04 University of Plymouth Research Theses

01 Research Theses Main Collection

2022

# A socio-economic and ecological approach to informing sustainable marine management in Jersey, Channel Islands

# Blampied, Samantha Robyn

http://hdl.handle.net/10026.1/19400

http://dx.doi.org/10.24382/983 University of Plymouth

All content in PEARL is protected by copyright law. Author manuscripts are made available in accordance with publisher policies. Please cite only the published version using the details provided on the item record or document. In the absence of an open licence (e.g. Creative Commons), permissions for further reuse of content should be sought from the publisher or author.

This copy of the thesis has been supplied on condition that anyone who consults it is understood to recognise that its copyright rests with its author and that no quotation from the thesis and no information derived from it may be published without the author's prior consent.



A socio-economic and ecological approach to informing sustainable

marine management in Jersey, Channel Islands

by

Samantha Robyn Blampied

A thesis submitted to the University of Plymouth in partial fulfilment for the degree

of

DOCTOR OF PHILOSOPHY

School of Biological and Marine Sciences

March 2022

#### Acknowledgments

Where to start! I would not have completed this thesis without the help of so many wonderful people. Thank you to my supervisors Emma, Sian, Francis and Martin for all their advice and help that has shaped me into the scientist I am today. To Paul, who has been amazingly helpful in many aspects of this PhD from GIS training to second opinions on worm identifications. And of course, to Morven at BLUE for taking a chance on me all those years ago!

Team aMER (aka Team Sheehan), it has been a pleasure to work with you, even if I rarely got to see you all in person! Special thanks to Rich Hartley in the lab for answering countless questions and to Mark Haggar for being the most laid-back health and safety officer I have ever met. To all those that spent many an hour at sea with me; especially the Marine Resources team for their commitment to seeing through three summers of fieldwork, and Kevin at Jersey Marine Conservation for stepping in when no one else could. Many thanks are also owed to Josh, Gibby and Toby for not only taking me out on their fishing boats in less than favourable conditions but for also providing the muscles to lift the kit when my arms all too quickly wore out.

Alterations to the towed video setup and development of the baited video cameras would not have been possible without the help of local expert 'DIY-ers' Dan and Chris. Shout out to Eddie from JFS tackle shop for always having a stash of frozen mackerel for me. And to all the students that assisted me in the lab or in the field, your help has been invaluable. Special thanks to Izzy for bringing the guns during baited video surveys, Max and Xander for their relentless patience identifying critters

under the microscope, and Rob and Meg for taking on countless hours of video footage.

To all of my friends and family for believing in me (especially Georgie and Paige for proof reading) and to my grandparents, Brian, Joan and Maureen, who I lost during this journey, I wish I could share this achievement with you. And of course, to my parents, Paul and Jane for igniting my love of the ocean, for your unconditional love and support, and for putting up with my rants when research was not going as planned or when the wind blew a force 6 when I wanted to go to sea. And finally, to Bede – thank you for giving me the confidence to believe I could do this, and for your relentless patience in teaching me to code, I will always be grateful that this PhD brought you into my life.

This work would not have been possible without the various funding bodies that have stood behind me. Many, many thanks to the Blue Marine Foundation, the Howard Davis Farm Trust, the Countryside Enhancement Scheme, the Ecology Trust and the Marine Biology Section of the Société Jersiaise.

#### Author's Declaration

At no time during the registration for the degree of Doctor of Philosophy has the author been registered for any other University award without prior agreement of the doctoral college quality sub-committee. Work submitted for this research degree at the University of Plymouth has not formed part of any other degree either at the University of Plymouth or at another establishment.

Throughout the course of this PhD, there were multiple opportunities to communicate the purpose and findings of this study to the local community through various media outlets. Radio interviews (BBC Jersey) were attended several times a year, while TV interviews (ITV), and magazine and newspaper articles (Rural magazine and Jersey Evening Post) were typically one-off occurrences. These interviews provided a useful platform to disseminate my research and establish a line of communication with the public. In addition to this, relevant scientific seminars and conferences were regularly attended at which work was often presented:

- Coastal Futures (London 2018)
- Bay of Granville Meeting and Mollusc Working Group (Jersey 2018)
- Marine Parks conference (Plymouth 2018)
- International Rhodolith Workshop (Roscoff 2018) P
- IMBeR ClimEco 6 (Yogyakarta 2018) P
- Inter Island Environmental Meeting (Jersey 2018) P
- Seminar for Jersey International Centre for Advanced Studies (Jersey 2018) P
- Plymsef (Plymouth 2019) P
- Inter Island Environmental Meeting (Alderney 2019) P
- MPA symposium (Fishmongers Hall, London 2020)

- National marine parks social and economics workshop (online conference organised by the University of Western Australia - 2020)
- IMCC6 (2 week online international conference 2020) P Best Poster
  Award on 'The Value of Marine Habitats'
- Inter Island Environmental Meeting (online conference 2020)
- Lecture on Marine Policy in Jersey for JICAS (Jersey 2021) P
- Inter Island Environmental Meeting (online conference 2021) P

## \***P** = Presentation given

### Fieldwork and data collection

The field element of this research spanned three summers (May-October) and approximately 400 hours at sea to collect data using the various underwater sampling and recording equipment. All fieldwork was led by myself but was facilitated by both the Government of Jersey Marine Resources team, Jersey Marine Conservation, local fishers and students. The majority of underwater sampling and recording equipment was constructed or modified locally. This included building a frame with lights and lasers to house an underwater video camera that would enable the seabed to be recorded over long transects (100-200m). Additionally, multiple (~20) baited remote underwater video (BRUV) units were built; these were designed to function in Jersey's waters while also being compact enough to allow for multiple units to be deployed from a 6 m rib.

Word count for the main body of this thesis: 56,764

Signed:

Date: 31st March 2022

#### Abstract

# A socio-economic and ecological approach to informing sustainable marine management in Jersey, Channel Islands

#### Samantha Robyn Blampied

Marine ecosystems support human life in multi-faceted ways and are depended on for food and income by many coastal communities. However, marine habitats are being altered by anthropogenic pressures, influencing the diversity and distribution of species, including species that are relied upon by commercial fisheries. Destructive fishing practices such as dredging and trawling can have adverse impacts on the seabed and associated species through the degradation and fragmentation of benthic habitats. Spatial management of marine areas is required to mitigate impacts, with Marine Protected Areas (MPAs) being advocated as tools that can not only protect and restore biodiversity but also improve fisheries sustainability and protect fisher livelihoods. The levels of protection vary and many MPAs are feature-based in that they only restrict damaging activities from specific evidenced conservation features within their boundaries. MPAs that exclude bottomtowed fishing from all habitats within their boundaries, a form of whole-site MPA designation, have shown benefits in terms of increased biodiversity and biomass beyond the benefits demonstrated by feature-based MPAs.

In Jersey, Channel Islands, two offshore MPAs that are managed following the whole-site approach were designated in 2017. This provided an opportunity to assess both ecological and socio-economic outcomes of this form of marine

management. The objectives of MPAs relate to both conservation and fisheries but the impact of spatial management changes on local fisheries is rarely captured in post MPA designation monitoring, despite social and economic factors being known to influence the success of a MPA. To date, fishing impacts on rocky reef habitats have been the focus of ecological MPA research, while the impacts of fishing on mixed sediments that also support biodiversity have received less attention.

Firstly, this thesis aims to add to the understanding of socio-economic impacts through engagement with the local fishing fleet to understand impacts to fishers' livelihoods; and secondly it aims to address the ecological knowledge gap regarding the response of mixed sediments following the exclusion of bottom-towed fishing through a combination of survey methods (baited video, towed video, and grab surveys), both inside and outside the MPAs. For the key commercial fishery species (lobster, Homarus gammarus; brown crab, Cancer pagurus; spider crab, Maja brachydactyla; scallop, Pecten maximus; and whelk, Buccinum undatum), multiple habitats in Jersey's waters were found to contribute to their annual economic value to Jersey fisheries (~ £7.5 million). Also identified is the strong economic dependence of local fishers on shellfish, particularly crab and lobster, as the majority of the fleet comprises static potting vessels. However, the MPAs had yet to have a noticeable benefit in terms of increased catch or improved fisher wellbeing. Ecologically, it was found that mixed sediment habitats, such as those targeted by bottom-towed fishing practices, had greater numbers of mobile and infaunal taxa inside the MPAs compared to Open Controls. Although for infaunal taxa, this was only statistically significant for the oldest MPA surveyed. This highlights the importance of MPA age, especially when considering ecological success. Within the MPAs there was no increase in the abundance of any of the key commercial

crustacean species (lobster, brown crab and spider crab). This may have been related to the continuation of static fishing within the MPAs.

Emphasised in this thesis is the importance of considering species' life histories and their habitat requirements in management plans. Also discussed is the need for further spatial management of fisheries outside of the current MPAs and the introduction of gear limits within the MPAs to secure sustainable fishing livelihoods. Critically, this research provides the first insight into species assemblage composition, diversity and abundance in response to the exclusion of bottom-towed fishing across a range of sedimentary habitats and the results could be used to better inform future MPA management.

# Contents

Acknowledgments	3
Author's Declaration	5
Abstract	7
List of Figures	. 14
List of Tables	. 18
1. Chapter 1 – Introduction	. 20
1.1. Marine biodiversity and ecosystem services	. 20
1.2. Human pressures on the marine environment	. 23
1.2.1. Fishery impacts on biodiversity	. 23
1.2.2. Impacts of mobile fishing	. 24
1.2.3. Impacts of static gear	. 27
1.3. Fisheries management and conservation	. 27
1.3.1. Ecosystem based approach to fisheries management	. 28
1.3.2. Marine Protected Areas – definitions and variations	. 29
1.3.3. Effectiveness of MPAs for ecological objectives	. 32
1.3.4. Effectiveness of MPAs for societal objectives	. 35
1.4. Jersey	. 36
1.4.1. Background	. 36
1.4.2. Designation of two offshore Marine Protected Areas	. 39
1.4.3. Fishing Activity and Management	. 44
1.5. Research aims	. 48
1.6. Thesis structure	. 49
2. Chapter 2 - The Value of Coastal Habitats to Commercial Fisheries in Jersey, Englis	sh
Channel, and the Role of Marine Protected Areas	. 51
Author Contributions	. 51
Abstract	. 52
	. 53
2.2. Study location	. 57
2.3. Materials and Methods	. 60
2.3.1. Commercial Landings Value	. 60
2.3.2. Habitats and essential life history stages for commercial species value	. 61
2.3.3. Apportioning commercial fisheries values to essential life history	. 63
2.4. Results	. 64

2.4.1. Identifying Commercially Important Species to Jersey Fisheries	64
2.4.2. Commercial Landings Value	65
2.4.3. Habitat Value and Value Protected	66
2.5. Discussion	70
2.6. Conclusion	77
3. Chapter 3 – The socio-economic impact of Marine Protected Areas in Jersey; a	
fishers' perspective.	78
Author Contributions	78
Abstract	79
3.1. Introduction	80
3.2. Case Study Site	85
3.3. Methodology	87
3.3.1. Data collection	87
3.3.2. Data analysis	90
3.4. Results	91
3.4.1. Catch composition of the fleet	91
3.4.2. Interviews	94
3.5. Discussion	102
3.6. Conclusion1	110
4. Chapter 4 – Removal of Bottom-Towed Fishing from Whole-Site Marine Protected Areas (MPAs) in Jersey's Territorial Waters (Channel Islands, UK) Promotes Mobile	
Species Biodiversity	112
Author Contributions	112
Abstract1	13
4.1. Introduction	14
4.2. Case study1	121
4.3. Materials and Methods1	124
4.3.1. Survey design 1	124
4.3.2. Site selection1	124
4.3.3. Data collection and equipment1	126
4.3.4. Video analysis 1	128
4.3.5. Data analysis1	128
4.4. Results	129
4.4.1. Total diversity (taxa)1	130
4.4.2. Total Abundance 1	L31
4.4.3. Total Assemblage 1	132
4.4.4. Indicator taxa 1	L34

4.5. Discussion	142
4.6. Conclusion	147
5. Chapter 5 – The response of epibiotic and infaunal assemblages to the removal of bottom-towed-fishing from Marine Protected Areas in Jersey, Channel Islands, UK	149
Author Contributions	149
Abstract	150
5.1. Introduction	151
5.2. Methods	158
5.2.1. Survey design	158
5.2.2. Data collection	160
5.2.3. Data analysis	166
5.3. Results	169
5.3.1. Epibiota	169
5.3.2. Sediment properties and Infauna	175
5.3.3. Infauna	177
5.3.4. Infaunal assemblage and sediment property responses by habitat	181
5.4. Discussion	186
5.4.1. Epibiota	187
5.4.2. Sediment properties	189
5.4.3. Infauna	190
5.4.4. Maerl habitat	191
5.4.5. Seagrass habitat	192
5.4.6. Coarse sediment habitat	193
5.5. Conclusion	195
6. Chapter 6 – The response of commercial crustacean populations to the removal of bottom-towed fishing within Marine Protected Areas in Jersey, Channel Islands, UK	196
Author Contributions	196
Abstract	197
6.1. Introduction	198
6.2. Case study	201
6.3. Study design	205
6.4. Methodology	205
6.4.1. Site selection	205
6.4.2. Data collection	206
6.2.3. Data analysis	208
6.5. Results	209
6.5.1. Historic Potting Surveys	209

6.5	6.5.2. Ecrehous and Minquiers MPA Potting Survey		
6.6. Discussion			
6.6.1. Lobster			
6.6.2. Spider crab 22			
6.6	.3. Brown crab	229	
6.7.0	Conclusion	231	
7. Dis	cussion	233	
7.1.	Overview	233	
7.2.	Towards an Ecosystem Based Approach to Fisheries Management	235	
7.3.	Indicators of biodiversity recovery	237	
7.4.	Shortfalls in MPA placement for fisheries	239	
7.5.	MPAs alone are not enough	240	
7.6.	Promoting sustainable fisheries	241	
7.7.	Global context	242	
7.8.	Implications for management	244	
7.9.	Ongoing monitoring and further research	247	
7.10.	Concluding remarks	248	
8. Ap	pendix A	250	
9. Ap	pendix B – Fisher Questionnaire	254	
10. A	Appendix C	262	
11. A	Appendix D	279	
Referen	Ces	280	

# List of Figures

Figure 1.1 a) An example of a dredge designed for catching scallops taken from (Boulcott & Howell, 2011), and observations of the seabed following disturbance from dredging from b) (Wood, 2018), and c) (Hall-Spencer & Moore, 2000) showing a scallop (arrow) that has been displaced from it's usual position in the seabed
Figure 1.2. Location of Jersey within the Normano-Breton Gulf showing the maritime boundaries and management measures
Figure 1.3. Timeline of spatial fisheries management designations
Figure 2.1. Location of Jersey within the Normano-Breton Gulf. Green filled polygons are areas which are protected from bottom-towed fishing gear. The light green shaded area shows the extent of the Bay of Granville which is an area of shared access rights for both Jersey and French vessels. The dashed green line shows the international boundary between Jersey and France, everything within this boundary is Jersey's territorial waters.
Figure 2.2. The spatial distribution of the five habitat groups in Jersey's territorial waters. MPAs are outlined by transparent polygons
Figure 2.3. Commercial landings (kg) between 2015 and 2018 showing the top 15 species that contribute to 99% of landings by weight. The remaining 1% of commercial landings consists of 36 species. Outliers are shown by filled circles
Figure 2.4. Mean landings (£) per year (2015-2018) for each species fishery and nation (France/Jersey) from Jerseys territorial waters
Figure 2.5. The contribution of each species to the habitat value (£) for each nation (Jersey and France)
Figure 2.6. The total value per habitat group for each nation (Jersey and France). Hatched areas show the value protected within the MPAs
Figure 3.1. Location of MPAs at the Ecrehous and Minquiers (transparent polygons) in relation to Jersey and France and each nations territorial boundaries
Figure 3.2. Commercial landings (kg) between 2015 and 2018 showing the top 15 species that contribute to 99% of landings by weight. The remaining 1% of commercial landings consists of 36 species
Figure 3.3. Landings value (£) per year (2015-2018) for the top five contributing species to Jersey's fisheries value, landed by Jersey vessels from the three primary sectors (mobile, static and diving)
Figure 3.4. Likert scale responses for each stakeholder group. Responses to the questions: a) to what extent do you support the MPAs (1 = not at all; 10 = completely), b) what do you perceive the levels of non-compliance are within the MPAs (1 = complete compliance; 10 = complete non-compliance), c) to what extent have the MPAs changed where you fish (1 = not at all; 10 = completely) and d) to what extent have the MPAs changed how often you fish (1 = not at all; 10 = completely). Filled in circles show outliers.
Figure 3.5. Likert scale responses for each stakeholder group split for 2017 and 2018 for questions relating to well-being. Responses to the questions: a) how satisfied are you with your fishing income (1 = completely dissatisfied; 10 = completely satisfied), b) how

satisfied are you with your fishing (job satisfaction) (1 = completely dissatisfied; 10 = completely satisfied), c) what level of conflict did you experience (1 = no conflict; 10 = high levels of conflict), d) what level of stress did you experience (1 = no stress; 10 = high levels of stress).
Figure 3.6. Average annual turnover and b) average annual profit for the mobile and static respondents
Figure 3.7. Mean percentages of catch sold through various routes as reported by mobile and static respondents
Figure 4.1. Location of MPAs within Jersey's territorial waters
Figure 4.2. Baited remote underwater video survey sites in relation to the MPAs
Figure 4.3. a) Profile of a baited underwater remote video unit and b) view of bait pole and attached bait cage within the field of view of the camera, with two cuttlefish (Sepia officinalis) observed on a sandy substrate
Figure 4.4. Total numbers of a) taxa and b) individuals for Location and Treatment. Black crosses and error bars show fitted GLMM means and standard errors of individual abundance predicted using GLMMs with Treatment (MPA and Open Control) and Location (Ecrehous, Southeast and Minquiers) as fixed effects and Year (2019 and 2020) and Site (n=15) as random effects
Figure 4.5. nMDS illustrating the dissimilarities in assemblage composition between Locations and Treatments. Each point represents a replicate site
Figure 4.6. Indicator taxa. a) <i>Maja brachydactyla</i> b) <i>Labrus bergylta</i> (example of Labridae spp.) c) <i>Spondyliosoma cantharus</i> d) Juvenile <i>S. cantharus</i> e) <i>Galeorhinus galeus</i> (example of shark species) f) <i>Raja undulata</i> (example of ray species)
Figure 4.7. Total numbers of Maja brachydactyla individuals for Location and Treatment. Black crosses and error bars show fitted GLMM means and standard errors of individual abundance predicted using GLMMs with Treatment (MPA and Open Control) and Location (Ecrehous, Southeast and Minquiers) as fixed effects and Year (2019 and 2020) and Site (n=15) as random effects
Figure 4.8. Cumulative presence of Labridae spp. at each Location and Treatment 138
Figure 4.9. Cumulative presence of a) adult and b) juvenile <i>Spondyliosoma cantharus</i> at each Location and Treatment
Figure 4.10. Cumulative presence of IUCN listed near threatened or vulnerable shark species at each Location and Treatment
Figure 4.11. Cumulative presence of ray species at each Location and Treatment 141
Figure 5.1. Location of Jersey within the Normano-Breton Gulf. The light green shaded area shows the extent of the Bay of Granville which is an area of shared access rights for both Jersey and French vessels. The dashed green line shows the international boundary between Jersey and France, everything within this boundary is Jersey's territorial waters. MPAs are areas which are protected from bottom-towed fishing gear
Figure 5.2. Survey sites in relation to the three MPAs (Ecrehous, Minquiers and Southeast)
Figure 5.3. Field sampling apparatus. a) Side view diagram and b) front view image (right) of the towed video array showing the underwater video system (a), lights (b), lasers (c),

buoyancy tubes (d), chain weight (e), cable (f), and tether (g). c) 0.2 m2 Van Veen grab.

Figure 6.1. Location of MPAs at the Ecrehous and Minquiers (transparent polygons) in relation to Jersey and France and each nations territorial boundaries
Figure 6.2. Management measure boundaries and 500 x 500 m potting sites within each treatment at a) the Ecrehous and b) the Minquiers
Figure 6.3. Diagram of a Parlour Pot showing A) the first chamber with the mouth entrance at the top, this is also the chamber which holds the bait, B) the second chamber, which is linked to chamber one via an opening (illustrated by a blue oval), and C) the hinged released door, which is secured by a hook on a bungee cord. Chamber two would normally have an escape gap via which juvenile crab and lobster may exit, but these were closed for the purpose of this study
Figure 6.4. Historic data from annual potting surveys conducted by the Government of Jersey Marine Resources team between 2014 and 2020. Lines show mean number of a) lobster, b) brown crab and c) spider crab per 10 pots and shading shows standard error, split for above MLS (dark blue) and below MLS (green)
Figure 6.5. Modelled mean estimates of lobsters per 10 pots (string) at the Minquiers and Ecrehous per year and treatment. Grouped by Total (all lobsters), above MLS lobsters and below MLS lobsters
Figure 6.6. Carapace length frequency distribution of lobster per Treatment (MPA and Open Control) and Year (2018, 2019 and 2020) in 5 mm classes for a) the Minquiers and b) the Ecrehous. The dashed line indicates the minimum landing size of 87 mm
Figure 6.7. Modelled mean estimates of spider crabs per 10 pots (string) at the Minquiers and Ecrehous per year and treatment. Grouped by Total (all spider crabs), above MLS spider crabs and below MLS spider crabs
Figure 6.8. Carapace width frequency distribution of spider crab per Treatment (MPA and Open Control) and Year (2018, 2019 and 2020) in 10 mm classes for a) the Minquiers and b) the Ecrehous. The dashed line indicates the minimum landing size (MLS) of 120 mm
Figure 6.9. Modelled mean estimates of brown crabs per 10 pots (string) at the Minquiers and Ecrehous per year and treatment. Grouped by Total (all brown crabs), above MLS spider crabs and below MLS brown crabs
Figure 6.10. Carapace width frequency distribution of brown crab per Treatment (MPA and Open Control) and Year (2018, 2019 and 2020) in 10 mm classes for a) the Minquiers and b) the Ecrehous. The dashed line indicates the minimum landing size (MLS). Note, MLS increased from 140 mm to 150 mm in 2019
Figure C.10.1. a) the mean particle size (µm), and b) the proportion of organic carbon, for each Habitat, Location and Treatment. Note that not all Habitats occur in all Treatments or Locations
Figure C.10.2. Proportion of particle size categories (Mud, Sand, and Gravel) for each Location and treatment where maerl occurred. Each bar represents an individual sample. 270
Figure C.10.3. Weight of a) Live maerl, and b) Dead maerl in kg per litre of sediment for Treatment (MPA and Open Control) and Location (Ecrehous and Southeast). Filled points represent outliers

Figure C.10.4.	nMDS illustrating the dissimilarities in assemblage composition between	
Locations and	Treatments2	273

# List of Tables

Table 1.1. The various types of MPA in the UK as listed by the JNCC and a description of what they are designated to protect
Table 1.2. Metiers employed by 186 vessels between 2015 and 2020. 45
Table 2.1. Habitat table detailing the EUNIS code habitats that have been grouped together
Table 2.2. Commercial shellfish species habitat usage as spawning areas (S), nursery areas (N), and feeding areas (F). Full references are listed in Appendix A (Table. A.1)67
Table 2.3. The total area of each habitat within Jersey's territorial waters, the area contained within the MPAs, the total annual value (£) and the value (£) protected within the MPAs for Jersey and French fisheries
Table 3.1. The primary fishing method of fishers interviewed presented as a percentage of fishing effort for each sector averaged over 2017 and 2018. The percentage of fleet was calculated from number of vessels interviewed compared to all vessels. Many fishers w.89
Table 4.1. List of the recorded taxa in all Locations and Treatments
Table 4.2. General Linear Mixed Effects Model outputs for baited video taxa and abundance as a function of Location and Treatment with Year and Site as random effects.Figures in bold denote a significant result.132
Table 4.3. PERMANOVA of assemblage based on Bray-Curtis dissimilarities. a) Main test and b) Pairwise tests for the interactions Location (Lo) and Treatment (Tr). Data were square root transformed. Figures in bold denote a significant result
Table 4.4. SIMPER analysis results showing the differences in average abundance(Av.Abund) and dissimilarity (Av.Diss) between MPA and Open Control treatments at eachLocation (Ecrehous, Southeast and Minquiers). Species contribution cut-off was set at95%.134
Table 4.5. General Linear Mixed Effects Model outputs for baited video abundance of Majabrachydactyla as a function of Treatment and Location with Year and Site as randomeffects. Figures in bold denote a significant result
Table 4.6. General Linear Mixed Effects Model outputs for baited video presence of Labridae spp. as a function of Treatment and Location with Year and Site as random effects. Figures in bold denote a significant result
Table 4.7. General Linear Mixed Effects Model outputs for baited video presence of adult and juvenile Spondyliosoma cantharus as a function of Treatment with Year and Site as random effects. Figures in bold denote a significant result
Table 5.1. General Linear Mixed Effects Model outputs for towed video proportions ofstructure forming organisms as a function of Treatment, Location, and Year with Site as arandom effect. Figures in bold denote a significant result.170

Table 5.2. General Linear Mixed Effects Model outputs for towed video Taxa andAbundance as a function of Treatment, Year, and Location with Site as a random effect.Figures in bold denote a significant result.173
Table 5.3. PERMANOVA of assemblage based on Bray-Curtis dissimilarities for the interactions of Year (Yr), Treatment (Tr) and Location (Lo). No transformation was applied to the data. Figures in bold denote a significant result
Table 5.4. Pairwise tests on the interaction Year x Treatment for each Location. Note therewere no Open Control replicates for the Southeast in 2018 and so no comparison hasbeen made. Figures in bold denote a significant result.174
Table 5.5. General Linear Mixed Effects Model outputs for grab sample mean grain size $(\mu m)$ and organic carbon proportion as a function of Location and Treatment, with Year and Site as random effects for the Ecrehous and Minquiers and Site as a random effect for the Southeast. Figures in bold denote a significant result
Table 5.6. General Linear Mixed Effects Model outputs for infaunal grab taxa (>1 mm) as a function of Location and Treatment with Year and Site as random effects for the Ecrehous and Minquiers and Site as a random factor for the Southeast. Figures in bold denote a significant result
Table 5.7. PERMANOVA of assemblage based on Bray-Curtis dissimilarities for the interactions of Location and Treatment. Year was a random factor for the Ecrehous and Minquiers Locations. The Southeast Location was only tested in 2020 due to a lack of Open Control replicates in 2018. Data were fourth root transformed. Figures in bold denote a significant result
Table 6.1. Key management measures in place for the major commercial crustaceanspecies in Jersey.203
Table 6.2. General Linear Mixed Effects Model outputs for change in Above MLS and Below MLS lobster, brown crab and spider crab abundance over time (2014-2020).Results are modelled as a function of Year with Location as a random effect. Bold denotes a significant result.210
Table 6.3. General linear model outputs for Lobster abundance at the Minquiers and Ecrehous. Total results are modelled as a function of Year, Treatment. Size Class results are modelled as a function of Year, Treatment and Size Class. Bold denotes a significant result
Table 6.4. General linear model outputs for spider crab abundance at the Minquiers and Ecrehous. Total results are modelled as a function of Year, Treatment. Size Class results are modelled as a function of Year, Treatment and Size Class. Bold denotes a significant result
Table 6.5. General linear model outputs for brown crab abundance at the Minquiers and Ecrehous. Total results are modelled as a function of Year, Treatment. Size Class results are modelled as a function of Year, Treatment and Size Class. Bold denotes a significant result.
Table A.1. Literature identified during data collection on life history and habitat      associations
Table C.1. Infaunal taxa recorded in the grab samples

#### 1. Chapter 1 – Introduction

This chapter will outline the importance of marine biodiversity and how marine resources support human well-being before highlighting some issues facing the marine environment and the current mitigation methods that are employed. Also detailed are the various shortfalls in current management measures and where further research is needed to inform best practice, using the example of marine management measures in Jersey, Channel Islands.

