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Ecosystem service benefits and costs of deep-sea ecosystem restoration

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ABSTRACT

Deep-sea ecosystems are facing degradation which could have severe consequences for biodiversity and the livelihoods of coastal populations. Ecosystem restoration as a natural based solution has been regarded as a useful means to recover ecosystems. The study provides a social cost-benefit analysis for a proposed project to restore the Dohrn Canyon cold water corals and the deep-sea ecosystem in the Bay of Naples, Italy. By incorporating ecosystem service benefits and uncertainties related to a complex natural-technological-social system surrounding restoration activities, the study demonstrated how to evaluate large-scale ecosystem restoration activities. The results indicate that an ecosystem restoration project can be economic (in terms of welfare improvement) even if the restoration costs are high. Our study shows the uncertainty associated with restoration success rate significantly affects the probability distribution of the expected net present values. Identifying and controlling the underlying factors to improve the restoration successful rate is thus crucial.

1. Introduction

Deep-sea ecosystems are among the most extensive habitats in the world and provide important ecosystem goods and services (Thurber et al., 2014) but face pressures from both climate change and other anthropogenic factors such as trawling, mining, oil and gas exploitation, and marine litter that are seriously threatening their integrity (e.g. Dailianis et al., 2018; Da Ros et al., 2019; Taviani et al., 2019; Gerovasileiou et al., 2019; Smale et al., 2019; Bekkby et al., 2020, Danovaro et al., 2020). Marine ecosystem degradation could have severe consequences on biodiversity and the livelihoods of coastal populations. Reversing this trend requires the identification of the ecological variables critical for conservation and restoration in order to develop achievable marine planning objectives (Danovaro et al., 2020).

On March 1, 2019, the UN General Assembly declared 2021–2030 the "UN Decade on Ecosystem Restoration.". It calls for actions to accelerate the global restoration of degraded ecosystems (Waltham et al., 2020). The UN Decade on Ecosystem Restoration coincides with the UN Decade of Ocean Science for Sustainable Development. Both decades aim to turn the tide on the loss of ecosystems and provide society with a more sustainable future (Ryabinin et al., 2019; Waltham et al., 2020; Aronson et al., 2020). Only with healthy ecosystems can we enhance people's livelihoods and combat climate change impacts and increase resilience. Restoration is also a priority of the European Green Deal and Nature-based solutions, such as restored coral reefs functioning as important habitats for marine species and as natural breakwaters to protect vulnerable shoreline properties, are seen as important means to deal with climate change induced hazards (European Commission, 2020). Marine restoration experiments have been carried out across the European seas, testing different approaches and solutions but deep-sea ecosystem restoration is still largely at an experimental stage (Barbier et al., 2014; Boch et al., 2019; Danovaro et al., 2021). Restoration can improve marine ecosystems (or natural capital assets) (Gordon et al., 2020), their associated ecosystem services and their values (De Groot et al., 2013; Pendleton et al., 2010; Aronson et al., 2020) but can also involve a high cost especially for deep-sea habitats where expensive technologies are often required to achieve the restoration goals (e.g. Da Ros et al., 2019; Bayraktarov et al., 2019, 2016).

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Cost benefit analysis (CBA) is an important tool to support decision-making and is a common means of appraising investments in various marine industries such as oil, gas, and offshore windfarms. It is also common to carry out a social cost benefit analysis (SCBA) where the social consequences are considered (Boardman et al., 2018; Atkinson and Mourato, 2008). SCBA is a method aimed at ensuring the most economic use of resources by selecting investment projects or prioritizing policy measures that achieve the highest benefit cost ratios (Boardman, 2018). SCBA has been widely applied in environmental conservation (World Bank, 2005) and is regarded as a useful tool in evaluating the societal impacts resulting from changes in natural capital assets (Pearce et al., 2006). If the total benefits exceed the total costs of restoration, it can be considered that the restoration has net economic benefits to society (Pendleton et al., 2010).

SCBAs have been used for various coastal and marine governance problems, for example for multi-usage of the sea (Chen et al., 2021; Koundouri et al., 2017), for deep-sea mining (e.g. Wakefield and Myers, 2018; EU, 2016) and for marine protected areas (e.g. OECD, 2017; Pascal et al., 2018). Prioritisation through SCBA of restoration projects can also assist policy makers to make informed decisions when the demand for ecosystem restoration is high and resources are scarce (Blignaut et al., 2014). Consideration of the wider economic impact of restoration activity through ecosystem service accounting is also starting to become a priority internationally (UNCEEA, 2021; Chen et al., 2020).

Using data from published studies, De Groot et al. (2013) estimate the costs and benefits for ecosystem services bundles for restoration of various habitats including coral reefs and coastal ecosystems. Costs for coral reefs and costal ecosystem restorations was found to exceed the benefits in most of the scenarios. Stewart-Sinclair et al. (2020) conducted a global spatial cost-benefit analysis to determine whether the monetary value provided by ecosystem services of four marine habitats (coral reef, mangrove, saltmarsh and seagrass) exceeds the cost of restoration. They use costs in the published restoration case studies from Bayraktarov et al. (2020) and an adjusted-value-transfer method to assign benefit values to these case studies. Benefits were found to outweigh the costs for restoration for all the four ecosystems. Gregr et al. (2020) studied the cascading social-ecological costs and benefits of kelp forest recovery following introduction of predator species sea otters that help control the sea urchins that destroy the kelp forests. They found the values of tourism, fisheries and carbon capture from recovered kelp forest outweighed the harvesting losses of shellfish which sea otters also prey on. Other marine related CBA examples can also be found in the review by Saunders et al. (2020) about the bright spots in coastal marine ecosystem restoration.

Research on social cost benefit analysis for deep sea ecosystem restoration is however limited. This is partly because most of these projects are pilot and small-scale. Also, because of the uncertainty associated with restoration success rates, it is difficult to provide estimates of the ecosystem services and their values delivered following the final scaled-up implementation. It is still important however to put the SCBA framework in place even for pilot and small-scale projects so as to facilitate learning, the identification of data gaps and to inform monitoring and data collection. Folkersen et al. (2018a) carried out a systematic review and meta-analysis for the valuation of deep sea ecosystem services. Only 45 studies were identified. A few studies have estimated the value of restoring deep-sea habitats using non-market valuation methods (e.g. Jobstvogt et al., 2014; Aanesen et al., 2015; Aanesen and Armstrong, 2019; Xuan et al., 2021). O'Connor et al. (2020a, 2020b) estimated the willingness to pay for restoring the deep-sea ecosystem of the Dohrn Canyon at around 35 EUR per Italian citizen. A recent study by Hynes et al. (2021a, 2021b) that repeated a choice experiment examining deep-sea ecosystem management found relatively stable environmental preferences and willingness to pay both pre and post the peak of the first wave of the Covid-19 pandemic. The authors conclude that this suggests strong support in terms of societal priorities regarding the conservation of deep-sea environments far removed from direct use or experience.

