Investigating public preferences for the management of native and invasive species in the context of kelp restoration

Authors:

Isobella Grover,
Tasmanian School of Business & Economics,
University of Tasmania, Private MailBag 84, Sandy Bay, TAS 7001, Australia.
Isobella.Grover@utas.edu.au

Mark Tocock,
Tasmanian School of Business & Economics,
University of Tasmania, Private MailBag 84, Sandy Bay, TAS 7001, Australia.
Mark.Tocock@utas.edu.au

Dugald Tinch,
Centre for Marine Socioecology,
Tasmanian School of Business & Economics,
University of Tasmania, Private MailBag 84, Sandy Bay, TAS 7001, Australia.
Dugald.Tinch@utas.edu.au

Darla Hatton MacDonald,
Centre for Marine Socioecology,
Tasmanian School of Business & Economics, University of Tasmania,
Private MailBag 84, Sandy Bay, TAS 7001, Australia.
Darla.HattonMacDonald@utas.edu.au

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Abstract

The Southern Rock Lobster (*Jasus edwardsii*) is key to management of the Long Spined Sea Urchin (*Centrotephanus rodgersii*). The southern shift of the urchin has created 'urchin barrens' of the kelp forests (*Ecklonia radiata* and *Macrocystis pyrifera*) in coastal waters off the East Coast of Tasmania. Consistently high fishing efforts of large Southern Rock Lobster in the region has reduced natural predation of the urchin. A number of restoration options exist but these options have not been tested for acceptability with the public to date. In this study we estimate the willingness to pay by households in Tasmania for Southern Rock Lobster habitat and management. The willingness to pay estimates can be used in benefit-cost analysis of the various combinations of restoration options supporting decision-making in this Australian marine planning context. Results indicate that direct replanting of kelp is worth $37 per household per year for five years and a 1% increase in the area of marine reserves is worth $2 per household per year for five years. Direct intervention to control sea urchin populations by reintroducing Rock Lobsters or capturing and crushing urchins is worth $33 and $31 respectively per household per year for five years. Households did not support artificial kelp beds installations or subsidising urchin farming operations.

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1 Introduction

Multiple anthropogenic pressures, including climate change, overfishing, coastal development and runoff, have resulted in shifts in the distribution of invasive species and loss of marine plant communities [1,2]. Shifts in major global ocean current systems have led to reduced marine biodiversity as well as reductions in the size and structure of macroalgae-dominated habitats [3], combined with range modifications of key habitat-modifying species [2–5]. These interactions present a set of potential risks affecting the resilience of ecosystems and socio-economic systems [6]. As a result, the management of marine resources becomes increasingly complex, requiring the implementation of a series of resource management programmes and actions. Often sustained action requires public support which can be difficult to secure where the ecological functions and interdependencies are complex, financial costs more immediate and the broader economic values difficult to quantify. The identification of the benefits and values associated with ecosystems can enhance understanding of the level of environmental management that is warranted and would be supported in a given context [7,8]. To this end, this study uses stated preference techniques [9] to investigate the socially acceptable trade-offs in marine resource management using a case study of the Southern Rock Lobster (*Jasus edwardsii*), the Long Spined Sea Urchin (*Centrotephanus rodgersii*) and Kelp forests (common kelp (*Ecklonia radiata*) and giant kelp (*Macrocystis pyrifera*)) in the coastal waters of the East Coast of Tasmania.

The Southern Rock Lobster is an aquatic crustacean, native to the temperate marine environments of South Eastern Australia. Whilst the conservation status of Southern Rock Lobsters is classified as ‘least concern’ under State and Commonwealth legislation in Australia, the species is considered regionally vulnerable in the coastal waters of Tasmania (see Fig. 1). It is estimated that the

Rock Lobster regional population in the North West and the East Coast regions of Tasmania are currently at approximately 10% and 11% of the unfished biomass respectively [10]. The Southern Rock Lobster is an ecologically valuable species within Tasmania's relatively cool, rocky reef, marine environment and is a principal predator of a key invasive species, the Long Spined Sea Urchin [11]. Long Spined Sea Urchins have been associated with catastrophic over-grazing of macroalgae species, resulting in cascading impacts on marine habitats [5,12]. The Long Spined Sea Urchin has undergone southward range expansion as a result of the East Australia Current shifting poleward resulting in a significant reduction in common kelp and giant kelp [12,13]. The reduction of these macroalgae species has established 'urchin barrens' ecosystems throughout Bass Strait and Tasmanian East Coast [5].

Fig 1 about here;

Efforts to control commercial and recreational overfishing of the Southern Rock Lobster through catch and bag limits have been unsuccessful to date [14] with flow-on consequences for kelp habitat. Minimum size limits have served to remove large mature Rock Lobsters and resulted in the remaining individuals being too small to successfully control the Long Spined Sea Urchin population. Although numerous mitigation strategies have been proposed and trialled, including kelp habitat restoration, reintroduction of large Rock Lobster and invasive species control, Rock Lobster stock levels have not yet grown to a level where they can prevent further ecosystem decline. A combination of strategies and sustained effort are required in order to manage ecosystems to achieve the objective of Rock Lobster habitat regeneration and repopulation in East Coast Tasmanian waters. Management options such as the reintroduction of large rock lobsters, the expansion of marine reserves, and alternatives for the management of sea urchins were identified as being applicable to the East Coast Stock Rebuilding Zone (ECSRZ). Both recreational and commercial fishing have been severely limited in the ECSRZ using a cap on the seasonal catch.

While each of the management options have been trialled and shown to be effective in improving habitat and reducing the loss of species, to date there is no definitive evidence in the form of biophysical modelling or trials to judge which combination of policy alternatives is likely to be most effective. However, policy makers often require a better understanding of public acceptance of policy alternatives prior to allocating budgets for larger trials and/or broader suite of actions, particularly in the context of the current study [13].