#### 1.1. Marine biodiversity and ecosystem services

The seafloor covers 71% of the Earth's surface and is a mosaic of connected habitats that support a diverse range of species assemblages (Gray, 1997). Marine sediments support high faunal biodiversity (Snelgrove, 1998) that play important roles in ecosystem processes and services (Heery et al., 2017; Snelgrove, 1998; Thrush & Dayton, 2002; Woodin et al., 2016). They also sequester and store vast quantities of carbon (Epstein and Roberts 2022; Sala et al. 2021). Marine sedimentary habitats can be further categorised based on the biological communities associated with them such as ecosystem engineers (e.g. seagrasses, maerl, polychaetes, and bivalves) that add structural complexity to marine sediment habitats (Jones et al., 1994). Seagrass and maerl in particular are IUCN red list species and OSPAR (Convention for the Protection of the Marine Environment of the North East Atlantic) threatened habitats (OSPAR 2002).

Seagrass (Figure 1.1. a) is a term for a group of angiosperms (flowering plants) which grow in the intertidal and shallow sub-tidal environments, both stabilising the

sediment and providing a foraging and nursery ground for many species (Lugendo et al 2006). Seagrass is an internationally recognised habitat (Tullrot 2009) that plays an important role in coastal ecosystems and ecosystem services (Duarte et al. 2005) and there is concern for global decline of this habitat (Borum et al., 2004; Cullen-Unsworth & Unsworth, 2013; Green et al., 2021). They are also nursery habitats for many species, including those of importance to fisheries (Unsworth et al. 2019).

Maerl (Figure 1.1. b) is a loose coralline red algae that accumulates in dense quantities to form beds that are also a nursery ground for many marine species, including those of commercial importance, such as King scallop (*Pecten maximus*) and Queen scallop (*Aequipecten opercularis*) (Hall-Spencer & Moore, 2000; Kamenos et al., 2004b). Maerl beds are known to support high diversity of infaunal and epifaunal organisms (Barbera et al. 2003), but the slow growth rate (<1 mm/y<sup>-1</sup>) and fragility of maerl makes it sensitive to disturbance (Wilson et al. 2004).



Figure 1.1. examples of a) seagrass (green blades) with other algal species growing amongst the blades of seagrass and bare sand in the foreground and b) maerl with a mix of live (pink) and dead (white) maerl amongst shell and sand material. Photo credit Sam Blampied.

Marine biodiversity is the basis of ecosystem function that maintains the services that the ocean provides to humanity (Duarte, 2000). Coastal communities in particular often depend on the ocean where fishing may be a primary source of income or food provision (UNEP, 2006b). Human well-being and economy depend on the marine environment in multi-faceted ways (Beaumont et al., 2007):

"People are part of ecosystems and ecosystem services (ES) are the benefits that people obtain from ecosystems" (UNEP, 2006a).

Ecosystem services (ES) are the supporting, regulating, provisioning and cultural services supplied by the habitats within the ecosystem (Maass et al. 2005), and ES supply is the full potential of an ecosystem to provide a service, irrespective of whether humans actually use or value that function (Cesar, 2002). This can range from basic human needs such as oxygen and food generation, to higher level needs such as culture, tourism and well-being. These services can only be supplied to humans if ecosystem function is conserved (Duarte, 2000), so that natural systems can continue to support human needs, either directly or indirectly, through the supply of ecosystem services (Fletcher et al., 2012; Groot et al., 2002; Harvey et al., 2017; Rees et al., 2013a). In a world with a growing human population, the oceans are a source of food security for many nations and in 2018 global capture fisheries production reached 96.4 million tonnes and the primary source of this production was marine systems (FAO, 2020). Fisheries increasingly support human well-being through job creation. In 2018, approximately 59.5 million people were involved in the primary sector of fisheries and aquaculture globally (FAO, 2020). This dependence on the marine environment provisioning service, however, is also one

of the key threats to healthy ecosystem functioning and the delivery of ES (Worm et al., 2006).

#### 1.2. Human pressures on the marine environment

The ocean faces a number of threats, with greater anthropogenic pressure being put on ecosystems in terms of fishing pressure, climate change, pollution and habitat loss (Halpern et al., 2008, 2019; Hughes et al., 2003; Pauly et al., 1998, 2005; Sumaila et al., 2016; Worm et al., 2006). Ecosystem degradation and the resultant biodiversity loss reduces the ecosystem services that human health and well-being depend upon (Chapin et al. 2000; Hooper et al. 2005; UNEP 2006; Worm et al. 2006). As the exploitation of marine resources increases to meet human demand, species diversity is lowered and seabed integrity is compromised, leading to loss of habitat (Dobson et al., 2006), collapse of food webs (Folke et al., 2004; Scheffer et al., 2005), and lowered delivery of ecosystem services and mapping them to their habitats is one of the core actions of the EU Biodiversity Strategy for 2030, with the aim to protect nature and reverse the degradation of ecosystems (European Commission, 2020).

#### 1.2.1. Fishery impacts on biodiversity

Wild capture fisheries largely fall into four main groups: mobile fishing, which involves dragging gear, such as dredges and trawls; static fishing, such as pots, set nets and lines; diving practices, such as diving for scallops; and intertidal fishing, such as digging for bivalves. Between 2010 and 2014, global marine capture fishery discards were estimated to be 9.1 million tonnes annually, with 46 percent of these discards originating from bottom trawls (Pérez Roda et al., 2019), primarily

comprising non-target species (bycatch). This fishing mortality is negatively impacting marine biodiversity and conservation efforts (Wilcox & Donlan, 2007). Further, highly selective fishing can truncate lifespans of commercial species (Berkeley et al., 2004; Beukers-Stewart et al., 2005; Thurstan et al., 2010). The resulting reduced adult populations consisting of smaller individuals can impede recruitment into subsequent generations due to lowered fecundity of smaller individuals (Barneche et al., 2018; Hislop, 1988; Ramirez, 2002). In addition to population level effects, fishing can also damage the seabed, with some methods having a greater environmental impact than others (Eno et al., 2013; Hooper et al., 2017).

#### 1.2.2. Impacts of mobile fishing

Mobile fishing involves dragging nets through the water column or along the seabed. Those that are dragged along the seafloor have the greatest potential to damage marine habitats due to direct contact with the seabed. This form of fishing puts pressure on more than just the targeted fish stocks as they impact the seabed itself, altering the structure, function and associated trophic dynamics (Bradshaw et al. 2001; Thrush and Dayton 2002; Sheehan et al. 2013; Howarth and Stewart 2014; Stewart and Howarth 2016; Kaiser et al. 2018). Scallop dredging is a form of mobile fishing where heavy nets and chains are dragged along the seabed to collect scallops (Figure 1.2. a-c). This form of fishing damages and homogenises the seabed (Bradshaw et al. 2001; Jennings et al. 2001; Thrush and Dayton 2002; Hall-Spencer et al. 2003; Sheehan et al. 2013) in addition to catching non-target fish which are typically discarded (Putten et al., 2019; Silva & Ellis, 2019; Stratoudakis et al., 2001). This form of fishing can be described as an example of 'bottom-towed

fishing' and this term will henceforth be used in this thesis to refer to mobile fishing gears that have contact with the seabed.



Figure 1.2. a) An example of a dredge designed to be towed along the seafloor to catch scallops taken from (Boulcott & Howell, 2011), and observations of the seabed following disturbance from dredging from b) (Wood, 2018), and c) (Hall-Spencer & Moore, 2000) showing a scallop (arrow) that has been displaced from its usual recessed position in the seabed.

Benthic habitats are important for coastal fisheries, but are being degraded by bottom-towed fishing practices (Kritzer et al., 2016). The sensitivity of habitats and

associated benthic biota has a bearing on the level of potential degradation. A metastudy (Hiddink et al., 2019) showed differing recovery rates of long-lived vs short lived organisms following a trawl event, with considerable reductions of benthic invertebrates that had a longevity of more than one year. Homogeneity of the seabed as a result of bottom-towed fishing leads to reduction in habitat complexity and species richness (Watling & Norse, 1998; Halpern et al., 2008). This reduction in species richness is particularly pronounced after the first occurrence of a dredging or trawl event (Cook et al., 2013), critically with large areas of the seabed already exposed to bottom-towed fishing, the natural state of the seabed and level of diversity is unknown for the majority of the worlds ocean (Jackson et al., 2011).

Mobile gear has the potential to impact all aspects of a species' life cycle. Juveniles, including those of commercial species, use several habitats but tend to be more selective for one or the other (Elliott et al., 2017b), and loss of particular habitats through mobile gear will reduce the survivability of habitat specialists (Berger, 2021; Clavel et al., 2011). For example, seagrass has greater abundances of juvenile fish than surrounding sand and mud habitats (Jackson et al., 2002; Lilley & Unsworth, 2014), and certain species of damsel fish are strongly associated with specific coral species (Bonin et al., 2015). Other complex habitats with relatively high epifaunal diversity have been shown to support greater abundances of commercial species, such as scallop (Howarth et al., 2011). Scallops prefer structurally complex habitats (Bradshaw et al., 2001, 2003; Howarth et al., 2011; Kamenos et al., 2004b); maerl creates a habitat that provides this complexity and the association of scallops on maerl habitat has been widely documented (Hall-Spencer 1998; Hall-Spencer and Moore 2000; Barbera et al. 2004; Kamenos et al. 2004; Kamenos et al. 2004b; Hall-Spencer et al. 2010; Howarth et al. 2011;

Sheehan et al. 2015). It is for this reason that maerl habitats have historically been targeted by scallop dredgers.

#### 1.2.3. Impacts of static gear

Static fishing gears include pots, fish traps, longlines, set nets and any other fishing gear that is stationary that fish either swim into or are attracted to with the use of bait. The impact of static gear on seabed fauna and flora is less studied. Early evidence on the subject suggested that pots did not negatively impact the seabed (Coleman et al., 2013; Eno et al., 2001; Stephenson et al., 2017). However, there has been criticism of this broad theory concerning the lack of information regarding habitat type and sensitivity to potting (Walmsley et al., 2015). Eno et al. (2013) found a few habitats that were highly sensitive to potting, whereas many habitats showed high sensitivity to scallop dredging, highlighting the importance of habitat type when considering fishery impacts. More recent studies that have focussed directly on the impact of potting on sensitive habitats have shown the potential of pots to negatively impact the benthic biota on reef habitat (Gall et al., 2020; Rees et al., 2021). Another study in Qatar has observed declines in corals, seagrass and oysters in areas subject to increased fish trap fishing (Ibrahim et al., 2018). Rees et al. (2021) found the intensity of potting was a factor in the level of damage observed to the benthic biota, with a noticeable decline in some sessile species in areas with the highest potting intensity.

#### 1.3. Fisheries management and conservation

Fisheries can be manged in a variety of ways depending on the species of concern and well managed fisheries have been shown to reduce fishing pressure and increase stocks (Hilborn et al., 2020). Management of both mobile and static

fisheries has primarily been aimed at reducing pressure on species, not habitats, through the use of quotas or restrictions on gear types. For fishing communities that rely on specific species as a primary source of income, management specific to that species may be employed. For example, crab and lobster fishing supports many coastal fisheries in the UK and, to reduce juvenile mortality in these species, it is common practice to enforce legislation that requires pots to be fitted with escape gaps that allow smaller individuals to escape (Bakke, 2019; Bullimore et al., 2001; Cefas, 2020a; Marine Resources, 2019). Species with seasonal migrations or breeding patterns can be managed through the use of closed seasons (Dunn et al., 2011), whereas relatively stationary local populations can be manged using guotas (Baudron et al., 2020), temporary closures (Gnanalingam et al., 2021), bag limits (ICES, 2018), or equipment modifications, such as minimum mesh sizes of nets to improve selectivity for larger individuals (Suuronen et al., 2007). However, species specific management strategies do not take into account the wider ecosystem within which the fishery exists. The potential of species-specific management to achieve sustainability may be undermined if damaging fishing methods are permitted on benthic habitats that support fisheries species at other points in their life history (e.g. juvenile or spawning areas) (Seitz et al., 2014; Solandt et al., 2020). Instead, effective fisheries management needs to be ecosystem based, taking into account all parts of an ecosystem that support a fish species though to adulthood (and therefore commercial viability). In this sense, fisheries and conservation are intrinsically linked (Rees et al 2020).

1.3.1. Ecosystem based approach to fisheries management Historically, fisheries management has been focussed on single species and sustainability measured using Maximum Sustainable Yield (MSY) (Pikitch et al., 2004). There has since been a shift in policy towards an ecosystem based management approach to fisheries, where the whole ecosystem within which the fishery exists is considered (Long et al. 2015; Pikitch et al. 2004), with an aim of achieving sustainable use of ecosystem goods and services to support societal values (Laffoley et al., 2004; Thrush et al., 2016).

Current policies that recommend an ecosystem-based approach to fisheries management are the UK Fisheries Act (2020), the European Union Marine Strategy Framework Directive (MSFD), Common Fisheries Policy (CFP), the Integrated Maritime Policy (IMP), and Goal 14 of the UN Sustainable Development Goals (SDGs). This change in ambition for sustainable fisheries has resulted in new approaches to management to promote the recovery of marine ecosystems (Pikitch et al., 2004). One tool that can be used to combat over-fishing and habitat degradation is to close areas of seabed to certain fishing metiers, particularly those such as dredging which are known to damage the seabed (Thrush et al. 1995; Long et al. 1996; Thrush and Dayton 2002; Sheehan et al. 2013b), typically through the designation of Marine Protected Areas (MPAs).

#### 1.3.2. Marine Protected Areas – definitions and variations

The term 'Marine Protected Area' varies in its definition but is described by the IUCN as 'a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values' (Day et al., 2012, page 2). MPAs encompass a spectrum of protection types due to variations in management and legislation used to create the MPA. Approximately 7.7% of the

world's oceans are currently within MPAs (Figure 1.3), but only 2.4% are highly protected (Marine Conservation Institute, 2022).



Figure 1.3. Global coverage of various Marine Protected Areas (Marine Conservation Institute, 2022). Blue = Highly/Fully Protected, Green = Less Protected/Unknown, Red = Designated & Unimplemented.

The legislation that underpins the management of MPAs varies globally and in the UK the Marine and Coastal Access Act (2009) commits the UK government to deliver a network of MPAs in English and Welsh territorial waters (Mangi et al., 2011). This has resulted in several different types of MPA (Table 1.1). Within UK MPAs, the legislation is primarily targeted at protecting the known extent of conservation features within the MPA boundaries (Solandt et al., 2020). The key exceptions are the MPAs in Lyme Bay, Port Erin, Isle of Arran and Lundy, which have legislation to

exclude bottom-towed fishing from the whole MPA (Beukers-Stewart et al., 2005; Howarth et al., 2015; Sheehan et al., 2013a; Watson et al., 2016). Additionally, there are fishery closure areas, such as Rockall, UK, (Hall-Spencer et al., 2009), that exclude bottom-towed fishing. It is the role of regional fisheries authorities to manage fisheries impacts on conservation features within MPAs with their duties described under the Marine and Coastal Access Act 2009.

Table 1.1. The various types of MPA in the UK as listed by the JNCC and a description of what they are designated to protect.

Type of MPA	Description
Special Areas of	Designated to protect habitats and species of European
Conservation	importance (listed on Annex I and II of the European
(SACs)	Habitats Directive).
Special Protection	Areas protected for bird species of European importance
Areas (SPAs)	and migratory birds.
Marine	Designated to protect nationally important species,
Conservation	habitats, ecological processes and features of geological
Zones (MCZs)	importance.
Sites of Special	Areas of special interest that are designated to protect
Scientific Interest	flora, fauna, geological or physiographical features. These
(SSSIs)	are typically coastal and terrestrial but with some sites
	extending to marine features.
Ramsar Sites	Wetlands of international importance designated under the
	Ramsar Convention.

Five key attributes that underpin the ability of a MPA to meet conservation goals are outlined by Edgar et al. (2014): level of protection, enforcement, age, size and isolation. The global extent of MPAs is increasing but many do not adequately protect biodiversity to achieve conservation goals (Claudet et al., 2020; Klein et al., 2015; Kuempel et al., 2019; Mazaris et al., 2018). This shortfall in protected biodiversity will have implications for commercial species, either directly through overfishing and habitat loss, or indirectly through the breakdown of food webs (Dobson et al., 2006; Scheffer et al., 2005; Thurstan & Roberts, 2010). Additionally, there are very few MPAs that have been established over sufficient time periods to allow a climax community to develop and a plateau of biodiversity to be reached. Edgar et al. (2014) suggests that increases in fish biomass are more likely to be observed in older MPAs (>10 years), but only for MPAs that are appropriately placed and subsequently enforced.

#### 1.3.3. Effectiveness of MPAs for ecological objectives

The level of protection a MPA is afforded is determined by the objectives for which the MPA has been designated for and the success of the MPA will ultimately come down to its ability to meet these objectives (Gaines et al., 2010). Those which restrict all forms of extractive activities, both commercial and recreational, are termed No Take Zones (NTZs). NTZs are considered by some to be the most effective protected areas (Edgar et al., 2014; Roberts et al., 2005; Sala & Giakoumi, 2018), resulting in greater fish biomass compared to MPAs that allow some forms of fishing (Sala & Giakoumi, 2018). A review by Lester et al. (2009) discusses the idea that NTZs in temperate settings can have a similar or greater positive effect compared to those in the tropics. However, NTZs are rare and currently make up 1.6% of MPAs globally (Lubchenco & Grorud-Colvert, 2015). The reality is that the level of protection afforded to a MPA varies greatly (Day et al. 2012), and the majority of MPAs are multiple use, allowing certain fishing activities.

It may not always be possible to achieve a no-take level of protection within a MPA and in many cases lower levels of protection may be more appropriate for balancing ecological and socio-economic objectives (Agardy, 2000). Many MPAs in the UK are feature-based; where only the evidenced extent of a conservation feature is

protected from damaging fishing activities, such as bottom-towed fishing, within the MPA boundary. However, this level of protection has been shown to be less effective in the recovery of biodiversity compared to MPAs where there are spatial restrictions across the whole MPA for fishers using bottom-towed gear (and still allowing other forms of fishing), which protects not just the conservation features but also the interstitial habitats (Solandt et al., 2020). This whole-site approach to MPA management is promoted as a means to enhance wider ecosystem recovery both in research (Davies et al., 2021a; Rees et al., 2013c) and in policy (HM Government, 2018) to support the objectives of the ecosystem based approach to fisheries management (Pikitch et al., 2004; Pitcher et al., 2009; Solandt et al., 2020).

Research from the Lyme Bay MPA (Figure 1.4.) has shown that the whole-site approach can improve ecological condition and, through monitoring the recovery of reef associated species, can reveal the true functional extent (the area of seabed capable of supporting reef associated species) of reef habitat (Sheehan et al. 2013a). Also demonstrated were increases in species richness despite other, low impact, extractive fishing methods being permitted to continue (Davies et al., 2021a; Sheehan et al., 2013a). Further, there was increased resilience of the reefs to storms, where the trend of recovery in MPA assemblages and diversity following storms in 2013 and 2014 was quicker than the recovery following the cessation of bottom-towed fishing in 2008 (Sheehan et al., 2021). This provides strong evidence for the benefits of maintaining the integrity and health of benthic habitats as they are the basis of a functioning ecosystem in which biodiversity thrives, including species of commercial interest.



Figure 1.4. The location of Lyme Bay in the UK showing local towns and the outline of the Lyme Bay MPA and various seafloor substrates. Taken from (Rees et al., 2013)

With increasing research showing the benefits for biodiversity of protecting all habitats within MPAs via the whole-site approach (Davies et al., 2021a; Sheehan et al., 2013b; Solandt et al., 2020), there is increasing pressure on marine governing bodies to extend the protection to all habitats within the MPA boundary (HM Government, 2018). As many species use multiple habitats across their life cycle (Kritzer et al., 2016; Seitz et al., 2014), this whole-site approach to MPA management can support the dual objectives of conservation and sustainable fisheries. However, there are clear socio-economic trade-offs within the fishing industry that will need to be addressed if whole-site approach MPAs are to be an effective means to successfully meet these two goals. This is essential not only to

sustain the supply of provisioning ecosystem services from marine systems but also all the additional ecosystem service benefits that can be derived from healthy, functioning marine ecosystems (Fletcher et al., 2011; Leenhardt et al., 2015; Moran et al., 2008; Sheehan, Bridger, & Attrill, 2015).

#### 1.3.4. Effectiveness of MPAs for societal objectives

Clearly defined objectives for a MPA are essential to enable effective management (Halpern and Warner 2003). There is a crux between the primary ecological objectives for a MPA designation (to broadly support ecosystem services to human kind, through the protection of conservation features within a network of MPAs) and the localised societal objectives (Agardy, 2000). Whilst MPAs can support fisheries management (Fletcher et al., 2010; Gell & Roberts, 2003; Hilborn et al., 2004), unless societal fisheries management objectives are built into the criteria for MPA effectives then the two (conservation and sustainable fisheries management) will never be shared and realised objectives. There is evidence that MPA effectiveness can be driven by social factors, with stakeholder engagement, enforcement and clear management plans required (Hattam et al., 2014; Mazaris et al., 2018), especially where small-scale fisheries are concerned (Di Franco et al., 2016; Rees et al., 2010).

The human dimensions of MPAs are complex with many potential socio-economic benefits, namely improved provisioning and regulating services but also increased benefits to non-consumptive sectors such as eco-tourism (Charles & Wilson, 2009). There are also costs associated with MPAs and these are primarily felt in the fishing sector, but they are not equally distributed (Hattam et al., 2014), with losses in catch and increased distances to fishing grounds experienced by some sectors (Mangi et
al., 2011), while others may benefit from reduced fishing pressure within the MPA boundary (Rees et al., 2021). MPAs which do not effectively meet their objectives risk a breakdown of trust between stakeholders with obvious consequences for the original intended conservation objectives (Pendleton et al., 2018).

#### 1.4. Jersey

## 1.4.1. Background

Jersey is the largest of the Channel Islands situated in the Normano-Breton Gulf on the north coast of Brittany, with a marine territorial area of approximately 2,500 km<sup>2</sup> (Figure 1.5 and 1.6 a-b). As a self-governing dependency of the British Crown, Jersey is not represented in the UK parliament and as such has its own fiscal, administrative and legal systems. Jersey has its own democratically elected parliament called the States Assembly that is responsible for all aspects of island government, except foreign affairs that are deferred to the UK. They are also responsible for electing the Ministers (one Chief Minister and 10 other Ministers) who will deliver strategic priorities, as set by the Government of Jersey, relating to economy, community and environment. It is through this system that policies underpinning the resource use of Jerseys natural environment are set.

Jersey's marine environment ranges from 0-50 m depth and consists of rocky reef, boulders, cobbles, gravel, sand and biogenic habitats, such as maerl (*Lithothamnion coralloides* and *Phytomatolithon calcareum*) and both intertidal (*Zostera noltei*) and subtidal (*Zostera marina*) seagrass. The large tidal range experienced in Jersey (up to 12.2 m) also creates a large intertidal area (~30 km<sup>2</sup>), contributing to a diverse habitat composition supporting a wide array of species.



Figure 1.5 Admiralty chart (UK Hydrographic Office) showing the Island of Jersey, the offshore reefs, the extent of Jersey's territorial waters (black line), and nautical navigation marks (purple). Land mass = yellow, intertidal = green, dark blue = 0-10 m subtidal contour, light blue = 10-20 m subtidal contour, white = subtidal deeper than 20 m.



Figure 1.6. a) Admiralty chart and b) aerial images showing the land mass of Jersey, the extent of the intertidal areas and the topography of Jersey's coastal areas. Aerial image provided by Dr Paul Chambers, pers. comm. Marine Resources.

### 1.4.2. Designation of two offshore MPAs

To protect areas of fragile habitat and high biodiversity associated with two offshore reefs in Jersey's territorial waters, two MPAs which excluded mobile fishing gear were designated in 2017 (Chambers et al. 2020) (Figure 1.7). These two MPAs, along with several other previously established inshore MPAs, resulted in 6.5% of Jersey's territorial waters protected from mobile gear (both bottom-towed gear and mid-water trawls). These MPAs follow the whole-site approach to marine management as all habitats within their boundary are afforded the same level of protection (Solandt et al., 2020). Currently these are the only management measures specific to protect benthic habitats within the Normano-Breton Gulf. Of the offshore MPAs, one is located 20 km to the south of Jersey around a large offshore reef, Les Minquiers (pronounced 'minkies' and hereafter referred to as the Minguiers). The other was designated around another offshore reef, Les Écréhous (pronounced eck-re-hoes and hereafter referred to as the Ecrehous), which is found approximately 10 km off the north-east coast of Jersey (Figure 1.7). Both the Minguiers and the Ecrehous are shallow reefs with subtidal habitats ranging from 0 to 15 m below Chart Datum within the MPA boundaries. Both are exposed to strong tidal currents, caused by the large tidal ranges experienced around Jersey, creating a diverse array of habitats and species found across the reefs and it was for this reason that they were both established as Ramsar sites in 2005.



Figure 1.7. Location of Jersey within the Normano-Breton Gulf showing the maritime boundaries and management measures. Inset shows the location of the Normano-Breton Gulf in the British Isles.

The plateau of the Minquiers (Figure 1.8. a) is over 300 km<sup>2</sup>, making it bigger than the island of Jersey itself, and 47.5km<sup>2</sup> of the central part of this reef is protected by the MPA. An extensive study was carried out at the Minquiers over several years by local researchers (Chambers & Binney, 2016) which documented the geology, habitats, species and ecosystem services provided by the reef. A total of 613 species have been recorded there, both recently and historically, on 37 intertidal and 25 subtidal biotopes. The diversity of habitats, especially the intertidal habitats, found there is likely owed to a combination of a diverse topography, a range of exposures to currents and large tidal ranges, which sees the area of reef exposed increase from 0.01 km<sup>2</sup> at high tide to 21 km<sup>2</sup> at low tide.