Elsewhere, Bayraktarov et al. (2016) estimated the average cost of the restoration of five types of marine habitats including coral reefs, seagrass, mangroves, saltmarshes and oyster reefs using data collected from a comprehensive literature review. The average reported costs were around 1.2 million EUR per hectare (2010 value). A review of the cost of coral reef restoration worldwide indicated that the average cost depends strongly on the restoration method (Bayraktarov et al., 2019). Artificial reefs were found to be among the most expensive methods, with costs reaching 107 million EUR per hectare (2010 values). Reguero et al. (2018) compares the cost effectiveness of nature-based green, grey and policy adaptation measures for current and future risks under climate change. The investment in the green adaptation was found to be cost effective. Cost for deep-sea restoration is likely to be at least two to three times more expensive than coastal restoration (Barbier et al., 2014). Costs for restoring the Darwin Mounds stony cold water corals situated off the North West coast of Scotland and the Solwara 1 hydrothermal vent in the Bismark Sea were estimated at 6000 million EUR per hectare and 574,928 million EUR per hectare respectively (Van Dover et al., 2014). It also needs to be considered that costs might not be the most appropriate comparator if the effectiveness of restoration varies.

Surprisingly few studies have quantitatively compared potential ecosystem benefits with the restoration costs for deep-sea ecosystems. Van Dover et al. (2014) and Da Ros et al. (2019) pointed to the remoteness of the deep sea, the lack of public awareness, the uncertainty of outcomes, and the potential for high costs and long recovery periods, especially for passive restoration, as factors that have limited the economic assessment of deep-sea restoration. Cooper et al. (2013) explored whether the benefits of physical seabed restoration after dredging could justify the costs. When costs for various restoration measures are quantified, the benefits of various ecosystem services are largely unknown. In practice, decisions on whether or not to carry out restoration projects are often made focusing on either the costs or the ecological benefits. Although most deep-sea restoration projects are still at their experimental stage, SCBA analyses may assist managers in deciding, which projects should be scaled up in the future.

There are usually several major uncertainties related to restoration activities, including the restoration costs, the societal values, and the restoration success rate (e.g. Van Dover et al., 2014; Da Ros, 2019; Bayraktarov et al., 2019; Ounanian et al., 2018). Ounanian et al. (2018) mentioned three kinds of uncertainties related to marine ecosystem restoration, namely: incomplete knowledge, unpredictability, and ambiguity. Incomplete knowledge results from lack of data and limitations on data accessibility and quality. Knowledge gaps are common concerning the structure, function, biodiversity and interactions in marine ecosystems and these may increase the research costs. Unpredictability is a common feature arising from the complex, dynamic and non-linear behaviour of natural, technical, and social systems, and should be considered in the governance of marine restoration. Ambiguity could stem from different perceptions on the same phenomenon by different actors. Elsewhere, in a review of the literature Deely et al. (2020) identified over fifty barriers that could hinder restoration efforts being used as nature-based solutions. Therefore, SCBA for marine restoration projects needs to consider all the relevant uncertainties in what complex natural-technical-social systems are.

Uncertainties have been dealt with by various methods in the environmental literature including sensitivity analysis (e.g. Stewart-Sinclair et al., 2020), Monte Carlo simulation (Wallhead et al., 2018; Benke et al., 2007), Bayesian Belief Network models (Phan et al., 2016; Landuyt et al., 2013) and Neural Network-Based Analysis (Li et al., 2020). The Markov model has been frequently used in the decision analysis of stochastic economic systems (Stokey et al., 1989), adaptive resource management (Mranda and Fackler, 2002), and economic evaluation of healthcare interventions (Markov Model, 2016). As with the treatment of sick patients, ecosystem restoration provides interventions for an ecosystem falling 'ill'. Markov models use discrete states to represent all possible consequences of the restoration intervention. These states are mutually exclusive and exhaustive during any given period. Markov models assume that transition from the present state depend only on the present state, but not the past. Therefore, a simple Markov model suffices when only one restoration cycle is considered, while multiple restoration cycles may require, more complex models with 'memory' of cumulative impacts, or else an expanded set of Markov states.

A key challenge in the application of Markov models is how to assess the transition probabilities between states and their uncertainties. These could be estimated empirically using historical data for comparable systems subjected to comparable restoration treatments, but such data are rarely available for deep sea marine ecosystem restoration. An alternative approach is to rely on expert opinion. The expert opinion has been used widely as inputs for models to handle problems associated with high levels of uncertainties of human activities on the environment, ecosystems and their services. Expert knowledge is regarded as helpful to capture uncertainty inherent in the complex socio-ecological system (Phan et al., 2016; Barton et al., 2012). For example, expert opinions have been used in the Bayesian Belief Networks to construct the model and to evaluate the uncertain consequences of potential actions in water resource management (Phan et al., 2016 and reference therein) and in ecosystem services modelling (Landuyt et al., 2013). Expert knowledge is useful to understand different stages of deep-sea ecosystem recovery at particular sites and how adaptive measures can be applied (Morato et al., 2018).

As the deep-sea ecosystems and their responses to restoration interventions are only known by a few experts who have carried out restoration experiments, expert opinions are considered in this study to be the best available knowledge to estimate the transition probabilities between states. This study therefore uses a Monte Carlo simulation approach to account for the 'total' uncertainties (Benke et al., 2007) in expert-based Markov transition probabilities, as well as in the estimation for costs, benefits, and interest rates. By combining the Markov and Monte Carlo methods, our study aims to address all three types of uncertainties described by Ounanian et al. (2018).

The study contributes to the literature by 1) providing an example to demonstrate how SCBA can be applied to evaluate large-scale ecosystem restoration activities and implementation, 2) taking into consideration the uncertainties that will invariably surround complex restoration situations including not only uncertainties associated with costs and restoration success rate but also the uncertainties associated with benefits, 3) developing a Markov model to tackle the uncertainties associated with restoration success rate when historical data for comparable systems subjected to similar restoration treatments are unavailable, 4) showing that the social benefits can outweigh the restoration costs in the case of large scale deep sea ecosystem restoration.

This study aims to provide a social cost-benefit analysis for a proposed project to restore the Dohrn Canyon deep-sea ecosystem, including a comprehensive treatment of uncertainty. In what follows section 2 presents the background of the restoration project in the Gulf of Naples in Italy. Section 3 describes the SCBA methods and the methods used to tackle the uncertainties. Section 4 describes the data and section 5 presents the results. Section 6 outlines the results of a sensitivity analysis while section 7 provides a discussion around the findings and offers some conclusions.