As such, public support of policies, regulations and management actions is often key to successful implementation. Communication and engagement with stakeholders in the development of knowledge requires ongoing exchanges of information and views. However, as is the case for most environmental management actions, support can be difficult to secure where there is uncertainty over outcomes [15]. Similarly, there are inherent difficulties for decision-makers developing management plans due to the complex relationship between the economic (use) and environmental (non-use), and socio-cultural benefits that coastal ecosystems provide [16,17]. As one step in on-going consultation and engagement, we developed a survey to elicit public preferences for coastal ecosystem management policies that affected the Southern Rock Lobster. Using a species familiar to our respondents, and its habitat, allowed for the development of realistic and tangible resource management options that summarised complex concepts for the respondent. Options described in management plans can then be interpreted in terms of the broad scale protection of ecosystem services and functions to which they relate.

This study addresses the issues identified above by eliciting public preferences for management options identified as having quantifiable influence on a coastal ecosystem. From the family of stated preference techniques, we employ a discrete choice experiment (DCE)[1] [18] to elicit values by constructing potential management scenarios. These values are useful in decision-support

[1] The main arguments for using a DCE, over say a contingent valuation exercise (where a single scenario is considered), is the potential to get more information from each respondent. A DCE yields a series of implicit prices and relative preferences through the consideration of trade-offs among attributes. A DCE is often more useful in the policy process [71,72]

tools such as benefit-cost analysis [19], regulatory impact assessment, and broader assessments of public policy.

1.1 Background Literature

The non-market valuation literature includes a wide and diverse range of applications of revealed and stated preference techniques in marine and terrestrial settings. This literature is motivated, in part, to provide estimates of the benefits and costs associated with changes in environmental quality or ecosystem services. Stated preference techniques are required when the non-use values are of particular importance e.g. the intrinsic value of a habitat area or unfamiliar marine assets [20]. Stated preference techniques can be used to generate welfare measures consistent with the economic theory that underpins benefit-cost analysis. For this reason, much effort has been expended on constructing defensible, replicable estimates [21] with researchers concerned with survey construction, experimental design and the potential for redressing different sources of biases (see Johnston et al., [22] for a recent summary on current issues).

Approaches to eliciting non-use values have taken a number of directions in the stated preference literature with a subset of studies focusing on terrestrial threatened and endangered species (e.g. Cummings et al., [23]), marine mammals (e.g. Loomis et al., [24]; Langford et al., [25]; Boxall et al., [26]) as well as the recovery of freshwater and marine fish species [27,28]. Early in the litany of studies, Loomis and White [29] suggested that a habitat-based approach, rather than threatened and endangered species, could be more useful in decision-support. In particular, the focus on threatened and endangered species may overlook the "ecological complementarity among species … and substitution effects (both in the utility function and in the budget constraint)" [29]. For an overview of non-market valuation studies of threatened and endangered species see Subroy et al., [30].

With the emergence of the ecosystem services framework [31] which summarised information, data and values at a global and sub-continental level, came a concomitant surge in valuation efforts [32,33]. The ecosystem service approach shifted the emphasis from single, threatened and endangered species to valuation exercises which linked biodiversity with landscapes and habitat areas [34]. This literature is also concerned with the communication and integration of ecosystem services and economic values in planning processes such as regional coastal ecosystems in relation to conservation areas e.g. the Great Barrier Reef [35,36], Californian coastal areas [21], Baltic marine areas [37] or the Norwegian sea [2,38]. Emerging marine planning paradigms in the UK and USA for marine planning areas have converged with both countries [39] employing a wide range of non-market values in priority setting and again the unifying theme of environmental benefit-cost analysis [20]. Practical applications remain challenging as the scale of particular ecosystem functions and their related use and non-use values expressed by people may be unrelated to the spatial scale of political units [21] at which data is gathered or policies targeted [40,41].

2 Survey Development

The marine environment treatment presented in this article was a part of a multi-treatment DCE conducted across two Australian states (Tasmania and Victoria). The overall objective of the project was to elicit household preferences for species management in the context of changing habitat conditions and the presence of an invasive species. This study focuses on the Tasmanian sample who participated in the marine environment treatment. Respondents across all treatments were recruited with a general invitation to answer an online survey. The survey was designed to present the resource management problem in plain simple language, viewed as consequential in terms of being paid for through a tax and as an input into public policy development.

2.1 Survey

In the first part of the survey, respondents completed an informed consent section (Ethics Approval H0018065) and faced warm-up questions where they ranked threats to Australian species. Section two provided background information regarding the threats to the Tasmanian Rock Lobster population resulting from the over-harvesting of large Rock Lobsters and a loss of habitat due to Long

Spined Sea Urchins. Respondents were presented with a set of management options that would increase the size of existing marine reserves, control measures for urchin populations and several approaches for re-establishing kelp forests. Each of these management options were presented to respondents as having been successfully demonstrated in a Tasmanian context through small-scale pilot projects. However, due to uncertainties around scaling up of each management option, it was not possible to provide estimates as to the probability of successful reductions to habitat and species loss. To ensure that respondents were carefully reading information, respondents were asked 'true/false' questions at the end of each screen of information to test their understanding of each management option. It was emphasised that no single strategy on its own was likely to be successful in restoring Rock Lobster habitat in Tasmanian waters.

The resource management programmes would have a cost to the household, presented as an increase in taxes over a five-year time frame. Respondents were reminded of their individual budget constraints and a 'cheap talk script', following Morrison and Brown [42]**,** served to remind respondents that they needed to treat each choice as if they really had to pay. Respondents were informed the relative effectiveness of attribute levels to confirm that selecting a higher cost alternative would reflect the selection of management options that may be more effective. A combination of management attributes was presented in six choice tasks completed in section three. Respondents were asked to treat each task as an independent referendum decision. A few debriefing questions followed the choice tasks. Both sections four and five repeated the previous two sections with another species and a new set of choice tasks. The respondents could not go backwards in the survey or change responses once a response was submitted. The sixth section asked a series of socio-demographic questions including age, gender, income, education, and participation in environmental charities. The final section asked a series of attitudinal questions used to capture support for the development of different private/public sector projects in Australia. All choice data analysed in this study are from section three of the survey where management actions for Rock Lobster are presented first.