The Ecrehous reef (Figure 1.8. b) covers a far smaller area than the Minquiers and the MPA even less at just 15 km<sup>2</sup>. There has been sporadic research carried out locally on the Ecrehous marine environment including Seasearch surveys, studies of the maerl beds in 2014 and 2015 (Blampied et al. 2015 – unpublished bachelors thesis), habitat mapping and species recording (Chambers & Binney, 2016). Substrate types across the Normano-Breton gulf were surveyed in 1979 that included areas surrounding both the Ecrehous and Minquiers (Retière, 1979), and another study examined the physical characteristics of the sediments (Le Hir, 1986), but there have been no other wide scale classifications of the sediments since. There have also been reports on the use of the Ecrehous and the Minquiers, both recreational and commercial, and their political history, which highlights the value of these reefs to both Jersey and France (Falle & Humphreys, 1886; Fleury & Johnson, 2015).



Figure 1.8. Images of a) the Minquiers looking out to sea from the main island (Maîtresse-Île) at low tide, photo credit Sam Blampied, and b) the Ecrehous at low tide showing the houses in the central area of the reef, the exposed pebble bank, and subtidal reef areas, photo credit James Bowden and the Blue Marine Foundation.

Seagrass beds in Jersey were devastated by a disease in the 1930s and, based on aerial photos from 1933 and 1944, an estimated 90% of coverage was lost (pers. comm. Dr Paul Chambers). In recent decades, aerial photos have shown the expansion of both intertidal and subtidal seagrass beds (aerial photos taken at low tide when there is good visibility allows for observations of subtidal seagrass). Seagrass is found across the east and southeast coast and also in small, but relatively healthy patches at the Minquiers currently. However, in 2013, aerial photos showed dredge tracks across an area of seagrass at the Minquiers which now no longer exists (pers. comm. Dr Paul Chambers). This particular area has now been closed to mobile gear and the close proximity of other seagrass beds may serve to seed this area (Balestri et al., 2017; Hosokawa et al., 2015).

The Minquiers is a hazardous reef with many rocks unmarked on modern charts, making it difficult to access, and it is for this reason that much of this reef has been untouched by mobile gear (Chambers et al., 2016). The wide range of habitats and associated biodiversity found here formed part of the argument to protect it as a precaution against future mobile fishing exploits. Areas of the Ecrehous are much more accessible to mobile gear with large patches of maerl between reef heads providing good scallop grounds. It is for this reason that areas of maerl at the Ecrehous have been damaged by scallop dredging. By introducing a MPA here, it is hoped that the maerl beds will be able to recover and provide a healthy habitat for species to live and take refuge on. Due to the difference in habitat composition and accessibility to mobile gear prior to designation, it is expected that the ecological changes in response to the exclusion of mobile gear will differ between the two MPAs.

In addition to the MPAs at the Minquiers and the Ecrehous, Jersey also has several inshore MPAs, which have been established over a longer period than the two recent offshore MPAs (Figure 1.9). Of these, the Southeast MPA (Figure 1.7) has been included as a comparison site for the two recent MPAs. The Southeast MPA was

first established in 2010, with an extension to the east in 2014 after towed video surveys identified areas of maerl that were accessible by bottom-towed gear and in relatively poor health on the east coast (Sheehan et al., 2015). The extent of dense areas of maerl were later mapped by the Société Jersiaise (Chambers & Binney, 2015) to determine the extent of the extension of the MPA. The Southeast MPA contains the same habitats of interest (seagrass and maerl) as the Minquiers and the Ecrehous MPAs and provides a comparison of seabed that has been protected over different time scales to the two offshore MPAs.



Figure 1.9. Timeline of spatial fisheries management designations.

#### 1.4.3. Fishing activity and management

There have been various fisheries in Jersey throughout the previous centuries, with several key fisheries of note. In the 1800s, Jersey's fishing fleet was primarily composed of vessels designed for long voyages across the North Atlantic to catch cod. Also in the early 1800's, fishers exploited oyster beds on the east coast of Jersey. Oysters were targeted by both Jersey and French fishers, resulting in conflict between the two nations which led to the original Bay of Granville Agreement being established in 1839 (Chambers et al., 2020). However, due to over exploitation, this fishery had completely collapsed by 1871. Various fish species were targeted in the years that followed, namely crab, lobster and finfish (primarily flatfish) but many were over exploited due to a lack of fisheries regulations from the late 1860s and by 1950

there were no commercial fishing boats left. In the early 1960's Jersey fisheries started up once more and have since heavily relied on benthic shellfish such as crab, lobster, scallop and whelk. Today, Jersey's fishing fleet employs a mixture of metiers, with many vessels equipped for operating several different fishing gears to diversify their approach to fishing.

Since 2015 there have been roughly 186 vessels registered as commercial fishing boats, but the majority of these are part-time. Table 1.2 shows the distribution of fishing metiers across the fleet; note that the percentages total over 100 due to several boats employing mixed metiers. The static fishery makes up the biggest portion of the fleet, which is primarily owed to the large number of vessels using pots to target crab and lobster. As with many other small-scale fisheries, accurate spatial fishing information is not available for most of the fleet as vessels under 12 m in length are not required to have Vessel Monitoring Systems (VMS) on board.

Metier	Number of vessels	% of fleet		
Mobile	11	6		
Static	173	93		
Diving	22	11.8		
Low water fishing	7	3.7		

Table 1.2. Metiers employed by 186 vessels between 2015 and 2020.

The management of fisheries in Jersey waters, in addition to balancing conservation objectives, sits with the Government of Jersey. The Government of Jersey is split into various departments within which there are several groups (Figure 1.10). The Marine Resources team sits within the Natural Environment group of the Infrastructure, Housing, and Environment. To deliver the management of the marine sector in Jersey, the Marine Resources team covers research, policy, legislation,

regulation, licencing, education, and stakeholder engagement. There is regular communication with Ministers, in particular the Environment Minister, to ensure there is effective delivery of policy goals relating to the marine sector.





Jersey is not committed to protect its marine habitats under the Marine and Coastal Access Act 2009 as Jersey is not part of the UK, neither is it a member of the EU and so is not included in the Natura 2000 network. As discussed earlier, there are several UK policies that provide the framework for implementing ecosystem based management of marine resources. While Jersey is not committed to these policies, Jersey has its own legal systems and commitments to protecting its marine resources and has an Integrated Coastal Zone Management (Government advocates for an ecosystem-based approach to marine management (Government of Jersey, 2008). Jersey is also committed to protecting it's marine environment as a signatory to various conventions and agreements (States of Jersey 2013), such as OSPAR (2002), The Bern Convention (Convention on the Conservation of European Wildlife and Natural Habitats) and Convention on Biological Diversity and Ramsar (Convention on Wetlands of International Importance). While these do not provide any legislative protection, they provide the legal framework for signatory nations to implement their own protection measures.

# *1.4.3.1. Management implications of an international*

### treaty

Historical fishing agreements between France and Jersey (between 2004 and 2021), resulted in shared fishing grounds (Figure 1.7) defined and jointly managed through the Bay of Granville Agreement (Chambers et al., 2020; Fleury, 2011), which granted access to both Jersey and French vessels in possession of a permit (Figure 1.7). Both the Minquiers and Ecrehous reefs are located within the shared fishing grounds. As a result, the politics and laws surrounding the offshore reefs are complex, making them a challenge to manage effectively (Chambers et al., 2020; Fleury, 2011; Fleury & Johnson, 2015).

Evidence of the high biodiversity and priority habitats found at the Ecrehous and Minquiers, were used to argue the need to put management measures in place to protect them. The simplest and more readily enforceable solution was to exclude mobile gear access to both Jersey and French vessels, creating what are now known as the MPAs. Negotiations for the MPAs lasted five years before an agreement was made and it is through the continued work of both local scientists and fisheries officers that this was possible. This is because, until very recently, decisions on management within the Bay of Granville had to be agreed upon during

tri-annual meetings where a combination of government officials, fisheries officers, fishers and scientists, from both Jersey and France were required to attend.

Management of Jersey waters has recently changed due to the Bay of Granville Agreement being abolished in 2021 following changes related to Brexit. After months of negotiations, a new deal with France called the Trade and Cooperation Agreement (TCA) was signed on 30<sup>th</sup> December 2020 that would allow the continuation of shared access to Jersey and French boats based on their track record. However, the abolishment of the Bay of Granville Agreement in 2021 has led to major disputes between Jersey and France. In March 2021, Jersey fishers blockaded the main port of St. Helier in Jersey to protest fishing rights following Brexit that prevented them landing their catch into France. Not long after the French fishing fleet blockaded the same port of St. Helier on 6<sup>th</sup> May 2021 to protest their access rights to the shared fishing area. The boundaries that define the Bay of Granville, and the various reporting zones and management measures within it, remain the same, with the key changes concerning vessel licencing and management, control of which now resides with the Jersey authorities.

# 1.5. Research aims

The purpose of this thesis is to investigate whether MPAs that follow a whole-site approach can deliver benefits to both biodiversity and society (social and economic), by gathering information on habitat composition, species assemblages and fisheries data. This thesis aims to answer two broad questions;

1. What are the socio-economic impacts to the local fishing industry of MPAs that follow the whole-site approach in Jersey?

2. What are the differences in benthic habitat composition and associated species assemblages within MPAs that exclude bottom-towed fishing compared to Open Controls?

# 1.6. Thesis structure

This thesis comprises 7 chapters and each data chapter (chapters 2 through 6) follows the structure of a standalone paper and therefore there is some unavoidable repetition of background information. Current MPA literature is explored in chapter 1 to identify knowledge gaps and the opportunities presented by the MPAs in Jersey to further this field of research. In chapter 2, the value of marine habitats to local fisheries is examined in the context of how the MPAs are safeguarding this value. Landings could not be attributed directly to the MPAs so interviews were conducted with local fishers to determine the impact of the MPAs on their fishing frequency, location and well-being (chapter 3).

The ecological and physical changes following the exclusion of bottom-towed fishing are assessed in chapters 4 (mobile species) and 5 (epibiota, sediment properties and infauna). Mixed sediment habitats, such as those targeted by bottom-towed fishing practices, were selected for study inside the MPAs and in nearby fished control sites to assess the potential contribution of these interstitial habitats towards the recovery of biodiversity in MPAs. Chapter 6 bridges the gap between socio-economics and ecology by investigating the response of key commercial crustacean species to the MPAs through the use of potting surveys. A final discussion chapter (chapter 7) summarises the findings and discusses implications for management.

Chapter 1: Introduction of the literature, knowledge gaps and case study area.

Chapter 2: The Value of Coastal Habitats to Commercial Fisheries in Jersey, English Channel, and the Role of Marine Protected Areas. In review. *Fisheries Management and Ecology*.

Chapter 3: The socio-economic impact of Marine Protected Areas in Jersey; a fishers' perspective. In prep for submission to *Marine Policy*.

Chapter 4: Removal of Bottom-Towed Fishing from Whole-Site Marine Protected Areas (MPAs) in Jersey's Territorial Waters (Channel Islands, UK) Promotes Mobile Species Biodiversity. In review. *Estuarine, Coastal and Shelf Science*.

Chapter 5: The response of epibiotic and infaunal assemblages to the removal of bottom-towed-fishing from Marine Protected Areas in Jersey, Channel Islands, UK. In prep for submission to *Estuarine, Coastal and Shelf Science*.

Chapter 6: The response of commercial crustacean populations to the removal of bottom-towed fishing within Marine Protected Areas in Jersey, Channel Islands, UK. In prep for submission to the *Regional Studies in Marine Science*.

Chapter 7: Discussion – a summary of the findings from this research and their contribution to the understanding of the effectiveness of MPAs that follow the whole-site approach in temperate waters.

 Chapter 2 - The value of coastal habitats to commercial fisheries in Jersey, English Channel, and the role of Marine Protected Areas

# Author Contributions

Samantha Blampied was responsible for the conception and design of this research. She organised the fieldwork, undertook the analysis, interpretation of data and writing of the chapter. Francis Binney assisted in the acquisition of data from government sources and all supervisors contributed to the critical revision of manuscripts and have given their approval for inclusion in this thesis. This manuscript is in review with the *Journal of Fisheries Management and Ecology*.

Author	Institution	Contribution
Samantha Blampied	University of Plymouth	65%
Sian Rees	University of Plymouth	15%
Martin Attrill	University of Plymouth	5%
Francis Binney	Government of Jersey	5%
Emma Sheehan	University of Plymouth	10%

#### Abstract

Fisheries are in decline worldwide and Marine Protected Areas (MPAs) are being advocated as tools that can not only protect and restore biodiversity but also improve fisheries sustainability and protect fisher livelihoods. This is achieved by implementing spatial management measures within MPAs to exclude destructive fishing methods from sensitive benthic habitats that are protected as a conservation feature. Whilst fisheries benefits are not the main driver of conservation measures in MPAs, many features of conservation interest are also important for the essential life history stages of commercial fish species, and so the two objectives of sustainable fisheries and marine conservation are linked. To understand the current role of MPAs in underpinning commercial fisheries, this study demonstrates the economic value (first sale (£), wet weight (kg) of commercial landings) of Jersey's benthic habitats for five dominant commercial species (Homarus gammarus, Cancer pagurus, Maja brachydactyla, Pecten maximus, Buccinum undatum) that are targeted by both French and Jersey fishers. This value is apportioned between habitats that support these commercial species across all their essential life history stages within the defined fishing area that comprises Jersey's territorial waters. Much of the Jersey territorial waters are currently open to bottom-towed fishing which has long term and adverse impacts on benthic habitats. This study analyses the proportional economic value that is protected within MPAs that have spatial management measures that exclude bottom-towed fishing from areas of seabed. For each species, it was found that multiple habitats across Jersey's territorial waters, contribute to their economic value of £7,521,275 to Jersey fisheries. For Jersey's fisheries £2.7 million of the economic value is protected within MPAs that exclude bottom-towed fishing. The economic value of individual habitats to Jersey and French fisheries differed but

subtidal sediments were the most valuable to both Jersey (£2.1 million) and French (£2.5 million) fisheries annually. Subtidal sediments were also the least protected habitat from bottom-towed fishing (2.73%). The findings from this study support the ecosystem-based approach to fisheries management and emphasizes the importance of considering species life histories, and their habitat requirements, in their management.

### 2.1. Introduction

Fisheries are a source of global food security, but many fish stocks have been overexploited (Pauly, 2009), Corresponding

habitat destruction caused by fishing gear has exacerbated global declines in ecosystem health, fish biomass, and economic value (Steneck & Pauly, 2019; Sumaila et al., 2012; Worm et al., 2006). As humans and ecosystems are intrinsically linked (UNEP, 2006b), the effective management of ocean resources is essential to support human well-being (Millennium Ecosystem Assessment, 2005) and to achieve the interlinked sustainability objectives defined in the United Nations Sustainable Development Goals (Rees et al., 2018; UN General Assembly, 2015). The processes of natural environments in supporting human well-being have been described as ecosystem services (Beaumont et al., 2007; UNEP, 2006b), which are categorised into provisioning, regulating, cultural and supporting services (Millennium Ecosystem Assessment, 2005). To raise the profile of the importance of ecosystem services to human wellbeing there have been research programs and projects that have sought to value ecosystem services (Costanza et al., 1996, 2014; Millennium Ecosystem Assessment, 2005; TEEB, 2010).

Traditionally, measurements of fisheries sustainability have focused on population based metrics, such as Maximum Sustainable Yield (Gemert & Andersen, 2018; Ulrich et al., 2017). However, these methods do not take into account the wider ecosystem that is supporting the fishery. Many fish species rely on multiple habitats across their life cycle (Seitz et al. 2014). They specifically rely on 'essential habitat' defined as waters and substrates that are necessary for feeding, growth and reproduction of fish species in their various life stages (Rosenberg et al., 2000) Habitats in which adults are caught provide evidence of habitat use in their adult stages, but just as vital to their life history are the habitats they rely on in their spawning and juvenile stages. Spawning areas are those where adults migrate to release eggs, and nursery areas are characterised by high densities of juveniles and support increased growth and survival compared to surrounding habitat (Beck et al., 2001).

Recently there have been attempts to further incorporate habitat information into fisheries management (Brown et al., 2019) to move towards Ecosystem Based Fisheries Management (Halpern et al., 2010; Pikitch et al., 2004; Rosenberg & McLeod, 2005). Benthic habitats in particular are important for fisheries as they provide shelter, foraging grounds and breeding grounds for species targeted by commercial fisheries (Howarth et al., 2011; Kritzer et al., 2016; Laurel et al., 2009). The definition of habitat is varied in the literature (Elliott et al., 2016); here the term 'habitat' refers to the physical characteristics of benthic substrates, as can be classified using the hierarchical European Nature Information System (EUNIS). The essential life history stages of fish species are supported by the multiple dimensions of the marine environment (Elliott et al., 2017a; Kritzer et al., 2016). Therefore, an understanding of species-habitat associations is central to this approach to identify

essential habitats that support the life stages of exploited species and are therefore of high priority for management (Seitz et al., 2014).

Commercial fisheries do not operate in isolation of the wider ecosystem. Across the board marine and coastal habitats are threatened by multiple anthropogenic impacts (Lotze et al., 2006), including fishing practices (Halpern et al., 2007). Bottom-towed fishing practices, such as trawling and scallop dredging, cause extensive damage to the seabed (De Grave & Whitaker, 1999; Sheehan, Cousens, et al., 2013; Thrush & Dayton, 2002). This can create conflict with other fisheries targeting the same habitats through direct damage to target species (Beukers-Stewart & Beukers-Stewart, 2009b; Kaiser & Spencer, 1995) or indirectly through habitat degradation and disruption of trophic dynamics (De Grave & Whitaker, 1999; Thrush & Dayton, 2002).

Marine conservation through MPAs and sustainable fisheries are intrinsically linked (Rees et al., 2020). Marine Protected Areas are not fisheries management tools; they are 'clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values' (Day et al., 2012). However, it is often the case that legal instruments available to fisheries managers (such as bylaws) that exclude specified fishing metiers from spatial areas are employed to protect specific features within MPAs. There is increasing evidence that MPAs with fishery management measures to exclude bottom-towed gear can both protect and support the recovery of protected features (Sheehan, Stevens, et al., 2013; Solandt et al., 2020) and provide social and economic benefits (Rees et al., 2021a; Rees et al., 2021b). Additionally, where the structural integrity of the seabed

can be maintained or improved, it enhances the ability of a system to support fishery species (Davies et al., 2021a; Elliott et al., 2017a; Howarth et al., 2011). At present, the majority of fisheries bylaws to protect conservation features fall within national networks of MPAs and it has been noted that the potential of a MPA to improve the sustainability of fisheries will be limited if habitats that are important to the survival of commercially targeted species fall outside of boundary of protection (Klein et al., 2015). It is broadly recognised that there is now a need to think beyond MPAs as the sole tool to underpin all the demands that society requires from marine ecosystems, moving towards sustainable use of oceans for a wide range of societal benefits (Laffoley, 2021; Rees et al., 2020a).

As well as supporting food security, fisheries also provide, through direct market values at the point of landing, a source of jobs and income (FAO, 2020). Economic valuation applied to ecological systems is proving to be a useful tool to progress debate and discussion as to the costs and benefits of fisheries management and conservation (Ovando et al., 2016). Several studies make use of first-sale wet weight market values at the point of port landings to serve as an indicator of the value of fisheries to a coastal communities (Bastardie et al., 2010; Jackson et al., 2015; Rees et al., 2021). However, there is a movement towards an ecosystem-based approach to marine management recognised in both policy (e.g. Food and Agriculture Organization of the United Nations (Staples & Funge-Smith, 2009); Marine Strategy Framework Directive (European Parliament and Council, 2008); and Centre for Environment, Fisheries and Aquaculture (Cefas, 2012)) and practice (Marshall et al., 2019) as a tool to link essential habitats both inside and outside MPAs.

There remains a lack of research that links the role of essential habitat in underpinning the economic value of fisheries, including consideration of current levels of habitat protection through MPAs. In this paper we apply an interdisciplinary, mixed method approach to further bridge the gap between ecology and economics. We test this in a case study area.

### 2.2. Study location

Jersey is a self-governing dependency of the United Kingdom situated in the Normano-Breton Gulf off the west coast of Normandy. The marine territorial area (~ 2,500km<sup>2</sup>) has a maximum depth of 50 m deep consists of rocky reef, boulders, cobbles, gravel, sand and biogenic habitats, such as maerl and seagrass. The large tidal range experienced in Jersey (12.2 m) also creates a vast intertidal area. As an island community, Jersey depends on its marine estate, which supports tourism, recreation and commercial exploits, especially fisheries. Commercial catch comprises a mixture of shellfish and finfish, with 94% of commercial landings attributed to five shellfish species: lobster (Homarus gammarus); brown crab (Cancer pagurus); spider crab (Maja brachydactyla); scallop (Pecten maximus); and whelk (Buccinum undatum). Most commercial fishing by Jersey vessels for these five shellfish species (>90%) takes place within Jersey's territorial waters from a combination of static and bottom-towed fishing vessels, and SCUBA diving vessels. Based on Landings Per Unit Effort (LPUE), there have been local declines in lobster since 2015 and brown crab since 2012, while increases in LPUE have been observed for spider crab since 2013. Whelk LPUE has shown a slight decline since 2016 and scallop LPUE has been declining for dredge fisheries since 2013 but increasing for dive fisheries since 2015 (Marine Resources, 2019). With fisheries largely dependent on benthic species, management of benthic habitats is key to

ensuring the sustainability of these stocks. While there will be movement of adult and larval stages of commercial species across the boundary of Jersey's territorial waters, detailed spatial coverage of habitats outside of this area is currently unknown.

The majority of Jersey's fishing fleet comprises vessels under 12 metres and are therefore not required to have Vessel Monitoring Systems (VMS) that track their fishing positions. Instead fishers are required to submit logbooks that report their catch and the fishing zones the catch came from. Landings data are not attributable to the MPAs as fishers are not required to state whether they fished in the MPA and the reporting zones are larger than the MPAs. Jersey fishing vessels operate from a number of harbours and marinas across the island, with the majority landing their catch into the local port of St. Helier, although vessels are also permitted to land their catch in France. Historical fishing agreements between France and Jersey have resulted in shared fishing grounds defined in the Bay of Granville Treaty (Fleury 2011), which grants access to both Jersey and French vessels in possession of a fishing permit (Figure 2.1). To protect areas of fragile habitat and high biodiversity within Jersey's territorial waters that are shared with France, two offshore MPAs were designated in 2017 (Chambers et al. 2020) to exclude the use of bottom-towed fishing gears (Figure 2.1). This, in addition to the multiple coastal MPAs already in place within Jersey's exclusive fishing zone, has resulted in 6.5% of Jersey's territorial waters being protected from bottom-towed fishing. Currently these are the only management measures specific to protecting benthic habitats within the Normano-Breton Gulf.





The aim of this study was to assess the availability of essential fish habitat that supports the supply of commercially important species, with the habitats investigated being those that support adult, juvenile and spawning stages. Here, the economic value (landings value  $(\pounds)$ ) is assumed to be supported equally by all habitats used in a species life cycle. This is a standard approach in economic and ecological valuation studies where there is not sufficient ecological evidence of the availability and suitability of other habitats to fill a gap (Jackson et al., 2015; Seitz et al., 2014). It is also not the purpose of this research to highlight the economic value

of fishing grounds (adult life history stages) over and above other areas important for the availability of species to the commercial market.

An understanding of which habitats in Jersey contribute to the final economic value of a fishery, and whether these habitats are currently covered by spatial management measures that exclude bottom-towed fishing, can be used to better inform decisions in Marine Spatial Planning (MSP). It is therefore hypothesized that, within the defined fishing area of Jersey's territorial waters, multiple habitats will contribute to the final value of exploited species and the value safeguarded for each fishery species by the MPAs will differ depending on their habitat associations.

#### 2.3. Materials and methods

To determine how the marine habitats in Jersey contribute to the value of commercial fisheries and assess sustainability of current management approaches to secure this value, a novel mixed methods approach was used. This involved combining landings data, primary sales information, spatial habitat and MPA data, and life history information.

#### 2.3.1. Commercial landings value

Commercial fishers in Jersey are required to submit logbooks, which state their catch in kilograms for every fishing trip. Data for trips between 2015 to 2018 were obtained from the Government of Jersey Marine Resources team and used to calculate mean yearly landings for all commercial species and identify key fisheries. Whilst commercial fishing activity can and does take place in shared waters, the majority of landings into Jersey ports (~94%) are caught within the island's territorial waters (Marine and Coastal Manager, Jersey Marine Resources, pers. comm.).

Data on primary fish sales in Jersey were based on quarterly buying prices (wet weight) from a local vendor (The Fresh Fish Co) between 2015 and 2018. Landings value (first sale wet weight) in Jersey was calculated by multiplying the quarterly landings weight by the quarterly primary sale value. Quarterly values were summed for each year and then averaged across the four years to give a mean landings value (£) per year for each fishery.

As French vessels target the same species in Jersey's territorial waters as Jersey vessels, it was also necessary to quantify French landings. Landings data between 2015 and 2018 for French vessels were supplied by the Government of Jersey Marine Resources team in tonnes per year and value per year ( $\in$ ) based on information shared by a French research institution, IFREMER (The Institut Français de recherche pour l'exploitation de la mer). These data were filtered for the same key fisheries as Jersey and converted to kilograms. Values were converted ( $\in$  to £) using exchange rates from statista.com on 4<sup>th</sup> February 2021 (De Best, 2021). From this, mean yearly landings values (£) were calculated.

#### 2.3.2. Habitats and essential life history stages for commercial species value

Spatial habitat information was obtained from the Government of Jersey Marine Resources team. This habitat information is a combination of modelled and survey data and represents best available evidence of the location of benthic habitats in Jersey's territorial waters (Chambers in peer review). Habitats are defined using the European Nature Information System (EUNIS) habitat classification system. The EUNIS classification system is a publicly accessible database that provides reference information for European habitats and is widely compatible with peer reviewed literature as well as legislation and marine spatial planning across Europe

(Davies et al., 2004). To establish how different shellfish species utilise habitats throughout their life history stages a species-habitat matrix was created from literature complied primarily through Google Scholar, using search terms that included each species name and all known habitat types. Literature was searched using specific EUNIS habitat nomenclature and also broad habitats groups to account for lack of detail on species habitat usage in the literature (Seitz et al. 2014). When information could not be found for certain life stages of a species, literature from laboratory studies or closely related species were used (Howard & Bennett, 1979a; Kamenos et al., 2004b).

