Ecological Economics 179 (2021) 106833

Contents lists available at ScienceDirect

Ecological Economics

journal homepage: www.elsevier.com/locate/ecolecon

Analysis

Valuing the ecosystem service benefits from kelp forest restoration: A choice experiment from Norway

Stephen Hynes[a,*], Wenting Chen[b], Kofi Vondolia[b], Claire Armstrong[c], Eamonn O'Connor[a]

[a] Socio-Economic Marine Research Unit, Whitaker Institute, National University of Ireland, Galway, Ireland
[b] Norwegian Institute for Water Research (NIVA), Oslo, Norway
[c] The Artic University of Norway, Tromso, Norway

ARTICLE INFO

Keywords:
Kelp forest restoration
Ecosystem service benefit values
Choice experiment

ABSTRACT

Habitat loss and degradation are recognised as the most important causes of species decline and extinction in marine ecosystems. It is also widely recognised that a range of restoration actions are now essential to halt further decline. From a policy perspective, demonstration that restoration activity is in the interest of society is an important goal. In this paper, the welfare impacts of restoring Norwegian kelp forests to areas where they once were dominant but which now lie barren are estimated using the discrete choice modelling approach. The paper also examines if more direct contact with the environmental good under investigation influences respondents' willingness to pay to restore ecosystem features. The results indicate a positive and significant marginal societal willingness to pay for the ecosystem services associated with kelp forest restoration. The enhanced biodiversity levels as a result of the restoration activity are the most highly valued by the Norwegian public although the size of the area restored is more highly valued by respondents who are active marine environment users. It is argued that without incorporating these non-market values into the decision making process marine policy decisions may be made that are not in fact in the best interest of society.

1. Introduction

Kelp forests are extensive, underwater habitats that dominate subtidal shallow rocky coasts contributing to their production, biodiversity and functioning in temperate to polar parts of the world (Araújo et al., 2016; Filbee-Dexter and Wernberg, 2018). They are known to be one of the most productive natural ecosystems on the planet supporting complex food webs in coastal zones and providing food, shelter and habitat for a variety of invertebrates, fish, mammals and seabirds (Christie et al., 2009; Graham, 2004; Norderhaug et al., 2005; Teagle et al., 2017). As well as these intrinsic biodiversity values kelp forests generate direct use value through the kelp harvesting, commercial and recreational fishing, and tourism activities that they support (Vasquez et al., 2014; Bennett et al., 2016; Blamey and Bolton, 2018). Kelp forests also provide many supporting and regulating ecosystem services that benefit society indirectly (Pascual et al., 2010; Pendleton, 2010). The role of kelp forests acting as a carbon sink is also currently an active area of research (Smale et al., 2016; Filbee-Dexter and Wernberg, 2020). Kelp forests also take up extra nutrients in the water and hence also provide a bioremediation function, something that is often mentioned in connection with increased aquaculture production (Gundersen et al., 2016). The forests have also been found to support other adjacent and diverse ecosystems through the export of kelp detritus (Krumhansl and Scheibling, 2012).

Healthy kelp forests can also play an important role in mitigating the impacts of storm surges on vulnerable coastal areas by dampening the intensity of the wave forms generated before they reach land (Lovas and Tørum, 2001). Such threats are expected to increase with climate change and there is a growing interest internationally in the adoption of these nature based managed realignment approaches or what is sometimes referred to as blue green infrastructure (Luisetti et al., 2011; Ghofrani et al., 2017; Deely and Hynes, 2020). As discussed by Esteves and Williams (2017) managed realignment approaches involving the restoration of coastal habitats such as saltmarshes, mangroves and oyster reefs are increasingly being considered as “a no-regret option bringing social and environmental benefits” which can improve the long-term sustainability of coastal management strategies. Therefore, restoring kelp forest habitat would not only improve the ecosystem and ecosystem function of the seabed but also the ecosystem service benefits received by coastal communities and society more generally.

Kelp forests are known however to be on the decline at a global scale (Fredriksen et al., 2020; Wernberg et al., 2019). This decline is

* Corresponding author.
E-mail address: stephen.hynes@nuigalway.ie (S. Hynes).

https://doi.org/10.1016/j.ecolecon.2020.106833
Received 22 November 2019; Received in revised form 21 August 2020; Accepted 27 August 2020
Available online 08 September 2020

linked to anthropogenic induced impacts such as ocean warming, eutrophication, and overfishing that has led to sea urchin abundance and overgrazing. This decline has serious consequences for the coastal societies that rely on kelp for the many ecosystem goods and services provided by these habitats. There is also evidence that these human impacts on kelp forests may be accelerating and large-scale restoration efforts may be required to halt the loss of these valuable ecosystems (Krumhansl et al., 2016). Fredriksen et al. (2020) review alternative approaches to restoring kelp forests including fishing and hunting restrictions; sea urchin removals by commercial harvest, quick liming, or culling by divers; the creation of artificial kelp forests by adding boulders to sandy bottoms, seeding techniques and methods to enhance natural recruitment. Restoration of natural kelp forests is now viewed by many as being necessary not only to ensure its future availability for possible use as a food source in multi-trophic aquaculture, for alginate extraction and for biofuel production but also as an environmental purifier (Vasquez et al., 2014) and a vehicle for carbon storage (Gundersen et al., 2010), and has been studied in countries such as the Australia (Layton et al., 2020), the US (Claisse et al., 2013) and Italy (Tamburello et al., 2019).

Restoration rather than preservation is also seen as vital for kelp recovery in Norway where the issue of sea urchin overgrazing has been of particular concern. For the last four decades, dense sea urchin populations have destructively grazed kelp forests over extensive areas, particularly along the coast of three northern counties; Nordland, Troms and Finnmark (Christie et al., 2009; Norderhaug and Christie, 2009). In these areas barrens have resulted - a desert-like seabed consisting almost entirely of sea urchins. Sea urchin barrens have low productivity and support few other ecosystem functions (Christie et al., 2009). If the urchin population were reduced, kelp forests could recover (Filbee-Dexter and Scheibling, 2014). Rapid kelp recovery following sea urchin mortality has been seen in Norway in a small-scale experiment (Leinaas and Christie, 1996) and at a larger scale in nature (Norderhaug and Christie, 2009). Fig. 1 shows the known distribution of kelp along the Norwegian coast as well as the sea urchin barrens along the Northern coast and the location of kelp forest that is threatened by eutrophication and ocean warming.

Recognising the ecosystem services provided by kelp forest restoration, quantifying them and finally valuing the benefits to society from the additional level of services provided enables policy makers to take such values into account when assessing policies which may affect kelp forest habitats, and can also assist decision makers to decide on which restoration projects should be prioritised. With this in mind current paper presents a choice experiment study that was carried out amongst the Norwegian population to assess the preferences and willingness to support kelp forest restoration activities in Norwegian coastal waters. While other studies have used secondary sources of information to quantify the values of a number of kelp forest ecosystem service benefits to society none have conducted a primary analysis to examine what the values from the restoration of such an ecosystem might be. As such this study provides important information related to the benefit values derived through the restoration of marine ecosystems as well as a better understanding of societal preferences for alternative kelp forest policy options. We also examine if being an active marine recreationalist may influence respondents' willingness to pay to restore kelp forests.