Rock Lobsters are managed for the purposes of harvesting and their habitat area is monitored for the distribution of invasive species. The management options in the survey were developed based on assessment of the current threats to Rock Lobster habitat in Tasmania in addition to current and proposed management options [43,44]. The options are expected to reduce the threat to the regional Rock Lobster population as well as being viable alternatives in a Tasmanian context. At the time of survey, all management actions presented were deemed feasible but had not been rolled out as a state-wide strategy largely due to cost and uncertainty regarding public support. The options presented have been discussed with local researchers, trialled in Tasmanian waters and shown to lead to ecosystem recovery or trialled with a similar species (e.g. Purple Sea Urchins, *Strongylocentrotus purpuratus*, in northern California/Oregon [45]). The range of management actions, levels associated with the current situation and potential environmental quality improvements are summarised in Table 1.

The management options were presented as quasi-unlabelled alternatives in the conventional language of DCEs in each choice task (see Fig. 2). Option A and B involve a combination of management actions which result in an improvement to the marine environment in terms of reserve area, habitat or predator control while the third option represented the Current Situation or Status Quo (Status Quo alternative, hereafter).

Table 1 about here;

Fig 2 about here;

2.2 Experimental Design

The experimental design for the DCE was generated using Ngene software version 1.01.02 [46]. The design included 48 choice tasks split into eight blocks with each respondent completing six choice tasks. Fig. 2 shows an example of one of the six choice tasks presented to respondents. An initial design was generated utilising priors from the DCE literature (i.e. cost and habitat [47]) and the coefficients on the effects-coded, categorical variables for which no values existed were assigned an initial estimate to balance utility. This experimental design approach is useful for avoiding unrealistic or dominated choices. The initial design was piloted with 50 Tasmanian respondents. Parameter estimates based on the pilot data were then used to update the simulation. The final Bayesian efficient design was selected based on the design with the lowest D-error (D-error = 0.013). The D-error is the determinant of the asymptotic variance- covariance matrix of an associated design. Designs with a lower D-errors are more efficient in that they minimise the standard errors of the parameters included within the design [48] and used as a selection criterion among designs.

3 Econometric Specification

In this study it is assumed that respondents are evaluating resource management policies consistent with the Random Utility Model [49]. The utility U obtained from selecting an alternative (a combination of resource management policies) is composed of both a deterministic component V and an unobservable random component expressed through the error term ε. From every choice situation s, respondent n selects one of j alternatives for the management of Rock Lobsters and habitat area. Each respondent evaluated six unique choice tasks representing different alternatives which represented a vector of policy attributes X_{snj}. By specifying a linear function for the deterministic component, the utility of a particular alternative can be expressed as:

$$U_{snj} = \alpha_{snj} + \beta X_{snj} + \gamma s_n + \varepsilon_{snj} \quad (1)$$

where β is the estimated preference parameters for each of the policy attributes included within each alternative and α is an alternative-specific constant. In this study the Status Quo and middle alternative (Option B) include a constant term to capture unobserved heterogeneity that affects the propensity to select each alternative. The interactions between the Status Quo alternative and sociodemographic characteristics s_n are estimated through the parameter γ. For this study age as a continuous variable as well as three dummy variables are included to explain the propensity to select the Status Quo alternative. The three variables identify respondents who donated to a nature fund in the last year, whether they could correctly identify a Southern Rock Lobster from several images, and if they stated they had a year 10 or below education.

Assuming a logit specification for the error term, the probability of selecting a particular combination of management policies in an alternative can be shown as:

$$P_{snj} = Prob\left(V_{snj} + \varepsilon_{snj} > V_{sni} + \varepsilon_{sni}\right) s \in S, \forall j, j \neq i \quad (2)$$

where P_{snj} is the probability that the utility of a selected alternative is maximised for the respondent based on the specific set of policies on offer. The error terms are assumed to be drawn from an extreme value distribution, which assumes that the terms are identically and independently distributed [50].

A Mixed Multinomial Logit (MMNL) with an error-component is estimated to allow for preference heterogeneity and differences in substitution patterns between alternatives. Both a mean and standard deviation parameter is estimated for each policy attribute (Table 1), with the latter parameter determining the shape of the distribution for each attribute. For each non-cost attribute, a normal distribution is specified, and a lognormal distribution is specified for the cost attribute. The error component is included to address potential correlation in the non-Status Quo alternatives [51]. In addition, full correlation was allowed between all random parameters included in the model. The probability expressed in (2) is estimated using maximum simulated likelihood [52] with 5,000 Modified Latin Hypercube Sampling draws [53].

As the aim of this study is to obtain implicit prices for each of the policy attributes included in the DCE the model is estimated directly in Willingness to Pay (WTP) Space [54,55]. By separating out the cost attribute p_{snj} utility can be expressed as:

$$U_{snj} = -\beta_p p_{snj} + \delta X_{snj} + \gamma s_n + \theta_{n|AB} + \varepsilon_{snj} \tag{3}$$

where β_p is the estimated cost parameter and δ is defined as the ratio of each non-cost attribute marginal utilities with cost, γs_n estimates the effect of sociodemographic factors, and $\theta_{n|AB}$ is the error-component for option A and B. Using this specification, the results reported for each estimated parameter are expressed as implicit prices for marginal changes in attribute levels.[2] The final model was estimated using Python Biogeme [56] with supporting code from Rose and Zhang [57].

4 Results

The survey was administered online through the Online Research Unit (ORU) during December 2019. The sample was stratified by age and gender and restricted to persons over 18 years of age and persons residing at a Tasmanian address. An initial invitation was sent via email and up to three email reminders. Responses were collected from 294 Tasmanian respondents with an overall response rate of 12.6%. Respondents were given financial compensation for completing the survey in the form of

[2] Based on this specification both the alternative-specific constants, the sociodemographic interactions with the status quo alternative, and error-component parameters are also expressed as a ratio with cost.

airline points/shopping cards by ORU. Since each respondent was presented six choice tasks, there were 1,764 observed choices for the entire sample.