EUNIS habitat data for Jersey's territorial waters were grouped into broad categories (Table 2.1), to account for lack of available information on species habitat usage in the literature (Seitz et al. 2014). Jersey's large intertidal area provides essential habitat for both juvenile brown crab and lobster, and all habitats above the low water mark have been grouped into an intertidal habitat category. All rock and boulder categories below the low water mark were grouped into subtidal hard substrate, and sedimentary habitats below the low water mark were categorised into subtidal sediment. Seagrass and maerl have been separated from subtidal sediment as they provide added structural complexity, which support distinct species assemblages (Howarth et al., 2011; Jackson et al., 2002).

Table 2.1. Habitat table detailing the EUNIS code habitats that have been grouped together.

Habitat Group	EUNIS Codes	Description
Intertidal sand and rock	A1, A2	All hard and soft substrates in the intertidal zone.
Subtidal hard substrate	A3.12, A3.214, A4.13	Bedrock and boulders below the low water mark.
Subtidal sediment	$ \left  \begin{array}{cccc} A5.133, & A5.135, & A5.137, \\ A5.141, & A5.142, & A5.145, \\ A5.231, & A5.431, & A5.451 \end{array} \right  $	Coarse and mixed sediments below the low water mark.
Maerl beds	A5.51	Coralline red algae.
Seagrass meadows	A5.53	Zostera marina and Zostera noltei.

# 2.3.3. Apportioning commercial fisheries values to essential life history

To provide the annual value to commercial fisheries linked to Jersey's habitats, and the proportion of value protected from physical impacts from bottom-towed fishing gear, fishery species landing values were apportioned to habitats used in their life cycle. Habitat information was available on a 250 m resolution, which provided areas of habitat across Jersey's territorial waters and within the MPAs (Figure 2.2). For each commercial species, their landings value was spread equally across the multiple habitats used in their life cycle. Values were divided equally as all habitats used by a species were assumed to contribute to their ability to reach their adult stage and therefore their economic value. This was calculated separately for Jersey and French landings values. The value of benthic habitat safeguarded through the exclusion of bottom-towed fishing gear was calculated by multiplying the proportion of each habitat within the MPAs by the total value of the habitat.





Habitat Type

- Intertidal sand and rock
- Maerl beds
- Seagrass beds
- Subtidal hard substrate
- Subtidal sediment

Figure 2.2. The spatial distribution of the five habitat groups in Jersey's territorial waters. MPAs are outlined in black.

# 2.4. Results

# 2.4.1. Identifying commercially important species to Jersey fisheries

The majority of Jersey's fisheries are dependent on shellfish (Figure 2.3), with approximately 1.5 million kg (94%) of landings per year attributed to five shellfish species: Lobster, Homarus gammarus; Brown crab, Cancer pagurus; Spider crab, Maja brachydactyla; Scallop, Pecten maximus; and Whelk, Buccinum undatum. All other species (n=46) combined accounted for ~140,000 kg (6%) of landings, much of which was incidental catch, such as catsharks (Scyliorhinus spp.), a bycatch of the netting and potting fisheries. Individually these species do not influence overall fisheries value and so have not been included for further investigation.



Figure 2.3. Commercial landings (kg) between 2015 and 2018 showing the top 15 species that contribute to 99% of landings by weight. The remaining 1% of commercial landings consists of 36 species. Outliers are shown by filled circles.

## 2.4.2. Commercial landings value

The combined value of shellfish landings from Jersey and French vessels operating in Jersey's territorial waters resulted in an average value of £14,664,729 ± £969,105 per year in primary sales (wet weight) per year, with £7,521,275 ± £1,366,282 of this total attributed to landings from Jersey vessels. The value of each fishery varied between Jersey and France (Figure 2.4). Jersey obtained the majority of its value from lobster (£4.1 million per year), whereas France obtained the majority of its value from a combination of scallop (£1.9 million per year), spider crab (£2 million per year) and whelk (£2.4 million per year). Scallop and whelk contributed less to Jersey fisheries compared with France (Figure 2.4). France reported brown crab and lobster as contributing relatively low values to their overall fishery (Figure 2.3). Jersey scallop landings can be differentiated between dived and dredged (Figure 2.3), and this shows the majority of scallop value in Jersey was attributed to dredging practices.



Figure 2.4. Mean landings (£) per year (2015-2018) for each species fishery and nation (France/Jersey) from Jerseys territorial waters.

### 2.4.3. Habitat value and value protected

Based on current literature, habitat use varies per species and all habitats are used by multiple species (Table 2.2). Literature evidence highlights maerl as a foraging habitat for all species, except lobster, and is also a nursery and spawning habitat for scallop. Scallops use maerl and subtidal sediment for all three life history stages. Seagrass is used as a foraging habitat by all three crustacean species, but is not currently known to be used by the two mollusc species. Intertidal sand and rock is a nursery habitat for brown crab and lobster. Subtidal hard substrate is both a foraging and nursery ground for all three crustacean species, and a spawning ground for whelk. Subtidal sediment is a foraging ground for all species, but only a nursery ground for lobster and whelk and a spawning ground for brown crab. Of the five species considered, crustaceans utilise more habitats than molluscs (Table 2.2). Brown crab is particularly ubiquitous and uses all habitats, while scallops only use two. All species are able to use more habitats for foraging purposes than for nursery or spawning.

Table 2.2. Commercial shellfish species habitat usage as spawning areas (S), nursery areas (N), and feeding areas (F). Full references are listed in Appendix A (Table. A.1).

Species	Intertidal sand and rock	Subtidal hard substrate	Subtidal sediment	Maerl	Seagrass	References	
Lobster (Homarus gammarus)	N	N,F	N,F		F	Seitz et al. (2014); Howard and Bennett (1979); Wahle and Ste- neck (1991); Wilson (2008); Lin- nane et al. (2000); Lawton and Lavalli (1995); Galparsoro et al. (2009); Unsworth and Cullen- Unsworth (2015)	
Brown Crab ( <i>Cancer pagurus</i> )	N	N,F	F,S	F	F	Seitz et al. (2014); Sheeny and Prior (2008); Hall et al. (1993, 1991); Brown and Bennett (1980); Neal and Wilson (2008); Hauton et al. (2003); Unsworth and Cullen- Unsworth (2015)	
Spider Crab ( <i>Maja brachydactyla</i> )		N,F	F	F	F	Hosie (2009); Gonzalez-Gurriaran and Freire (1994); Corgos et al. (2011); Unsworth and Cullen- Unsworth (2015)	
Scallop (Pecten maximus)			S,N,F	S,N,F		Marshall and Wilson (2008); Howarth et al. (2011); Kamenos et al. (2004); Irlandi et al. (1999)	
Whelk (Buccinum undatum)		S	N,F	F		Seitz et al. (2014); Ager (2008); Valentinsson et al. (1999); Blunden et al. (1977)	

Value differed between habitats and between nations depending on the number of species that used the habitat and the level of exploitation of each species by the two nations (Figure 2.5). Subtidal sediment had the highest economic value to both

nations as it was the only habitat used by all species (Figure 2.5). The second most valuable habitat for Jersey's commercial fisheries was subtidal hard substrate, whereas for France it was maerl. Maerl beds were of greater value to the current French fisheries compared to Jersey due to the association of scallop and whelk with this habitat that are exploited to a greater level by vessels from France (Figure 2.5). Seagrass and intertidal habitats were of greater value to Jersey compared to France due to the association of lobster with these habitats as essential to survival to adulthood (Figure 2.5, Table 2.3).



Figure 2.5. The annual (2015-2018) contribution of each species to the habitat value  $(\pounds)$  for each nation (Jersey and France).

Several habitats of high value to both Jersey and French fisheries were largely unprotected by the MPAs (Figure 2.6), with a total economic value of £4,127,999 protected annually between the nations (Table 2.3). Cumulatively, subtidal sediment was the most valuable habitat to both nations (£2.12 million to Jersey and £2.47 million to France annually) but was also the least protected habitat (Figure 2.6) with

just 2.73 % safeguarded within the MPAs (Table 2.3). The value protected of maerl and subtidal hard substrate was similarly low (14.78 and 8.91 % respectively). Consequently, due to their greater reliance on scallop and whelk that are supported by habitats with greater coverage outside of the MPAs, habitats supporting French fisheries have less of their value protected than Jersey fisheries. There was a substantial economic value for seagrass (97.04 %) and intertidal rock (88.49 %) protected within the MPAs, although to both nations these two habitats had an overall lower contribution to the economic value of fisheries (Figure 2.6, Table 2.3).



Figure 2.6. The total value per habitat group for each nation (Jersey and France). Hatched areas show the value protected within the MPAs.

A large proportion of subtidal hard substrate and subtidal sediment are not protected, but these habitats have a greater total area within Jersey's territorial waters, meaning that the total area of these habitats protected by the MPAs is actually greater in comparison to seagrass, maerl and intertidal (Table 2.3). The value of each fishery species protected within the MPAs from bottom-towed fishing gear differed depending on the area of their associated habitats falling within the MPAs (Table 2.3). Lobster had the most value protected within the MPAs (49.29 %), followed closely by brown crab (42.38 %), due to the association of these species with seagrass and intertidal habitat, of which the majority is within MPAs. The value of scallop (8.75 %) and whelk (8.8 %) protected was similarly low (Table 2.3), and this stems from their association with fewer habitats, the majority of which fall outside of the MPA boundaries (Table. 2.2, Fig 2.5).

Table 2.3. The total area of each habitat within Jersey's territorial waters, the area contained within the MPAs, the total annual value (£) and the value (£) protected within the MPAs for Jersey and French fisheries.

Habitat group	Total Area	Area inside	Jersey Total	French Total	Jersey Value	French Value	Value (%)
	$(km^2)$	$MPA (km^2)$	Value (£)	Value (£)	(£) Protected	(£) Protected	Protected
Intertidal sand and rock	29.10	25.75	1,222,024	188,721	1,082,739	167,211	88.49
Maerl	56.70	8.38	1,101,490	2,291,821	162,734	338,593	14.78
Seagrass	4.06	3.94	1324117	701075	1,283,375	679,504	97.04
Subtidal hard substrate	806.00	71.81	1750998	1494921	155,961	133,152	8.91
Subtidal sediment	1440.00	39.25	2,122,646	2,466,915	57687	67043	2.73
Habitat Total	-	-	$7,\!521,\!275$	$7,\!143,\!454$	$2,\!742,\!496$	$1,\!385,\!503$	-
Species							
Brown Crab	-	-	1,004,335	68138	425,685	28,880	42.38
Lobster	-	-	4,084,626	700,375	2,013,209	345,197	49.29
Scallop	-	-	743,297	$1,\!943,\!987$	65,007	17,0018	8.75
Spider Crab	-	-	408,376	2,049,415	125,904	631,843	30.83
Whelk	-	-	1,280,641	$2,\!381,\!539$	112,691	209,565	8.80
Species Total	-	-	$7,\!521,\!275$	$7,\!143,\!454$	2,742,496	$1,\!385,\!503$	-

# 2.5. Discussion

The objective of this research was to determine how coastal habitats support the economic value of key fisheries in Jersey and France and whether these habitats are appropriately safeguarded within the MPAs. The fishery species studied here utilised multiple habitats, as has been shown in previous studies (Kritzer et al., 2016; Seitz et al., 2014). This current work has expanded on this to show the economic value of habitats to local fisheries by distributing landings values (first sale (£) wet weight (kg)) across habitats associated with each species life history stage. It was found

that multiple habitats contribute to the value of fisheries, but the majority of this value is not safeguarded within MPAs that have specific management measures to protect the seabed from bottom-towed fishing practices. The five key fishery species are worth almost £7.5 million a year in primary wet weight sales to Jersey fisheries and £7.1 to French fisheries. £2.7 of this value to Jersey fisheries is safeguarded within the MPAs. A lower value (£1.4) is safeguarded for the value it provides to French fisheries due to their reliance on species that primarily utilise habitats for their essential life history stages that are underrepresented within the MPAs.

Previous efforts to value the marine environment have focussed on single habitats (Jackson et al., 2015) or fishing grounds (Bastardie et al., 2010; Lee et al., 2010), and those that have considered multiple habitats have not done so in monetary terms (Kritzer et al., 2016; Seitz et al., 2014). Here a method has been provided in which life history information can be used to better understand how the economic value of fisheries is related to broad habitats by analysing the market value of species according to the habitats necessary to reach commercial viability (adult). However, habitat value is not fixed and these results only provide a snapshot of the current habitat value. As markets change, landings values will also change to reflect demand.

Habitat value varied across the species and was of differing economic value to France and Jersey's fisheries; however, subtidal sediments were the most valuable to both nations. Kritzer et al. (2014) identified soft sediments as supporting the greatest range of species due to its greater availability, meaning that more species are able to utilise it. This is supported in the findings of the current study which has shown subtidal sediments, the most prevalent habitat, to support all five of the key
commercial species and was therefore also worth the greatest value to fisheries as a whole. What is vital here is the evidence that this habitat supports all the studied commercial species through to adulthood and also that only a fractional area of subtidal sediment (and therefore value) is currently protected from destructive fishing practices (namely scallop dredging). This may undermine the potential for this value to the broader fishing community to be safeguarded and even improved for future generations. With no clearly defined thresholds for what may be considered 'acceptable' levels of impact to sediment habitats (Hooper et al., 2017) there arises an ethical consideration as to whether one sector of an industry can place at risk the ability of another sector to continue to benefit.

Species move between habitats during their life cycle (Seitz et al., 2014) and therefore their economic value is dependent on multiple habitats. Protection from activities that cause impact to the structure and function of habitats that are essential in supporting the life stages of commercial species is fundamental to maintaining the economic value of fisheries. This raises several points for discussion with regard to a more integrated approach to fisheries management, whereby multiple habitats are protected from the impacts of bottom-towed fishing in order to allow for habitat connectivity between life stages. While every habitat group in Jersey was represented within the MPAs, the degree to which each habitat group was protected varied; for example, 97% of seagrass as opposed to 3% of subtidal sediment was protected. Specific habitats may be a limiting factor for species that are not able to substitute one habitat for another in certain life stages (Halpern, 2004; Parrish & Polovina, 1994; Wahle & Steneck, 1991). If, for example, seagrass habitat was lost, it is possible that many species would be able to utilise adjacent habitats as substitutes, as has been observed in intertidal boulder fields (Chapman, 2012). But

survivability may be greatly reduced if substituted habitats are of low quality in comparison to their original habitat (Godet et al., 2018). These results have shown seagrass to have a combined value of £2 million annually to both Jersey and French fisheries and loss of seagrass habitat could result in substantial economic costs to fisheries if species are not able to survive on near-by habitat. Additionally, the associated changes to the food web through loss of habitat could have negative consequences for many species or result in altered communities (Komyakova et al., 2019).

Nursery habitats are characterised by high densities of juveniles (Beck et al., 2001) and may be a demographic bottleneck for many species (Nagelkerken et al., 2015). Soft sediments are important nursery habitats, especially in the Mid-Atlantic and North Atlantic (Kritzer et al., 2016), including commercially important species (Seitz et al., 2014). Subtidal sediment was the only habitat utilised by all five of the fishery species studied here and is also a habitat primarily targeted by bottom-towed fishing gear due to France's resource demand for scallop (*P. maximus*). This habitat is of more value to lobster fisheries than scallop fisheries and dredging for scallops may be negatively impacting the lobster fishery, reducing nursery habitat integrity (Thrush & Dayton, 2002) and thus supporting a less valuable fishery. Subtidal sediment makes up the vast majority of Jersey's subtidal area and there is a need for future discussion with regard to further spatial management of the scallop fleet, guidance for which may be sought through the development of indicators of Good Environmental Status for seabed integrity within the European Union's MSFD (European Parliament and Council, 2008). To achieve an ecosystem-based approach to fisheries management in Jersey, a balance needs to be sought that recognises the commercial needs of the scallop fleet but also enables long-term

sustainability of the lobster fishery; this will require that some areas of subtidal sediment are protected to maintain the integrity of the seabed and improve connectivity between life stages of lobster. With no thresholds available as to how much area of a habitat is needed to the essential life history of commercial species, a precautionary approach is advised as the value of this habitat to both nations is considerable.

Economic values apportioned to habitats differed between the two nations, as French fishers exploited the same species differently to Jersey fishers, and therefore the value protected within the MPAs differed between the nations. At the time of study, maerl was of greater value to French fisheries due to higher levels of exploitation of scallops. Notably maerl was also one of the least protected habitats via MPAs that exclude bottom trawling. Structurally complex habitats may promote scallop spat settlement and juvenile abundance (Howarth et al., 2011; Kamenos et al., 2004b) and large areas of maerl can provide this structural complexity (Kamenos et al., 2003; Sheehan, Bridger, & Attrill, 2015). Maerl habitats were largely unprotected from bottom-towed fishing gear, but were also economically valuable to multiple fisheries. The use of MPAs to restrict bottom-towed fishing gear could improve the sustainability of the scallop fisheries by protecting the integrity of maerl habitat. As scallops are broadcast spawners, fertilisation success is increased with greater densities of scallops (Vause et al., 2007), and protecting areas of seabed that support dense aggregations of scallops will result in increased spawning and improve recruitment to areas both within and beyond the protected areas (Beukers-Stewart & Beukers-Stewart, 2009a).

Species living in areas affected by bottom-towed fishing exhibit increased scavenging behaviour (Shephard et al., 2014) and the population of necrophagus whelks may benefit from the availability of dredge-damaged prey. In terms of landings weight, whelk is the largest fishery in Jersey, but due to its low market value this species is currently not as valuable (£1.3 million per year) as lobster (£4 million per year). Whelk has been fished in Jersey's waters by both French and Jersey vessels for decades, but there has been a recent expansion since 2018 (Marine Resources, 2019) in response to an increase in market demand through exports. This highlights the importance of considering the economic value of species in the socio-economic management of marine resources as well as a discussion on whether an emergent fishery is benefitting from a degraded ecosystem, as has been seen in the Firth of Clyde where over exploitation of benthic fish through bottom trawling resulted in a collapse of the fisheries until only the Norway lobster (*Nephrops norvegicus*) and scallop (*Pecten maximus*) fisheries remained (Thurstan & Roberts, 2010).

Effectiveness of governance and enforcement will have implications in the success of a MPA (Bennett & Dearden, 2014). Differing fishery regulations, such as quotas and closed seasons, create complications for management of shared resources, whereas clear simple measures that cover a seascape, such as a MPA, rather than a feature allow for more efficient and accurate enforcement (Chambers et al., 2020). As all of the key commercial species in Jersey are benthic, they are strongly associated with seabed habitats and therefore there is much to be gained from spatial management that protects this seabed. While reliance on certain species differs between Jersey and France due to differing markets, both nationalities are dependent on species that are using multiple habitats and therefore both

nationalities will benefit from having areas of mixed habitats protected from bottomtowed fishing gear. This potential fishery benefit has been shown in Lyme Bay where commercially exploited species increased in abundance following the removal of bottom-towed fishing pressure from a mosaic of habitats (Davies et al., 2021a). When economic values and livelihoods become part of the discussion with regard to MPAs the motivations for the protection of benthic habitats becomes more tangible as a fisheries benefit. The MPAs will consequently gain commercial support from those most likely to benefit from management that ensures sustainability (Rees et al., 2021; Rees et al., 2021).

This research has considered habitat use across a species' life history but has not captured the relative importance of one habitat over another or temporal variability of habitat use. It is also important to note that the species included were a subset of those found in Jersey's waters and were chosen based on their direct importance and value to commercial fisheries. The inclusion of other species may change the value of habitats, but would not diminish the key finding that multiple habitats contribute to fisheries value. Jersey's marine estate will also support other industries which are dependent on a healthy functioning ecosystem, such as recreational fishing, diving and tourism. Jackson et al. (2015) found that seagrass had a higher direct value to recreational fishing as opposed to commercial fishing in the Mediterranean. The vast intertidal area in Jersey supports several aquaculture concessions, further adding to the economic value of this habitat. There are other, less tangible, ecosystem services and benefits from marine ecosystems such as regulation of climate and water quality, educational and cultural values, all of which feed into human wellbeing (Costanza et al., 1996, 2014; Millennium Ecosystem Assessment, 2005; TEEB, 2010). The value of these ecosystem services need to be

considered in addition to the value that habitats provide to fisheries, especially if considering the case of allowing bottom-towed fishing which is known to alter the functional diversity and ecological processes of benthic communities (Tillin et al., 2006).

#### 2.6. Conclusion

Marine ecosystems are a source of income for coastal communities and the value of a fishery is not simply attributed to fishing grounds but is dispersed across a mosaic of habitats. This study demonstrates the value of coastal habitats in Jersey to commercial fisheries and evidences the importance of protecting multiple habitats to improve life stage connectivity and support fishery yield. The ability of a species to switch to a different habitat, should their primary habitat be degraded or lost, is not well understood and indirect use of habitats needs to be considered to fully understand the interactions between habitats and species survival, such as habitats that support the prey of target species, which may not be used by the target species itself. There is a need for fisheries and conservation to integrate an ecosystem-based approach to management that focusses on broader marine seascapes, not just singular habitats, to maximise habitat connectivity. These results should be applied to conservation and management to support long term commercial interests and the sustainability of a broad range of ecosystem services which underpin human wellbeing. Where shared waters are concerned, it can be difficult to make management decisions that will benefit both nations, especially when there are differing demands for fishery species. MPAs will protect habitats that contribute to fisheries value of both nations and by understanding how habitats contribute to individual fisheries MPA spatial coverage can be tailored to maximise economic value.

 Chapter 3 – The socio-economic impact of Marine Protected Areas in Jersey; a fishers' perspective.

# Author Contributions

Samantha Blampied was responsible for the conception and design of this research. She organised the data collection, undertook the analysis, interpretation of data and writing of the chapter. Francis Binney assisted in the acquisition of data from government sources and all supervisors contributed to the critical revision of manuscripts and have given their approval for inclusion in this thesis.

Author	Institution	Contribution
Samantha Blampied	University of Plymouth	75%
Sian Rees	University of Plymouth	10%
Martin Attrill	University of Plymouth	5%
Francis Binney	Government of Jersey	5%
Emma Sheehan	University of Plymouth	5%

#### Abstract

Marine Protected Areas (MPAs) are advocated as tools to support sustainable fisheries and biodiversity by excluding the most destructive fishing practices. Some MPAs protect the whole-site of habitat mosaics from bottom-towed fishing but most only restrict damaging activities from specific evidenced conservation features. Social and economic factors influence the success of a MPA but the impact of spatial management changes to local fisheries is rarely captured in post MPA designation monitoring. In Jersey, Channel Islands, two offshore MPAs that are managed following the whole-site approach were designated in 2017. This study provides an insight into the overall economic importance of key fishery species (whelk (Buccinum) undatum), brown crab (Cancer pagurus), scallop (Pecten maximus), lobster (Homarus gammarus), and spider crab (Maja brachydactyla) to local fishing fleets and the impact of the MPAs on local, small-scale fishers in Jersey. Total landings (kg and £) in Jersey pre and post MPA designation were calculated from logbook and primary sales data and a structured interview was developed and tested as a research tool to document the impact (positive and negative) of the MPA designation on local fishers. Specific questions were designed to elicit both quantitative and qualitative data relating to the participants (n=21) fishing activity; support for the MPAs; income and job satisfaction; subjective well-being and current sales strategies. The results demonstrate that one mobile gear vessel respondent was displaced from traditional fishing grounds as a result of the MPA designation. Fishers also reported an increase in static fishing inside the MPAs, post designation. Generally, there are high levels of support for the MPA from static fleet respondents. In the first year since designation there have been no reported changes to catch (kg, £). Mobile gear fishers reported a lower subjective wellbeing and satisfaction with

their job and income than static gear fishers, despite greater profits. The major concern reported by the mobile fleet that contributed to their stress levels, was not attributed to the MPA designations but related to conflict with French static net fisheries operating outside of the MPAs. Ongoing support for Jersey's fishers and securing a sustainable and fulfilling livelihood will require further spatial management of fisheries outside of the MPAs with a possibility of setting gear or effort limits within the MPAs.

## 3.1. Introduction

The ocean provides a range of ecosystem services that underpin human wellbeing (Millennium Ecosystem Assessment, 2005; UNEP, 2006b). Demand on ocean resources has been increasing, particularly on seafood as a source of food and income, and in 2018 approximately 59.5 million people employed globally in the primary sector of fisheries (FAO, 2020; UNEP, 2006b). The sustainable management of ocean resources is essential to support human well-being (Reimer & Devillers, 2021; UN General Assembly, 2015). The United Nations Sustainable Development Goals (SDGs) have outlined a number of targets to achieve sustainable development of natural resources (UN General Assembly, 2015). SDG Goal 14 in particular applies to the sustainable use of oceans, seas and marine resources. The implementation of Marine Protected Areas (MPAs) to conserve biodiversity that underpins social and economic development is integral towards achieving this goal.

Fisheries are a socially and economically valuable industry that operate both inside and outside MPAs. Ultimately, fisheries depend on ecosystem functions and

processes to enable the exploitation of commercial fish species (Dobson et al., 2006; Staples & Funge-Smith, 2009) and it is only within MPAs where these functions and processes are protected. It remains that the exploitation of fish species above ecologically sustainable limits is one of the biggest threats to this resource (Pauly et al., 2005), and despite both national and international management measures, there are still declines in fish stocks (Claudet et al., 2020; Halpern et al., 2008). Along with persistent fishing effort, there are various fishing methods globally that also damage the habitats that fish depend upon at various points in their lifecycle. Mobile fishing gear such as trawls and dredges that drag along the seafloor are considered amongst the most destructive due to the abrasion of the seabed that can have negative impacts to both habitats and associated species (Hall-Spencer & Moore, 2000; Hiddink et al., 2017; Thrush & Dayton, 2002; Tillin et al., 2006). Mobile bottom-towed fishing gears (hereafter referred to as mobile gear) can also cause conflict with other fishing sectors by intersecting spatially with static gear fisheries (Hattam et al., 2014), causing direct damage to target species (Beukers-Stewart & Beukers-Stewart, 2009a; Kaiser & Spencer, 1995), and through degradation of habitats that support other commercially important species (De Grave & Whitaker, 1999; Stewart & Howarth, 2016; Thrush & Dayton, 2002).

The restoration and recovery of marine biodiversity can be realised though the exclusion of mobile fishing gears inside MPAs (Davies et al., 2021a; Sheehan et al., 2013a). Typically, MPAs are established to protect or enhance biodiversity (Lester et al., 2009; Zupan et al., 2018) but they are also important for securing socio-economic benefits through increased ecosystem services (Costanza et al., 1996; Dasgupta, 2021) and by conserving ecosystem function that can support greater fish biomass, including commercial species biomass (Beukers-Stewart et al., 2005;

Brown et al., 2010; Willis et al., 2003). While highly protected MPAs are most beneficial to improve the status of biodiversity (Edgar et al., 2014; Lester & Halpern, 2008; Sala & Giakoumi, 2018), the level of protection afforded to MPAs varies greatly (Day et al., 2012; Greathead et al., 2020). Despite the evidence, it may not always be socially or politically acceptable to achieve a high level of protection, given the range of ocean livelihood dependencies.