While many ecosystem services are provided by kelp forests the cognitive burden of completing a choice experiment means that only a limited number of attributes can be included in the options presented to respondents in a choice experiment. Based on discussions with marine ecologists in the Norwegian Institute for Water Research and other marine scientists in the EU MERCES project, and a review of the literature, three key restoration attributes were included; one relating to biodiversity, one related to the nursery function played by kelp forests for juvenile fish and one on the size of the ecosystem to be restored. The study also examines how respondents make trade-offs between the

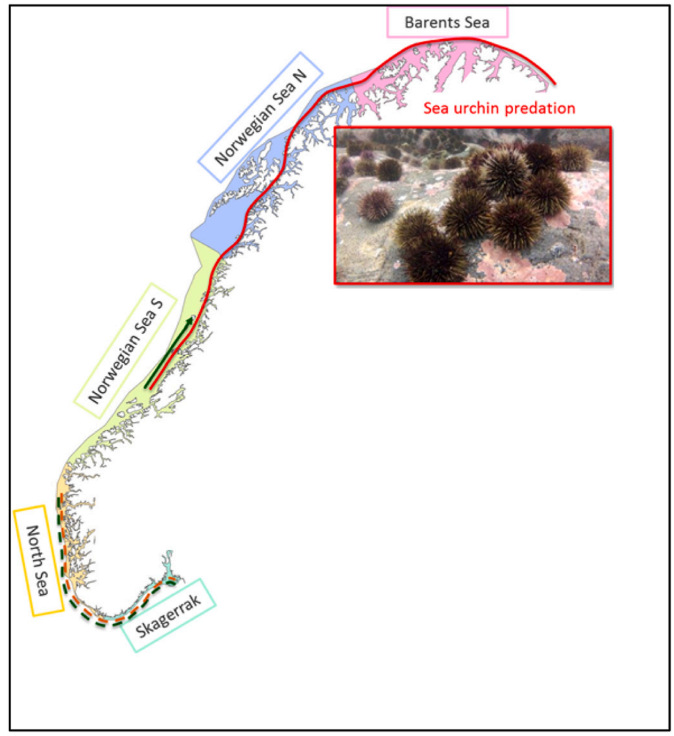

Fig. 1. Kelp distribution along the Norwegian coast.
The red line indicates the spread of the sea urchin barrens along the Northern coast. The green line indicates the presence of healthy kelp forest in mid-Norway. The dotted line in the south indicates the presence of kelp forest that is threatened by eutrophication and ocean warming. The map is adapted with permission from Gundersen et al. (2016). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

possible levels of ecosystem services that could be provided by restoring the kelp forest on the urchin barrens in Northern Norway. Using the responses to the choice experiment we estimate the Norwegian population's marginal willingness to pay for ecosystem service attributes associated with kelp forest restoration, namely biodiversity improvements and the nursery services that such forests provide to juvenile fish. The relevance of the size of the area to be restored to respondents is also assessed in the choice experiment and in the welfare estimations. Model results are also employed to estimate the welfare impact[1] of a number of alternative restoration policy options that vary in terms of the ecosystem service levels delivered.

In what follows we first briefly review previous efforts at valuing the restoration of different habitat types and report on the limited number of studies that have looked at valuing kelp forest ecosystem service benefits. Section 3 then outlines the design of the survey. Following that Section 4 presents the discrete choice modelling methodology. Model results and the WTP estimates are presented in Section 5. Finally, Section 6 concludes with some recommendations for further research.

2. Valuing ecosystem restoration

There have been a number of studies that estimate the values of restoration of different habitats using mostly stated preference methods. Some of these studies use the methods of contingent valuation

[1] Welfare impact in environmental valuation exercises generally refers to the values that people assign to marginal changes in the supply of environmental goods. In a choice experiment setting the welfare impact is indirectly inferred from the trade-offs that people are willing to make when choosing their preferred alternatives in the choice sets.

and choice experiments (Endo et al., 2012; Park et al., 2013; Dissanayake and Ando, 2014). Others use market data (Jenkins et al., 2010; Vasquez et al., 2014). However, most of these valuation studies are on wetlands (Milon and Scrogin, 2006; Westerberg et al., 2010; Glenk and Martin-Ortega, 2018), riparian or river ecosystems (Ojeda et al., 2008; Loomis et al., 2000; Grazhdani, 2013), and terrestrial ecosystems such as grassland (Dissanayake and Ando, 2014). There are also a number of coastal ecosystem restoration valuation studies including applications to tidal flats (Kim et al., 2017) and coastal lagoons (Stone et al., 2008; De Wit et al., 2017; de Rutger et al., 2017). De Wit et al. (2017) examined the WTP for the ecological restoration of coastal lagoons along with waterside facilities. Interestingly, they found that local populations were willing to pay €25 per person per year for restoration of the ecosystems but were only willing to pay €5 for additional footpaths and hides, suggesting a higher willingness to pay for non-use benefit values from restoration than use values.

While a number of studies have estimated the ecosystem service values associated with the conservation of off-shore marine ecosystems (for example Jobstvogt et al., 2014; Norton and Hynes, 2014; Armstrong et al., 2017; Aanesen et al., 2018) those that have estimated the value of restoring such ecosystems are more limited. Borger and Piwowarczyk (2016) conducted a choice experiment to value non-market benefits of seagrass restoration in the Gulf of Gdansk in Poland. The benefits were estimated for a reduction in filamentous algae in the water and on the beach, access to seagrass meadows for boating and diving, and improvement in water clarity. In another offshore study Tonin (2018) estimated the economic value of restoration efforts to improve biodiversity in coral forming habitats in the Northern Adriatic Sea using a contingent valuation survey of Italian households. They found that household willingness to pay (WTP) for biodiversity restoration and conservation ranges between €10.30 and €64.02 depending on the assumptions underlying the different models employed. In a rare example of a deep-sea restoration valuation study O'Connor et al. (2020) used the contingent valuation method to elicit the Italian population's willingness to pay for the restoration of a deep-sea canyon ecosystem in the Bay of Naples. The authors argue that such valuation exercises are critical when decision makers are faced with multiple restoration needs and limited budgets and can help to ensure that the most efficient restoration projects are chosen.

While no study to date has estimated the marginal welfare impact from the possible restoration of kelp forests, Vasquez et al. (2014) did use several economic indicators to estimate market values of various goods and services that utilize the biomass of kelp. These values included the extraction of alginic acid, the market values of economic species associated with kelp, the value as a source of scientific information, the value of kelp as a climate buffer in terms of CO_2 capture and release of O_2, the value of biodiversity emanating from non-commercial kelp species, value as cultural heritage and value as a reservoir of biodiversity. Moreover, Vasquez et al. (2014), using contingent valuation, estimated the willingness of citizens to pay and work without payment to preserve kelp ecosystems in Chile. All these components were combined to estimate the total economic value of kelp beds in Chile to be approximately €600 million. Blamey and Bolton (2018) also used secondary sources to estimate the current value of kelp forests and associated temperate reefs that dominate 1000 km of the nearshore subtidal zone in the southern Benguela region of South Africa. The authors estimated that the direct use value of this ecosystem was €391 million per year with ecotourism contributing almost 40% of this, recreational fishing 28%, and commercial and illegal fishing approximately 15–16% each. The authors also estimated that the indirect ecosystem service benefits due to services such as nutrient cycling could be valued at €130 million per year.

Elsewhere, Bennett et al. (2016) also assessed the direct use values associated with the kelp dominated ‘Great Southern Reef’ in Australian waters. They estimated that recreational fisheries and the two most valuable commercial fisheries (rock lobster and abalone) in the area of the kelp covered reef were worth €726 million while total tourism expenditure in coastal areas immediately adjacent to the kelp reef generated over €6.7 billion in revenues to the local economies. In another study, Rebours et al. (2014) underscore the importance of the sustainable harvesting of kelp and other seaweed for wealth creation and sustainable livelihoods amongst the coastal communities on different continents. Finally, based on the results of Blamey and Bolton (2018), Bennett et al. (2016) and Vasquez et al. (2014), Filbee-Dexter and Wernberg (2018) estimate that kelp forests may provide ecosystem service benefits worth in the region of €442,000 to €885,000 per kilometer of coastline.

This study adds to the above literature by estimating the welfare implications of restoring kelp forests to areas where they once were dominant but which now lie barren. In doing so the Norwegian population's marginal willingness to pay for a number of ecosystem service attributes associated with kelp forest restoration are estimated, namely biodiversity improvements and the nursery services that such forests provide to juvenile fish. The size of the area restored is another attribute considered in the choice experiment. We also examine if increased use of the marine environment through participation in water based activities has an influence on a person's preferences for kelp forests and their associated attributes. The paper also contributes to the very limited literature related to valuation of restoration in marine environments more generally.