The sample demographics are presented in Table 2. There are some differences in the sample compared to the census characteristics. Although our sample was fairly representative of the gender ratio, our respondents were older and had higher annual income than Tasmanian population estimates. The mean sample age is higher than the Tasmania mean ($t=11.67$, $p<0.01$) and the proportion of the sample with an income less than $649 per week is proportionally less than the census population ($t=-5.53$, $p<0.01$) and proportionally more who selected the income group $1,500 to $3,000 or more per week ($t=-3.80$, $p<0.01$). However, it is noted that 17.12% of the sample selected 'prefer not to say' for the income question. The sample also tended to be more educated compared to Tasmanian population estimates, with a higher proportion of post-high school qualifications including Diplomas, Certificates (I, II, III or IV) or Undergraduate degree ($t=4.13$, $p<0.01$) as well as Post Graduate degree/Professional Qualification ($t=16.85$, $p<0.01$).

Table 2 about here;

Table 3 summarises the results for the MMNL estimated in WTP Space. Starting with the mean parameters, WTP with the exception of the *Artificial Kelp Beds* and *Subsidise Sea Urchin Farms* attributes are positive and statistically significant. The most preferred policy overall was the *Replanting Kelp Beds*, with an average WTP of $37.29 for every replanted bed. The *Reintroduction of Rock Lobsters* was the preferred policy for controlling predator populations, followed by the policy *Capture and Crush*. The *Extension to Marine Reserves* was positive and significant, suggesting that for every 1% increase in marine reserves respondents were WTP $2.65. Finally, the cost coefficient was negative and significant, suggesting that cost was important when trading off competing alternatives.

Table 3 about here;

The interactions with the Status Quo ASC are all statistically significant except for the low-education binary variable. Based on the sign of the interaction parameters respondents who were older, donated to any nature-based charity in the last year, and could correctly identify a picture of a Southern Rock Lobster were less likely to select the Status Quo alternative. Despite including these sociodemographic characteristics the negative and statistically significant alternative-specific constant for the Status Quo suggests that there other factors, on average, lead respondents to being less likely to select the Status Quo alternative. Possible explanations for this heterogeneity may include status quo bias [58], the respondents perception of the status quo [59], or naive yeah-saying [60].

The Option B alternative-specific constant is statistically insignificant (p=0.07). The significant error-component confirms there is a difference in substitution patterns when comparing the Status Quo alternative and environmental quality improving alternatives.[3] Finally, the standard deviation parameters are statistically significant for all attributes.[4] This suggests that there is preference heterogeneity within the sample, with a proportion of respondents having negative WTP values for each attribute. For example, some respondents might prefer a smaller area of marine reserve in exchange for other attributes. Fig. 3 illustrates the simulated distributions for each attribute with significant mean and standard deviation parameters. For all significant attributes the proportion of positive values exceeds the proportion of negative values. The *Reintroduce Rock Lobster* attribute has the lowest relative proportion of negative WTP with approximately 12% of simulated WTP values being negative. In contrast the *Capture and Crush* attribute has the largest proportion of negative WTP with approximately 29% of simulated WTP values being negative

Fig 3 about here;

A set of aggregate welfare measures are provided in the Appendix. The sample is re-weighted to Tasmanian census characteristics.

4 Policy Discussion and Conclusions

The effect of invasive marine species has increased in many regions due to changing climate, poleward shifts in currents and the introduction of species via marine transport. Within the ecological management and policy literature, sea urchins (*Echinoidea*) and lionfish (*Pterois miles and Pterois volitans)* have received attention due to their destructive interaction with native species [2,61,62]. The need for appropriate non-market values to assess the costs and benefits of invasive species control has been identified locally [13] and more generally in a global context [8]. Costs for

[3] The alternative-specific constants and error-component are expressed in Willingness to Pay Space. Converting these parameters to preference space results in estimated coefficients of -2.41 (Status Quo), 0.23 (Option B) and 6.70 (Error Component).
[4] Parameters and standard errors for lower-triangular matrix available from authors on request.

management at a large scale are currently uncertain, however, one of the key barriers identified [13] to a fuller implementation of a series of management options relates to a lack of knowledge of public acceptability.

Our results reveal the importance of coastal and marine ecosystems and a willingness on the part of Tasmanians to support management efforts to control an invasive species, replant kelp in urchin barrens and to extend the geographic area of existing marine reserves. The mean values (and standard errors) indicate a significant and positive preference for the majority of respondents for at least one of the identified management actions. However, there is heterogeneity in these preferences which may reflect some of the recreational use values of a portion of the sample. The simulated distributions of WTP estimates illustrate that a proportion of the simulated population has a negative WTP for each management action.

The habitat and fish nursery functions of kelp forests provide important ecosystem function and increase the flow of ecosystem services including direct use values related to recreational and commercial fishery harvests. The statistically significant coefficients on habitat provision indicate concern for kelp protection and the risk of the formation of urchin barrens created by the invasive Long Spined Sea Urchin. It was found that households had a WTP of approximately $37 ($AUD) per household per year for five years to introduce a kelp replanting programme for the East Coast of Tasmania. Results, however, indicated that households had a WTP that was not statistically different from zero for the installation of artificial kelp beds. This result was unexpected given respondents were informed that artificial beds provided better anchoring for replanted areas to better encourage kelp growth. One possibility is that respondents may have had negative connotations associated with the term 'artificial' in relation to environmental management, whether that was due to perceptions of high costs or difficulty in successfully implementing an 'artificial' management action. It is also noted that at the time of the survey there was extensive press coverage of the issue of marine plastics. For example a major art installation, designed to raise public awareness, depicting sea creatures made from recovered marine plastics was widely reported in early December 2019 [63]. This may have influenced perceptions as artificial kelp beds utilise a plastic substructure fixed to the ocean floor. This may have created uncertainty over the future removal of the artificial kelp beds and contributed to the insignificant result.

The results of this survey indicate that Tasmanians are now aware and concerned with the expansion of the Long Spined Sea Urchin population and are willing to pay to reduce that population. The positive and significant results for most management actions for Long Spined Sea Urchin reductions were not unexpected due to the media attention on the problem [43]. The reintroduction of large Rock Lobster, which prey on Long Spined Sea Urchins, had the highest stated WTP associated with the invasive species control attribute ($24 per household per year for five years). Urchin control through predatory pressure has been identified as one of the key drivers of successful kelp reestablishment in urchin barrens in the North Atlantic [62]. The reintroduction of large Rock Lobster would also provide additional ecosystem services in terms of increasing potential yields from both recreational and commercial Rock Lobster harvest in the medium to long term; this may explain the higher relative value the public place on this management action.