Fisheries and MPAs have traditionally been in conflict (Hattam et al., 2014; Rees et al., 2013b), with social costs of MPA management measures felt in the fishing sector. These costs are not equally distributed (Hattam et al., 2014), with some sectors experiencing losses in catch and increased distances to fishing grounds (Mangi et al., 2011), compared to others that will benefit from reduced fishing pressure within the MPA boundary (Rees et al., 2021). Conflict may increase between stakeholders where there is a perceived benefit to one group over another (Agardy et al., 2011). MPAs that have excluded mobile gear while allowing other extractive activities to continue have resulted in low levels of support from mobile fishers (Rees et al., 2013a). There are clear synergies between conservation objectives of MPAs and fisheries management (Rees et al., 2020). It is argued that linking the social and ecological systems in MPA placement and purpose with small scale fisheries objectives may improve the ecological and socio-economic outcomes for both fisheries and conservation (Rees et al., 2020; Reimer & Devillers, 2021). Setting both social and ecological baselines within a long term MPA monitoring programme is therefore key to determine whether MPAs are meeting principle objectives. This is crucial to the long term success of a MPA, as MPAs that do not meet their objectives risk losing trust from stakeholders in the management, with detrimental effects on desired conservation objectives (Pendleton et al., 2018).

The most immediate and documented impact of MPAs is the shift in fleet dynamics, through the displacement of fishing, changes to gear types and effort; and this can have implications on social factors in the form of increased conflict, loss of or changes to expected income or increased competition for the same resource (Mangi et al., 2011; Rees et al., 2021). These changes in fleet dynamics may have consequences for fish stocks and benthic habitat health, with increased fishing effort of excluded gear types outside of the MPAs and a proliferation of permitted gear types within the MPA. Movement of permitted fishing methods into the MPAs may reduce the expected benefits of a MPA, such as increased abundance of exploitable species (Davies et al., 2021a), and therefore expected income. Increased potting effort inside a MPA may result in reduced catch of commercial crustaceans (Mangi et al., 2011; McClanahan & Mangi, 2000; Rees et al., 2021), masking potential fishery economic benefits to potting sectors from the exclusion of bottom-towed fishing. Quantifying such shifts in small scale fleets is problematic due to a lack of VMS data or accessible AIS (Automatic Identification System) data from small (<12 m) vessels, and therefore relatively little spatial information is available both pre and post MPA designation. It is therefore necessary to obtain this information through other means to improve the monitoring and management of MPAs.

It has been shown that stakeholder engagement and support can influence the ability of a MPA to meet its conservation and socio-economic objectives (Gall & Rodwell, 2016; Giakoumi et al., 2018; Rees et al., 2013a, 2013b), as effective stakeholder participation in the designation process can help manage expectations of MPAs (Pomeroy et al., 2005). In order to align the objectives of MPA and fisheries management, there is a need to track the performance of MPAs and associated socio-economic impacts that may underpin the ability of a MPA to provide both

conservation and fishery benefits. Understanding the attitudes of key stakeholders is an important consideration of ongoing monitoring and will aid in the determination of MPA success, and effective management and enforcement (Gall & Rodwell, 2016).

To understand how MPAs impact local fishing communities and identify where these impacts can be minimised, improved engagement and communication with the fishing community is essential. This study aims to set a baseline of the socioeconomic impacts (positive and negative) of a MPA designation to better inform fisheries and MPA co management approaches that will provide long term benefits for biodiversity and linked livelihoods. We undertake this study in a case study area of Jersey where two MPAs have recently been designated. Specifically, we aim to:

- 1) Characterise the primary sectors (mobile and static) of the fishing fleet and their economic value (pre and post MPA designation).
- Document the levels of support for and perceived compliance within the MPAs.
- Quantify the impact of the MPAs to local fishers in terms of fishing frequency and location.
- Assess the impact of the MPAs on fisher wellbeing across mobile and static sectors.
- 5) Investigate differences in turnover and profit between static and mobile fleets.
- Determine the sales strategies of each sector (pre and post MPA designation).

#### 3.2. Case study site

Jersey's marine environment supports mobile, static and Self-Contained Underwater Breathing Apparatus (SCUBA) diving fisheries, targeting a range of benthic, demersal and pelagic fish, but primarily benthic shellfish comprising lobster (*Homarus gammarus*), brown crab (*Cancer pagurus*), spider crab (*Maja squinado*), scallop (*Pecten maximus*), and whelk (*Buccinum undatum*). To protect areas of fragile habitat and high biodiversity associated with two offshore reefs in Jersey's territorial waters, two Marine Protected Areas (MPAs), which excluded mobile fishing gear, were designated in 2017 (Chambers et al. 2020) (Figure 3.1). Within the MPAs, static fishing (pots, nets, and lines), scallop diving, and low water fishing are permitted. Unlike many MPAs in the UK, Jersey's MPAs are not 'feature' based in that only specific features of conservation interest are managed within the MPA (Solandt et al., 2020). Instead the MPAs follow a whole-site approach whereby mobile fishing is not permitted anywhere within the MPA boundaries, representing an ecosystem based approach to fisheries management (Pikitch et al., 2004; Pitcher et al., 2009; Solandt et al., 2020).





The majority of the static fleet are made up of fishers using pots, targeting either crab and lobster, or whelk. The highest value fishery in Jersey is the static crab and lobster fishery, which comprises approximately 70% of all fisheries landings value (Marine Resources, 2019). Crab and lobster are caught using a mixture of pots (inkwells, creels, D-pots and parlour pots). The mobile fishing sector is the smallest in terms of number of vessels (5% of all vessels) and completely comprises vessels equipped for dredging for scallops. The MPAs are located in an area of shared

waters that is fished by both Jersey and French vessels but fall within Jersey's territorial waters and are therefore under Jersey's jurisdiction. Static gear fishing and scallop diving occurs both inside and outside the MPAs in Jersey. As a result of the MPAs being fished locally by small vessels without Vessel Monitoring Systems (VMS), and by French vessels from which landings data were not attributable to the MPAs, it was unknown how many fishers were operating in the MPAs prior to the closures.

## 3.3. Methodology

# 3.3.1. Data collection

The research used a combination of quantitative and qualitative approaches to capture information on the social and economic impacts experienced by stakeholders in Jersey following the designation of the two MPAs. The data only represents the Jersey perspective. French stakeholders were not approached due to ongoing political sensitivity over fishing rights in Jersey's territorial waters as a result of the UK's withdrawal from the European Union that resulted in previous fishing agreements between Jersey and France being legally void.

# 3.3.1.1. Catch composition of the Jersey fleet

Landings data and primary sale prices supplied by the Government of Jersey Marine Resources team was used to calculate overall landings weight and value for the whole Jersey fleet. It was not possible to attribute this information to the MPAs and is instead used to highlight the key commercial species targeted by the fishing community.

#### 3.3.1.2. Interviews

A guestionnaire was developed to collect data on fisher perceptions, change in fishing activity, well-being, and sales strategies resulting from the exclusion of mobile gear from the two MPAs (Appendix B). Questionnaires were used to guide face-toface interviews (n=21), but postal surveys were also used when fishers were not available to attend an interview, postal surveys were only included where answers were sufficiently detailed (n=2). Quantitative data were gathered by asking respondents to give scores on a Likert-type scale between 1 and 10. Qualitative data was generated from several open-ended questions relating to changes in their fishing strategies and views on current management. Information relating to income was obtained by asking respondents to state annual turnover and estimated percentages of associated annual costs from which profit could be estimated. Sales strategies were assessed by asking respondents, as a percentage, where each landed species was sold to, and whether this had changed between 2017 and 2018. Responses were recorded on paper during the interview and, where respondents permitted, interviews were recorded using a dictaphone to aid in transcribing responses post-interview. Ethical approval was granted from University of Plymouth prior to conducting work involving human participants.

### *3.3.1.3.* Fishing sectors

The respondents were split into mobile and static sectors depending on their primary fishing method. The static sector is a combination of potting, netting and diving fishers due to the tendency of netting and diving being used to supplement potting activities rather than being a sole fishing method. At the time of the survey, there were 149 vessels registered as commercial fishing vessels in Jersey. Only primary owners of vessels were surveyed. While only 21 fishers were surveyed, many vessels

registered as commercial are part time and do not rely on fishing as their primary source of income. It was not possible to ascertain which fishers predominantly operate in the vicinity of the MPAs from vessel logbooks as the areas they must report their catch from are large and that are not specific to the MPAs. Instead, fishers operating close to the MPA were identified though liaison with the Government of Jersey (Marine Resources pers. comm), and through communication with local fishers. To confirm how representative these fishers were in representing the broader fishing interests beyond the MPAs we calculated the contribution of these fishers to overall fishing effort in Jersey's waters through matching the vessel identifier (PLN) with the number of reported trips taken. We surveyed 57% of the mobile gear fleet (n=4), who undertake 67% of all Jersey registered mobile gear fishing in Jersey's waters (Table 3.1). We surveyed 12% of the static gear fleet (n=17), who undertake 19% of all Jersey registered static gear fishing in Jersey's waters (Table 3.1).

Table 3.1. The primary fishing method of fishers interviewed presented as a percentage of fishing effort for each sector averaged over 2017 and 2018. The percentage of fleet was calculated from number of vessels interviewed compared to all vessels. Many fishers work part time and so the percentage of fishing activity was derived from trips per year of interviewed vessels compared to total trips for all vessels from the Government of Jersey Marine Resources database\*.

Sector	Number of	Number	of	Percentage	of	Percentage	of
	registered	interviews fleet		fishing activity*			
	vessels						
Mobile	7	4		57		64	
Static	142	17		12		19	

### 3.3.2. Data analysis

## 3.3.2.1. Catch composition of the Jersey fleet

Catch composition was calculated from landings data of all Jersey vessels. Commercial fishers in Jersey are required to submit logbooks, which state their catch in kilograms for every fishing trip. Data for trips between 2015 to 2018 were obtained from the Government of Jersey Marine Resources team and used to calculate yearly landings for commercial species that contribute 99% of landings by weight (kg).

Landings value (first sale wet weight) of key target species in Jersey was calculated by multiplying the landings weight by the primary sale value to give an annual landings value (£) per target species for mobile, static and diving vessels. Data on primary fish sales in Jersey were based on quarterly buying prices (wet weight) from a local vendor (The Fresh Fish Co) between 2015 and 2018.

# 3.3.2.2. Interviews

The Likert scores from respondents were used to assess the distribution of responses in relation to the overall average, stated values are the mean, ± standard deviation. Income and associated costs were assessed by calculating averages of percentage costs separately for the mobile and static sectors. The percentage of profit was used to estimate the monetary value of profit from each respondent's annual turnover. Sales strategies were assessed by averaging responses for each sector. For all analyses, fishers were categorised into two groups: static (potting and diving) or mobile, representing the predominant gear type used.

## 3.4. Results

# 3.4.1. Catch composition of the fleet

The primary species targeted by the fleet are shellfish (Figure 3.2), consisting of whelk (*Buccinum undatum*), brown crab (*Cancer pagurus*), scallop (*Pecten maximus*), lobster (*Homarus gammarus*), and spider crab (*Maja squinado*). Whelk landings have been increasing overall over the four years, with the largest increase observed in 2018 (838,942 kg), compared to 2017 (345,999 kg). Lobster landing were in consistent decline over 2015 to 2018, with the largest decrease observed between 2017 and 2018 where landings dropped from 237,837 kg to 196,019 kg (Figure 3.2). Spider crab landings increased each year, with 207,449 kg landed in 2017 and 293,569 kg in 2018. Brown crab landings peaked slightly in 2016 (348,500 kg) but declined in subsequent years, with the lowest landed weight observed in 2018 (292,987 kg). Overall scallop landings were relatively uniform across the years (Figure 3.2).



Figure 3.2. Commercial landings (kg) between 2015 and 2018 showing the top 15 species that contribute to 99% of landings by weight. The remaining 1% of commercial landings consists of 36 species.

These five species constitute 94% of all fisheries landings and are worth approximately £7.5 million annually over the period considered (2015-2018). Of these, lobster is notably the most economically valuable species with an average of £4.1 million per year in primary sales (Figure 3.3). The primary sale price of fish to vendors has been relatively stable between 2015 and 2018 (Figure 3.3), with changes in total value a reflection of the change in landed weight. Over the time period surveyed in the interviews (2017 and 2018), there was an increase in the landings, and therefore value, of spider crab (£444,748 in 2017 and £657,700 in

2018), whereas there was a decline of almost £1 million in lobster landings value from £4,622,932 in 2017 to £3,731,496 in 2018 (Figure 3.3). Scallops caught by the mobile sector showed a small increase in landings value from £673,084 in 2017 to £747,697 in 2018, as did dived scallops from £164,734 in 2017 to £252,571 in 2018, the highest value of dived scallops in the four years presented (Figure 3.3). Brown crab landings value increased from £731,046 in 2017 to £1,005,712 in 2018 but this was due to a dip in landings in 2017, with landings value in 2018 in line with those from 2015 and 2016 (Figure 3.3).



Figure 3.3. Landings value (£) per year (2015-2018) for the top five contributing species to Jersey's fisheries value, landed by Jersey vessels from the three primary sectors (mobile, static and diving).

#### 3.4.2. Interviews

Static fishing vessels primarily pot for crab and lobster but some fishers also use nets, lines, and diving to supplement their catch. Only two respondents solely targeted scallop (*Pecten maximus*) through diving and so these were grouped with the static sector. The mobile sector was represented by four respondents who primarily employed dredges to target scallops. Of all the respondents, 17 were found to actively fish in the vicinity of at least one of the MPAs.

## 3.4.2.1. Support for the MPAs

Amongst the respondents to the survey (n=21), support for the MPA was almost identical for both MPAs (Ecrehous and Minquiers) except for one respondent that answered 10 (strongly support) for the Minquiers and 8 for the Ecrehous, with the reason provided relating to the small size of the Ecrehous MPA. The results presented are overall support for both MPAs together, with the support averaged for the one respondent which gave differing results (Figure 3.4a).

The majority of static fishers supported the MPAs, with a mean of 8.9 ( $\pm$  2.3), whereas mobile gear fishers responded with varied, but typically low, levels of support for the MPAs with a mean of 4.8 ( $\pm$  3.9) (Figure 3.4a). While support in the static sector was high, three fishers raised concerns that the MPAs were too small and another that the Minquiers MPA was not well placed as the majority of seabed there had not previously been targeted by mobile gear fishers and therefore was considered largely a MPA which would secure the long-term protection of the habitat but offers no overall biodiversity gain.

#### 3.4.2.2. Perceived compliance

The level of compliance with local fishing regulations was generally perceived by fishers to be good (1 = complete compliance and 10 = complete non-compliance) by both static (mean  $2.8 \pm 2.2$ ) and mobile (mean  $2 \pm 1.2$ ) sectors. The mobile sector responses showed a general consensus that compliance levels were high (Figure 3.4b), with just one respondent stating that one or two were not following the rules, and another observing that the compliance of French vessels is high due to most having Vessel Monitoring Systems (VMS) which enables remote observation of the location of vessels at sea. Where there were perceptions of non-compliance with the MPA byelaws the responses were varied. These included a perception that part-time and recreational fishers were being less compliant with the rules than full-time fishers, that compliance is declining over time, that undersized catch was being landed, and that mobile gear vessels were 'pushing the limits' of the MPA boundaries. Two static fishers commented that they perceived compliance with the MPA byelaws was lower amongst French vessels, which is counter to the view of the mobile fisher. This may reflect the higher numbers of smaller French vessels using static gear which do not require VMS, compared to the majority of mobile vessels which do. Many of the comments from static fishers related to noncompliance of broader regulations, such as the size of catch, rather than the specific MPA regulations that exclude the use of mobile gear within the MPA boundaries.

# *3.4.2.3. Fishing location and frequency*

Reported changes in fishing location were used as a proxy for displacement. Four static fishers reported a change to their fishing location following the designation of the MPAs, but where this occurred, the change was reported to be minimal (mean =  $1.6 \pm 1.2$ ) (Figure 3.4c). Three of the four static fishers reported this change was

related to moving their pots into the MPAs or spending more time scallop diving inside the MPAs. Only one static fisher reported a change in fishing location due to displacement of mobile fishing into potting areas outside of the MPAs. One mobile fisher reported a minimal change in fishing location and another commented that their fishing location had not changed but stated they would have started dredging for scallops at the Ecrehous had the MPA not been in place. Fishing frequency was unchanged for all but one static fisher who reported diving more frequently in the Ecrehous MPA since it had been designated, reporting that the scallops were of better quality (Figure 3.4d).



Figure 3.4. Likert scale responses for each stakeholder group. Responses to the questions: a) to what extent do you support the MPAs (1 = not at all; 10 = completely), b) what do you perceive the levels of non-compliance are within the MPAs (1 = complete compliance; 10 = complete non-compliance), c) to what extent have the MPAs changed where you fish (1 = not at all; 10 = completely) and d) to what extent have the MPAs changed changed how often you fish (1 = not at all; 10 = completely). Filled in circles show outliers.

## 3.4.2.4. Job and income satisfaction

There was no discernible change in reported job and income satisfaction between 2017 and 2018 for either sector (Figure 3.5a and 3.5b) and satisfaction for mobile respondents were low (job satisfaction mean per year = 3.8 ± 2.8; income satisfaction mean per year =  $3.8 \pm 2.1$ ) compared to static respondents (job satisfaction mean per year =  $7.1 \pm 2$ ; income satisfaction mean per year =  $6.8 \pm 1.8$ ). Reasons for low levels of job satisfaction from mobile fishers related to low profits, too many restrictions and displacement from fishing grounds due to French netting vessels. Static respondents generally reported moderate to high levels of job and income satisfaction in both 2017 and 2018. Reasons provided included enjoying the lifestyle and being happy with their levels of catch. There was a strong overlap in comments relating to job satisfaction and income satisfaction, with the primary reasons of lowered job satisfaction relating to changes in income over this time period. There was no general consensus in the reasons given by mobile fishers for low levels income satisfaction, and these included: perceived high overheads, less profits than expected and restrictions in exporting scallops to France during the French closed season. Key concerns communicated by several static respondents that had reported reductions in their job and income satisfaction were: competition created by high levels of potting from other vessels, an increase in spider crab combined with a decrease in brown crab and lobster resulting in less profitable catch, restrictions on bass fishing, an inability to diversify their catch, and boat maintenance and loan repayments reducing their profits.

### *3.4.2.5.* Conflict and stress

In response to the question: what level of conflict did you experience (1 = no conflict; 10 = high levels of conflict), reported levels of conflict were greater on average in the mobile respondents (mean per year =  $5.9 \pm 2.7$ ), than the static respondents (mean per year =  $3.4 \pm 2.7$ ) (Figure 3.5c). All four mobile fishers reported this was related to conflict with other gear, primarily nets but also pots, from French vessels only. There was no conflict perceived within the local mobile gear sector nor with other local vessels from other sectors. Where conflict was perceived by static fishers, five stated this was due to conflict within the static sector from both Jersey and French vessels targeting the same ground with pots. Three reported conflict with netting vessels from France and another three reported conflict with mobile sectors, from both Jersey and French mobile vessels, which have resulted in static fishers having to relocate their pots.

Reported stress levels (1 = no stress; 10 = high levels of stress) were high for mobile fishers (mean per year =  $8.3 \pm 1.3$ ) (Figure 3.5d) for various reasons, including too many restrictions, concerns over large French vessels reducing stocks and issues relating to boat maintenance. Static responses were variable but similar overall in both years and lower than mobile responses (mean per year =  $4.4 \pm 2.3$ ). Four static fishers reported high stress levels as a result of conflict with other fishers, three as a result of too many restrictions and another three due to concerns over declines in catch, particularly declines in brown crab relating to a recent increase in the regulation for Minimum Landing Size. Other factors affecting stress levels were licence repayments and costs associated with boat maintenance. Neither the mobile nor static respondents reported the designation of the MPAs as contributing to their stress levels.



Figure 3.5. Likert scale responses for each stakeholder group split for 2017 and 2018 for questions relating to well-being. Responses to the questions: a) how satisfied are you with your fishing income (1 = completely dissatisfied; 10 = completely satisfied), b) how satisfied are you with your fishing (job satisfaction) (1 = completely dissatisfied; 10 = completely satisfied), c) what level of conflict did you experience (1 = no conflict; 10 = high levels of conflict), d) what level of stress did you experience (1 = no stress; 10 = high levels of stress).

# 3.4.2.6. Fleet income breakdown

The average self-reported turnover of the mobile respondents was almost double than that of the average self-reported turnover of the static respondents. The mobile respondents also reported 5% higher associated costs on average (Table 3.2). The average annual profit of the mobile sector is approximately £50,000, which is much greater than the static sector which had an average annual profit of £30,000 (Figure 3.6). There was one exception from the static sector which had an annual turnover

and profit that were double that of the mean for the mobile sector and more than three times the average of the static sector.

Table 3.2. Average annual turnover for each sector and the average percentage of turnover given to the various associated costs.

Sector	Average	Avg.	Avg.	Avg.	Avg.	Avg.	Other	Avg.
	turnover	Fuel	Harbour	Mainten-	crew	Capital	(insurance	profit
	(£)	costs	dues (%)	ance	costs	payments	/bait/parki	(%)
		(%)		costs (%)	(%)	(%)	ng etc.)	
Mobile	157,500	12	5	28	7	5	11	32
Static	80,000	12	3	20	5	8	15	37





# 3.4.2.7. Sales strategies

The sales strategies for lobster, brown crab and spider crab were almost identical. The majority of crustacean catch from both mobile and static sectors is sold to local vendors, with the majority of the rest being exported to France and a small portion through private sales (Figure 3.7). Private sales refer to catch that is sold directly to restaurants or the public, typically for a greater value than can be achieved by selling catch to vendors. Sales strategies for scallop in the mobile sector were more varied, with around a third each sold to local vendors, exported to France or sold through private sales. Static catch of scallop (dived scallop) was primarily sold privately, and the majority of the rest was sold to local vendors. Catch can be exported to France through local vendors or directly landed in France by fishers but this was not discernible from the interview responses. Finfish from the static sector were almost all sold though vendors, with a small portion being sold through private sales. Sales strategies remained relatively unchanged for static respondents between 2017 and 2018, with one respondent reporting a reduction in direct landings to France, and another reporting an increase in direct landings to France. Sales strategies remained unchanged for mobile respondents.



Figure 3.7. Mean percentages of catch sold through various routes as reported by mobile and static respondents.

#### 3.5. Discussion

This study has identified a range of perceptions towards the newly established MPAs within a fishing community, and the social impacts experienced by representatives of different sectors. Jersey fisheries are dominated by static fishers, particularly those using pots to target crab and lobster. Through assessment of annual landings weight and value, lobster was identified to hold the greatest value to fisheries. The levels of support for the MPA varied between static and mobile fleets, while compliance was generally reported to be high across the sectors. In the first year since designation there have been no reported changes to catch (kg, £), and only one mobile gear vessel respondent reported being displaced from traditional fishing grounds as a result of the MPA designation. Fishers also reported an increase in static fishing inside the MPAs, post designation. Mobile gear fishers reported a lower subjective wellbeing and satisfaction with their job and income than static gear fishers, despite greater profits. The major concern reported by the mobile fleet that contributed to their stress levels, was not attributed to the MPA designations but related to conflict with French static net fisheries operating outside of the MPAs. There was no indication that the MPAs had affected fisher income, which is likely related to few fishers having to change their fishing location or fishing frequency following the designation of the MPAs. Sales strategies primarily involved selling catch to local vendors for both fleets and was unchanged following the designation of the MPAs.

The MPAs were primarily designated to protect habitats of ecological importance, not as fisheries management tools, though the objectives are not dissimilar. Due to the exclusion of mobile gear from the MPAs and the continuation of static fisheries, the MPAs were expected to be better received by static fishers. As anticipated, the

fishers' perceptions of the MPAs differed between mobile and static fleets, with greater support for the reported from static respondents. This was also shown by Rees et al (2021; 2013) where static respondents typically scored more positively than mobile respondents to the exclusion of mobile gear from the MPA in Lyme Bay. However, not all mobile gear respondents had completely negative perceptions of the MPA in Jersey, with one respondent in full support of the MPAs, highlighting that perceptions are not only different between sectors but between individuals within sectors also.

It is important to understand landings across long timescales and the economics of the fleet as a baseline for future comparison following the designation of MPAs. This may also provide an insight into which sectors may be economically affected by changes in spatial management, which may allude to perceptions and support for the MPA from different fishing sectors. In time, a MPA can be expected to improve landings as the protection of essential fish habitat can support a greater abundance of commercial species (Davies et al., 2021a). Jersey fisheries are valuable (~£7.5 million annually) but as there is no accurate spatial information for most vessels it is impossible to evaluate the impact of the MPAs on fishing activity or the subsequent economic value from landings. In previous studies where this information is available, there have been economic benefits to static fisheries due to increases in catch of high value species within the MPA following the exclusion of mobile gear (Rees et al., 2021). There was a strong reported dependence on shellfish for both sectors but static fishers who took part in this research have indicated that there has been a decline in brown crab and lobster catches that is affecting their livelihoods. They indicate that the long-term sustainability of their fishing activity should be a primary concern for local fisheries managers. The intensification of some fishing activity 103

within MPAs is not new as the spatial management measures for MPAs often partially benefit some sectors of the fishery over others (Rees et al., 2021b). Without pre-emptive fisheries management, it can be seen that potting activity can increase within MPAs which have excluded mobile gear (Mangi et al., 2011; Rees et al., 2021a; Rees et al., 2021b). While only three static respondents reported an increase in their fishing effort inside the MPAs one year after their designation, there may be more instances of displacement into the MPAs that is unaccounted for in the small sample size of this study. Improved monitoring of spatial fishing effort is needed to allow for accurate assessments of fishing displacement.

The decrease in lobster landings across Jersey's waters, not just from those fishers reporting to fish within the MPAs, plus an increase in fishing effort (Marine Resources, 2019), is a concern for the fleet as the majority of vessels rely on lobster as their main source of income. The reports of declines in lobster in Jersey does not reflect stock assessments of lobster on the southwest coast of the UK that indicate lobster landings are relatively stable in the southwest English Channel (Cefas, 2020b; Cornwall IFCA, 2018b; Southern IFCA, 2020), suggesting that the decline in lobster is localised to Jersey, and potentially a reflection of the high potting levels reported by respondents. Further reducing the profitability of catch is the reported increase in low value spider crab landings in addition to a decline in brown crab. The decrease in brown crab landings is also being experienced by Jersey's fishers operating inside and outside the MPAs. More recent reports from Jersey (Marine Resources, 2019) and the UK (Cefas, 2020a) suggest that brown crab catches are also declining in the English Channel and may not be solely related to exploitation levels. At present, local fishers do not attribute any changes in landings to the MPAs. More accurate spatial fishing information is needed to monitor levels of potting and

determine a threshold limit that will reduce overexploitation of crab and lobster and also reduce pressure on benthic habitats for which the MPA is designated to protect.