3. Survey design and choice experiment

An online survey was carried out in the spring of 2018 to obtain information relating to the Norwegian public's preferences for kelp forest restoration. Included in the survey was a choice experiment to generate data for the estimation of the public good benefit value of such restoration. Extensive discussions with kelp ecologists led to the choosing of the relevant attributes and levels that should be used in the choice experiment. This was followed up with focus groups to make sure that the attributes were described in a manner that was also understood by the general public.

Three focus groups were carried out in total. The first was with 10 bachelor students from UiT, The Arctic University of Norway and was used to finalise the first complete draft of the survey instrument and to test general understanding of the information provided. The following two focus groups involved nine participants in each group, selected by the market research company Norstat to be close to representative of the population. Observations from the focus group discussions were also used to refine the language, descriptions and other questions asked in the survey instrument. The same market research company was then employed to collect the survey data using their established online panel of the general public. Pilot testing of the survey instruments was also conducted with 90 randomly selected panel members prior to the main survey. While no further issues were raised with the survey instrument in the pilot test phase, the responses were used to establish the choice sets for the main survey.

In the final survey instrument, respondents were given some background information on the status of kelp forests in Norwegian waters and then asked a series of questions related to their attitudes towards marine ecosystems, marine ecosystem restoration as well as questions that retrieved respondent's experience with the marine environment. A number of socio-demographic questions were also asked related to age, gender, marital status, occupation, working status, income, number of persons in household and education. Finally, a series of 6 choice cards were presented to each respondent that examined their preferences for a set of attributes associated with kelp forest ecosystems and possible restoration projects. The surveys resulted in 1102 complete observations.

To generate the choice cards used in the survey, a Bayesian efficient design was employed (Hess et al., 2008; Scarpa and Rose, 2008). The design for the main survey was generated using the NGENE software

Table 1
Restoration attributes and levels used to describe choice alternatives.

Attribute	Description	Levels
Biodiversity	Number of species present per m^2	Low abundance (approx. 10 species) Medium abundance (approx. 75 species). High abundance (approx. 250 species).
Nurseries for juvenile fish	Juvenile fish abundance per m^2	Low abundance (max 10 juveniles) Medium Abundance (max 20 juveniles) High Abundance(max 30 juveniles)
Area restored	Total area of kelp forest restored	40,000m^2 (5.5 soccer pitches) 20,000m^2 (3 soccer pitches) 10,000m^2 (1.5 soccer pitches) None
Cost	Amount paid per person per year through higher tax payments.	€0, €5, €10, €20, €30, €45, €60

and the value of the D-Error for the main design was 0.55 (mean value). In order to generate an efficient design, priors for the unknown parameters must be established by the researcher. Following best practice (Hoyos, 2010), the priors used in the pilot study were developed based on literature review and the experience of the authors in previous marine ecosystem valuation exercises. Data from the pilot was then analysed and the resulting parameter estimates used as priors to a more efficient design for the final survey.

For the choice experiment, respondents were first informed that: “This survey is concerned with your opinions about the characteristics of kelp forest restoration in Norwegian waters. For the purposes of this study, we think about kelp forest restoration in terms of four characteristics”. Respondents were then presented with a description of the 4 characteristics used in the choice cards; species biodiversity, nurseries for juvenile fish, area that is to be restored and the price of each restoration option. Species biodiversity refers to the species diversity and abundance in an area, and was defined by the number of species present per m^2, as well as the composition of the abundance. Nurseries for juvenile fish referred to the number of juvenile fish present per m^2. The area attribute indicated the size of the area to be restored and was presented in m^2. To improve understanding the restored area was also presented as the approximate number of soccer pitches it was equivalent to. Finally, the cost of each option (the price) was presented in the form of an annual increase in personal income tax. The cost of each option on the choice cards was shown in Norwegian Kroner but all subsequent analysis was carried out using the Euro equivalent to allow for comparison to other studies and WTP estimates are also presented in Euro. The restoration attributes and levels used to describe the choice alternatives are also shown in Table 1.

Following the presentation of the attributes, the respondent was then informed that “in each card different combinations of the restoration's characteristics are shown describing how the kelp beds might change in the future if restoration actions are taken to improve matters”. Furthermore, they were told “The cards also show the associated cost on you and your household of such actions. Please consider your own budget and ability to pay when considering each option”.

Respondents were also asked to remember that some people say they are willing to pay more in surveys for these types of improvements in ecosystems than that they actually would pay if the situation were real. They were further informed that this is because when people actually have to part with their money, they take into account that there are other things they may want to spend their money on. Respondents were therefore asked to “Please consider the impacts on you and your family of improving the kelp bed ecosystems and to imagine yourself actually paying the amounts specified for the next 10 years”.

An example choice card (Fig. 2) was then presented and described so that the respondents would fully understand the task ahead. The 6 choice cards presented three restoration alternatives and asked them to choose their most preferred option. The third option on each card always represented the status quo alternative. The status quo represents the absence of a restoration policy and means no additional financial cost to respondents but would also result in the absence of a kelp forest ecosystem over time. The first and second options of management/restoration represent alternatives leading to improvements in the kelp forest ecosystems and thus were associated with a positive cost.

4. Methodology

The basis for the analysis of the response data to the kelp restoration options presented in the choice cards is the standard random utility modelling (RUM) framework (McFadden, 1974). According to this framework, the indirect utility function for each respondent is made up of a deterministic component determined by the attributes of the alternatives in the choice experiment and characteristics of the respondent, and a stochastic component which represents unobservable influences on individual choice. The RUM model can be specified in different ways depending on the distribution of the stochastic component of the model, i.e. the error term (Hynes et al., 2008). If the error terms are independently and identically drawn from an extreme value distribution, the RUM model is specified as the Conditional Logit (CL) (McFadden and Train, 2000).

The CL model relies on the independence of irrelevant alternatives (IIA) assumption which states that the ratio of choice probabilities between any two alternatives in a choice card is not affected by the introduction of removal of additional alternatives. The CL model also assumes homogenous preferences across respondents since it estimates a single (mean) attribute parameter for each choice attribute. These are strong assumptions and Train (2003) discusses practical estimation techniques based on simulation methods to overcome these shortcomings of the CL. One of those approaches is referred to as the random parameter logit (RPL)[2].

The RPL generalizes the CL by allowing the β coefficients of observed variables to vary randomly over people rather than being fixed as in the CL model; thereby accounting for preference heterogeneity. The RPL model also allows the error components of different alternatives to be correlated. The unknown parameters of the RPL model are distributed across the population according to a specified distribution function (McFadden and Train, 2000). In this paper, the RPL has a fixed cost parameter but assumes normally distributed parameters for the other kelp restoration management attributes, with mean?? and standard deviation σ. The fixed cost coefficient was used to avoid convergence issues and to facilitate the calculation of the implicit prices for the restoration attribute levels (Revelt and Train, 1998; Wielgus et al.,

[2] For a more in-depth presentation of the RUM modelling framework and the utility specification of the CL and RPL models the interested reader is directed to Train (2013) and Hynes et al. (2008). For a more descriptive introduction to the approaches see Hanley and Barbier (2009).

	Option A	Option B	No Change
Biodiversity (abundance of macroinvertebrate species) per m2	Medium abundance (max. 75 species).	Medium abundance (max. 75 species).	Low abundance (max. 10 species)
Nurseries for juvenile fish: Juvenile fish abundance per m2	Medium Abundance (max 20 juveniles)	High Abundance (max 30 juveniles)	Low abundance (max 10 juveniles)
Total area of kelp forest restored	20,000m^2 (3 soccer pitches)	40,000m^2 (5.5 soccer pitches)	None
Annual increase in personal income tax	NOK 450	NOK 600	NOK 0

Fig. 2. Example choice card.

2009). The RPL model is estimated by simulated maximum likelihood.

For any of the choice models described above, the marginal utility estimates for changes in the level of each attribute can be converted to the marginal willingness to pay for the particular change in each attribute. These marginal values are derived by dividing the β parameter for an attribute by the β parameter for the price attribute, since the resultant term expresses the scaled marginal utility associated with a change in an attribute in monetary units. In estimating the marginal effects using the RPL the expected measure needs integration over taste distribution in the population which is computed by simulation from draws of the estimated distributions for the random parameters (Scarpa and Thiene, 2005). The marginal willingness to pay calculations allow the researcher to compare the relative importance of changes in one attribute to changes in another attribute within the choice set design (Hynes et al., 2013).