The public were willing to pay for programmes to capture and crush the invasive Long Spined Sea Urchins ($20 per household per year for five years) indicating support for a direct policy of removal. However, there was a level of heterogeneity identified in the simulated distribution of WTP for this attribute. Similar results have been shown for invasive lionfish in the Mediterranean Sea where findings from informal interviews suggested that 48% of stakeholders did not support a policy to cull the species. The key driver identified for resistance towards lionfish culling amongst the sample was lack of knowledge regarding the ecological effects and beliefs that invasive species contributed to increasing biodiversity [61]. In the case of Tasmania, there are divergent social-ecological movements, ranging from “Eat the Problem” [64] through to vocal opposition to the expansion of the Tasmanian aquaculture industry. The latter has generated substantial media coverage and a Senate Inquiry investigating sustainable aquaculture regulation in Australia [65].

Globally marine reserves and other marine protected areas have expanded to cover important marine habitats [66]. The extent to which the ecosystem services are protected is more contentious. Some research has indicated that the use of marine reserves highlights the tension that exists among key stakeholders including government and non-government conservation and fisheries organisations in terms of the efficacy of marine reserves relative to alternative management options [67,68]. This may be due to inconsistencies among management objectives which require long-term protection of ecosystems and revenue from fishery harvests in the short term. In an Australian context the need for better integration of spatial scales of conservation and fishery management has been identified for more unified management of marine resources [69]. In this case study, geographical expansion of Tasmanian marine reserves would have ramifications for both recreational and commercial operators. Whilst this attribute reflects tougher fishing restrictions, the increase in marine reserves is associated with a positive and significant WTP ($2 per household per year for five years for each 1% expansion of reserves). Recent research regarding social licence for marine reserves identified a general consensus amongst Tasmanians of the need to further protect the marine environment [70]. The level of heterogeneity found in our results may reflect that there is a continuum of views on the appropriate mechanisms for protection [70]. Weighing up the benefits and costs of expansion requires information on preferences and acknowledgement of the different points of view.

A key message from the survey is that preferences for the management of marine and coastal ecosystems are not homogeneous. A MMNL model was used to allow for heterogeneity amongst respondents and the significant standard deviation parameters indicate that attitudes are heterogeneous. Those attributes with a positive and significant mean coefficient indicate support for particular management actions to be taken. While acknowledging the limitations in the representativeness of online samples, a large proportion of our sample support increased management actions in coastal and marine waters, but the support is for specific actions. Some Tasmanians may have limited appreciation of the need to protect and enhance ecosystem services and functions or alternatively, may not be supportive of some of the management actions presented in the context of this experiment. The identification of heterogeneity across respondents reinforces that consensus can be challenging to achieve. These results are consistent with the literature and parallel results found in the Northern hemisphere (e.g. large, statistically significant standard deviation parameters for all significant attributes in a MMNL) with the restoration of urchin barrens in Norwegian waters [2].

The results suggest that there are use and non-use values associated with supporting Southern Rock Lobster populations and kelp forest restoration. This case study demonstrates an approach which could be utilised where marine ecosystems degradation from changing environmental conditions, harvest pressures and invasive species can be observed. The research utilises the familiarity of the local population with management for a particular species to develop scenarios where the value of more complex issues of ecosystem services and function can be investigated. It develops values that can be used as part of a fuller consideration of costs and benefits of management action [8,13]. Reintroduction of a natural predator and replanting of kelp are supported by a larger proportion of respondents with few having a negative WTP. This suggests that management actions which are perceived as rebalancing the natural system (i.e. redeveloping multitrophic interactions) and enhancing ecosystem function may garner wider public support in the context of the protection of coastal ecosystems.

References

[1] S.E. Reeves, N. Kriegisch, C.R. Johnson, S.D. Ling, Reduced resistance to sediment-trapping turfs with decline of native kelp and establishment of an exotic kelp, Oecologia. (2018). https://doi.org/10.1007/s00442-018-4275-3.

[2] S. Hynes, W. Chen, K. Vondolia, C. Armstrong, E. O'Connor, Valuing the ecosystem service benefits from kelp forest restoration: A choice experiment from Norway, Ecol. Econ. 179 (2021) 106833. https://doi.org/10.1016/j.ecolecon.2020.106833.

[3] D.A. Smale, Impacts of ocean warming on kelp forest ecosystems, New Phytol. (2020). https://doi.org/10.1111/nph.16107.

[4] R.L. Kordas, C.D.G. Harley, M.I. O'Connor, Community ecology in a warming world: The influence of temperature on interspecific interactions in marine systems, J. Exp. Mar. Bio. Ecol. (2011). https://doi.org/10.1016/j.jembe.2011.02.029.

[5] S.D. Ling, Range expansion of a habitat-modifying species leads to loss of taxonomic diversity: A new and impoverished reef state, Oecologia. (2008). https://doi.org/10.1007/s00442-008-1043-9.

[6] M.C. Pelletier, J. Ebersole, K. Mulvaney, B. Rashleigh, M. Nicole, M. Chintala, A. Kuhn, M. Molina, M. Bagley, C. Lane, Resilience of aquatic systems : Review and management implications, Aquat. Sci. 82 (2020) 1–25. https://doi.org/10.1007/s00027-020-00717-z.

[7] H. Glenn, P. Wattage, S. Mardle, T. Van Rensburg, A. Grehan, N. Foley, Marine protected areas-substantiating their worth, Mar. Policy. (2010). https://doi.org/10.1016/j.marpol.2009.09.007.

[8] N. Hanley, M. Roberts, The economic benefits of invasive species management, People Nat. (2019). https://doi.org/10.1002/pan3.31.

[9] W. Adamowicz, P. Boxall, M. Williams, J. Louviere, Stated Preference Approaches for Measuring Passive Use Values: Choice Experiments and Contingent Valuation, Am. J. Agric. Econ. (1998). https://doi.org/10.2307/3180269.