The MPAs are offshore and primarily fished by small vessels that do not require VMS (real time monitoring of the location of fishing effort) making the enforcement of regulations is problematic. Other studies have shown that compliance with MPA regulations is improved when there is support from local stakeholders (Gall & Rodwell, 2016; Giakoumi et al., 2018; Read et al., 2011; Rees et al., 2013a) and this is particularly important in Jersey where there is limited ability to detect illegal fishing activity within the MPAs. Compliance levels were generally reported by respondents to be high, particularly for larger mobile gear vessels that are required to have VMS. However, it was also reported that mobile gear vessels were 'pushing the limits' of the MPA boundaries, which may have consequences for expected benefits, such as overspill of fishery species into surrounding fishing areas. MPAs that are not perceived to be meeting prior expectations may lose support of stakeholders (Gall & Rodwell, 2016; Giakoumi et al., 2018; Rees et al., 2013a), so it is important to communicate the expected benefits and costs and to consider compliance when assessing the progress of a MPA.

Overall, the stress levels reported by mobile fishers was higher on average, and job satisfaction lower than average, compared to their static counterparts, but this was not reported to be attributed to the MPAs. Similar to results from Lyme Bay (Rees et al., 2021b), the mobile fishers reported lower levels of income satisfaction on average than static fishers, despite reporting overall higher profits on average than static fishers. In previous studies, the implementation of coastal MPAs in which mobile gear is restricted has resulted in strong social impacts to fishers operating

mobile gear vessels, including the need to travel further to fishing grounds outside of the MPAs (Hattam et al., 2014; Rees et al., 2013b), increasing travel expenses and raising safety concerns in the event of bad weather while far from a home port. However, the two MPAs in Jersey have not had this effect due to the fact they are located away from the mainland coastline (7-20 km), with fishing grounds open to mobile gear both closer to Jersey and beyond the MPAs which means increased travel costs are unlikely to be contributing to stress levels or income satisfaction. Impacts on subjective well-being (stress and job satisfaction) is largely reported, in this case study area, to be linked to conflict to other fishers, particularly French static net fisheries outside of the MPAs, and financial concerns (licence and boat repayments). The reported fisher conflict outside of the MPAs suggests that additional spatial management could improve the well-being of the mobile fleet by reducing overlap with other sectors and securing 'patches' of desirable ground to maintain their income. However, as mobile fishing degrades ecosystems overtime (Stewart & Howarth, 2016), this may result in areas of unviable fishing grounds if fishing levels are not managed sustainably, further reducing the well-being outcomes for mobile fishers.

The exclusion of mobile gear, in particular dredging practices, can improve habitat integrity (Sheehan et al., 2013a) which can lead to an improved supply of ecosystem services (Rees et al., 2020). Improved ecosystem function and flow of ecosystem services are most likely to benefit the static gear sector as the MPAs protect the reef habitats on which their target species depend. Additionally, the MPAs provide a space where they can set their gear that is not in conflict with mobile gear. Benthic shellfish, including those of commercial importance, can be physically damaged by mobile fishing gears (Beukers-Stewart & Beukers-Stewart, 2009a; Kaiser &

Spencer, 1995), decreasing their densities (Veale et al., 2000). It is expected that benthic shellfish would benefit from the exclusion of mobile fishing but the time frame within which change can be expected must also be considered. At the time of interviews, the MPAs had been established for one year, and so changes in species abundances in response to the protection measure was unlikely to have occurred, and therefore social benefits relating to changes or increases in catch will yet to have been realised. One static fisher reported an improvement in the quality of scallop but not in the abundance of scallop, which is expected as they take approximately four years to reach Minimum Landing Size (MLS) (Marshall & Wilson, 2008). The same can be expected of crab and lobster, with lobster in particular taking up to seven years to reach MLS (Schmalenbach et al., 2011). Older MPAs have greater human well-being benefits associated with them (Ban et al., 2019), and with time the MPAs in Jersey may gain support from stakeholders if they are deemed to be improving fisheries sustainability or at the very least not negatively impacting on fisher wellbeing. However, these benefits may not be realised if levels of permitted fishing within the MPAs are not managed sustainably, and the concerns of mobile gear fishers linked to resource rights and space outside the MPA are ignored.

Static gears are not without their impacts on benthic communities. High levels of potting can have negative impacts on both fisheries, in terms of reduced catch, and benthic species in terms of abrasion to the seabed causing declines in long-lived sessile species, that fundamentally underpin the food chain (Rees et al., 2021a). Highly selective scallop diving activities have the potential to significantly reduce stocks (Tracey & Lyle, 2011), further exacerbated by the targeted selection of large, highly fecund individuals (Beukers-Stewart & Beukers-Stewart, 2009a), reducing recruitment both to MPAs and open fishing grounds. Fishers may have expectations
of 'spillover' from MPAs due to increased densities of target species inside (Leleu et al., 2012; Russ et al., 2004) but in Jersey scallop dive fisheries potentially pose a threat to the benefit of MPAs to scallop populations, and therefore to scallop dredge fisheries. While dive pressure in the UK is not thought to be high enough to significantly impact scallop stocks (Beukers-Stewart & Beukers-Stewart, 2009a), it is an important fishery to monitor as divers select for larger scallops, which are typically those that have the greatest contribution to spawning success (Beukers-Stewart & Beukers-Stewart, 2009a). Both potting and diving need to be managed effectively to avoid the over exploitation of their respective target species.

To avoid conflict with stakeholders many MPAs are placed in areas to provide protection to specific features (habitats and species) of conservation importance but avoiding areas of commercial economic value, providing few opportunities for the restoration and recovery of ecosystems. These MPAs have been described as residual, conferring no new or additional benefits to biodiversity (Devillers et al., 2015, 2020). Only one mobile fisher reported that the MPAs had displaced them from previous fishing grounds. This raises the question of whether the placement of the MPAs in Jersey has added additional protection to the seabed, much of which are composed of bedrock and boulders that would have previously deterred some mobile fishing efforts. However, mobile vessels deploying dredges will target scallops in-between reef features (Pikesley et al., 2021) which can impact both on the condition of the intact reef and also prevent opportunities for reef associated species to recolonise areas where the sediment is a veneer covering a harder substrate below (Sheehan et al., 2013b).

Uncaptured in this study is the displacement caused to French mobile gear vessels from the MPAs, which is likely to have been considerable for some vessels (personal comm. Government of Jersey). Additionally, with improving technology on fishing vessels, it is not to say that fishing effort and the pressure from mobile gear would remain the same, so the MPAs, and the whole-site management measures may be a future proofing mechanism from further incursions into the reefs. Given the uncertainty in the current use of the reefs by French vessels and potential future demand and access for the shellfish within the reef systems protection of these critical ecosystems within the MPAs has immediate benefits for long term sustainability as long as there is continued compliance. One of the points raised by several static fishers was that the MPAs are currently too small. In this case study area, original proposals for larger MPAs were rejected by several Jersey and French stakeholders due to their overlap with commercially important fishing grounds (Dr Paul Chambers pers. comm. Marine Resources). It is here that the disconnect between sustainable fisheries and areas protected for biodiversity (hosting areas important for the life history of commercial fish species) is most apparent. Whilst the ecological effectiveness (relating to biodiversity metrics) of small MPAs is debated in the literature (Edgar et al., 2014; Turnbull et al., 2018), the well-being outcomes (relating to social benefits) of small MPAs can be more positive than those of large MPAs (Ban et al., 2019).

According to fishers interviewed for this research, the primary route to market for crab and lobster catch was to local vendors as this is considered the most reliable and convenient route for many fishers. By selling to local vendors (wholesale) the fisher receives the first sale market price for the goods and then the vendors finds secondary markets to trade goods for profit. Jersey fishers also report that this sales

strategy may achieve less per kg than if they were to sell to a private buyer such as a restaurant. However, this restaurant market is limited. Mobile gear caught scallops were sold to a combination of local vendors, exported to France and sold privately. Dive caught scallops was primarily sold through private sales. Elsewhere, creating a premium for traceable and sustainable seafood has benefitted small scale fishers operating sustainably within MPAs (Blue Marine Foundation, 2016; Rees et al., 2021b). In the current economic climate following the UK's exit from the EU, export of seafood to France is becoming increasingly difficult due to the customs requirements of the 'third country' status now imposed on Jersey that has restricted markets. Alternative sales strategies with a local sustainability brand may provide a mechanism to support Jersey's small-scale fishers and should be explored further if traceability of catch from the MPAs can be improved.

### 3.6. Conclusion

Effectively managed MPAs are needed to help mitigate human impacts on the marine environment and support sustainable fisheries. In doing so, pressure will be placed on fishing communities that may be displaced from traditional fishing grounds or be subject to increased conflict with other fishing gears due to changes in fishing activity that result from change in management. To understand where and when negative impacts can be reduced, and positive impacts can be highlighted, engagement and communication with the fishing community is needed. This research has provided a snapshot of fisher perceptions from a subset of the fishing community in Jersey. It was found that both mobile and static sectors depend on shellfish species but that the high value species that static fisheries depend on (lobster and brown crab) are in decline. There were no reports of increased catch relating to the MPAs, which was expected due to the young age of the MPAs. The

greater level of support for the MPA reported by static fishers may pertain to reduced conflict with mobile fishers, but the reports of increased static fishing in the MPAs may prevent the long-term sustainability of static fishery target species, ultimately negatively impacting the static fishery. The MPAs were identified through interviews as having had little effect on fisher wellbeing, but the information gathered during the interviews has highlighted key issues being experienced by the fleet beyond the MPAs, regarding declines in target species and conflict with fishing gear from French vessels. Future work involving interviews with the French fleet is needed to broaden the scope of this research. While the level of protection afforded the MPAs in Jersey may be high compared with many other MPAs in Europe, the observed decline in fisheries landings and value over time suggests that habitat conservation and fisheries management are not sufficiently aligned, even though the goals are common. The MPAs alone are not sufficient to create a sustainable fishery and these results indicate that additional management, focussed on recovering target species, is needed to compliment the MPAs.

 Chapter 4 – Removal of bottom-towed fishing from whole-site Marine Protected Areas (MPAs) in Jersey's territorial waters (Channel Islands, UK) promotes mobile species biodiversity

# Author Contributions

Samantha Blampied was responsible for the conception and design of this research. She organised the fieldwork, undertook the analysis, interpretation of data and writing of the chapter. Francis Binney assisted in fieldwork logistics and all supervisors contributed to the critical revision of manuscripts and have given their approval for inclusion in this thesis. This manuscript is in review with the *Journal of Estuarine, Coastal and Shelf Science*.

Author	Institution	Contribution	
Samantha Blampied	University of Plymouth	70%	
Sian Rees	University of Plymouth	5%	
Martin Attrill	University of Plymouth	10%	
Francis Binney	Government of Jersey	5%	
Emma Sheehan	University of Plymouth	10%	

#### Abstract

Marine habitats are being altered by anthropogenic pressures, influencing the diversity and distribution of species. Marine Protected Areas (MPAs) are increasingly used as spatial management tools to mitigate these human impacts on marine systems, but levels of protection vary. MPAs that exclude bottom-towed fishing activities from all habitats within their boundaries, a form of 'whole-site' MPA, have shown benefits in terms of increasing biodiversity and biomass. In this study we use baited videos to quantify differences in species assemblage, diversity and abundance inside and outside multiple 'whole-site' MPAs of differing ages (3 and 6 years) within Jersey's territorial waters. Mixed sediment habitats, such as those targeted by bottom-towed fishing practices, were selected for study inside the MPAs and in nearby fished control sites to assess the potential of these interstitial habitats to contribute to the recovery of biodiversity in MPAs. Greater numbers of taxa were recorded within all three surveyed MPAs compared to nearby unprotected areas. No difference was observed in the number of total individuals, but several indicator species (Labridae Spp., adult and juvenile Spondyliosoma cantharus, and IUCN threatened shark) were detected in higher numbers within the MPAs than the nearby unprotected areas. Not all species responded positively to the MPAs, with scavenging species such as Maja brachydactyla and ray species detected in higher numbers in nearby fished areas, highlighting the need to consider coverage of key habitats used by species of concern or commercial interest within the MPAs and regulation of other fishing metiers within the MPAs. BRUV surveys that have been used to survey MPAs in Europe are sparse and have primarily focussed on reef habitat. This study provides the first insight into species assemblage, diversity and

abundance in response to MPA designation across a variety of habitats and further evidences the merit of a whole-site based approach to MPA management.

#### 4.1. Introduction

The oceans face a number threats in an increasingly populated world, with greater anthropogenic pressure being put on ecosystems through fishing, climate change, pollution and habitat loss (Halpern et al., 2008, 2020; Hughes et al., 2003; D Pauly et al., 1998, 2005; Sumaila et al., 2016; Worm et al., 2006). Fisheries are an economically valuable resource as well as an important food source across the globe (FAO, 2020). Underpinning fisheries, are multiple and interlinked food webs, cross cutting with the processes and functions of the ecosystem. (Dobson et al. 2006; Staples and Funge-Smith 2009). Despite this dependency, the fishing activity itself poses one of the largest threats to the sustainability of this food resource (Pauly, Watson, and Alder 2005), though causing direct physical impacts to marine habitats (Hall-Spencer & Moore, 2000; Thrush & Dayton, 2002). Despite both national and international management measures in place to protect this interdependency between marine habitats and fisheries, there are still global declines in habitat condition (Dobson et al., 2006; Klein et al., 2013), biodiversity (Sala et al., 2021; Worm et al., 2006) and fish stocks (Claudet et al. 2020; Halpern et al. 2008). Ecosystem degradation and the resultant biodiversity loss may negatively impact human well-being through lowered ecosystem services (Chapin et al. 2000; Hooper et al. 2005; UNEP 2006; Worm et al. 2006). This has resulted in new approaches to management that promote the recovery of marine ecosystems (Pikitch et al., 2004). Marine Protected Areas (MPAs) are a common management tool employed to conserve habitats and species of conservation importance, but the level of

protection afforded to a MPA varies greatly (Day et al. 2012). Approximately 7.7% of the world's oceans are currently within MPAs, but only 2.4% are highly protected (Marine Conservation Institute, 2022). Research suggests that highly protected MPAs that exclude the most destructive human activities are the most beneficial for biodiversity (Edgar et al., 2014; Lester & Halpern, 2008; Sala & Giakoumi, 2018), but it may not always be possible to achieve this level of protection due to societal constraints (Halpern et al., 2008). It has been argued that increasing the size and number of MPAs needs to align with an agenda for sustainable lives and livelihoods (Rees 2020). In this framing only fishing methods deemed to negatively impact the conservation goals of the MPA are prohibited. In the UK, many MPAs have been designated with the intention to conserve a single species or marine feature, but this management approach has been shown to be less effective in the recovery of marine biodiversity than an ecosystem-based approach (Solandt et al., 2014). This is particularly ineffective where bottom-towed fishing is permitted in the areas between the protected features, which is typically the case as interstitial habitats between reef habitat are generally considered of low conservation importance and are rarely protected (Solandt et al., 2020). Currently in English waters, MPAs cover 64,980 km<sup>2</sup> and within this there is an area of 45,763 km<sup>2</sup> of sedimentary habitats (mixed sediments (6,049 km<sup>2</sup>), mud (6,537 km<sup>2</sup>), sand (33,174 km<sup>2</sup>) and seagrass (3 km<sup>2</sup>) that are not protected from bottom-towed fishing (MPA Reality Check, 2019), but these habitats have the potential to contribute to the recovery of MPA biodiversity for species of both conservation and commercial interest. The fragmentation and loss of habitats is contributing to biodiversity loss in the oceans (Dobson et al., 2006; Haddad et al., 2015), and bottom-towed fishing gears reduce habitat complexity through habitat fragmentation and homogenisation (Thrush &

Dayton, 2002; Veale et al., 2000). Bottom-towed fishing gears have the potential to negatively impact marine species directly, through damage to species (Beukers-Stewart & Beukers-Stewart, 2009b; Kaiser & Spencer, 1995), or indirectly through habitat degradation and disruption of trophic dynamics (De Grave & Whitaker, 1999; Stewart & Howarth, 2016; Thrush & Dayton, 2002). For example, many crustaceans are physically damaged by bottom-towed fishing gears (Kaiser & Spencer, 1995), leading to decreases in densities (Veale et al., 2000).

The whole ecosystem that the fishery exists within should be considered for management (Long et al., 2015; Pikitch et al., 2004), with an aim of improving ecological sustainability to support the fishery and broader societal values (e.g. recreation interest) (Thrush et al., 2016). This can be achieved by protecting the full suite of habitats and species across a defined area of seabed from fishing activities which threaten seabed integrity, creating a "whole-site" MPA that will support wider ecosystem recovery (Rees et al., 2013c; Sheehan et al., 2013a; Solandt et al., 2020). The exclusion of bottom-towed fishing gears from all features within a MPA, particularly those towed along the sea bed ("bottom-towed"), protects seabed integrity and ecosystem function, which will support greater fish biomass including commercial species (Beukers-Stewart et al., 2005; Brown et al., 2010; Willis et al., 2003). This will benefit some species more than others depending on their life histories (Kaplan et al., 2009; Tillin et al., 2006), and will improve the connectivity of life stages for species that use multiple habitats in their life cycle.

Juveniles, including those of commercial species, use several habitats but tend to show selectivity (Elliott et al., 2017b), and loss of particular habitats through bottom-towed fishing will reduce the survivability of such habitat specialists (Berger, 2021;

Clavel et al., 2011). For example, seagrass has greater abundances of juvenile fish than surrounding sand and mud habitats (Jackson et al., 2002; Lilley & Unsworth, 2014), and certain species of damsel fish are strongly associated with specific coral species (Bonin et al., 2015). Other complex habitats with relatively high epifaunal diversity or structural complexity, such as maerl, have been shown to support higher abundances of commercially targeted organisms, such as commercially exploited scallop species (Howarth et al., 2011; Kamenos et al., 2004b). Further, postsettlement survival of some benthic species, such as crustaceans and bivalves, are thought to be higher in more complex habitats (Howarth et al., 2011; Palma et al., 1998; Robinson & Tully, 2000; Stevens & Kittaka, 1998; Tonk & Rozemeijer, 2019). Therefore, the exclusion of bottom-towed fishing may improve larval recruitment of certain species as benthic habitats recover, allowing the growth of more complex habitat structures (Howarth et al., 2011; Wahle & Steneck, 1991).

There are several factors that underpin a MPA's ecological success and these typically relate to level of protection, enforcement, size, age and isolation (Edgar et al., 2014). Highly protected marine areas, such as those which protect all habitats from destructive fishing methods and No Take Zones where all extractive activities are excluded, have the potential to contribute to the recovery of previously exploited species (Davies et al., 2021a; Edgar et al., 2014; Lester et al., 2009). Increases in top predators may also be observed and facilitate an enhancement of ecological complexity that underpins healthy ecosystems (Shears & Babcock 2002; Byrnes et al. 2006). Monitoring changes in biological communities following the designation of protection measures can provide important information on the effectiveness of MPAs (Pomeroy et al., 2005) and their ability to protect certain species (White et al., 2013). Change is likely to be most observable in benthic and demersal species as

they rely on benthic habitats for a combination of life stages (Kritzer et al., 2016). Depending on a species life cycle, and its reliance of particular habitats or habitat complexity, the response, and time taken to respond, will differ (Kaplan, 2009; Lotze et al., 2011). Long-lived species tend to sexually mature later and so recovery of these species is expected to take longer compared to short-lived species (Hiddink et al., 2019). Reproductive strategies will also factor into a species ability to recover following the implementation of a MPA (Coleman et al., 2000). Species that have large ranges, seasonal migrations or spawn by larval dispersal will not benefit from MPAs as much as sedentary and sessile species, which may live out their whole life within the bounds of a MPA (Maxwell et al., 2015). There are also very few species that will use just one habitat in their life cycle, so habitat connectivity within a MPA can be expected to improve the survivability of many marine species (Abecasis et al., 2009; Krueck et al., 2017; Lipcius et al., 2008).

Areas with reduced fishing pressure may allow better resilience of habitats and species to environmental change (Hiddink et al. 2017; Hughes et al. 2003; McLeod et al. 2009; Micheli et al. 2012) and protecting multiple habitats within a MPA may allow a more natural state to develop (Sheehan et al., 2013b), helping to understand shifting baselines in both biodiversity (Little et al., 2017; Villnas & Norkko, 2011) and fishery stocks (Bunce et al., 2008; Ulman & Pauly, 2016) in comparison to areas that continue to be fished. An example of this observed by Sheehan et al. (2013a) following the exclusion of bottom-towed fishing in Lyme Bay, UK, sessile reef species colonised the sedimentary habitats within the reserve, showing that the baseline for some marine habitats had shifted as a consequence of bottom-towed fishing (Braeckman et al., 2014; Collie et al., 2000; Jackson et al., 2011). Also observed in Lyme Bay were increases to sessile, sedentary and mobile taxa (Davies et al., 2020;

Sheehan et al., 2013a). By protecting whole areas of seafloor habitats from bottomtowed fishing, structural integrity and functioning of the ecosystem can be maintained or enhanced. Habitat complexity promotes primary productivity and biodiversity (Worm & Duffy, 2003), and the high seafloor productivity supports higher trophic level species (Hunt & McKinnell, 2006). Therefore, over time, changes in mobile species assemblages can be expected following the protection of habitats from activities that impact on structural complexity (Davies et al. 2021a).

How species are distributed within an ecosystem is an important consideration in conservation and fisheries management (Colton & Swearer, 2010), and it is particularly important to understand species distributions in response to fishing pressures (Baudron et al., 2020). There are a variety of scientific sampling methods that can be used to measure changes in biological communities in response to MPAs. Commonly used methods include: Underwater Visual Census (UVC) (Edgar & Samson, 2004; Russ et al., 2004), extractive sampling methods, such as trawls (Day et al., 2020; Jackson et al., 2002), and both baited and un-baited remote video surveys (Bradley et al., 2017; Cappo et al., 2006; Langlois et al., 2012; Sheaves et al., 2016). The advantages and disadvantages of these different methods have been debated, with the chosen method being appropriate to the communities being sampled (Cappo et al., 2004; Edgar et al., 2004; McIntyre et al., 2015; Mclean et al., 2015; Willis et al., 2000). Baited Remote Underwater Videos (BRUVs) provide a non-extractive way of assessing the mobile fauna associated with a habitat or location over short time periods. While BRUVs cannot be used to measure absolute abundance, they provide a way of understanding differences in mobile species assemblages between locations using relative abundance (Cappo et al., 2004). While the use of bait may confound results towards predators and scavengers,

baited set-ups have been compared to non-baited and the use of bait increased attraction of all species and made for a clearer distinction between fish assemblages (Watson et al., 2005).

Baited video surveys have primarily been used on hard reef habitat in MPAs, especially in tropical systems where biotic reef has been the focus (Goetze et al., 2021). In northern Europe this has also been the case (Davies et al., 2021a) and those that have considered non-reef habitat have not done so in relation to MPAs (Elliott et al., 2017b; Rhodes et al., 2020; Unsworth et al., 2014). The use of baited videos in temperate settings is increasing but their use in surveying assemblage changes in MPAs that have excluded bottom-towed fishing gear is relatively sparse as this form of MPA is few in number in the UK (Solandt et al., 2020). This study aims to assess the effectiveness of three whole-site MPAs to support more diverse mobile assemblages by surveying a range of sedimentary habitats inside each MPA and outside in nearby fished controls, using the temperate waters in Jersey as a case study. This research aims to assess the following hypotheses:

- Species richness and abundance of mobile species is greater in MPAs compared to Open Controls.
- Species richness and abundance of mobile species is greater in older MPAs compared to young MPAs.
- 3) Species assemblage differs between MPAs and Open Controls.
- 4) Species assemblage differs between old and young MPAs
- Presence of indicator species will be greater within the MPAs compared to Open Controls.

#### 4.2. Case study

Jersey is a self-governing dependency of the United Kingdom situated in the Normano-Breton Gulf off the north coast of Brittany, with a marine territorial area of approximately 2,500km<sup>2</sup>. Jersey's marine environment is primarily shallow water coastal seascape consisting of rocky reef, boulders, cobbles, gravel, sand and biogenic habitats such as maerl and seagrass. The large tidal range (up to 12.2 metres) experienced in Jersey also creates a vast intertidal area, contributing to a diverse habitat composition supporting a wide array of species. Maerl and seagrass are habitats of international importance, both of which are IUCN red list species and OSPAR (Convention for the Protection of the Marine Environment of the North East Atlantic) threatened habitats (OSPAR, 2002). As a signatory to OSPAR, The Bern Convention on the Conservation of European Wildlife and Natural Habitats), Convention on Biological Diversity, Ramsar (Convention on Wetlands of International Importance), and ASCOBANS (Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas), Jersey is committed to protecting its marine habitats (Government of Jersey, 2013).

The majority of Jersey's territorial waters (78%) are subject to shared fishing access with French commercial vessels, and there are several habitats of international importance that are located within these shared fishing grounds. As a result of this shared access, the politics and laws surrounding the conservation of these habitats is complex, making them a challenge to manage effectively (Chambers et al., 2020; Fleury, 2011; Fleury & Johnson, 2015). Vessels from both Jersey and France employ bottom-towed fishing methods, and conflict over management between nations can result in measures that are politically driven rather than evidence based, which could lead to unsustainable fisheries. To protect areas of fragile habitat and high

biodiversity associated with two offshore reefs in Jersey's territorial waters, two Marine Protected Areas (MPAs) which excluded mobile fishing gear, including bottom-towed, were designated in 2017 (Chambers et al., 2020) (Figure 4.1). These two MPAs, along with several other previously established inshore MPAs, resulted in 6.5% of Jersey's territorial waters protected from mobile fishing gear. These MPAs are considered to follow the whole-site approach to marine management as all habitats within their boundary are afforded the same level of protection (Solandt et al., 2020). In addition to the MPAs at the Minguiers and the Ecrehous, Jersey also has several inshore "whole-site" MPAs, which have been established over a longer period than the two recent offshore MPAs. Of these, the Southeast (SE) Corner MPA (Figure 4.1) has been included as a comparison site for the two recent MPAs. The SE Corner was first established in 2010, with an extension to the east in 2014. The Minguiers, Ecrehous and Southeast are all shallow reefs with subtidal habitats ranging from 0 to 15 m below chart datum within the MPA boundaries. All are exposed to strong tidal currents, caused by the large tidal range experienced around Jersey.