In addition, the welfare impact, as measured by Compensating Variation (CV), of a restoration project that leads to specified changes in ecosystem service provision, as described by multiple changes in the series of attributes, may be calculated (Hoyos, 2010). The average WTP to move from the state of the world given in the baseline (the status quo scenario with the barren marine habitats) to the state of the world that results with the highest level of each attribute in the choice experiment is therefore estimated. Finally, respondents' experiences with environmental goods will influence personal attitudes, as well as the degree on which they benefit from a specific ecosystem service. Furthermore, this may influence their willingness to pay to conserve, or in this case restore, ecosystem features (Krupnick and Adamowicz, 2006). We test for this latter point in the choice models by interacting a dummy variable for active marine users with the attribute levels.

5. Results

Table 2 provides an overview of the mean summary statistics for the sample of the 1102 Norwegian respondents to the survey. The average age in the sample (adults aged 18 plus) is 47 while 47% were female and 64% had a third level qualification. There is a higher number of persons in the sample with third level education that what is in the population. According to Statistics Norway the third level education attainment of the population is 33.4%. Given the on-line nature of the survey this was not an unexpected result. Eleven per cent of the sample were active students, 19% were retired and only 2% indicated that they were currently unemployed. These figures are broadly in line with the Census equivalents for the population.[3] Just under 10% are a member of an environmental organisation of some sort.

Table 2
Summary Statistics.

Variable	Mean	Std. Dev.
Age	47.072	17.657
Female (p)	0.468	0.499
Third level educated (p)	0.644	0.479
Married (p)	0.424	0.494
Single (p)	0.237	0.425
Have children (p)	0.581	0.494
Member of environmental organisation (p)	0.096	0.295
Employed fulltime (p)	0.458	0.498
Employed part time (p)	0.097	0.296
Student (p)	0.114	0.318
Retired (p)	0.190	0.392
Unemployed (p)	0.015	0.123
Household size	2.337	1.179
Has visited seashore in last 12 months (p)	0.625	0.484
Water User: Participates in water activities such as swimming, snorkelling, diving, sailing, boating, canoeing, kayaking, etc. (p)	0.549	0.498
Aware of any Marine Protected areas (MPA) in Norway (p)	0.288	0.453
Aware of any marine restoration activity in Norway (p)	0.086	0.281
Know of any kelp beds/forests (p)	0.196	0.397

p indicates variable is a proportion.

It is interesting to note that in the sample, 62% had visited the seashore in the previous 12 months while a further 55% had participated in water activities such as swimming, snorkelling, diving, sailing, boating, canoeing, kayaking, etc. during that same period. Norwegians also seem to be relatively well informed about marine protected areas with just under 30% of the sample indicating that they knew of an MPA area. Although not shown in Table 2, the survey results also indicated that the average Norwegian takes 7.5 trips to the seashore in the year, 3.8 trips to undertake 'in water' activities such as swimming, snorkelling, diving, and 5.3 trips to participate in 'on water' activities such as sailing, boating, canoeing, kayaking, etc. Only 9% of the sample indicated that they were aware of any marine ecosystem restoration

[3] Census information for Norway was obtained from Statistics Norway, the national statistical institute of Norway and the main producer of official statistics in the country (https://www.ssb.no/en/).

Table 3
Conditional Logit Model Results.

	Coef.	Std. Err.
Biodiversity: High abundance (approx. 250 species).	0.591*	(0.066)
Biodiversity: Medium abundance (approx. 75 species).	0.441*	(0.069)
Nurseries for juvenile fish: High Abundance(max 30 juveniles)	0.249*	(0.064)
Nurseries for juvenile fish: Medium Abundance (max 20 juveniles)	0.251*	(0.068)
Area Restored: 40,000 m^2 (5.5 soccer pitches)	0.341*	(0.087)
Area Restored: 20,000 m^2 (3 soccer pitches)	0.245*	(0.086)
Area Restored: 10,000 m^2 (1.5 soccer pitches)	−0.117	(0.086)
Cost	−0.022*	(0.001)
Interaction terms		
Water User*Biodiversity: High abundance (approx. 250 species).	0.012	(0.087)
Water User*Biodiversity: Medium abundance (approx. 75 species).	−0.053	(0.092)
Water User* Nurseries for juvenile fish: High Abundance(max 30 juveniles)	0.228*	(0.085)
Water User* Nurseries for juvenile fish: Medium Abundance (max 20 juveniles)	0.101	(0.090)
Water User*Area Restored: 40,000 m^2 (5.5 soccer pitches)	0.455*	(0.103)
Water User*Area Restored: 20,000 m^2 (3 soccer pitches)	0.358*	(0.108)
Water User*Area Restored: 10,000 m^2 (1.5 soccer pitches)	0.283*	(0.109)
Log likelihood	−6752	
Likelihood Ratio chi^2 (15)	918.000	
Observations	19,692	

Notes: Figures in parenthesis indicate the values of the standard errors. *indicates significant at 1%.

activity in Norway while 20% indicate that they were aware of kelp forests in Norwegian waters. More direct contact with the environmental good under investigation may influence respondents' willingness to pay to conserve, or in this case restore, ecosystem features (Krupnick and Adamowicz, 2006). We test for this in the choice models by interacting a dummy variable for active marine users with the attribute levels.

The results from the CL model are presented in Table 3. For the analysis, we restricted the sample to those respondents who did not serially choose the status quo option as a protest response; this left a sample size of 1094 respondents. The main protest reasons indicated by respondents as to why they always chose the status quo option was because they objected to paying taxes (1 respondent); for a further 3 individuals they stated that the government/local council should pay while a further 3 indicated that the reason they choose the status quo option on all 6 choice occasions was that they did not believe the restorations would actually take place. A number of the status quo picking respondents indicated that the reason they choose the status quo option was because restorations were not important to them or that the no change option was satisfactory (6 and 11 respondents respectively). These were considered legitimate reasons and these individuals were not excluded from the analysis.

For the *Biodiversity* and *Nurseries for Juvenile Fish* attributes, the level against which these estimates are compared in all models is the low abundance levels in each case (10 species or 10 juveniles, respectively). In terms of the *Area Restored* attribute the base level is *no area restored.* Restoration attributes and all associated levels were summarized in Table 1. As shown in Table 3, the magnitude and signs of the attribute coefficients in the CL model are in line with expectations. In particular respondents show a stronger preference for higher levels of biodiversity, area restored and juvenile fish abundance. In the latter case though, the medium level has a marginally higher coefficient than the high abundance level dummy. As expected the coefficient on cost is negative and significant, suggesting that ceteris paribus, respondents prefer to pay lower amounts of additional taxation. The attribute level dummies were also interacted with a binary variable that indicates whether a person participates in marine related activities that involve being in the water or on top of the water (henceforth referred to as a water user). The results highlight that being a water user is a positive and significant predictor of choosing a restoration option involving a larger area and for a policy option associated with the highest abundance level of juvenile fish.

Table 4 presents the results from the RPL model. A Hausman test showed that, as is usually the case with choice experiments, there was a breach of the IIA assumption in the CL model which suggests the need for an alternative specification such as the RPL model that relaxes this assumption. The parameters for the cost attribute and the water user interaction terms are specified as fixed. The fixed cost attribute is a restrictive assumption as it implies that the marginal disutility of income is the same for all respondents but similar to many previous valuation exercises using this model specification we do so to facilitate the calculation of welfare effects and reduce the possibility of retrieving extreme welfare estimates.

As is evident from Table 4 both the means and the standard deviations are significant for all random parameters bar the lower level area restored parameter associated with 10,000m^2 where the mean parameter is insignificant. The mean coefficients show the same pattern as in the CL case except for the area restored attribute levels. In this case the medium area restored has a higher mean coefficient than the highest level offered in the choice cards. There is however a wide distribution in the preferences for this attribute as seen in the magnitude of the standard deviation coefficients. In fact the size of the standard deviation coefficients relative to the mean values across all attributes indicates that there is substantial heterogeneity in the preference of respondents.