[10] K. Hartmann, C. Gardner, D. Hobday, Fishery Assessment Report: Tasmanian Rock Lobster Fishery 2017/18, 2019. https://imas.utas.edu.au/__data/assets/pdf_file/0011/1245458/RL_Stock_Assessment_2017-19_Final_June_2019.pdf. (Accessed 8th July 2020).

[11] S.D. Ling, C.R. Johnson, S.D. Frusher, K.R. Ridgway, Overfishing reduces resilience of kelp beds to climate-driven catastrophic phase shift, Proc. Natl. Acad. Sci. U. S. A. (2009). https://doi.org/10.1073/pnas.0907529106.

[12] C.R. Johnson, S.C. Banks, N.S. Barrett, F. Cazassus, P.K. Dunstan, G.J. Edgar, S.D. Frusher, C. Gardner, M. Haddon, F. Helidoniotis, K.L. Hill, N.J. Holbrook, G.W. Hosie, P.R. Last, S.D. Ling, J. Melbourne-Thomas, K. Miller, G.T. Pecl, A.J. Richardson, K.R. Ridgway, S.R. Rintoul, D.A. Ritz, D.J. Ross, J.C. Sanderson, S.A. Shepherd, A. Slotwinski, K.M. Swadling, N. Taw, Climate change cascades: Shifts in oceanography, species' ranges and subtidal marine community

dynamics in eastern Tasmania, J. Exp. Mar. Bio. Ecol. (2011). https://doi.org/10.1016/j.jembe.2011.02.032.

[13] C. Layton, M.A. Coleman, E.M. Marzinelli, P.D. Steinberg, S.E. Swearer, A. Vergés, T. Wernberg, C.R. Johnson, Kelp Forest Restoration in Australia, Front. Mar. Sci. (2020). https://doi.org/10.3389/fmars.2020.00074.

[14] C. Lyle, JM and Hartmann, K and Mackay, M and Yamazaki, S and Ogier, E and Revill, H and Pearn, R and Rizzari, J and Tracey, S and Gardner, Rebuilding East Coast Rock Lobster Stocks: Developing an Effective Management Framework for Recovery, Hobart, Tasmania, 2020.

[15] R.A. Huber, M.L. Wicki, T. Bernauer, Public support for environmental policy depends on beliefs concerning effectiveness, intrusiveness, and fairness, Env. Polit. (2020). https://doi.org/10.1080/09644016.2019.1629171.

[16] R. Lopes, N. Videira, Valuing marine and coastal ecosystem services: An integrated participatory framework, Ocean Coast. Manag. (2013). https://doi.org/10.1016/j.ocecoaman.2013.08.001

[17] M. Christie, K. Remoundou, E. Siwicka, W. Wainwright, Valuing marine and coastal ecosystem service benefits: Case study of St Vincent and the Grenadines' proposed marine protected areas, Ecosyst. Serv. (2015). https://doi.org/10.1016/j.ecoser.2014.10.002.

[18] R.T. Carson, J.J. Louviere, A Common Nomenclature for Stated Preference Elicitation Approaches, Environ. Resour. Econ. (2011). https://doi.org/10.1007/s10640-010-9450-x.

[19] R. V. Salm, J.R. Clark, Erkki Siirila, Marine and Coastal Protected Areas: A Guide for Planners and Managers, 2000. IUCN, Gland, Switzerland and Cambridge, UK. (2000) https://doi.org/10.2305/iucn.ch.2000.13.en.

[20] K.J. Davis, M. Burton, A. Rogers, A. Spencer-Cotton, R. Pandit, Eliciting public values for management of complex marine systems: An integrated choice experiment, Mar. Resour. Econ. (2019). https://doi.org/10.1086/701303.

[21] N. Raheem, S. Colt, E. Fleishman, J. Talberth, P. Swedeen, K.J. Boyle, M. Rudd, R.D. Lopez, D. Crocker, D. Bohan, T. O'Higgins, C. Willer, R.M. Boumans, Application of non-market valuation to California's coastal policy decisions, Mar. Policy. (2012). https://doi.org/10.1016/j.marpol.2012.01.005.

[22] R.J. Johnston, K.J. Boyle, W. Vic Adamowicz, J. Bennett, R. Brouwer, T. Ann Cameron, W. Michael Hanemann, N. Hanley, M. Ryan, R. Scarpa, R. Tourangeau, C.A. Vossler, Contemporary guidance for stated preference studies, J. Assoc. Environ. Resour. Econ. (2017). https://doi.org/10.1086/691697.

[23] R.G. Cummings, P.T. Ganderton, T. McGuckin, Substitution Effects in CVM Values, Am. J. Agric. Econ. (1994). https://doi.org/10.2307/1243622.

[24] J.B. LOOMIS, D.M. LARSON, Total Economic Values of Increasing Gray Whale Populations: Results from a Contingent Valuation Survey of Visitors and Households, Mar. Resour. Econ. (1994). https://doi.org/10.1086/mre.9.3.42629085.

[25] I.H. Langford, M.S. Skourtos, A. Kontogianni, R.J. Day, S. Georgiou, I.J. Bateman, Use and nonuse values for conserving endangered species: The case of the Mediterranean monk seal, Environ. Plan. A. (2001). https://doi.org/10.1068/a348.

[26] P.C. Boxall, W.L. Adamowicz, M. Olar, G.E. West, G. Cantin, Analysis of the economic benefits associated with the recovery of threatened marine mammal species in the Canadian St. Lawrence Estuary, Mar. Policy. (2012). https://doi.org/10.1016/j.marpol.2011.05.003.

[27] K.P. Bell, D. Huppert, R.L. Johnson, Willingness to pay for local coho salmon enhancement in coastal communities, Mar. Resour. Econ. (2003). https://doi.org/10.1086/mre.18.1.42629381.

[28] E. Ojea, M.L. Loureiro, Valuing the recovery of overexploited fish stocks in the context of existence and option values, Mar. Policy. (2010). https://doi.org/10.1016/j.marpol.2009.10.007.

[[29] J.B. Loomis, D.S. White, Economic benefits of rare and endangered species: Summary and meta- analysis, Ecol. Econ. (1996). https://doi.org/10.1016/0921-8009(96)00029-8.