To assess the impact of the MPAs following the closure to mobile fishing, we tested the assumption that species diversity, abundance, and presence of indicator species would be greater inside the MPAs in comparison to Open Controls, and that assemblage would differ inside the MPAs compared to Open Controls that continue to be fished with mobile gear. The habitats assessed were the sedimentary interstitial habitats between the rocky reefs that would typically continue to be fished in a feature-based approach. Indicator taxa were selected based on their association with complex habitats and their likelihood of being sampled using BRUVs. High trophic level species, such as sharks, are indicators of ecosystem health (Yagnesh et al., 2020) and BRUVs provide useful information for helping design conservation strategies for rare or threatened elasmobranchs (White et al., 2013). Therefore, shark species listed as threatened by the IUCN were investigated, as the ability of MPAs to support these species is not well known in temperate waters.

### 4.3. Materials and methods

#### 4.3.1. Survey design

Three MPAs, two designated in 2017 (Ecrehous and Minquiers) and one in 2010 that was later extended to the east in 2014 (Southeast), were selected as survey locations. Comparable sites, consisting of similar mixed sediments and depth ranges between 5 and 15 m (Below Chart Datum), were selected both inside the MPAs and outside in Open Controls. BRUVs were deployed in July and August in 2019 and 2020. Eight sites were inside the MPAs: two at the Ecrehous, three at the Southeast and three at the Minquiers. Seven sites were Open Controls within 0.4 to 2.5 km from the MPA boundaries, with two at the Ecrehous, two at the Southeast and three at the Minquiers. Between two and six BRUV replicates were deployed at each site each year.

### 4.3.2. Site selection

To select sites, spatial analyses using a combination of benthic substrate, depth, historical fishing information, and the boundaries of the MPAs were used. The habitat

structure differs between the locations: with seagrass found in the Southeast and Minquiers MPAs but not the Ecrehous; and maerl found at the Southeast MPA and Ecrehous MPA but not the Minquiers. Coarse sediment without maerl or seagrass was also surveyed at all locations. Benthic substrate was determined from towed videos that were conducted in 2018 so that only mixed sediments, such as those that are targeted by bottom-towed fishing, were sampled. Depth was obtained from Admiralty charts. Historical fishing information was determined from Vessel Monitoring System (VMS) data between 2014 and 2018 that was supplied by the Government of Jersey Marine Resources team. VMS was only available for boats over 12 m in length and so complete spatial fishing information was unknown, but areas where bottom-towed fishing had previously occurred could be identified. The boundaries of the MPAs define areas where bottom-towed fishing gear is excluded and were used to define the boundary between impacted and unimpacted sites. All selected sites were located on mixed sediments (gravel, sand, shell and cobbles), between 5 and 15 m below chart datum.



Figure 4.2. Baited remote underwater video survey sites in relation to the MPAs.

## 4.3.3. Data collection and equipment

BRUV units consisted of a 0.8 m (8 mm diameter) fiberglass bait pole attached at a 90-degree angle to a 0.5 m (10 mm diameter) fiberglass base pole with weights attached to one end of the base pole (2 kg) and a float attached to the other to ensure the correct orientation of the unit (Fig 4.3a). A metal bait cage was attached

at the end of the bait pole (Fig 4.3b). A GoPro (Hero 4 or 6) was attached to the base pole underneath the bait pole, with the field of view angled at the bait cage. Each unit was attached to ~20 kgs of lead weight via a leading rope (~5 m) and a marker buoy attached to the lead weight with 20 m of leaded line. The bait cage was filled with ~100 g of Atlantic mackerel (*Scomber scombrus*); after each deployment, left over bait was discarded and replaced with fresh bait. Tidal currents may confound interpretation of BRUV data as the sampled area increases with increasing current velocity (Taylor et al., 2013). Strong tidal currents are experienced in the survey location of Jersey, and current speed is known to influence the number of species observed on BRUVs in Jersey's waters (Plaster, 2017). To account for the large tidal range in Jersey and the effect of current speed on bait plume size, all BRUVs were deployed on neap tides within a two-hour period around slack water. BRUVs were left to 'soak' for 40 minutes before being recovered and were spaced a minimum of 300 m from one another to ensure bait plumes were independent.



Figure 4.3. a) Profile of a baited underwater remote video unit and b) view of bait pole and attached bait cage within the field of view of the camera, with two cuttlefish (*Sepia officinalis*) observed on a sandy substrate.

#### 4.3.4. Video analysis

Videos were assessed for quality and were not analysed if the camera was out of focus, the seabed was not in view, or the view of the bait box was obscured. Videos were analysed for 40 minutes from the moment the BRUV was stationary on the seafloor. The number of individuals of each mobile species on screen were recorded every minute, to the highest taxonomic resolution possible. For every minute recorded, the greatest count of each species was taken as the MaxN (maximum number of individuals on screen), and the largest value over the 40 minute period was used. MaxN was used to decrease the chance of an individual being recorded more than once, giving an estimate of relative abundance rather than absolute abundance (Cappo et al., 2004). Small and cryptic benthic species, such as hermit crabs (*Pagurus* spp.), dog whelk (*Tritia reticulata*), and dragonet (*Callionymus* spp.), were not recorded as it was not possible to accurately record these species in high algal or high seagrass cover sites.

### 4.3.5. Data analysis

Generalised Linear Mixed Effects Models (GLMMs) were used to assess the response of taxa diversity and abundance (MaxN) in R (Wickham et al., 2019). Response variables were modelled as a function of Treatment (MPA and Open Control) and Location (Ecrehous, Southeast and Minquiers) with Year (2019 and 2020) and Site (n=15) as random effects. Counts of species and individuals were modelled using a Poisson distribution. When counts of individuals showed overdispersion, a negative binomial distribution was used. To assess the presence/absence of low abundance indicator taxa, a binomial distribution was used. Optimal models were determined using the Akaike Information Criterion (AIC) in which an iterative process was used, where predictor variables were sequentially

added and then dropped from the model, to determine variable contribution to model fit. Data manipulation and visualisation was carried out in the statistical program R (R Core Team 2018) using the packages in the tidyverse (Wickham et al. 2019).

Permutational Multivariate Analysis of Variance (PERMANOVA) in PRIMER v7 statistical software package (Clarke & Warwick, 2001), was used to test for species assemblage differences between Locations and Treatments. The fixed factors were Location and Treatment, and the random factors were Site and Year. Site was nested within Location and Treatment. There were three levels within the factor Location (Ecrehous, Minguiers, Southeast) and two levels within Treatment (MPA and Open Control), and abundances were pooled at the Site (n=15) and Year (2019) and 2020) levels for analysis. To test the species assemblages between Locations and Treatments, the multivariate data were square root transformed and a dummy species with an abundance of 1 was added to ensure that samples that are similarly devoid of species were considered as being similar (Clarke et al. 2006; Sheehan et al. 2013). Bray-Curtis dissimilarity indices (Bray & Curtis, 1957) were used to enumerate difference in assemblages between Location and Treatment. Analyses permutations (Anderson & Ter Braak, used 9999 2003). Non-metric Multidimensional Scaling ordinations (nMDS) were used to visualise the differences in similarity of assemblage between locations and treatments.

### 4.4. Results

A total of 36 taxa were identified across the 101 BRUVs that were retained for analysis following the removal of low quality videos and failed BRUVs (total 153 deployed). Actinopterygii were the most represented with 19 taxa, the most common of which was black seabream (*Spondyliosoma cantharus*). Malacostraca were

represented by eight taxa, most commonly spider crab (*Maja brachydactyla*), and Elasmobranchii by seven taxa, with catshark (*Scyliorhinus canicula*) the most observed species. There was only one species of Gastropoda (*Bucinum undatum*) and one species of Cephalopoda (*Sepia officinalis*) (Table. 4.1).

Table 4.1. Full list of taxa recorded on the BRUVs across all Locations (Ecrehous, Minquiers and Southeast) and Treatments (MPA and Open Control).

#### Actinopterygii

Labrus bergylta Symphodus melops Ctenolabrus rupestris Labridae spp. Dicentrarchus labrax Mullus surmuletus Diplodus vulgaris Trisopterus luscus Trisopterus minutus Conger conger Trachurus trachurus Sparus aurata Chelon spp. Chelidonichthys lucerna Tripterygion delaisi Centrolabrus exoletus *Gobiusculus flavescens* Spondyliosoma cantharus Pollachius pollachius

Cephalopoda Sepia officinalis

### Elasmobranchii

Scyliorhinus canicula Scyliorhinus stellaris Galeorhinus galeus Dasyatis pastinaca Raja undulata Raja brachyura Mustelus spp.

#### Gastropoda

 $Buccinum \ undatum$ 

#### Malacostraca

Maja brachydactyla Cancer pagurus Homarus gammarus Liocarcinusspp. Necora puber Inachus spp. Decapoda spp. Ebalia spp.

## 4.4.1. Total diversity (taxa)

There was a significant treatment effect on the number of taxa, which was greatest in the MPAs ( $4.9 \pm 1.8$ ) compared to the Open Controls ( $3.9 \pm 1.7$ ) (Figure 4.4a, Table 4.2). There was no treatment effect on the number of taxa between Locations.



Figure 4.4. Total numbers of a) taxa and b) individuals for Location and Treatment. Black crosses and error bars show fitted GLMM means and standard errors of individual abundance predicted using GLMMs with Treatment (MPA and Open Control) and Location (Ecrehous, Southeast and Minquiers) as fixed effects and Year (2019 and 2020) and Site (n=15) as random effects.

## 4.4.2. Total abundance

There was no significant treatment effect on abundance (Table 4.2). While the mean number of individuals at the Minquiers was greater inside the MPA ( $23.8 \pm 15.2$ ) compared to Open Controls ( $12.4 \pm 10$ ), this was not significant. The same pattern was seen at the Ecrehous and Southeast, where there was no significant difference in the number of individuals between the MPA and Open Controls (Figure 4.4b, Table 4.2).

Table 4.2. General Linear Mixed Effects Model outputs for baited video taxa and abundance as a function of Location and Treatment with Year and Site as random effects. Figures in bold denote a significant result.

Terms	Estimate	Std. Error	z value	р
Taxa				
Intercept	1.65	0.157	10.5	$< 0.0001^{***}$
MPA - Open	-0.206	0.0984	-2.09	$0.036^{*}$
Southeast - Ecrehous	-0.217	0.129	-1.68	0.092
Southeast - Minquiers	-0.0891	0.111	-0.801	0.42
Ecrehous - Minquiers	0.127	0.123	1.04	0.3
Abundance	•			
Intercept	2.91	0.142	20.5	$< 0.0001^{***}$
MPA - Open	-0.283	0.208	-1.36	0.26
Southeast - Ecrehous	-	-	-	-
Southeast - Minquiers	-	-	-	-
Ecrehous - Minquiers	-	-	-	-

# 4.4.3. Total assemblage

A significant Treatment x Location interaction indicated that species assemblage differences between treatments varied across locations (p=0.009, Table 4.3; Figure 4.5). This was driven by the significant difference in assemblage between the MPA and Open Controls at the Minquiers (p=0.0002). While there was a strong trend in the assemblage differences between the MPA and the Open Control at the Southeast, this was marginally non-significant (p=0.0525). Assemblage compositions in the MPAs were dissimilar from the Open Controls across locations (Figure 4.5).

Table 4.3. PERMANOVA of assemblage based on Bray-Curtis dissimilarities. a) Main test and b) Pairwise tests for the interactions Location (Lo) and Treatment (Tr). Data were square root transformed. Figures in bold denote a significant result.

PERMANOVA				Pairwise						
Source	df	SS	MS	Pseudo-F	р	$\operatorname{perms}$	Groups	t	р	Unique perms
Tr	1	5031.6	5031.60	5.075	0.000***	9929	Minquiers Open Control - MPA	2.5254	$0.0002^{***}$	9944
Lo	2	5861.1	2930.50	2.956	$0.000^{***}$	9921	Ecrehous Open Control - MPA	1.2882	0.1171	9933
TrxLo	2	4203.8	2101.90	2.120	$0.009^{**}$	9923	Southeast Open Control - MPA	1.408	0.0525	9934
Pooled	95	94180.0	991.37							
Total	100	111940.0								



Figure 4.5. nMDS illustrating the dissimilarities in assemblage composition between Locations (Ecrehous, Minquiers and Southeast) and Treatments (MPA and Open Control). Each point represents a site and shows the average assemblage composition across the replicates within each site.

The average abundance of *S. cantharus*, *Maja brachydactyla* and *Scyliorhinus canicula* were high in all treatments and locations (Table 4.4). There was also a relatively high average abundance of *Trisopterus luscus* in both the MPA and Open Control treatments at the Southeast. *Mullus surmuletus* had higher abundances within MPAs compared to Open Controls at all locations. *Sepia officinalis* and *lnachus* spp. both had higher contributions to the Open Control assemblages than the MPA at the Ecrehous; similar trends were seen at the Minquiers and Southeast for these two species, but contributed less overall to assemblage composition. Two Labridae species (*Labrus bergylta* and *Symphodus melops*) were found to contribute to the average abundance in the assemblage compositions within the MPAs at both the Minquiers and Southeast compared to Open Controls.

Table 4.4. SIMPER analysis results showing the differences in average abundance (Av.Abund) and dissimilarity (Av.Diss) between MPA and Open Control treatments at each Location (Ecrehous, Southeast and Minquiers). Species contribution cut-off was set at 95%.

Species	MPA Av.Abund	Open Control Av.Abund	Av.Diss	$\mathrm{Diss}/\mathrm{SD}$	$\operatorname{Contrib}\%$	$\operatorname{Cum.\%}$
Ecrehous						
Spondyliosoma cantharus	2.34	2.47	6.84	1.23	19.50	19.50
Maja brachydactyla	0.79	1.36	5.71	1.17	16.28	35.78
Scyliorhinus canicula	1.74	1.82	4.17	0.93	11.88	47.66
Sepia officinalis	0.13	0.40	2.97	0.80	8.47	56.13
Inachus spp.	0.00	0.48	2.93	0.57	8.35	64.49
Trisopterus minutus	0.52	0.00	2.27	0.27	6.46	70.94
Mullus surmuletus	0.23	0.00	1.68	0.49	4.78	75.72
Homarus gammarus	0.13	0.09	1.30	0.49	3.71	79.43
Cancer pagurus	0.20	0.00	1.23	0.48	3.51	82.94
Necora puber	0.13	0.00	0.74	0.38	2.10	85.04
Labrus bergylta	0.09	0.00	0.69	0.26	1.98	87.02
Liocarcinus spp.	0.00	0.13	0.67	0.31	1.92	88.93
Raja brachyura	0.00	0.09	0.66	0.31	1.87	90.80
Ctenolabrus rupestris	0.07	0.00	0.51	0.26	1.46	92.27
Dasyatis pastinaca	0.07	0.00	0.51	0.26	1.46	93.73
Trisopterus luscus	0.07	0.00	0.51	0.26	1.46	95.19
Southeast						
Spondyliosoma cantharus	2.10	2.08	5.79	0.97	13.13	13.13
Trisopterus luscus	0.54	0.65	5.66	0.60	12.81	25.94
Maja brachydactyla	1.01	1.46	5.54	1.26	12.55	38.49
Scyliorhinus canicula	1.71	2.09	3.77	1.26	8.55	47.04
Mullus surmuletus	0.49	0.17	3.71	0.87	8.41	55.44
Sepia officinalis	0.25	0.34	2.78	0.78	6.29	61.73
Trachurus trachurus	0.00	0.42	2.68	0.57	6.06	67.80
Labrus bergylta	0.32	0.17	2.51	0.72	5.68	73.48
Symphodus melops	0.35	0.00	2.16	0.72	4.90	78.39
Pollachius pollachius	0.22	0.00	1.36	0.47	3.08	81.46
Scyliorhinus stellaris	0.20	0.00	1.34	0.49	3.04	84.50
Buccinum undatum	0.05	0.10	0.91	0.39	2.05	86.56
Galeorhinus galeus	0.12	0.00	0.89	0.33	2.01	88.57
Trisopterus minutus	0.00	0.14	0.73	0.33	1.66	90.23
Conger conger	0.10	0.00	0.66	0.33	1.50	91.73
Inachus spp.	0.00	0.10	0.60	0.33	1.36	93.09
Raja brachyura	0.00	0.10	0.54	0.33	1.23	94.32
Dicentrarchus tabrax	0.09	0.00	0.44	0.23	1.01	95.33
Minquiers						
Spondyliosoma cantharus	3.03	1.28	16.05	1.35	25.54	25.54
Scyliorhinus canicula	0.98	1.43	7.24	1.18	11.53	37.07
Maja brachydactyla	1.49	1.50	6.66	0.98	10.60	47.67
Mullus surmuletus	0.73	0.00	4.61	0.79	7.33	55.00
Labrus bergytta	0.50	0.04	3.42	0.88	5.44	60.45
Inachus spp.	0.11	0.49	3.14	0.66	4.99	65.43 70.01
Chalan area	0.51	0.08	2.88	0.44	4.58	70.01
Cheton spp.	0.38	0.00	2.80	0.40	4.46	74.47
Sympnoaus melops	0.38	0.04	2.54	0.64	4.04	78.01
Trachurus trachurus	0.20	0.24	2.48	0.44	3.94	82.45
Septa officinatis	0.25	0.19	2.00	0.07	3.20	00.11
Sculiophique stellerie	0.18	0.12	1.04	0.40	2.02	00.55
Trisonterus minutus	0.05	0.15	1.40	0.40	2.22	02.16
Dasuatie nastinaac	0.05	0.10	0.00	0.25	1.01	03.50
Cancer nominaca	0.05	0.00	0.90	0.30	1.40	93.39
Pollachius pallachius	0.07	0.00	0.50	0.43	0.79	95.76
· ormentation productionad	0.01	0.00	0.00	0.40	0.10	00.10

## 4.4.4. Indicator taxa

Indicator taxa were selected based on their association with complex habitats and their likelihood of being sampled using BRUVs, these included: *Maja brachydactyla*,

Labridae spp., *Spondyliosoma cantharus*, Juvenile *S. cantharus* and sharks and rays listed as threatened on the IUCN red list (Nieto et al., 2015; Walker et al., 2020) (Figure 4.6).



Figure 4.6. Indicator taxa. a) *Maja brachydactyla* b) *Labrus bergylta* (example of Labridae spp.) c) *Spondyliosoma cantharus* d) Juvenile *S. cantharus* e) *Galeorhinus galeus* (example of shark species) f) *Raja undulata* (example of ray species).

## Maja brachydactyla

*Maja brachydactyla* (Figure 4.6a) was the only species of the five key commercial species that could be assessed using BRUV data and was recorded in high enough numbers to assess differences in relative abundance. *M. brachydactyla* was recorded in greater abundance in Open Controls ( $2.5 \pm 1.8$ ) compared to MPAs ( $1.9 \pm 2.4$ ), except for at the Minquiers where there was marginally greater abundance inside the MPA ( $3 \pm 2.9$ ) compared to Open Controls ( $2.7 \pm 2$ ) (Figure 4.7, Table 4.5). Abundance of *M. brachydactyla* was greater at the Minquiers compared to both the Ecrehous and Southeast, suggesting there was an effect of geographical location.





Table 4.5. General Linear Mixed Effects Model outputs for baited video abundance of *Maja brachydactyla* as a function of Treatment and Location with Year and Site as random effects. Figures in bold denote a significant result.

Terms	Estimate	Std. Error	z value	р
Maja brachydactyla				
Intercept	-0.167	0.316	-0.529	0.6
MPA - Open	0.940	0.381	2.470	$0.014^{*}$
Ecrehous - Minquiers	1.260	0.341	3.710	$< 0.0001^{***}$
Ecrehous - Southeast	0.605	0.363	1.670	0.095
Minquiers - Southeast	-0.658	0.260	-2.530	$0.012^{*}$
MPA - Open*Ecrehous - Minquiers	-1.080	0.441	-2.450	$0.014^{*}$
MPA - Open*Ecrehous - Southeast	-0.504	0.490	-1.030	0.3
$\operatorname{MPA}$ - $\operatorname{Open}^*\operatorname{Minquiers}$ - Southeast	0.575	0.380	1.510	0.13

## Labridae species

There was a significant treatment effect on the presence of Labridae spp. (Figure 4.6b), with Labridae recorded on 23 videos within the MPA compared to just three in Open Controls (Figure 4.8, Table 6). While there was no location effect on Labridae spp. presence, there were few sites with wrasse present at the Ecrehous compared to the other two locations (Table 4.6).





Table 4.6. General Linear Mixed Effects Model outputs for baited video presence of Labridae spp. as a function of Treatment and Location with Year and Site as random effects. Figures in bold denote a significant result.

Terms	Estimate	Std. Error	z value	р
Labridae spp.				
Intercept	-1.490	0.685	-2.180	$0.029^{*}$
MPA - Open	-2.580	0.720	-3.580	$< 0.0001^{***}$
Ecrehous - Minquiers	1.580	0.812	1.940	0.052
Ecrehous - Southeast	1.500	0.819	1.840	0.066
Minquiers - Southeast	-0.071	0.639	-0.111	0.91

## Spondyliosoma cantharus

While there were more sites where adult *S. cantharus* (Figure 4.6c) was present within the MPAs (47 sites) compared to Open Controls (34 sites), this was not significant (Fig 4.9a, Table. 4.7). This differed from the juveniles (Figure 4.6d) where

a significantly greater presence within the MPAs (23 sites) was observed compared to Open Controls (8 sites) (Figure 4.9b, Table 4.7). There was no effect of location on the presence of adult or juvenile *S. cantharus*.



Figure 4.9. Cumulative presence of a) adult and b) juvenile *Spondyliosoma cantharus* at each Location and Treatment.

Table 4.7. General Linear Mixed Effects Model outputs for baited video presence of adult and juvenile *Spondyliosoma cantharus* as a function of Treatment with Year and Site as random effects. Figures in bold denote a significant result.

Terms	Estimate	Std. Error	z value	р
Adult Spondy	liosoma co	an thar us		
Intercept	2.640	0.788	3.350	$< 0.0001^{***}$
MPA - Open	-0.801	1.050	-0.766	0.44
Juvenile Spon	dyliosoma	can thar us		
Intercept	-0.232	0.279	-0.830	0.41
MPA - Open	-1.350	0.478	-2.830	$0.0047^{**}$

### IUCN threatened shark species

Of the shark species observed on the BRUVs, three were listed as threatened by the IUCN. These species were: nursehound *Scyliorhinus stellaris* (near threatened), smoothhound *Mustelus* spp. (vulnerable) and tope *Galeorhinus galeus* (critically endangered) (Figure 4.6e). IUCN threatened shark species were only recorded at eight sites within the MPAs over the two years, and at three sites in the Open Controls (Figure 4.10). *G. galeus* is an IUCN critically endangered species as of 2020 (Walker et al., 2020) and was only recorded once within a MPA, where there were two individuals. While there were more sites where shark species were present within the MPAs compared to Open Controls, it was not possible to statistically test this difference due to the small sample size.



Figure 4.10. Cumulative presence of IUCN listed near threatened or vulnerable shark species at each Location and Treatment.

### IUCN threatened ray species

Of the ray species observed on the BRUVs, three were listed as threatened by the IUCN. These species were Blonde ray *Raja brachyura* (near threatened), Undulate ray *Raja undulata* (near threatened, Figure 4.6f) and Common stingray *Dasyatis pastinaca* (vulnerable). These three ray species were recorded at four sites within the MPAs over the two years, and at six sites in the Open Controls (Figure 4.11). While there were more sites where ray species were present in the Open Controls compared to the MPAs, it was not possible to statistically test this difference due to the small sample size.



Figure 4.11. Cumulative presence of ray species at each Location and Treatment.

#### 4.5. Discussion

Between 2014 and 2017, three MPAs were designated in Jersey's territorial waters. These MPAs provide protection to all the habitats and species within the MPA boundary (rather than just the features of conservation interest) and provide a timely opportunity to undertake research to understand how this form of management can benefit biodiversity. The results from this study show that after three to six years of protection there appears to be significant differences in mobile benthic communities inside the MPAs compared to Open Controls. Not only was the number of taxa greater within all MPAs, but assemblage composition also indicated that the MPA species assemblages were distinct from the Open Controls. The MPAs were characterised by greater proportions of Labridae species, *M. surmuletus* and juvenile *S. cantharus* compared to the Open Controls, which had greater proportions of *Inachus* spp. and *S. officinalis*. Further surveys are needed to monitor future changes as the literature would suggest that this difference will strengthen over time as communities become established within the MPAs (Davies et al., 2021a; Edgar et al., 2014).

Understanding fish distributions is a key component of effective management to ensure that space and habitat requirements of species are accounted for. Results suggested that assemblages varied across the treatments and this was most noticeable at the Minquiers where the mobile species assemblage was significantly different inside the MPA compared to Open Controls. While three species were found to dominate in all treatments and locations (*S. cantharus*, *M. brachydactyla* and *S. canicula*), there was a greater diversity of species contributing to the assemblage composition at the Minquiers. A previous BRUV survey also found these species to be dominant in inshore areas in Jersey (Plaster, 2017), suggesting they

are prominent across the whole of Jersey's territorial waters. The Minquiers MPA is thought to have been the least affected by bottom-towed fishing prior to its designation due to the numerous unchartered reef heads which pose a hazard to bottom-towed fishing gear. It may be that the relatively unimpacted condition of this MPA contributes to greater habitat heterogeneity, complexity and integrity that is supporting a greater range of species. While the presence of seagrass at the Minquiers may be a factor in the species assemblage there, the fact that species assemblage at the Southeast MPA was not different to the Open Controls suggests the presence of seagrass is not the only driving factor, as seagrass is also present in the Southeast MPA. It may be that species assemblages in the southern region of Jersey's territorial waters are distinct from those further north.

The response of a species to protection from bottom-towed fishing will vary depending on their life history and it was therefore appropriate to investigate indicator species which had been chosen for their predicted biological response to protection measures. Within three years of the MPAs being established, two of the five indicator taxa (Labridae spp. and juvenile *S. cantharus*) showed strong associations with the MPAs, one (*M. brachydactyla*) showed greater abundance in Open Controls and the other two indicator taxa (IUCN sharks and rays) were not detected in high enough numbers to assess. Of the five indicator taxa, only *M. brachydactyla* was recorded in high enough numbers to model abundance, which showed greater abundance in Open Controls except for at the Minquiers. *M. brachydactyla* was the only indicator species that experienced high levels of fishing pressure in the form of potting, with Landings Per Unit Effort (LPUE) known to be increasing in recent years (Marine Resources, 2019). As reported in other MPAs that have excluded bottom-towed fishing (Burton et al., 2016; Mangi et al., 2011; A.
Rees et al., 2021), there may have been increased potting effort within the MPA boundaries due reduced conflict with bottom-towed fishing. This may in part explain why *M. brachydactyla* abundance is lower in the MPAs overall compared to Open Controls. However, the observation of greater abundances in the southern MPA of the Minquiers, compared to the Ecrehous in the northeast, and the Southeast MPA, suggests there is a location effect. This could be due to a number of reasons, such as habitat composition, currents, exposure, or potting fishing effort.