The largest standard deviation coefficient is associated with the high abundance level of biodiversity. A possible explanation for the strength of diversity/heterogeneity in the preferences for this attribute may be that some respondents believe that such abundance levels involve invasive species in the marine environment or that it may interfere with other uses of the area. Norway, whose economy is very reliant on sea related industries such as oil and gas, sea fisheries and aquaculture may have less interest in high levels of biodiversity and more interest in the protection of juvenile commercial fish species for example.

Examining the interacted non-random parameter coefficients, we find that being a water user has no significant effect on a person's preference for high biodiversity levels but once again is found to positively influence preference for the highest level of juvenile fish abundance. Also, and as in the CL model, water users show a stronger preference for larger areas of kelp forest restoration than non-water users.

In Table 5, the marginal WTP estimates calculated based on both the CL model and random parameter logit models are presented along with their 95% confidence intervals. The marginal values have been

Table 4
Random Parameter Logit Model Results.

Random parameters in utility functions	Mean of coefficient	Standard deviation of coefficient
Biodiversity: High abundance (approx. 250 species).	0.618* (0.142)	2.129*(0.123)
Biodiversity: Medium abundance (approx. 75 species).	0.645* (0.110)	1.048*(0.120)
Nurseries for juvenile fish: High Abundance(max 30 juveniles)	0.312** (0.128)	1.485*(0.123)
Nurseries for juvenile fish: Medium Abundance (max 20 juveniles)	0.326* (0.111)	0.954*(0.125)
Area Restored: 40,000 m^2 (5.5 soccer pitches)	0.356** (0.171)	2.461*(0.136)
Area Restored: 20,000 m^2 (3 soccer pitches)	0.485* (0.138)	1.395*(0.117)
Area Restored: 10,000 m^2 (1.5 soccer pitches)	0.006 (0.129)	−1.046*(0.121)
Non-Random Parameters in Utility Functions		
Cost	−0.035* (0.002)	
Water User*Biodiversity: High abundance (approx. 250 species).	0.293 (0.181)	
Water User*Biodiversity: Medium abundance (approx. 75 species).	−0.026 (0.147)	
Water User* Nurseries for juvenile fish: High Abundance(max 30 juveniles)	0.427** (0.169)	
Water User* Nurseries for juvenile fish: Medium Abundance (max 20 juveniles)	0.108 (0.149)	
Water User*Area Restored: 40,000 m^2 (5.5 soccer pitches)	0.813* (0.208)	
Water User*Area Restored: 20,000 m^2 (3 soccer pitches)	0.362** (0.172)	
Water User*Area Restored: 10,000 m^2 (1.5 soccer pitches)	0.318** (0.156)	
Log likelihood	−6034	
Likelihood Ratio chi^2 (7)	1434.82	
Observations	19,692	

Figures in parenthesis indicate the values of the standard errors. *indicates significant at 1%, ** indicates significant at 5%.

Table 5
Attribute marginal willingness to pay and 95% confidence intervals (€ per person per year).

Attribute level	Conditional logit	Random parameter logit
Biodiversity: high abundance (approx. 250 species)	26.94* (22.36 31.53)	23.11* (17.33, 28.89)
Biodiversity: medium abundance (approx. 75 species)	18.56* (14.04, 23.08)	15.92* (11.46, 20.38)
Nurseries for juvenile fish: High Abundance(max 30 juveniles)	17.07* (13.03, 21.12)	15.32* (10.05, 20.60)
Nurseries for juvenile fish: Medium Abundance(max 20 juveniles)	13.97* (9.60, 18.34)	12.14* (7.54, 16.73)
Area restored: 40,000m^2 (5.5 soccer pitches)	26.05* (21.42, 30.68)	20.68* (14.74,26.62)
Area restored: 20,000m^2 (3 soccer pitches)	19.36* (14.63, 24.09)	21.22* (16.43, 26.01)
Area restored: 10,000m^2 (1.5 soccer pitches)	0.93 (−4.64, 6.52)	5.78** (0.54, 11.02)

Figures in parenthesis indicate 95% confidence intervals. ***indicates significant at 1%, ** indicates significant at 5%.

estimated using the Krinsky and Robb (1986) procedure.[4] The estimates produced by the two models are similar albeit those associated with the RPL model are smaller in magnitude than those from the CL. The highest estimated marginal WTP figure is for a high abundance of biodiversity in both models (€26.94 and €23.11 respectively), followed by the highest possible level for area restored in the CL model (€26.05) but the moderate level for area restored in the RPL model (€20.68). The lowest level of area restored (1.5 soccer pitches) is associated with the lowest marginal WTP in both models.

The results in Table 6 present the estimates of the compensating surplus (CS) associated with two possible kelp forest restoration projects. The first project results in the conversion of barren grounds from the lowest ecosystem service levels of the attributes, as shown in the status quo alternative on each choice card to the highest level of each attributes. This could be considered a fully restored kelp forest ecosystem. That is, the CS measure associated with the best standards of all the attributes. We also estimate the compensating surplus associated with a kelp forest restoration project that achieves the medium levels of all the attributes.

As expected, due to the magnitude of the coefficient estimates in Tables 3 and 4, the results show that the estimated compensating surplus measures are higher for the CL model compared to the RPL model. However, the estimates are not significantly different between the models as indicated by the overlapping confidence intervals. The welfare impact for scenario 1 (full restoration to the highest possible level of all attributes) is significantly larger than for the medium level restoration of scenario 2 based on the results of the CL model (€70.70 versus €51.89). The difference is not as great in absolute terms (or statistically) when the RPL results are used to estimate the scenario welfare effects.

6. Discussion and conclusions

Ecosystem restoration implies policies that focus on remediating environmental degradation (where possible) or other costs imposed on society. Restoration in the marine environment however can be particularly challenging and there are various factors that influence the success of marine ecosystem restoration activities. Van Dover et al. (2014) list the three main categories of factors; that is socio-economic, ecological and technological factors. Ecosystem service benefit value estimation can be used to highlight the societal impacts of restoration activities and thus needs to be considered in the restoration planning process. Though estimation of implementation and maintenance costs has been largely recognised and implemented in restoration planning and decision making, it is less clear if the economic value to society associated with marine restoration projects, through an increase in ecosystem service benefits, are being taking fully into account (Papadopoulou et al., 2017).

Although the restoration of degraded marine ecosystems can be expensive and a lengthy process, as Mitsch (2014) points out working 'with' nature and using simple ecological engineering approaches may provide cost-effective solutions to a range of societal challenges. In relation to kelp restoration in particular Fredriksen et al., 2020

[4] The Krinsky-Robb procedure estimates the empirical distribution of the WTP estimates based on N random drawings from the multivariate normal distribution defined by the coefficients and covariance matrix estimated from the model (Krinsky and Robb, 1986). This technique is used as it allows for the skewness of the distribution of the marginal WTP estimates.

Table 6
Attribute levels and compensating surplus value estimates for two policy scenarios (€ per person per year).

	Status Quo	Scenario 1 (Full restoration)	Scenario 2 (Medium level restoration)
Biodiversity	Low abundance (max. 10 species)	High abundance (approx. 250 species).	Medium abundance (approx. 75 species).
Nurseries for juvenile fish	Low abundance (max 10 juveniles)	High Abundance(max 30 juveniles)	Medium Abundance (max 20 juveniles)
Area restored	None	40,000 m^2 (5.5 soccer pitches)	20,000 m^2 (3 soccer pitches)
Compensating Surplus (€/ person/year)			
Conditional logit		70.7 * (64.51, 75.63)	51.89 * (46.94, 56.85)
Random parameter logit		59.12* (51.29, 66.94)	49.28* (43.72, 54.84)

Figures in parenthesis indicate 95% confidence intervals. ***indicates significant at 1%, ** indicates significant at 5%.

compared alternative techniques and concluded that surface methods for collection of reproductive donor material from source kelp populations, and kelp seeded on 'green gravel' that was dropped from the surface grew as effectively as more costly and labour intensive diver based approaches. They also contend that the costs of up-scaling from pilot projects has been underestimated for many diving-intensive restoration approaches. From a welfare maximization perspective it is important to consider if the ecosystem service benefits of the restoration outweigh the costs associated with these alternative approaches. With that in mind the objective of this paper was to assess the Norwegian population's preferences and willingness to pay for kelp forest restoration activities. To achieve this a discrete choice experiment was carried out amongst the population using an online survey.