2. The Dohrn Canyon in the Gulf of Naples

The Dohrn Canyon is approximately 12 nautical miles offshore from the Naples metropolitan area (Fig. 1). It is the main canyon crossing the Gulf of Naples (Tyrrhenian Sea), eroding the slope down to 1000 m-depth and including a hotspot of deep-sea benthic biodiversity at approximately 400 m depth. The hard bottoms are characterized by a high abundance of charismatic species, such as the habitat forming cold-water corals (CWC) *Madrepora oculata*, *Lophelia pertusa*, and *Desmophyllum dianthus* in association with the large size bivalves *Acesta excavata* and *Neopycnodonte zibrowii* (Taviani et al., 2019). The cold-water corals are key ecosystem engineers in deep-sea habitats (Taviani et al., 2019). Over many decades, the canyon has been subjected to high-intensity human uses linked to coastal-zone pressures such as illegal dumping and fishery malpractices, as well as bottom trawling in shallower parts. This has resulted in environmental degradation and large amounts of litter including lost fishing gears and plastic waste along the canyon axis and walls (Taviani et al., 2019). The Canyon does not meet the good environmental status for most Descriptors under the EU Marine Strategy Framework Directive.

2.1. Pilot project of restoration in the Dohrn Canyon

In order to restore the deep-sea ecosystem in the Dohrn Canyon, an approach combining both passive and active measures are proposed. That is, a deep-sea marine protected area (MPA) will be created first, and then artificial reefs will be used to restore the cold-water coral habitats and the related adjacent ecosystems in the canyon.

A new device has been tested to facilitate the restoration of deep-sea degraded habitats based on the use of artificial substrates. The Artificial Structures for Deep-sea species recruitment and Ecosystem Restoration (ASDER), which are designed with a triangular-based structure (1 m × 1 m × 1 m) provide support for anchoring 3 to 6 Autonomous Reef Monitoring Structures (ARMS). ARMS are cubic, long-term collecting structures designed to mimic the structural complexity of a three-dimensional habitat and to attract colonizing invertebrates. In

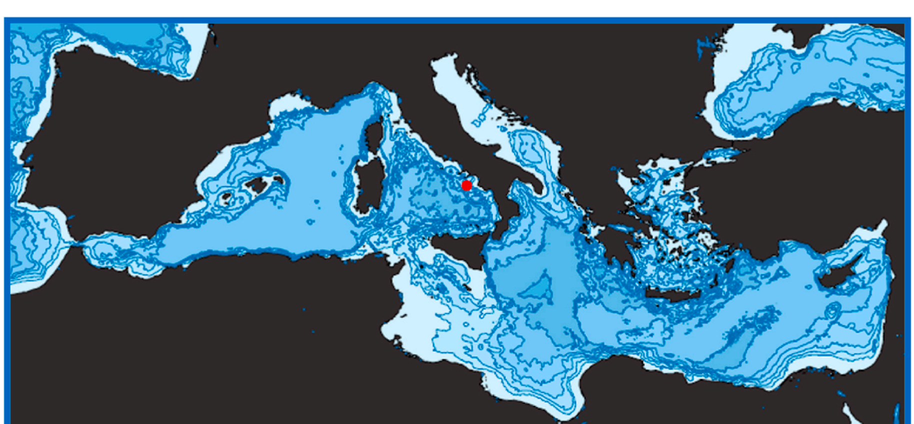

Fig. 1. Location of the dohrn canyon off the Gulf of Naples, (Tyrrhenian sea).

addition to the ARMS, the ASDER can host high resolution cameras to collect photos and short videos, a sensor to monitor environmental conditions and a hydrophone to sample the sound seascape. Once the lander is colonized by organisms (over 6–12 months) it can be transferred to degraded areas to promote recolonization of benthic organisms. ASDERs are potentially effective and low-cost devices to support active restoration initiatives in deep-sea ecosystems, which traditionally have very high costs.

Fig. 2 shows an example of the lander with 6 ARMS before deployment to the sea bottom and one ARMS unit two years after deployment. The figure shows an example from shallow water.

According to the scenarios proposed in the SCBA and suggested by the European Commission, a marine protected area (MPA) of 2 ha can be hypothetically identified at depth 500–600 m in the Dohrn Canyon. Commercial fishing trawling will be forbidden in the MPA. A buffer zone outside the MPA will also be created to achieve effective protection. Based on the literature related to the spatial distribution of cold-water corals inside the Dohrn Canyon (Taviani et al., 2019), the restoration of an area of approximately 2 ha will require 50 landers with 3 ARMS units in each. Fig. 3 shows the layout of the landers in the hypothetical MPA inside the canyon. The distribution and number of landers are estimated based on the seabed survey carried out in Taviani et al. (2019). While the technical coral habitat restoration with artificial reefs (ARMS) will cover only 2 ha, a conservative evaluation from H2020 MERCES project predicts the ecosystem benefits will cover the whole deep sea area of the Dohrn Canyon (an area between 52 and 104 km^2).

It is estimated that five cruises are needed for all restoration activities. The first and the second cruises will happen during the restoration phase. The first cruise is for the deployment of the 50 landers and lasts for about 8 days. After 12 months, once ARMS are colonized by organisms, the landers can be transferred to degraded areas in order to promote recolonization in these selected sites. The second cruise is for the landers transfer from the donor to the degraded site (cruise occurs between 12 and 36 months after initial deployment) and lasts for about 12 days. The third to the fifth cruises will be carried out to during the monitoring phase post-restoration to assess faunal recovery in the restored area and to monitor the effects of the ARMSs in the receiving areas. The third to fifth cruises are assumed to occur in year four, seven and ten respectively with a duration of 2 days each.

3. Methods

Various guidelines on SCBA/CBA have been produced at both international level (European Commission, 1997; 2015) and at national level (e.g. UK HM Treasury, 2011; New Zealand Treasury, 2015; Netherlands CPB/PBL, 2013). Following a simplified procedure for SCBA suggested by the European Commission (2015) this study first defines two relevant scenarios: a baseline (scenario 0) assuming business

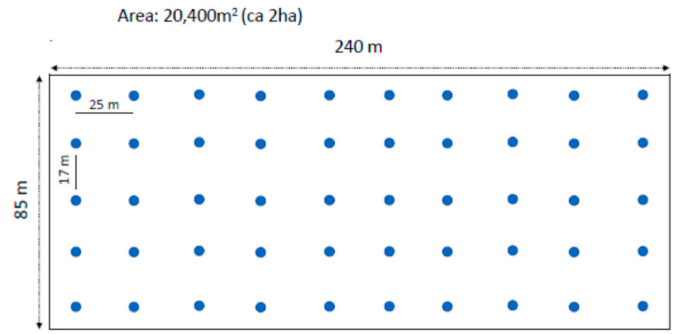

Fig. 3. The layout example for the 50 landers for the size of 2 ha.

as usual without any active or passive restoration, and scenario 1 assuming the creation of the MPA of 2 ha and the deployment of 50 landers hosting 150 ARMSs. Next, the potential physical and ecological improvements after the recolonization of the marine ecosystem are identified with the assistance of marine ecologists involved in the Dohrn Canyon deep sea restoration efforts. The improvements in ecosystem in the study is defined as returning to the pristine habitat levels for the canyon which include recovery of the habitats and high biodiversity (e.g. abundance of fish, starfish, corals, worms, lobsters, sponges and anemones).