[30] V. Subroy, A. Gunawardena, M. Polyakov, R. Pandit, D.J. Pannell, The worth of wildlife: A meta-analysis of global non-market values of threatened species, Ecol. Econ. (2019). https://doi.org/10.1016/j.ecolecon.2019.106374.

[31] Millennium Ecosystem Assessment (Program). Ecosystems and human well-being. Washington, D.C., Island Press (2005).

[32] (The Economics of Ecosystems and Biodiversity). TEEB, The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations, Earthscan, London and Washington, (2010).

[33] UK National Ecosystem Assessment (2011), The UK National Ecosystem Assessment: Technical Report. UNEP-WCMC, Cambridge (2011). Accessed from http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx. (Accessed 8th July 2020).

[34] I.J. Bateman, A.R. Harwood, D.J. Abson, B. Andrews, A. Crowe, S. Dugdale, C. Fezzi, J. Foden, D. Hadley, R. Haines-Young, M. Hulme, A. Kontoleon, P. Munday, U. Pascual, J. Paterson, G. Perino, A. Sen, G. Siriwardena, M. Termansen, Economic Analysis for the UK National Ecosystem Assessment: Synthesis and Scenario Valuation of Changes in Ecosystem Services, Environ. Resour. Econ. (2014). https://doi.org/10.1007/s10640-013-9662-y.

[35] N. Stoeckl, M. Farr, S. Larson, V.M. Adams, I. Kubiszewski, M. Esparon, R. Costanza, A new approach to the problem of overlapping values: A case study in Australia's Great Barrier Reef, Ecosyst. Serv. (2014). https://doi.org/10.1016/j.ecoser.2014.09.005.

[36] J. De Valck, J. Rolfe, Comparing biodiversity valuation approaches for the sustainable management of the Great Barrier Reef, Australia, Ecosyst. Serv. (2019). https://doi.org/10.1016/j.ecoser.2018.11.003.

[37] J. Sagebiel, C. Schwartz, M. Rhozyel, S. Rajmis, J. Hirschfeld, Economic valuation of Baltic marine ecosystem services: Blind spots and limited consistency, ICES J. Mar. Sci. (2016). https://doi.org/10.1093/icesjms/fsv264.

[38] M. Aanesen, J. Falk-Andersson, G.K. Vondolia, T. Borch, S. Navrud, D. Tinch, Valuing coastal recreation and the visual intrusion from commercial activities in Arctic Norway, Ocean Coast. Manag. (2018). https://doi.org/10.1016/j.ocecoaman.2017.12.017.

[39] T. Börger, N.J. Beaumont, L. Pendleton, K.J. Boyle, P. Cooper, S. Fletcher, T. Haab, M. Hanemann, T.L. Hooper, S.S. Hussain, R. Portela, M. Stithou, J. Stockill, T. Taylor, M.C. Austen, Incorporating ecosystem services in marine planning: The role of valuation, Mar. Policy. (2014). https://doi.org/10.1016/j.marpol.2014.01.019.

[40] T. Potts, The natural advantage of regions: linking sustainability, innovation, and regional development in Australia, J. Clean. Prod. (2010). https://doi.org/10.1016/j.jclepro.2010.01.008.

[41] T. Potts, D. Burdon, E. Jackson, J. Atkins, J. Saunders, E. Hastings, O. Langmead, Do marine protected areas deliver flows of ecosystem services to support human welfare?, Mar. Policy. (2014). https://doi.org/10.1016/j.marpol.2013.08.011.

[42] M. Morrison, T.C. Brown, Testing the effectiveness of certainty scales, cheap talk, and dissonance-minimization in reducing hypothetical bias in contingent valuation studies, Environ. Resour. Econ. (2009). https://doi.org/10.1007/s10640-009-9287-3.

[43] Wild Fisheries Management Branch Department of Primary Industries, Parks, Water and Environment: Proceedings of The 2018 Centrostephanus Forum, in: Centrostephanus Forum, Hobart, Tasmania, 2018. https://dpipwe.tas.gov.au/Documents/Centro%20workshop%20report%20FINAL.pdf. (Accessed 8th July 2020).

[44] Ling, J. Keane, Resurvey of the Longspined Sea Urchin (Centrostephanus rodgersii) and associated barren reef in Tasmania. Institute for Marine and Antarctic Studies Report. University of Tasmania, Hobart. (2018) https://www.imas.utas.edu.au/__data/assets/pdf_file/0005/1176026/129569-Resurvey-of-the-Longspined-Sea-Urchin-Centrostephanus-rodgersii-and-associated-barren-reef-in-Tasmania.pdf. (Accessed 8th July 2020).

[45] L. Rogers-Bennett, C.A. Catton, Marine heat wave and multiple stressors tip bull kelp forest to sea urchin barrens, Sci. Rep. (2019). https://doi.org/10.1038/s41598-019-51114-y.

[46] NGENE 1.1. 2 User Manual and Reference Guide: the cutting edge in experimental design C Metrics - USA: Choice Metrics, 2014 http://www.choicemetrics.com/NgeneManual120.pdf. (Accessed 8th July 2020).

[47] D. Hatton MacDonald, A. Ardeshiri, J.M. Rose, B.D. Russell, S.D. Connell, Valuing coastal water quality: Adelaide, South Australia metropolitan area, Mar. Policy. (2015). https://doi.org/10.1016/j.marpol.2014.11.003.

[48] R. Scarpa, J.M. Rose, Design efficiency for non-market valuation with choice modelling: How to measure it, what to report and why, Aust. J. Agric. Resour. Econ. (2008). https://doi.org/10.1111/j.1467-8489.2007.00436.x.

[49] D.L. McFadden, Conditional Logit Analysis of Qualitative Choice Behavior, in: Frontiers in Econometrics, ed. P Zarembka, New York: Academic Press, 1974.

[50] D.A. Hensher, J.M. Rose, W.H. Greene, Applied choice analysis, 2015. https://doi.org/10.1007/9781316136232.