Compared to cost effectiveness analysis which may skew the decision making towards smaller size restoration activities, economic valuation of marine ecosystem services can provide a more comprehensive picture in terms of restoration policy efficiency and economic trade-offs between alternative marine environment investments in general (Koundouri et al., 2017) and marine restoration scenarios in particular (Börger et al., 2014; Yin et al., 2013; Papadopoulou et al., 2017). Furthermore, while the value of some of the services provided by kelp forest restoration, such as increased fish populations, are somewhat easier to measure as they have established market prices, many of the benefits from such restoration, such as carbon sequestration, increased biodiversity and possible recreation opportunities, are not generally traded in markets and therefore do not generally command a price. Without incorporating these non-market values into the decision making processes these benefits may be ignored and decisions made that are not in fact in the interest of society. The choice experiment employed here estimated the value to society of a number of the non-market ecosystem services associated with kelp forest restoration.

The results from the analysis in this paper indicate a positive and significant marginal societal willingness to pay for the ecosystem services associated with kelp forest restoration. This result is consistent with prior studies that have valued other types of ecosystem restoration activities. It is difficult however to contextualise the magnitude of the results of this study as there are very few other studies that examine WTP for marine restoration efforts. Also, given that Norwegians enjoy one of the highest income per capita in the world along with a highly developed social welfare system there is a reasonable expectation that valuation studies from there will display a higher average WTP than other country studies. The strong association between Norwegians and the sea, as well as the fact that a larger proportion of the country's wealth is generated from sea based activity compared to most other developed countries will also have an influence on WTP for marine ecosystem restoration and add to the difficulty of comparing across studies internationally. Socio-demographic characteristics and experience with the sea through different mediums across different countries have been found to influence people's perceptions and willingness to pay in previous research (Zander and Feucht, 2018). As shown by Hynes et al. (2018), cultural differences can also significantly affect the transfer of a particular valuation exercise from one context to another. If the estimates here were to be used in value transfer these differences in culture, real incomes and attitudes would need to be accounted for.

The modelling output was used to explore how the Norwegian public make trade-offs across the identified attributes. Understanding attribute trade-off provides insight into non-market values that allow policy makers to prioritise projects based on their prospective delivery of these attributes. The results indicate that respondents placed a higher value on a high level of biodiversity relative to all other attributes, with the result consistent across various model specifications. This is an interesting result when compared with the relative importance of the attribute nurseries for juvenile fish as the nurseries function has a more tangible direct commercial impact through fisheries than general biodiversity increases. This may be reflective of increasing societal awareness of the importance of biodiversity amongst the Norwegian population which, as reported by Kaltenborn et al. (2016), is higher than the EU average.

Taste heterogeneity across the population for the various attributes was also examined. Firstly, systematic heterogeneity associated with direct use of the marine environment was examined by interacting the water user variable with the non-cost attributes. Being a water-user is a positive and significant predictor of an individual's preference for larger areas of kelp forest restoration and for high juvenile fish abundance. Users appear to have the same mean preferences for the other ecosystem service attribute levels as the rest of the population. It would appear that the size of the 'playground' is a key driver of this group's preferences when it comes to restoration activities. Indeed, while respondents generally displayed the highest marginal WTP for the high abundance biodiversity level, for water users the highest marginal WTP is instead shown for the highest possible area to be restored (40,000m^2 or 5.5 soccer pitches).

The second approach to examining the taste heterogeneity across the population was via the specification of the RPL model itself. The level of taste variation in the population's preferences for a particular attribute is approximated under the RPL model by examining the relationship between the estimated coefficients and their associated standard deviation coefficients. The results demonstrated significant unobserved preference heterogeneity amongst the Norwegian population for the attributes associated with kelp forest restoration with the highest variation associated with the biodiversity attribute. While only a limited number of attributes could be included in the choice experiment it would be interesting to explore respondents' marginal willingness to pay for other ecosystem service benefits generated by healthy kelp ecosystems. This could also be done within the framework of a choice experiment or, as the valuation literature expands in the area of marine ecosystem services, through a meta-analysis of valuation studies. This represents an avenue for future research.

The magnitude of the mean coefficient values for the biodiversity levels and the interaction terms do potentially point to how one might optimally prioritise kelp forest restoration. One may want to restore kelp over larger distances of coastline but with several smaller restoration projects, in order to secure species that are more comfortable in the south, north and central coast, rather than a single large restoration area in one place. Alternatively, near large urban centres or where a marine area is expected to see a high frequency of use by recreationalists a better strategy may be to restore the largest area possible in the one place. However, as pointed out by a reviewer on an earlier draft of the paper, these alternatives may not be identical in their biological outcomes and in particular many small areas may not deliver the desired biota and ecosystem services as one large one. How 'small' restoration projects can be and still deliver the required outcomes is also an area for further research.

It is important to note that while the results show a positive and significant societal benefit associated with kelp restoration, the derived estimates of WTP do not reflect the total derived ecosystem service benefits of kelp forest restoration. There are other service values that should also be taken into account by policy makers. For example, restored kelp forests act as carbon sinks and also provide coastal protection services by dampening the impact of storm surges. This limitation aside, the results here do give an indication of the possible benefit value from marine ecosystem restoration and a better understanding of societal preferences for alternative kelp forest policy options. As marine environmental policy shifts away from objectives solely related to ecosystem conservation to a position where the reality of the requirement for marine ecosystem restoration is fully recognised, information related to potential changes in all ecosystem service benefit values due to restoration activity will be vital for efficient decision making.

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.

Acknowledgements

This study has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 689518, project MERCES (Marine Ecosystem Restoration in Changing European Seas), the Fram Centre Flagship MIKON project ECOURCHIN (2015-2017) and the Norwegian Research Council project OPTIMAKELP (2018-2021) Project Number: 280732. The authors would also like to acknowledge the assistance of Hartvig Christie, Cristina Gambi and Roberto Danovaro in the preparation of the choice attributes and questionnaire and Hege Gundersen for the production of Fig. 1.

References

Aanesen, M., Falk-Andersson, J., Vondolia, G., Borch, T., Navrud, S., Tinch, D., 2018. Valuing coastal recreation and the visual intrusion from commercial activities in Arctic Norway. Ocean Coast. Manag. 153, 157–167.

Araújo, R., Assis, J., Aguillar, R., Airoldi, L., Bárbara, I., Bekkby, T., Christie, H., Davoult, D., Derrien-Courtel, S., Fernandez, C., Fredriksen, S., Gevaert, F., Gundersen, H., Le Gal, A., Lévêque, L., Mieszkowska, N., Norderhaug, K.M., Oliveira, P., Puente, A., Rico, J.M., Rinde, E., Schubert, H., Strain, E.M., Valero, M., Viard, F., Sousa-Pinto, I., 2016. Status, trends and drivers of kelp forests in Europe: an expert assessment. Biodivers. Conserv. 25, 1319–1348.

Armstrong, C., Kahui, V., Vondolia, G., Aanesen, M., Czajkowski, M., 2017. Use and non-use values in an applied bioeconomic model of fisheries and habitat connections. Mar. Resour. Econ. 32 (4), 351–369.

Bennett, S., Wernberg, T., Connell, S., Hobday, A., Johnson, C., Poloczanska, E., 2016. The 'great southern reef': social, ecological and economic value of Australia's neglected kelp forests. Mar. Freshw. Res. 67, 47–56.

Blamey, L., Bolton, J., 2018. The economic value of south African kelp forests and temperate reefs: past, present and future. J. Mar. Syst. 188, 172–181.