In addition, the marine litter on canyon floor should be reduced to a density of 0–1 item per km^2 (O'Connor et al., 2020a, 2020b). The ecosystem improvements are estimated to cover 52–104 km^2 seabed at about 400 m deep. Finally, the societal benefits and costs are identified. The *societal benefits* refer mainly to the non-market ecosystem services benefits with particular focus on the biodiversity and density of marine litter on the canyon floor after restoration (O'Connor et al., 2020a, 2020b). The Contingent valuation method (CVM) was used by O'Connor et al. (2020a, 2020b) to elicit the marginal ecosystem benefit value of the restoration of the Dohrn Canyon to Italian society. CVM is a survey-based stated preference method to estimate the economic value of nonmarket resources. It has been widely adopted in environmental valuation over the last 30 years. Participants were asked how much they were willing to pay (willingness to pay, WTP) for future improvement in the ecosystem and reduced marine litter in the Dohrn Canyon as a result of a 10-year restoration and monitoring plan involving an MPA of size 2 ha. Bid values of (€) 4, 8, 10, 15, 20, 30 were used. An online survey was carried out during two weeks in March 2019 with respondents throughout Italy. The restoration costs used in this current study include both material, equipment and labour costs for three phases, i.e. the preparation phase, the restoration phase and the monitoring phase post-restoration.

O'Connor et al. (2020a, 2020b) assumed certainty of restoration

Fig. 2. The Autonomous Reef Monitoring Structure Unit (ARMS) before and 2 years after deployment. This is an example from the shallow water.

success in their hypothetical scenario but noted in their discussion that "an interesting area for further research would be to explore the effect of alternative levels of risk for restoration action failure on the valuation of benefits through stated preference studies". In the current study, the uncertainty of restoration success is investigated using a Markov model. The model is constructed to estimate the transition of ecosystem/habitat status from one period to the next after restoration activities. A graphical representation of the model is shown in Fig. 4 where *F* denotes a status with deteriorated ecosystem condition. *S* denotes a successful status with improved ecosystem condition. *FS* denotes the *F*→ *S;* that is the ecosystem status changes from deteriorated to successful in the next period. And$p_{FS}$is the probability that the ecosystem status changes from deteriorated to successful in the next period. Similarly, $p_{SF}$is the probability of changing from successful to deteriorated status in the next period, p_{SS} is the probability of remaining in successful status (p_{SS}= 1 - p_{SF}), and $p_{FF}$is the probability of remaining in deteriorated status (p_{FF}= 1 - p_{FS}). The initial status is the deteriorated condition (*F*). The model here is based on the simplified assumption that there are only two states with no intermediate option between the two. Here it is also assumed that the probability of a state change remains constant.

The transition matrix M is defined between two periods by the stochastic matrix:

$$M = \begin{bmatrix} 1 - p_{FS} & p_{FS} \\ p_{SF} & 1 - p_{SF} \end{bmatrix} \text{ where } 0 \leq p_{FS}, p_{FF}, p_{SS}, p_{SF} \leq 1.$$

The restoration activity starts from the beginning of the program, but it assumed to require 3 years for full installation. Therefore, the stochastic process is only modelled for the latter 7 years of the 10-year program, when the measures are fully deployed. The system is assumed to remain in the deteriorated state during the first 3 years.

As there are uncertainties associated with costs and benefits in addition to restoration success, decisions regarding project investment should consider the probability distribution of the Net Present Value (NPV). This is defined as the total benefits minus costs of the project summed over *T* periods:

$$NPV_T = \sum_{t=0}^{T} \beta^t (B_t 1_S(X_t) - C_t) \tag{1}$$

where β is the social discount factor, B_t is the benefit during period *t*, C_t is the cost during period *t*, and $1_S(X_t)$ is the indicator function for success during period *t*, with a value of 1 if X_t is in the successful state and a value of 0 if X_t is in the deteriorated state. Note that NPV_T is a random variable with an uncertainty distribution defined by the stochastic process followed by the system state X_t, as well as the uncertainty distributions of β, B_t, C_t.

Typically, the first concern is the expected or mean value of the NPV; $ENPV_T = E[NPV_T]$. If the transition probabilities ($p_{FS}$and p_{SF}) are assumed to be perfectly known, then the state probability vector at time *t*, π_t, is given by the product of the initial state probability vector π_0 and the Markov transition matrix raised to the power *t* ($\pi_0 M^t$, where π_0 is a row vector). In our case, $\pi_0 = [1\ 0]$, since we assume that the system is in the deteriorated state during period 0 when the restoration measures become fully deployed. If we also neglect the uncertainty in the discount rate, and assume that the uncertainties in benefits and costs are independent of the stochastic process, then $ENPV_T$ can be written as a matrix vector product:

$$ENPV_T = \sum_{t=0}^{T} \beta^t \pi_0 M^t V_t \tag{2}$$

where $V_t = [-E[C_t];\ E[B_t - C_t]]$ is a column vector defined by expected benefits and costs during period *t*.

In this study the full uncertainty in the Net Present Value estimates is accounted for, including the uncertainty in transition probabilities (and therefore *M*) and social discount factor, as well as in the stochastic state of the system (for a given *M*) and costs/benefits. We do this because the transition probabilities, based on expert opinion, and the social discount factor, dependent on future economic developments, are subject to significant uncertainty. We use Monte Carlo simulations to generate a probability distribution of NPV and use this to evaluate ENPV as the mean over ensemble members. We also compare these results with those obtained when uncertainty in transition probabilities and discount factor is ignored, in which case Eq. (2) holds.

The potential annual ecosystem service benefit *B*t was estimated for the 2 ha MPA based on mean willingness to pay (WTP) data from O'Connor et al. (2020a, 2020b). The sample mean WTP (per person) and its standard error from O'Connor et al. (2020a, 2020b) were used to define a probability distribution for the true WTP (i.e. the total WTP of the entire Italian population). Data for potential benefits and the probability distribution are reported in section 4. A lognormal distribution was assumed here, thus ensuring non-negativity. The potential annual benefit was assumed to be constant during the 7 years following installation of the restoration measures. Benefits were assumed to be zero during the first 3 years, since the system was assumed to remain in the deteriorated state during this period. Although the expected ecosystem benefits will last much longer than the 10 year project period, WTP was elicited based on a 10 year annual payment. Therefore our estimation using 10 year WTP as social benefits will only provide a lower bound of the total benefits provided by the restoration project.

The annual costs *C*t were estimated for the three phases of restoration. The costs include labour costs, material costs, and equipment costs (e.g. cruise ship rent). Data for restoration costs are reported in section 4. These estimates were used to define the mean values of the lognormal uncertainty distributions for each year. An uncertainty estimate for the total final-year costs were obtained from the standard deviation of cost estimates from 22 marine restoration efforts within the EU MERCES project. This was used to calculate a logarithmic standard deviation for the cost uncertainty distribution (0.47, implying 47% uncertainty in the costs) which was assumed to be constant between years. Cost uncertainties were assumed to be independent of benefit uncertainties. Note that costs are non-zero during the initial 3-year installation period.