[51] Scarpa R., Ferrini S., Willis K. Performance of Error Component Models for Status-Quo Effects in Choice Experiments. In: Scarpa R., Alberini A. (eds) Applications of Simulation Methods in Environmental and Resource Economics. The Economics of Non-Market Goods and Resources, vol 6. Springer, Dordrecht. (2005) https://doi.org/10.1007/1-4020-3684-1_13

[52] K.E. Train, Discrete choice methods with simulation, 2003. https://doi.org/10.1017/CBO9780511753930.

[53] S. Hess, K.E. Train, J.W. Polak, On the use of a Modified Latin Hypercube Sampling (MLHS) method in the estimation of a Mixed Logit Model for vehicle choice, Transp. Res. Part B Methodol. (2006). https://doi.org/10.1016/j.trb.2004.10.005.

[54] Train K., Weeks M., Discrete Choice Models in Preference Space and Willingness-to-Pay Space. In: Scarpa R., Alberini A. (eds) Applications of Simulation Methods in Environmental and Resource Economics. The Economics of Non-Market Goods and Resources, vol 6. Springer, Dordrecht. (2005) https://doi.org/10.1007/1-4020-3684-1.

[55] R. Scarpa, M. Thiene, K. Train, Utility in willingness to pay space: A tool to address confounding random scale effects in destination choice to the Alps, Am. J. Agric. Econ. (2008). https://doi.org/10.1111/j.1467-8276.2008.01155.x.

[56] M. Bierlaire, Pythonbiogeme: a short Introduction. Report TRANSP-OR 160706. Series on Biogeme, Transport and Mobility Laboratory, (2016). https://transp-or.epfl.ch/documents/technicalReports/Bier16a.pdf. (Accessed 8th July 2020)

[57] J. Rose, John M; Zhang, An Introductory Guide to Estimating Discrete Choice Models Using Python Biogeme, (2017). https://www.uts.edu.au/sites/default/files/2018-03/BIDA-WP-1801.pdf. (Accessed 8th July 2020).

[58] J. Meyerhoff, U. Liebe, Status quo effect in choice experiments: Empirical evidence on attitudes and choice task complexity, Land Econ. (2009). https://doi.org/10.3368/le.85.3.515.

[59] D. Marsh, L. Mkwara, R. Scarpa, Do Respondents’ Perceptions of the Status Quo Matter in Non-Market Valuation with Choice Experiments? An Application to New Zealand Freshwater Streams, 3 (2011) 1593–1615. https://doi.org/10.3390/su3091593.

[60] K. Glenk, J. Martin-Ortega, M. Pulido-Velazquez, J. Potts, K. Glenk, J. Martin-Ortega, M. Pulido-Velazquez, J. Potts, Inferring Attribute Non-attendance from Discrete Choice Experiments: Implications for Benefit Transfer, Env. Resour. Econ. 60 (2015) 497–520. https://doi.org/10.1007/s10640-014-9777-9.

[61] C. Jimenez, V. Andreou, L. Hadjioannou, A. Petrou, R.A. Alhaija, P. Patsalou, Not everyone's cup of tea: public perception of culling invasive lionfish in Cyprus, Journal of the Black Sea/Mediterranean Environment, 23 (1) (2017).

[62] H. Christie, H. Gundersen, E. Rinde, K. Filbee-Dexter, K.M. Norderhaug, T. Pedersen, T. Bekkby, J.K. Gitmark, C.W. Fagerli, Can multitrophic interactions and ocean warming influence large-scale kelp recovery?, Ecol. Evol. (2019). https://doi.org/10.1002/ece3.4963.

[63] S. Schubert, Plastic pollution of Australia's beaches and oceans inspires unusual art installation - ABC News, (n.d.). https://www.abc.net.au/news/2019-12-06/plastic-pollution-of-beaches-and-oceans-inspires-artworks/11773902. (Accessed 3rd September 2020).

[64] Kaechele, K 2019, Eat the problem, Museum of Old and New Art, Berriedale, Tasmania.

[65] B. Haas, M. Phillipov, F. Gale, Media representations of seafood certification in Australia: Mobilising sustainability standards to attack or defend the value of an industry, Mar. Policy. (2020). https://doi.org/10.1016/j.marpol.2020.104126.

[66] P. Christie, N.J. Bennett, N.J. Gray, T. Aulani Wilhelm, N. Lewis, J. Parks, N.C. Ban, R.L. Gruby, L. Gordon, J. Day, S. Taei, A.M. Friedlander, Why people matter in ocean governance: Incorporating human dimensions into large-scale marine protected areas, Mar. Policy. (2017). https://doi.org/10.1016/j.marpol.2017.08.002.

[67] K.A. Alexander, A.J. Hobday, C. Cvitanovic, E. Ogier, K.L. Nash, R.S. Cottrell, A. Fleming, M. Fudge, E.A. Fulton, S. Frusher, R. Kelly, C.K. Macleod, G.T. Pecl, I. Van Putten, J. Vince, R.A. Watson, Progress in integrating natural and social science in marine ecosystem-based management research, Mar. Freshw. Res. (2019). https://doi.org/10.1071/MF17248.

[68] J.W. Turnbull, E.L. Johnston, G.F. Clark, Evaluating the social and ecological effectiveness of partially protected marine areas, Conserv. Biol. 0 (2021) 1–12. https://doi.org/10.1111/cobi.13677.

[69] P. Baelde, Interactions between the implementation of marine protected areas and right-based fisheries management in Australia, Fish. Manag. Ecol. (2005). https://doi.org/10.1111/j.1365-2400.2004.00413.x.

[70] R. Kelly, A. Fleming, M. Mackay, C. García, G.T. Pecl, Social licence for marine protected areas, Mar. Policy. 115 (2020) 103782. https://doi.org/10.1016/j.marpol.2019.103782.

[71] N. Hanley, S. Mourato, R.E. Wright, Choice modelling approaches: A superior alternative for environmental valuation?, J. Econ. Surv. (2001). https://doi.org/10.1111/1467-6419.00145.

[72] R.T. Carson, M. Czajkowski, The discrete choice experiment approach to environmental contingent valuation, in: Handb. Choice Model., 2014. https://doi.org/10.4337/9781781003152.00015.