Borger, T., Piwowarczyk, J., 2016. Assessing non-market benefits of seagrass restoration in the Gulf of Gdansk. J. Ocean Coast. Econ. 3 (1) Article1.

Börger, T., Beaumont, N., Pendleton, L., Boyle, K., Cooper, P., Fletcher, S., Haab, T., Hanemann, M., Hooper, T., Hussain, S., Portela, R., Stithou, M., Stockill, J., Taylor, T., Austen, M., 2014. Incorporating ecosystem services in marine planning: the role of valuation. Mar. Policy 46, 161–170.

Christie, H., Norderhaug, K., Fredriksen, S., 2009. Macrophytes as habitat for fauna. Mar. Ecol. Prog. Ser. 396, 221–233.

Claisse, J., Williams, J., Ford, T., Pondella II, D., Meux, B., Protopapadakis, L., 2013. Kelp forest habitat restoration has the potential to increase sea urchin gonad biomass. Ecosphere 4, 38.

de Rutger, W., Helene, R.V., Juliette, B., Vincent, O., Robert, L., 2017. Restoration ecology of coastal lagoons: new methods for the prediction of ecological trajectories and economic valuation. Aquat. Conserv. 27 (1), 137–157.

De Wit, R., Rey-Valette, H., Balavoine, J., Ouisse, V., Lifran, R., 2017. Restoration ecology of coastal lagoons: new methods for the prediction of ecological trajectories and economic valuation. Aquat. Conserv. 27 (1), 137–157.

Deely, J., Hynes, S., 2020. Blue-green or grey, how much is the public willing to pay? Landsc. Urban Plan. 203, 103909.

Dissanayake, S., Ando, A., 2014. Valuing grassland restoration: proximity to substitutes and trade-offs among conservation attributes. Land Econ. 90 (2), 237–259.

Endo, I., Walton, M., Chae, S., Park, G.-S., 2012. Estimating benefits of improving water quality in the largest remaining tidal flat in South Korea. Wetlands 32, 487–496.

Esteves, L., Williams, J., 2017. Managed realignment in Europe: a synthesis of methods, achievements and challenges. In: Bilkovic, D.M., Mitchell, M.M., Toft, J.D., La Peyre, M.K. (Eds.), Living Shorelines: The Science and Management of Nature-based Coastal Protection. CRC Press/Taylor & Francis Group, pp. 157–180.

Filbee-Dexter, K., Scheibling, R., 2014. Sea urchin barrens as alternative stable states of collapsed kelp ecosystems. Mar. Ecol. Prog. Ser. 495, 1–25.

Filbee-Dexter, K., Wernberg, T., 2018. Rise of turfs: a new battlefront for globally declining kelp forests. BioScience 68 (2), 64–76.

Filbee-Dexter, K., Wernberg, T., 2020. Substantial blue carbon in overlooked Australian kelp forests. Sci. Rep. UK 10, 12341.

Fredriksen, S., Filbee-Dexter, K., Norderhaug, K., Steen, H., Bodvin, T., Coleman, M., Moy, F., Wernberg, T., 2020. Green gravel: a novel restoration tool to combat kelp forests decline. Sci. Rep. UK 10, 3983.

Ghofrani, Z., Sposito, V., Faggian, R., 2017. A comprehensive review of blue-green infrastructure concepts. Int. J. Environ. Sustain. 6 (1), 15–36.

Glenk, K., Martin-Ortega, J., 2018. The economics of peatland restoration. J. Environ. Econ. Policy 7 (4), 345–362.

Graham, M., 2004. Effects of local deforestation on the diversity and structure of southern California giant kelp forest food webs. Ecosystems 7, 341–357.

Grazhdani, D., 2013. Applying contingent valuation survey to assess the economic value of restoring ecosystem services of impaired rivers: a case study in transboundary Buna River region, Albania. Int. J. Innov. Res. Sci. Eng. Technol. 2 (10), 5115–5123.

Gundersen, H., Christie, H., Rinde, E., 2010. Sea urchins – from problem to commercial resource. Estimates of sea urchins as a resources and an evaluation of ecological gains by sea urchin exploitation. NIVA report no. 6001–2010.

Gundersen, H., Bryan, T., Chen, W., Moy, F., Sandman, A., Sundblad, G., Schneider, S., Andersen, J., Langaas, S., Walday, M., 2016. Ecosystem Services in the Coastal Zone of the Nordic Countries. Report for Nordic Council of Ministers. TemaNord 2016. 552 ISSN 0908-6692.

Hanley, N., Barbier, E., 2009. Pricing Nature: Cost-Benefit Analysis and Environmental Policy. Edward Elgar Publishing Inc.

Hess, S., Smith, C., Falzarano, S., Stubits, J., 2008. Managed-lanes stated preference survey in Atlanta, Georgia: measuring effects of different experimental designs and survey administration methods. Transp. Res. Rec. 2049, 144–152.

Hoyos, D., 2010. The state of the art of environmental valuation with discrete choice experiments. Ecol. Econ. 69, 1595–1603.

Hynes, S., Hanley, N., Scarpa, R., 2008. Effects on welfare measures of alternative means of accounting for preference heterogeneity in recreational demand models. Am. J. Agric. Econ. 90 (4), 1011–1027.

Hynes, S., Tinch, D., Hanley, N., 2013. Valuing improvements to coastal waters using choice experiments: an application to revisions of the EU bathing waters directive. Mar. Policy 40, 137–144.

Hynes, S., Ghermandi, A., Norton, D., Williams, H., 2018. Marine recreational ecosystem service value estimation: a meta-analysis with cultural considerations. Ecosyst. Serv. 31, 410–419.

Jenkins, W., Murray, B., Kramer, R., Faulkner, S., 2010. Valuing ecosystem services from wetlands restoration in the Mississippi Alluvial Valley. Ecol. Econ. 15 (5), 1051–1061.

Jobstvogt, N., Hanley, N., Hynes, S., Kenter, J., Whitte, U., 2014. Twenty thousand Sterling under the sea: estimating the value of protecting deep-sea biodiversity. Ecol. Econ. 97, 10–19.

Kaltenborn, B., Gundersen, V., Stange, E., Hagen, D., Skogen, K., 2016. Public perceptions of biodiversity in Norway: from recognition to stewardship? Nor. Geogr. Tidsskr. 70, 54–61.

Kim, J., Lim, S.Y., Yoo, S.H., 2017. Public willingness to pay for restoring destroyed tidal flats and utilizing them as ecological resources in Korea. Ocean Coast. Manag. 142, 143–149.

Koundouri, P., Chen, W., Dávila, O.G., Giannouli, A., Brito, J., Kotoroni, E., Mailli, E., Mintenbeck, K., Papagianni, C., Souliotis, I., 2017. In: Nunes, Paulo A.L.D., Kumar, Pushpam, Svensson, Lisa E., Markandy, Anil (Eds.), A Socio-Economic Framework for Integrating Multi-Use Offshore Platforms in Sustainable Blue Growth Management: Theory and Applications, Handbook on the Economics and Management for Sustainable Oceans.

Krinsky, I., Robb, I., 1986. On approximating the statistical properties of elasticities. Rev. Econ. Stat. 68, 715–719.

Krumhansl, K., Scheibling, R., 2012. Production and fate of kelp detritus. Mar. Ecol. Prog. Ser. 467, 281–302.

Wessel, P., Smith, W., 2016. Global patterns of kelp forest change over the past half--century. PNAS 113 (48), 13785–13790. https://doi.org/10.1073/pnas.1606102113.

Krupnick, A., Adamowicz, W., 2006. Supporting questions in stated choice studies. In: Kanninen, B. (Ed.), Valuing Environmental Amenities Using Stated Choice Studies. Springer, pp. 43–65.