For the transition probabilities, uncertainty was modelled using Beta distributions (O'Hagan et al., 2006) with mean values and standard deviations derived from ranges of values obtained from expert opinion (Appendix 1). Uncertainty in these probabilities, due to expert fallibility, was assumed to be independent of the other uncertainties (system state and costs/benefits).

The Monte Carlo analysis was based on 10,000 simulations, sampling first the transitions probabilities and annual costs/benefits from their respective uncertainty distributions. Following that a stochastic process (within the sampled *M*) for the 7 years following deployment of the restoration measures was simulated. Finally, a sample value of NPV for the full 10 years was then calculated. Sensitivity analyses were used to study the effects of assuming a uniform distribution for transition probability. All simulations were performed using Python 3.8.

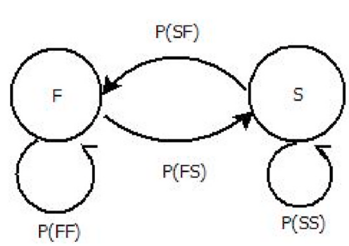

Fig. 4. Markov model for ecosystem transaction between degraded condition (F) and an improved states (S).

4. Data

4.1. Potential benefits (B_t)

Potential benefits were derived from a contingent valuation study of the Dohrn Canyon (O'Connor et al., 2020a, 2020b) where a national level Italian sample of 1060 respondents provided their willingness to pay via increased annual tax for the hypothetical ecosystem restoration project. Due to the high protest rate (about 20% of respondents indicated a protest zero WTP) O'Connor et al. (2020a, 2020b) used a sample selection model similar to Van de Ven and Van Praag (1981) and Petrolia et al. (2010) to estimate the ecosystem demand function. The estimated annual mean WTP per person for the 10 year period was €35 with 95% confidence interval ranging from €15.94 to €53.43. There are approximately 2.75 million adults in the Campania region. The annual WTP therefore amounts to €127 million if an additional regional environmental tax is levied.

4.2. Restoration costs (C_t)

Restoration costs are estimated based on experience from the restoration experiments carried out for the Dohrn Canyon in EU MERCES project. Table 1 presents the annual estimated costs related to the three phases of restoration including the preparation phase, restoration phase, and the post-restoration monitoring phase. For all the three phases, labour costs, material costs and equipment costs are considered. The preparation phase lasts a year. The restoration phase includes deployment of the 50 landers to the donor site in year 2 and deployment of the 50 landers to the restoration site in year 3. Post-restoration monitoring is estimated to occur in year 4, 7 and 10.

4.3. Restoration transition matrix (M^t)

Expert evaluation was used to estimate annual transition probabilities for the ecosystem state after full deployment of restoration measures (Appendix 1). The probability of transition from a deteriorated to an improved status was assessed as medium or low, while the probability of the system reverting from an improved to a deteriorated status was assessed as high. Various factors were considered to determine the transition probability. For example, the selection of suitable sites and depths within the canyon for deploying the artificial structures is fundamental to limit anthropogenic impacts. Whether trawling is present in the head of the canyon also can affect the success of the restoration activity. It was acknowledged that even when restoration can be regarded successful after one year, it may easily revert to a deteriorated state. The artificial structure for deep-sea species recruitment and ecosystem restoration can be removed or partially buried in the presence of trawling activities.

The presence of anthropogenic activities can also compromise the functionality of the device and contribute to the failure of the initiative. In this regard, the selection of the most suitable site for species recovery is a priority for restoration projects in deep-sea habitats. Coral habitats are vulnerable to climate change effects such as ocean warming, acidification and heat waves (e.g Hoegh-Guldberg, 2011). This may cause the ecosystem to flip between an improved status and deteriorated status. The expert evaluation is based on experiments carried out under the H2020 MERCES project. As the location has been considered as an important factor in restoration success, the choice of location in the MERCES experiments avoided areas of fishing trawling pressure. Therefore the estimates from the experiments provide an appropriate transition probability evaluation for the restoration upscaling with the creation of MPA where there is no fishing pressure. The low transition probability from failure to success may indicate not all the pressure has been removed by creating MPAs, such as coastal pollution. Estimates of transition probabilities are presented in Table 2. The estimates are derived from expert evaluation as shown in Appendix 1. Considering the uncertainties associated with these transition probabilities, Beta distributions with mean and standard deviations derived from ranges of these values are used in the Monte Carlo simulation to estimate the distribution of NPV.

Due to the large uncertainty associated with the probability estimates in Table 2, the values in column 3 are regarded as 95% confidence intervals for the beta distributions. The mean of the transition probabilities is $p_{FS} = 0.45$and $p_{SF} = 0.81$. The transition matrix over t year reads

$$M^t = \left(\begin{bmatrix} 0.55 & 0.45 \\ 0.81 & 0.19 \end{bmatrix} \right)^t$$

4.4. The social discount rate

The social discount rate reflects how future benefits and costs should be weighted today. The choice of social discount rate depends on the type of projects. Ecosystem restoration projects usually have long environmental impacts which justifies a low social discount rate. A discount rate of 3–5% is recommended by EU guidelines for investment projects with large environmental impacts (World Bank, 2005). The 3–5% range is regarded as a 95% confidence interval for the uncertain discount rate with a lognormal distribution. A 4% mean discount rate is therefore used in the ENPV estimation.

5. Results

5.1. ENPV with certain costs, potential benefits, and transition probabilities

We first apply the Markov matrix to calculate restoration success rates in each year (probability of being the successful state = $V0.M^t$) and calculate annual ENPV and total ENPV following Eq. (1) (see Table 3). Consistent results were also obtained by Monte Carlo simulation of the Markov process, allowing for error due to finite ensemble size (10,000).

Table 3 illustrates the annual ecosystem service benefits for 52–104 km^2 and total annual restoration costs for 50 landers. The ecosystem service benefits are calculated assuming a regional taxation policy in the region of Campania where the Dohrn Canyon is located is used to fund the restoration effort. The expected net present value amounts to €248 million.

5.2. ENPV with uncertain costs, potential benefits and social discount rate but certain transition probabilities

In this case we again apply Eq. (1), because only the expected values of costs, benefits and social discount factor impact the ENPV. However, to account for the uncertainty of costs, benefits and social discount factor, the Monte Carlo simulation approach is applied. The blue line in Fig. 5 shows the probability density of NPV derived from the Monte Carlo simulation. The mean ENPV is approximately €248 million. This is consistent with the value in Table 3. Given the large potential benefits relative to the restoration costs, the restoration project has a very high probability (98%) of having a positive NPV under this analysis. The net present value is about €2.38–4.77 per m2. This is similar in magnitude to estimates from Constanza et al. (2014).

