not possible, or is prohibitively expensive, compensation through other investments of social and environmental interest may be a more beneficial use of liability funds (Tinch & Van Den Hove, 2016).

The currently weak link between environmental science and valuation of ecosystem service benefits makes decision-making in ecological restoration even more complex. Cost information is an important factor in decisions regarding restoration choices, but many other economic aspects may contribute to improving the effectiveness and efficiency of restoration programs (Yin et al., 2013). It is the duty of restoration ecologists to highlight the importance of restoration for increasing natural capital and its significance for the flow of ecosystem services and benefits to society (Blignaut et al., 2013) without promising more than we can deliver (Aronson & Alexander 2013a). This will be done on the grounds of convincing evidence for increased benefits and through continuing efforts to advance methods and techniques that will make restoration projects affordable. As Blignaut et al. (2014b) argue “since the publication of the Millennium Ecosystem Assessment, an appropriate framework already exists to consider restoration not as a cost item on a project and/or government budget to undo wrongs of the past, but rather as a value-generating option”.

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