Layton, C., Coleman, M.A., Marzinelli, E.M., Steinberg, P.D., Swearer, S.E., Vergés, A., Wernberg, T., Johnson, C.R., 2020. Kelp forest restoration in Australia. Front. Mar. Sci. https://doi.org/10.1371/journal.pone.0224477.
Leinaas, H., Christie, H., 1996. Effects of removing sea urchins (Strongylocentrotus droebachiensis): stability of the barren state and succession of kelp forest recovery in the East Atlantic. Oecologia 105, 524–536.
Loomis, J., Kent, P., Strange, L., Fausch, K., Covich, A., 2000. Measuring the total economic value of restoring ecosystem services in an impaired river basin: results from a contingent valuation survey. Ecol. Econ. 33 (1), 103–117.
Lovas, S., Tørum, A., 2001. Effect of the kelp Laminaria hyperborea upon sand dune erosion and water particle velocities. Coast. Eng. 44, 37–63.
Luisetti, T., Turner, R., Bateman, I., Morse-Jones, S., Adams, C., Fonseca, L., 2011. Coastal and marine ecosystem services valuation for policy and management: managed realignment case studies in England. Ocean Coast. Manag. 54 (3), 212–224.
McFadden, D., 1974. Conditional logit analysis of qualitative choice behavior. In: Zarembka, P. (Ed.), Frontiers in Econometrics. Academic Press, New York.
McFadden, D., Train, K., 2000. Mixed MNL models for discrete response. J. Appl. Econ. 15, 447–470.
Milon, J., Scrogin, D., 2006. Latent preferences and valuation of wetland ecosystem restoration. Ecol. Econ. 56 (2), 162–175.
Mitsch, W., 2014. When will ecologists learn engineering and engineers learn ecology? Ecol. Eng. 65, 9–14.
Norderhaug, K., Christie, H., 2009. Sea urchin grazing and kelp revegetation in the NE Atlantic. Mar. Biol. Res. 5, 515–528.
Norderhaug, K., Christie, H., Fosså, J., Fredriksen, S., 2005. Fish-macrofauna interactions in a kelp (Laminaria hyperborea) forest. J. Mar. Biol. Assoc. UK 85, 1279–1286.
Norton, D., Hynes, S., 2014. Valuing the non-market benefits arising from the implementation of the EU marine strategy framework directive. Ecosyst. Serv. 10, 84–96.
O'Connor, E., Hynes, S., Chen, W., 2020. Estimating the non-market benefit value of deep-sea ecosystem restoration: evidence from a contingent valuation study of the Dohrn canyon in the Bay of Naples. J. Environ. Manag. 275, 111180. https://doi.org/10.1016/j.jenvman.2020.111180.
Ojeda, M.I., Mayer, A.S., Solomon, B.D., 2008. Economic valuation of environmental services sustained by water flows in the Yaqui River Delta. Ecol. Econ. 65 (1), 155–166.
Papadopoulou, N., Sevastou, K., Smith, C., Gerovasileiou, V., Dailianis, T., Fraschetti, S., Guarnieri, G., McOwen, C., Billett, D., Grehan, A., Bakran-Petricioli, T., Bekkby, T., Bilan, M., Boström, C., Carriero-Silva, M., Carugati, L., Cebrian, E., Cerrano, C., Danovaro, R., Eronat, E., Gagnon, K., Gambi, C., Kipson, S., Kizilkaya, I., Kotta, J., Linares, C., Milanese, M., Morato, T., Papa, L., Rinde, E., Sarà, A., 2017. State of the knowledge on marine habitat restoration and literature review on the economic costs and benefits of ecosystem service restoration. EU MERCES project report.
Park, S.-Y., Yoo, S.-H., Kwak, S.J., 2013. The conservation value of the Shinan tidal flat in Korea: a contingent valuation study. Int J Sust Dev World 20, 54–62.
Pascual, U., Muradian, R., Brander, L., Gómez-Baggethun, E., Martín-López, M., 2010. The economics of valuing ecosystem services and biodiversity. In: Kumar, P. (Ed.), The Economics of Ecosystems and Biodiversity (TEEB) Ecological and Economic Foundations. Earthscan, London and Washington, pp. 183–256.
Pendleton, L., 2010. Measuring and monitoring the economic effects of habitat restoration: a summary of a NOAA blue ribbon panel (Nicholas Institute for Environmental Policy Solutions, Restore America's Estuaries, and National Oceanic and Atmospheric Administration, 2010). http://www.era.noaa.gov/pdfs/NOAA%20RAE%20BRP%20Estuary%20Economics_FINAL.pdf.
Rebours, C., Marinho-Soriano, E., Zertuche-Gonzalez, J., Hayashi, L., Vasquez, J., Kradolfer, P., Soriano, G., Ugarte, R., Abreu, M., Bay-Larsen, I., Hovelsrud, G., Rødven, R., Robledo, D., 2014. Seaweeds: an opportunity for wealth and sustainable livelihood for coastal communities. J. Appl. Ecol. 26, 1939–1951.
Revelt, D., Train, K., 1998. Mixed logit with repeated choices: Households' choices of appliance efficiency level. Rev. Econ. Stat. 80, 647–657.
Scarpa, R., Rose, J., 2008. Design efficiency for non-market valuation with choice modelling: how to measure it, what to report and why. Aust. J. Agr. Resour. Econ. 52, 253–282.
Scarpa, R., Thiene, M., 2005. Destination choice models for rock-climbing in the north-eastern Alps: a latent-class approach based on intensity of participation. Land Econ. 81, 426–444.
Smale, D., Burrows, M., Evans, A., King, N., Yunnie, A., Moore, P., 2016. Linking environmental variables with regional-scale variability in ecological structure and standing stock of carbon within kelp forests in the United Kingdom. Mar. Ecol. Prog. Ser. 542, 79–95.
Stone, K., Bhat, M., Bhatta, R., Mathews, A., 2008. Factors influencing community participation in mangroves restoration: a contingent valuation analysis. Ocean Coast. Manag. 51 (6), 476–484.
Tamburello, L., Papa, L., Guarnieri, G., Basconi, L., Zampardi, S., Scipione, M., Terlizzi, A., Zupo, V., Fraschetti, S., 2019. Are we ready for scaling up restoration actions? An insight from Mediterranean macroalgal canopies. PLoS One 14 (10), e0224477.
Teagle, H., Hawkins, S., Moore, P., Smale, D., 2017. The role of kelp species as biogenic habitat formers in coastal marine ecosystems. J. Exp. Mar. Biol. Ecol. 492, 81–98.
Tonin, S., 2018. Economic value of marine biodiversity improvement in coralligenous habitats. Ecol. Indic. 85, 1121–1132.
Train, K., 2003. Discrete Choice Methods with Simulations. Cambridge University Press, New York.
Van Dover, C.L., Aronson, J., Pendleton, L., Smith, S., Arnaud-Haond, S., Moreno-Mateos, D., Barbier, E., Billett, D., Bowers, K., Danovaro, R., Edwards, A., Kellert, S., Morato, T., Pollard, E., Rogers, A., Warner, R., 2014. Ecological restoration in the deep sea: desiderata. Mar. Policy 44, 98–106.
Vasquez, J., Zuniga, S., Tala, F., Piaget, N., Rodriguez, D., Vega, J., 2014. Economic valuation of kelp forests in northern Chile: values of goods and services of the ecosystem. J. Appl. Ecol. 26, 1081–1088.
Wernberg, T., Krumhansl, K., Filbee-Dexter, K., Pedersen, M., 2019. Status and trends for the world's kelp forests. In: Sheppard, C. (Ed.), World Seas: An Environmental Evaluation, Vol III: Ecological Issues and Environmental Impacts. Academic Press, pp. 57–78.
Westerberg, V., Lifran, R., Olsen, S., 2010. To restore or not? A valuation of social and ecological functions of the Marais des Baux wetland in southern France. Ecol. Econ. 69 (12), 2383–2393.
Wielgus, J., Gerber, L., Sala, E., Bennett, J., 2009. Including risk in stated-preference economic valuations: experiments on choices for marine recreation. J. Environ. Manag. 90, 3401–3409.
Yin, R., Liu, T., Yao, S., Zhao, M., 2013. Designing and implementing payments for ecosystem services programs: lessons learned from China's cropland restoration experience. Forest Policy Econ. 35, 66–72.
Zander, K., Feucht, Y., 2018. Consumers' WTP for sustainable seafood made in Europe. J. Int. Food Agribusiness Market. 30 (3), 251–275.