5.3. ENPV with uncertain costs, potential benefits, social discount factor and transition probabilities

In this case Eq. (1) no longer strictly applies because the transition probabilities are uncertain and have a nonlinear impact on the state probability vector via the transition matrix raised to the power t. The probability densities of the transition probabilities are illustrated in Fig. 6. Monte Carlo simulation with transition probabilities drawn from the Beta distributions suggests that the ENPV is only slightly affected

Table 1
Costs of the restoration (50 landers).

Phase	Description of labour, material and equipment costs	Year	Labor (€1000/ year)	Material (€1000/year)	Equipment (€1000/year)
Preparation	Design and construction of the lander equipped with Autonomous Reef Monitoring Structures (ARMS), preparation of the documentation requested for the field work, general organization of the sampling cruise and the logistic (port of embarkation and disembarkation, scientific staff, equipment and devices for the sampling activity)	1	4	100	
Restoration	Deployment of the 50 landers to the donor site and maintenance	2	6.4	41.5	160
	Deployment of the 50 landers to the restoration site	3	9.6	48.6	240
Post-restoration monitoring	Remote monitoring and analysis of the field work outcomes	4,7, 10	1.6	20	47

Table 2
Estimated transition probabilities.

Transition probability	Description	Value	Source
p_{FS}	Annual transition probability from a deteriorated status to an improved status	0.4–0.5	Best estimate from expert evaluation
$1 - p_{FS}$	Annual transition probability of staying at a deteriorated state	0.5–0.6	Best estimate from expert evaluation
p_{SF}	Annual transition probability of ecosystem deteriorates after improving	0.7–0.9	Best estimate from expert evaluation
$1 - p_{SF}$	Annual transition probability of staying at the improved ecosystem	0.1–0.3	Best estimate from expert evaluation

(now €247 million, a 0.4% reduction). However, the uncertainty in transition probabilities has a very significant impact on the uncertainty distribution of NPV (see the orange line in Fig. 5). This illustrates the importance of considering the uncertainty associated with state transition probabilities derived from expert opinion.

As previously mentioned the transaction probabilities were calculated assuming a beta distribution. Fig. 7 a) and 7 b) demonstrate the results of a sensitivity analysis where the transition probabilities are assumed to have a uniform distribution. The outcomes do not differ significantly from those of the beta distribution assumption.

6. Discussion

In comparison to the deep-sea ecosystem restoration in Darwin Mounds and Solwara (Van Dover et al., 2014), the restoration costs for Dohrn Canyon is relatively low which may be attributed to the limited depth (500–600 m) and the short distance from the coast. The restoration cost in our case is in line with Bayraktarov et al. (2016, 2019). The relatively large aggregate willingness to pay associated with the ecosystem restoration project leads to high probability of positive ENPV disregarding the various uncertainties. In a review of existing economic assessments of coastal ecosystem restoration projects, De Groot et al. (2013) gave a range of benefit-cost ratios between 0.05 and 1.7. Although the benefit-cost ratios for coral reef restoration from Stewart-Sinclaire et al. (2021) meta study is about 4, the positive net benefits were only found for studies in the region of Indonesia, Thailand, the Philippines and the Great Barrier Reef in Australia. No positive net benefits for coral restoration have been found outside those regions. In contrast, the median benefit-cost ratios in our case is positive and is much higher than those found in De Groot et al. (2013) and Stewart-Sinclaire et al. (2021). The high benefit cost ratio may stem from the use of a non-market welfare measure to evaluate the ecosystem service benefits. Our study also highlights the importance of carrying out the SCBA for specific restoration projects rather than relying only on the outcomes from meta-analysis or global scale analysis.

The benefit derived from CV will include both use value (fish traded at market) and non-use value (bequest value for future generations and existence values for marine species) (O'Connor et al., 2020a, 2020b). There is however a need for caution in interpreting the results from O'Connor et al. (2020a, 2020b) as it arises from a very particular improvement scenario. It is also possible that the estimates may indicate a degree of insensitivity to scope, a common issue in CVM studies (Carson, 2012). Therefore, further research is required to verify the stability of preferences for marine restoration amongst the Italian population. If we used just exchange values, e.g. market price for fish or wastewater treatment fees, the estimated benefits would be expected to be much lower. When an improvement of natural capital via restoration covers a bundle of correlated ecosystem services, stated preference approaches which generate a hypothetical market will provide a better estimate of the aggregate welfare value for the service bundle. This is particularly the case for marine ecosystems where many of the associated services are non-market in nature. At the same time, the welfare measure is truncated at the 10-year mark as we assumed the restoration project only last for 10 years. This will lead to underestimate of the true ecosystem service benefits which on completion of restoration should last much longer than 10 years.

It is important to consider uncertainties related to a complex natural-technological-social system in evaluating the SCBA for restoration projects. This study shows the uncertainties associated with for example restoration success rate significantly affects the probability distribution

Table 3
ENPV with expected restoration successful rate.

	Year									
	1	2	3	4	5	6	7	8	9	10
Ecosystem service benefits (B_t) (€ million)										
	–	–	–	2105	2105	2105	2105	2105	2105	2105
Total costs (C_t) (€ million)										
50 landers	0.104	0.208	0.298	0.069	0	0	0.069	0	0	0.069
Discount rate (β)	0.04									
Restoration success rate (P_{FSt})	–	–	–	0.45	0.34	0.36	0.36	0.36	0.36	0.35
50 landers										
Annual $ENPV_t$ (€ million)	−0.100	−0.192	−0.264	49	35	37	34	33	32	30
ENPV (€ million)	248									

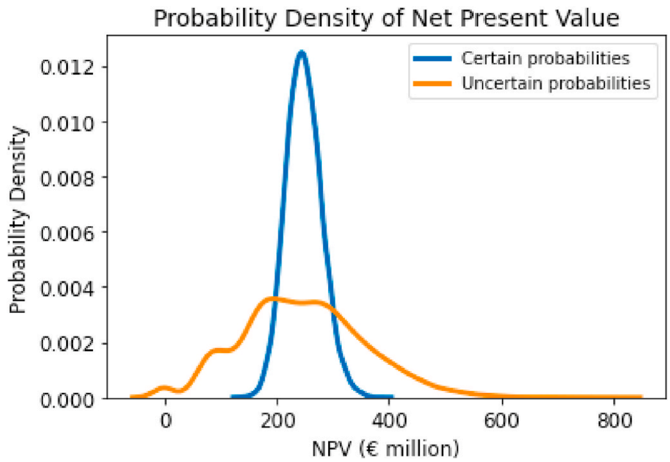

Fig. 5. Probability density of Expected Net Present Value: certain transition probability versus uncertain transition probability. Uncertainty associated with social benefits, restoration costs and social discount factor are considered in both cases.

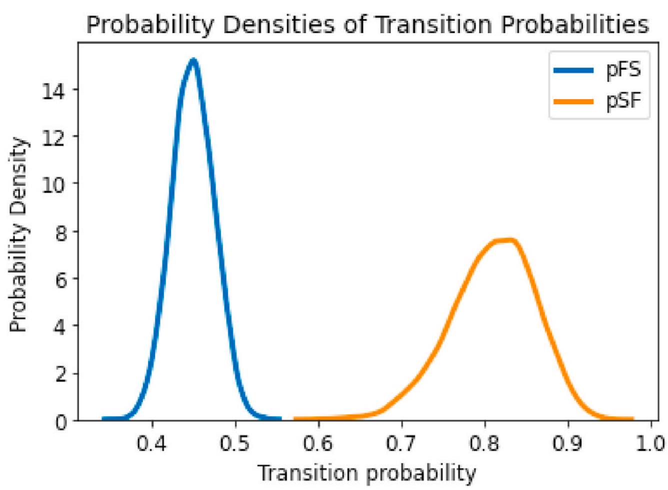

Fig. 6. Probability densities of transition probabilities. pFS for transition probability from a deteriorate ecosystem state to an improved ecosystem state. pSF for transition probability from a good ecosystem state to a deteriorate ecosystem state.

of the ENPV. Expert opinions were considered the most appropriate source of information to assess the restoration success rate in our case. However using expert opinion in modelling uncertainties has limitations. On the one hand, expert opinions can introduce additional ambiguity associate with the prediction even though Monte Carlo simulation can handle this type of ambiguity to a certain degree. On the other hand, expert opinions may not provide correct prediction for large scale and long-term restoration interventions as they are derived from small scale experiments. Therefore the restoration success estimates should be updated once better field data are available. The expert opinions on success rates used in this study are based on a four-year restoration experiment in the Dohrn Canyon carried out by a group of deep-sea ecologists. The expert opinion was collected from two representative biologists leading the project. It is not expected that the expert evaluation would be significantly different if more experts from the same project were involved as they may be subjective to systematic bias as a group. Future expert prediction on the deep-sea ecosystem response could follow the Delphi process (MacMillan and Marshall, 2006) in which expertise of individuals from *various* deep-sea restoration projects would provide their evaluations to improve the reliability when more

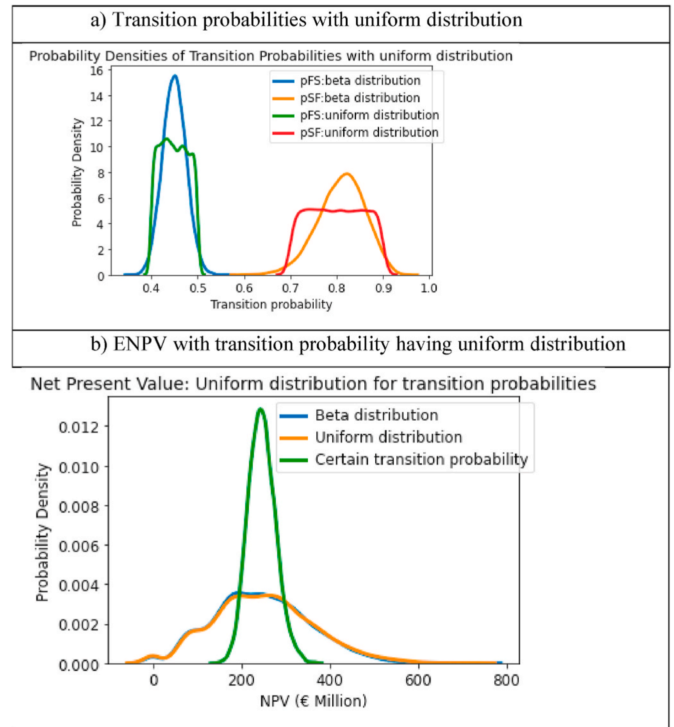

Fig. 7. Sensitivity analysis for probability density of ENPV with a) Probability Densities of transition probabilities with uniform distribution b) ENPV with transition probability having uniform distribution.

restoration projects start. Adaptive restoration strategies (learning by doing) and precautionary principles should always be incorporated in to the ecosystem restoration policy to provide rapid responses (Saunders et al., 2020).

Value pluralism and using multiple valuation methods for planning and policy have been called for in the Dasgupta Review (2021) and the ongoing IPBES Values Assessment (IPBES, 2021). In order to integrate biodiversity valuation in planning, the Convention on Biological Diversity (CBD) has referred to the valuation methods used in The Economics of Ecosystems and Biodiversity (TEEB) and SEEA EA (System of Environmental Economic Accounting - Ecosystem Accounting) (UNCEEA, 2021). As an integrated statistical framework, the SEEA EA aims to track the human activities on the ecosystem by organizing biophysical information about ecosystems, measuring ecosystem services, monitor and evaluate the benefit values generated by ecosystem services for society (UNCEEA, 2021). Ecosystem restoration has been mentioned as one area SEEA EA can be applied (UNCEEA, 2021). Plural values are partly addressed in ecosystem accounting through biophysical and monetary indicators of ecosystem services.

The Second Global Dialog for Ocean Accounting[1] showed that current experimental ocean accounts mostly focus on maritime industries and physical accounts of carbon storage/sequestration, shoreline protection, fish nursery and habitat and nutrient cycling. The growing evidence base will enable assessment of a broader range of impacts including the values of deep-sea ecosystem services (Chen et al., 2020b). This study showed the potential to value whole system changes via stated preference methods. As current SEEA EA only focuses on exchange values which is market oriented, it is important to consider various frameworks and include welfare values in the SCBA and policy appraisal to represent the actual values to society (Chen et al., 2020b). As not all the impacts of ecosystem services on human welfare can be

[1] https://www.oceanaccounts.org/second-global-dialogue-on-ocean-accounting/.

expressed in monetary terms, qualitative and non-monetary metrics indicating level of benefits should also be provided alongside the SCBA (Chen et al., 2020b).

In addition, it is important to bear in mind that the NPV estimated in a cost-benefit analysis is seldomly the only consideration in a political decision (Turner, 2007). As a normative analysis tool, cost-benefit analysis is done within a strictly defined anthropocentric and utilitarian framework, which limits considerations that are difficult to trade off against money, such as eco-centric, religious, or moral values (Groeneveld, 2020). In addition to anthropocentric values such as ecosystem services, restoration may also be motivated by idealistic or moral notions such as atonement for past ecosystem damage (Clewell and Aronson, 2006; Ounanian et al., 2018). Amkamah-Yeboah et al. (2020) find that respondents in Scotland and Norway hold eco-centric attitudes towards the marine environment. O'Connor et al. (2020a) find Italian and Norwegian respondents prefer on-site restoration over offsetting the damage by restoring an ecosystem elsewhere, which suggests that the benefits lost due to the damage represent non-substitutable socio-cultural values.

Recent systematic reviews have revealed the limited usage of ecosystem service values in decision-making (Laurans et al., 2013; Lautenbach et al., 2019; Mandle et al., 2021). Similar results have been found for spatial planning in coastal and marine waters (Kvalvik et al., 2020; Marre et al., 2016). These studies call for further research on processes and conditions to facilitate uptake of ecosystem service assessment and monetary valuation in decision-making. We need now to go beyond the conclusions that people support restoration and having some evidence of its monetary value, to explaining how that fits in to the current policy framework and can support restoration in the broader context of marine and coastal planning (Chen et al., 2020b). Future application needs to go beyond project-level SCBA to include strategic level target setting that is mainstreamed across the wider policy agenda (in terms of contribution to climate targets, blue growth and jobs, etc.). This should include natural capital accounting at all scales (from regional seas for monitoring purposes, to local assessments for coastal industries and landowners), developing business cases and financing as well as the potential use of Payments for Ecosystem Services (PES) or habitat banking (Chen et al., 2020b). In addition, marine and coastal restoration can act as important nature-based solutions to increase the resilience to the impacts of climate change. SCBA will also be an important tool in developing the adaptation strategies required under the European Commission's proposal for the first European Climate Law (Chen et al., 2020b).

The experience from terrestrial ecosystem restoration to date indicate that effective approaches to all restoration efforts require the engagement of all stakeholders including communities, scientists, policymakers, and land managers to successfully repair ecological damage and rebuild a healthier relationship between people and nature (Gann et al., 2019). Social awareness and the contribution of different stakeholders can play a fundamental role in the scaling up of restoration projects in marine ecosystems. Recent research outcomes from the H2020 MERCES project demonstrates that recognizing the public's current level of knowledge regarding marine ecosystems and restoration can assist in the development of educational tools and effective management policy, thus preventing future damages to marine ecosystems. Public support could be increased through campaigns to increase awareness of marine restoration activity including highlighting major advances, success stories and expected benefits (Chen et al., 2020b).

7. Future research prospective and field applications of current work

Despite numerous attempts to restore marine ecosystems globally in the last decade, the research in this area is still at a relatively early stage (Bayrakatarov et al., 2020; Saunders et al., 2020). Restoration of deep-sea communities in particular is a new frontier for ocean science (Da Ros, 2019; Boch et al., 2019). There are many research gaps for deep sea ecosystem restoration including testing new restoration techniques and tools, developing new monitoring equipment, upscaling and cross sectoral collaboration on restoration activities, better understanding the long term change in ecosystem function after restoration, and having legally binding high level commitment (Danovaro et al., 2017; Johnston et al., 2019; Yang et al., 2020; Hein et al., 2020, Smith et al., 2020). Folkersen et al. (2018a) point out that there is a lack of sufficient data on deep sea ecosystem functions and processes to accurately estimate the economic value of deep sea ecosystem services. Based on the finding from this study a number of potential future research prospective for economics of ecosystem restoration are proposed.

Mapping the spatial and temporal change of the ecosystem services benefits: Ecosystem restoration improves ecosystem function and ecosystem services. Linking the spatial and temporal change in ecosystem services use and benefits to the change in ecosystem function after restoration activities are crucial to evaluate the benefits of restoration. More research on mapping the spatial and temporal change of the ecosystem services use and benefits are needed. This will also require better data collection on the ecosystem services.

Project and policy appraisal, impact assessment: Upscaling of marine ecosystem restoration activities is important to achieve global targets for ecosystem restoration. The SCBA framework demonstrated in this study can be used in the decision support for upscaling of various coastal and marine ecosystem restoration projects and impact assessments of industry development as well as policy to assist in marine conservation. Relevant industries where the framework could be usefully employed include deep sea mining (Folkersen et al., 2018b, 2019, 2018b), bottom trawling (Pham et al., 2019), oil and gas development and the offshore renewable energy development (Chen et al., 2021).

Pricing decision and compensation for damage: Due to the technical challenges, deep-sea restoration can be very expensive especially during upscaling. Economic analysis of restoration interventions such as SCBA can provide useful information for pricing decisions post restoration. For example, it could be used to guide in the setting of MPA access fees for recreational diving. The analysis could also be useful to decide the compensation level for damage caused by industries.

Integrate valuation evidence in marine restoration financing: Long-term funding is needed to secure the continuity of marine restoration projects and restoration success. The high aggregate willingness to pay for the restoration project by the public in this study does imply that there is room for both public and private fund raising. Effective financing mechanism design in the future may need to connect to the outcomes of the restoration activities and the risks they face. Any such financial instruments should be underpinned by strong valuation evidence to establish appropriate levels for fees and payments. Innovative funding schemes such as public-private partnership, crowdfunding, biodiversity offsets, nutrients or carbon trading need to be tested. SCBA can demonstrate the value for money and to support fund seeking for restoration activities (Tinch et al., 2019). SCBA should be an integral part of building the business case for financing (Chen et al., 2020b).

8. Conclusions

The study presented an example of how to evaluate the economic benefits and costs of large-scale ecosystem restoration activities that account for the change in natural capital value that may or may not be traded in the market. The results indicate that an ecosystem restoration project can be economic (in terms of welfare improvement) even if the restoration costs are high. The study shows it is important to include uncertainty in the SCBA analysis. For example, the uncertainty associated with restoration success rate significantly affects the probability distribution of ENPV. The overall low restoration success rate with one intervention cycle implies a need for repeated restoration activities so as to increase the overall probability of success. Recovery of deep-sea ecosystems requires a long time horizon and passive recovery may need as long as four decades (Da Ros, 2019; Van Dover, 2014). The SCBA for longer period restoration projects will follow the same steps demonstrated in this study. Decision making for restoration projects with repeated interventions can be scaled up based on the current study and considering ecosystem dynamics in restoration. This will be explored in the future scale up of the Dohrn restoration project over the coming years.

Credit author statement

Wenting Chen and Philip Wallhead developed the methodology, did the programming, carried out the simulation and manuscript writing. Stephen Hynes, Rolf Groeneveld, Cristina Gambi, Roberto Danovaro, Rob Tinch, Nadia Papadopoulou and Chris Smith contributed to manuscript writing. Eamon O'Connor contributed to development of methodology.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix 1. Expert evaluation of transition probabilities for deep-sea ecosystem restoration in the Dohrn Canyon

Status transit (Time unit: one year)	Probability	Factors that determine the status transit (e.g. lack of control of eutrophication may lead to the status transit from success to failure)	How many years does the Dohrn Canyon restoration project last?
From Failure to Success	Medium/ low	The selection of site (inside the canyon) and depth where the artificial structures are deployed are fundamental to limit or exclude anthropogenic impacts. The presence/absence of trawling plays a key role for the failure or success of the restoration activity. A preliminary survey before the devise deployment can allow the identification of the best site for deep-sea species recruitment and ecosystem restoration. This can increase the chance of success of the restoration activity.	We have conducted a pilot action to test the efficacy of a new devise to facilitate the restoration of deep-sea degraded habitats based on the use of artificial substrates. We have deployed this structure for three months.
From success to failure	High	The artificial structure for deep-sea species recruitment and ecosystem restoration can be removed or and partially buried in case of the presence of trawling activities, especially in the head of the canyon where this can occur. The presence of anthropogenic activities can compromise the functionality of the devise and determine the failure of the initiative.	

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