Increased structural complexity of seafloor habitat promotes more productive and ecologically resilient assemblages that contribute to functional diversity, further increasing the diversity and abundance of species that are able to be supported by an ecosystem (Graham & Nash, 2013; Howarth et al., 2011). Pelagic-benthic coupling is stronger in coastal areas (Kopp et al., 2015) and systems with higher benthic primary productivity support higher trophic levels (Brown et al., 2010; Hunt & McKinnell, 2006). Increased primary productivity will improve trophic links and provide prey for commercially important fishes such as bream (S. cantharus). Juvenile bream were identifiable from adults based on small size in relation to the bait box, a more pronounced black band on their tail and the presence of numerous broken yellow stripes running the length of the body. High densities of juveniles are indicative of nursery areas (Beck et al., 2001). S. cantharus is a species of commercial value and the greater number of juveniles inside the MPA alongside no difference in adult bream indicates that there are areas within the MPAs that are important nursery areas for this species, not just that this species only occurs in the MPAs. Nursery areas tend to be highly productive (Beck et al., 2001; Heck et al., 1995) and complex habitat structures improve the survivorship of juvenile species (Bradshaw et al., 2003), indicating that habitats within the MPAs may be more

structurally complex. As this trend was seen across all locations, juvenile abundance is most likely driven by habitat integrity, rather than habitat type, as the habitats varied across the three MPAs. Juvenile pollack (*Pollachius pollachius*) and other unidentifiable juvenile fish were also recorded, but they were in two few numbers to assess. It may be that BRUVs are not an appropriate method to sample juveniles of many species as they may be deterred from BRUVs where larger individuals are congregated.

The success of marine reserves tends to be limited to sessile or sedentary species or those with high site fidelity (Kaplan, 2009; Klein et al., 2015). Labridae species, which are known to have strong site fidelity (Mucientes et al., 2019), were observed in greater presence in MPA sites than Open Control sites. Labridae species are relatively long lived: *Symphodus melops and Labrus bergylta* were the most commonly observed of the four species and they can live up to 9 and 29 years respectively (Skiftesvik et al., 2014; Treasurer, 1994). Reproductive success is lowered when populations are depleted and, particularly for long lived species, it will take time for larger species to be replenished (Lotze et al., 2011). No-take MPAs may favour large-sized, long-lived Labridae species (Coll et al., 2012), but considerable time-scales are needed for long-lived species to benefit from protection measures owing to their life-history traits (Lotze et al., 2011). While it was not significant, the presence of Labridae species was lower at the Ecrehous compared to the other two MPAs, suggesting there may also be an effect of habitat type or geographical location influencing the distribution of these species.

Sharks are indicators of ecosystem health and are in decline globally (Yagnesh et al., 2020), but it is unclear whether MPAs have the ability to support shark species,

most of which are long-lived with wide ranges. Vulnerable or near threatened shark and ray species were not observed in high enough numbers to assess which could either be related to an inappropriateness of BRUVs to detect these species, or generally low populations of these species in Jersey's shallow water marine environments. As BRUVs have been used in other locations to assess sharks and rays (White et al., 2013) it seems unlikely that the methodology is a factor in this and is most likely related to low population size, as suggested by the IUCN as they are in decline or at risk of being in decline (Nieto et al., 2015). MPAs alone are not sufficient to protect wide ranging species such as sharks and rays which will move beyond the boundaries of protection to forage, mate or migrate (Gaines et al., 2010; White et al., 2013). However, MPAs may provide improved foraging grounds for these species but species-specific management such as zero bycatch allowances or improvements in fishing techniques to decrease the capture of these species, or in the case of ray that are still legally targeted, decreased quotas, are needed to prevent further declines. In 2017, a zero-catch limit for G. galeus was implemented in Jersey, so this BRUV survey sets a baseline for this species against which future changes can be compared to. This ban on fishing combined with the MPAs may support the recovery of this species which is in global decline (Walker et al., 2020). It may be that the replication required to detect these species is much greater than for more abundant species.

Jersey's fishery is dependent on five key shellfish species (Lobster Homarus gammarus, brown crab Cancer pagurus, spider crab Maja brachydactyla, whelk Buccinum undatum and king scallop Pecten maximus), and all bar spider crab are not readily detected by BRUVs due to their life histories and behaviours. King scallop are sedentary filter feeders and so will not be observed on baited videos. Whelk are

attracted to the bait but are generally too slow to reach the unit during the deployment period. Lobster and brown crab are the more valuable of the three crustaceans but both are primarily active at night (Lawton & Lavalli, 1995; Skajaa et al., 1998) and so BRUVs deployed in the day will not provide accurate assessments of these species as only those with unusual behaviours are likely to be attracted to the bait in daylight. Spider crab is the one commercial species that is readily attracted to the baited videos and spider crab was found to be in lower abundance within the Southeast and Ecrehous MPAs compared to Open Controls. While spider crab are protected from mechanical damage from bottom-towed fishing gear inside the MPAs, they are not protected from potting. Potting levels are difficult to measure in Jersey as all potting vessels are less than 12 metres in length, meaning they are not required to have VMS and their spatial fishing activity unknown. It may be that potting levels have increased within the MPAs following the exclusion of bottomtowed fishing, as has been observed elsewhere (Mangi et al., 2011), and is preventing any changes in the spider crab populations being observed. Landings data from within Jersey's waters shows that both catch and effort of spider crab is increasing but this cannot be attributed to the MPAs.

### 4.6. Conclusion

This study provides the first baseline for mobile species assemblages across geographically separate MPAs in Jersey's waters, where the mobile species associated with sedimentary, interstitial reef habitat have responded positively to the exclusion of bottom-towed fishing. Following protection, the differences observed in benthic species assemblages are in line with results from similar studies worldwide and further adds to the evidence base that multi-use MPAs can improve ecological condition, provided bottom-towed fishing is excluded from all areas within the MPA

boundary. Given that roughly 70% sedimentary habitats within MPAs in English waters are unprotected from bottom-towed fishing gear (MPA Reality Check, 2019) there is scope to greatly improve the ability of MPAs to support the recovery of biodiversity. This initial observation of recovery can be expected to improve and species assemblage to continue to change over decadal time spans. The longest monitoring of a whole-site MPA in temperate waters detected change in biological communities 12 years after implementation (Davies et al., 2021a). Further field studies are needed to monitor changes as it may take several years, if not decades, for biological communities to stabilise within the MPAs. It is therefore recommended that monitoring is continued to further understand changes in species assemblages and distributions in response to protection measures and to provide reference areas to compare fished areas against, to inform best practice. Further, the impact of other fishing pressures may confound the ability of a MPA to improve commercial species stocks, and the implementation of NTZs is recommended to better understand more natural population structures and abundances of exploited species.

 Chapter 5 – The response of epibiotic and infaunal assemblages to the removal of bottom-towed-fishing from Marine Protected Areas in Jersey, Channel Islands, UK

# Author Contributions

Samantha Blampied was responsible for the conception and design of this research. She organised the fieldwork, undertook the analysis, interpretation of data and writing of the chapter. Francis Binney assisted in fieldwork logistics and all supervisors contributed to the critical revision of this manuscript and have given their approval for inclusion in this thesis.

Author	Institution	Contribution
Samantha Blampied	University of Plymouth	70%
Sian Rees	University of Plymouth	5%
Martin Attrill	University of Plymouth	10%
Francis Binney	Government of Jersey	5%
Emma Sheehan	University of Plymouth	10%

#### Abstract

Marine sediments support biodiversity on the seafloor but they are being altered by anthropogenic pressures, such as bottom-towed fishing, with consequences for the diversity and distribution of species. Marine Protected Areas (MPAs) are increasingly used as spatial management tools to mitigate human impacts on marine systems, but the level of protection varies. MPAs that exclude bottom-towed fishing activities from all habitats within their boundaries, a form of 'whole-site' MPA, have shown benefits in terms of increasing biodiversity but sedimentary habitats, and their infaunal assemblages, have rarely been studied. In this study we use a combination of methods to quantify change over time (2018-2020) in the coverage of structure forming organisms in sedimentary habitats; sediment composition; species assemblage; and diversity and abundance of both epibiota and infauna. These metrics have been assessed inside multiple 'whole-site' MPAs of differing ages (three and six years) within Jersey's territorial waters. Mixed sediment habitats, such as those targeted by bottom-towed fishing practices, were selected for study inside the MPAs and in nearby fished Open Control sites. The coverage of structure forming organisms was greater in the MPAs compared to Open Controls, but little change was observed across the three years except for a decline in coverage of structure forming organisms in the Open Control sites. The number of epibiota taxa and individuals was only greater in the MPAs compared to Open Control in the first year of study (2018), after which there was no significant difference between treatments. Sediment properties differed between habitat types (determined by substrate type and presence of structure forming organisms) with little observed effect of treatment. The number of infaunal taxa was greater within all three MPAs compared to the Open Controls but was only significant for the oldest MPA. Habitat

type was a key factor in the observed response of biodiversity metrics to protection from bottom-towed fishing. This study provides the first insight into species assemblages, diversity and abundance in response to MPA designation across a variety of sedimentary habitats and further evidences the need to protect these habitats within MPAs to support biodiversity.

### 5.1. Introduction

Marine sediments cover the majority of the seafloor and support the greatest faunal biodiversity globally (Snelgrove, 1998). The benthic communities associated with marine sedimentary habitats (muds, sands, gravels, and mixed sediments – EUNIS (Davies et al., 2004)) comprise microbial, meiofaunal and macrofaunal organisms that play important roles in the provision of ecosystem processes and services (Heery et al., 2017; Snelgrove, 1998; Thrush & Dayton, 2002; Woodin et al., 2016). For example, macrofaunal activity on the seafloor facilitates carbon storage, biogeochemical cycling, burial and metabolism of pollutants, and sediment transport (Snelgrove, 1998; Snelgrove et al., 2014). Marine sedimentary habitats can be further categorised based on the biological communities associated with them, such as ecosystem engineer species that add structural complexity to sediments by biologically and physically interacting with their environment. Ecosystem engineers such as seagrasses, maerl, polychaetes, and bivalves add structural complexity to marine sediment habitats (Jones et al., 1994), increase their value as nursery and feeding habitats to other species, and improve their ability to sequester carbon (Fourgurean et al., 2012; Githaiga et al., 2019). The biodiversity within marine sediments also provides a source of prey for many species (Carruthers et al., 2002; Hines et al., 1997), including those of commercial importance (Hall et al., 1991; Karnofsky et al., 1989).

In coastal areas where marine sediments support benthic fishery species, bottomtowed fishing practices are used that involve dragging heavy nets and chains along the seafloor. This form of fishing can have negative consequences for benthic biodiversity in the form of bycatch (Silva & Ellis, 2019; Stratoudakis et al., 2001). direct damage to species (Beukers-Stewart and Beukers-Stewart 2009; Kaiser and Spencer 1995), and habitat degradation (Hall-Spencer & Moore, 2000; Oberle et al., 2017; Thrush et al., 2001; Thrush & Dayton, 2002), potentially leading to shifts in benthic communities (De Grave & Whitaker, 1999). Recent studies have highlighted the sensitivity of marine sediments to fishing practices (Eno et al., 2013; Hooper et al., 2017), especially those characterised by habitat forming species that are temporally stable and allow for the development of diverse benthic communities (Bouma et al., 2009; Duffy, 2006; Van Hoey et al., 2008). The loss of species, ecosystem engineers, or functional groups that influence sediment biogeochemistry or habitat structure can have negative effects on biodiversity and ecosystem function (Coleman & Williams, 2002; Thrush et al., 2001, 2017; Waldbusser et al., 2004). For example, the removal of oyster beds in Scotland through bottom-towed fishing altered the species assemblage and resulted in a reduction of overall mollusc biomass and species richness (Thurstan et al., 2013). Activities that disturb the seabed, such as bottom-towed fishing, alter the granulometric properties and carbon content of sediments (Macreadie et al., 2019; Nayar et al., 2007), and therefore the biological community composition that is influenced by sediment composition (Cooper et al., 2011; Newell et al., 1998). Ecosystem engineers also alter sediment composition (Meadows et al., 2012), and loss of these organisms may result in reduced structural complexity or an alteration of sedimentary

biogeochemistry that may have implications for biodiversity, ecological processes and carbon storage (Coleman & Williams, 2002; Lima et al., 2019; Sala et al., 2021).

Habitat connectivity is important for species that rely on multiple habitats in their life cycle (Seitz et al., 2014). Juveniles, including those of commercial species, use multiple habitats but may also exhibit selectivity (Elliott et al. 2017b). For example, seagrass has higher abundances of juvenile fish than surrounding sand and mud habitats (Jackson et al. 2002; Lilley and Unsworth 2014). Therefore, the loss or fragmentation of habitats supporting essential life stages could have implications in the ability of an ecosystem to support species through to their adult stage. As different habitats support species in varying ways, with many species using multiple habitats for both conservation and fisheries (Kritzer et al., 2016; Sheehan et al., 2013a). The exclusion of bottom-towed fishing can enhance seabed integrity (Pikesley et al., 2021), which has been shown to support greater fish biomass, including commercial species (Beukers-Stewart et al., 2005; Brown et al., 2010; Willis et al., 2003), and this can be achieved through the use of Marine Protected Areas (MPAs).

MPAs are defined by the IUCN as a 'clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values' (Day et al., 2012). This form of spatial management can be used as a tool to mitigate fishing pressure on benthic habitats and species. However, the level of protection afforded a MPA varies from highly protected No Take Zones (NTZs), in which all extractive activities are prohibited (Sala & Giakoumi, 2018), to those that are

partially protected where only certain activities are excluded (Claudet, 2018). Many MPA regulations are not enforced strongly enough to have an ecological effect (Pieraccini et al., 2017). Currently, MPAs cover just 7% of the marine environment, with only 2.7% considered highly protected where destructive fishing practices are excluded (Marine Conservation Institute, 2022). The area of seafloor protected from bottom-towed fishing in English waters is 4,508 km<sup>2</sup> (MPA Reality Check, 2019), roughly 2%. Of this protected area, half (1% total) comprises sedimentary habitats (mud, mixed, sediment and seagrass), and the other half is reef habitat.

Many MPAs in the UK are feature based in that they have been designated with the intention to conserve specific species or features, but this approach has not been shown to be effective in the recovery of marine biodiversity (Solandt et al., 2014). The ecosystem-based, or "whole-site" approach to MPA management, in which all habitats within the MPA boundary are protected, is widely advocated but is rarely demonstrated in practice. Additionally, the ability to measure the impact of bottomtowed fishing on the seabed is confounded by shifting baselines (Jackson et al., 2011; Ulman & Pauly, 2016), with uncertainties regarding the true ecological condition of the seabed as most MPAs have previously been impacted by bottomtowed fishing (Cook et al., 2013; Solandt et al., 2020). The Lyme Bay MPA in southwest England is the largest, long-term example of the whole-site approach in the UK. Following the exclusion of bottom-towed fishing from the whole MPA, structure forming, reef associated species recovered on both the rocky reef and sedimentary habitats (Sheehan et al., 2013b). This surprising discovery suggested our perception of what species sedimentary habitats can support had been altered by bottom-towed fishing pressure and highlighting that sedimentary habitats can stabilise if left undisturbed and have a role in recovering biodiversity. Further, the

increase in sessile benthic species (Sheehan et al., 2013a; Sheehan et al., 2021) and mobile commercial species (Davies et al., 2021a) following the exclusion of bottom-towed fishing has demonstrated the benefit that this type of MPA can have for both conservation and fisheries alike.

MPAs managed using the whole-site approach are rare and research on benthic habitat recovery following the exclusion of bottom-towed fishing has focused on reef habitat. The implementation of new MPAs in Jersey, Channel Islands, that have adopted the whole-site approach and protect both the reef habitat and interstitial sedimentary habitat provides an opportunity to further understand how marine sediments recover following the exclusion of bottom-towed fishing. Namely, the benefit to benthic communities resulting from the exclusion of bottom-towed fishing that can be measured using biodiversity metrics, such as species richness and abundance. This study aims to determine how benthic communities respond following the exclusion of bottom-towed fishing using the to be open to bottom-towed fishing. Following the cessation of bottom-towed fishing, it is expected that changes will be observed in both the infaunal and epifaunal assemblages (Bradshaw et al., 2001; Davies et al., 2021a; Frojan et al., 2009; Sheehan et al., 2013a).

Jersey is an island located in the Normano-Breton Gulf (Figure 5.1) with subtidal habitats (~2,500 km<sup>2</sup>) that are a mosaic of rocky reef, kelp forests, boulder fields, maerl beds, seagrass beds (*Zostera marina*) and mixed/coarse sediments. Large areas of sedimentary habitat support economically valuable bivalve populations (scallop, *Pecten maximus*; amande, *Glycymeris glycymeris*; and praire, *Venus verrucosa*) that are targeted by bottom-towed fishing practices. The majority of

Jersey's territorial waters are shared with France in an area called the Bay of Granville, and both Jersey and French mobile gear vessels target scallop using scallop dredges; chain nets with metal teeth at the opening that are pulled across the seabed to flip scallops into the nets. French mobile gear vessels also target praire and amande using praire dredges that penetrate the sediment to a deeper depth than scallop dredges to obtain the bivalves that are buried deeper in the sediment.

Concerns over the health of the seabed, particularly in areas deemed to be intolerant of mechanical damage, such as maerl and seagrass, led to the expansion of MPAs into offshore areas in Jersey's territorial waters (Chambers et al., 2020). These MPAs add to a network of previously established coastal MPAs between 2001 and 2010, resulting in 6.5% of seabed protected from bottom-towed fishing in October 2017. The Ecrehous (northeast offshore), and the Minguiers (south offshore) MPAs were established in October 2017, and the southeast MPA (southeast inshore) was first established in 2010 and later extended in 2014 to the extent shown in Figure 5.1. Taking into account lessons learnt from the whole-site approach MPA in Lyme Bay, and in keeping with the pre-established coastal MPAs, the three most recent MPAs were designated to protect the whole area of seabed from bottom-towed fishing within the MPA boundaries, while other, low-impact fishing methods such as static fishing and scallop diving, were permitted to continue. This provided the opportunity to assess the whole-site approach across a network of MPAs in an offshore location to assess the recovery of benthic assemblages following the exclusion of bottom-towed fishing (Figure 5.1).



Figure 5.1. Location of Jersey within the Normano-Breton Gulf. The light green shaded area shows the extent of the Bay of Granville which is an area of shared access rights for both Jersey and French vessels. The dashed green line shows the international boundary between Jersey and France, everything within this boundary is Jersey's territorial waters. MPAs are areas which are protected from bottom-towed fishing gear.

The aim of this study is to test whether the exclusion of bottom-towed fishing has resulted in differences in sediment composition, epibiota and infaunal assemblages associated with sedimentary habitats in the MPAs compared to adjacent Open Control sites. The following hypotheses will be tested relating to epibiota, sedimentary and infaunal response to protection from bottom-towed fishing:

## Epibiota

 Over time, the proportional cover of structure-forming taxa (seagrass, maerl and sand-mason worm) is greater inside the MPAs relative to Open Control sites that remain open to bottom-towed fishing.

- Over time, richness of taxa and overall abundance are greater inside the MPAs relative to Open Control sites that remain open to bottom-towed fishing.
- Over time, the assemblage composition is more diverse inside the MPAs relative to Open Control sites that remain open to bottom-towed fishing.

## Sediment properties

- Sediment properties (particle size, organic carbon, proportion of live and dead maerl) change over time in the MPAs relative to the Open Controls and influence associated species metrics.

## Infauna

- Over time, richness of taxa and overall abundance are greater inside the MPAs relative to Open Control sites that remain open to bottom-towed fishing.
- Over time, the assemblage composition is more diverse inside the MPAs relative to Open Control sites that remain open to bottom-towed fishing.

# 5.2. Methods

# 5.2.1. Survey design

To collect data on the epibiota and infaunal assemblages, surveys were conducted on sedimentary habitats inside the MPAs and outside on Open Control sites between May and September each year from 2018 to 2020. The survey design considered two Treatments, 'MPA' and 'Open Control', at three locations (Ecrehous, Minquiers and Southeast) (Figure 5.2). The first surveys in 2018 began seven months after the designation of the MPAs and provide the baseline for the habitat composition in the areas surveyed. To ensure surveys took place on comparable sedimentary habitat, sites were selected based on depth (between 5 and 15 m below chart datum), previous survey data (Chambers et al., 2016; Chambers & Binney, 2015, 2016; Le Hir, 1986; Retière, 1979) and personal communication with the Government of Jersey Marine Resources team, the Marine Biology Section of the Société Jersiaise and Jersey Marine Conservation. The sedimentary habitats of interest were seagrass (Zostera marina), maerl spp, sandmason worm (Lanice conchilega) and bare sediment. Pilot surveys were conducted using a towed video to ground truth the location of sedimentary habitats prior to carrying out the full surveys. During the pilot surveys it became apparent that seagrass was not present in great enough coverage outside of the MPAs to be included as a habitat for this treatment and so the resulting design of the surveys is unbalanced in that not all habitats are present in every treatment. A combination of sightings data and VMS locations (pers. comm. Marine Resources) were used to determine areas that had been impacted by dredging outside of the MPAs. Due to the large tidal cycle experienced in Jersey, with up to a 12.2 m drop in tidal height and strong tidal currents of up to 3 m/s, the window for sampling each day was limited (~2 hours), so efficient sample methods were chosen to maximise data collection. A combination of methodologies was employed: towed videos were utilised between June and August each year to assess changes in epibiota assemblages over time (9 sites inside MPAs and 7 in Open Controls), and sediment samples were taken using grabs in May in 2018 and 2020 to assess differences in the infaunal assemblages (7 sites inside MPAs and 4 sites in Open Controls). Sites were nested within Treatment and Location, with three replicates at each site.





## 5.2.2. Data collection

## 5.2.2.1. Towed video

A towed video array was built to survey 100 m video transects of the seabed, the design of which has been adapted from Stevens et al. (2003) and Sheehan et al. (2010). The use of a towed video array is a cost effective way of surveying large areas of seabed that is also non-destructive to the seabed (Bicknell et al., 2016). The towed video array consisted of a real-time underwater video system (SpotX<sup>™</sup>)

Underwater Vision), housing a GoPro Hero4, connected to a console on the survey vessel via a cable to allow for live viewing of the seabed (Figure 5.3 a and b). This system was integrated into a bespoke frame to improve the stability of the video system and also enable the addition of two underwater lights (bigblue 1200 Lumen dive torch) for illumination of the seabed in low light conditions, plus two lasers (Z-Bolt<sup>®</sup> Green Dive Laser 5MW) of a known distance (0.2 m) apart to allow for scaling of images during video analysis. The array was maintained approximately 10 cm above the seabed and towing speed was kept below 0.25 knots, where conditions allowed, to improve image quality. Global Positioning System (GPS) locations were taken of the entry and exit points of the towed video.

### 5.2.2.2. Grab samples

Samples were taken from mixed sediments at depths between 5 and 15 m Below Chart Datum (BCD), using a 0.2 m<sup>2</sup> Van Veen Grab (Figure 5.3c) at sites inside and outside the MPAs. Van Veen grabs have been used in previous studies to sample surface sediments and associated infauna (Cabanellas-Reboredo et al., 2018; Juan et al., 2007; Villnas & Norkko, 2011). Sediment types sampled included maerl, seagrass, sediments characterised by sandmason worm (*Lanice conchilega*), and coarse sediments (with no biogenic characteristics). A grab was deemed unsuccessful if it had less than 10 cm depth of sediment or if it had not closed fully, in which case the sample was discarded and the grab redeployed. For successful grabs, first, a 150 ml subsample was taken for subsequent particle size analysis and organic matter content analysis. Subsamples were stored in a sealed bag in a freezer prior to analysis. Second, one litre of sediment was taken from the grab for infaunal assessment.



Figure 5.3. Field sampling apparatus. a) Side view diagram and b) front view image (right) of the towed video array showing the underwater video system (a), lights (b), lasers (c), buoyancy tubes (d), chain weight (e), cable (f), and tether (g). c) 0.2 m2 Van Veen grab.

# 5.2.2.3. Video processing

Video transects that extended past 100 m were edited to 100 m by calculating the ms<sup>-1</sup> from the submersion time and distance travelled. Videos were analysed in two stages. First, both mobile and sessile taxa were recorded by count for every 100 m transect video to obtain the number of taxa and individuals per transect. This was achieved by viewing the video at normal speed and recording each identifiable taxa visible in the field of view. Secondly, each 100 m transect video was processed in R to extract images at specified time intervals and overlay a digital quadrat onto the image. Intervals were set to one image every second and saved into a separate

folder. Any images that were blurred or angled away from the seabed were discarded and a subset of 10 images were randomly selected from the remaining images. Images were arranged four to a row within the extracted image folder and were selected by choosing the first image of each column in the folder and then the second, third and fourth columns until 10 images had been selected, this was to ensure a representative sample of the transect was analysed. Extracted images were used to record the coverage (%) of structure forming organisms. Additionally, the extracted images were used to obtain tube counts of sandmason worm (*Lanice conchilega*) and peacock worm (*Sabella pavonina*) as they were too numerous to count on the transect videos.

Due to technical issues with the lasers and flying angle of the camera in certain years and replicates, it was not possible to estimate the area inside the quadrat overlay and so no density estimates have been made. Instead, the percentage cover of structure-forming organisms (seagrass, maerl and *Lanice conchilega*) was calculated using the image analysis software imageJ. For seagrass and maerl, this was achieved by measuring the pixel size of the quadrat overlay and then drawing polygons around area covered by seagrass or maerl to determine their pixel area and then calculate the percentage covered of the total quadrat area. Maerl bleaches white when dead and for images in which maerl occurred, this method was repeated separately for live (pink) and dead maerl (white). For *Lanice conchilega* measures were achieved by determining the pixel sizes of individuals across randomly selected still images to obtain an average pixel size from which percentage cover per individual was derived. This percentage cover per individual was used to estimate total proportional cover of *L. conchilega* where they occurred.

### 5.2.2.4. Grab sample processing

The one litre sediment samples were analysed for infaunal diversity within 24 hours of being taken from the seabed, using methods adapted from Whomersley (2014). Each one litre sample was washed through a set of sieves (5 mm and 1 mm mesh size) on board to obtain a >5 mm fraction and a 1-5 mm fraction. Anything smaller than 1mm was washed away as the time frame from this study does not allow for thorough investigation of this size class. Samples were further processed in a laboratory where infauna were 'floated' from the 1-5 mm sediment by flushing with water and then pouring the water with the suspended infaunal species through a sieve with a mesh size of 1 mm. The infauna retained in the sieve were then transferred to a sample pot with 94% ethanol and stored in a fridge to be later identified. Taxa were identified to the highest taxonomic resolution possible using a microscope (6x-50x magnification).

## 5.2.2.5. Maerl samples

An extra step was carried out for maerl samples in order to determine weight of live and dead maerl. Maerl nodules within the >1 mm sediment fraction from the one litre samples were separated from the rest of the sediment and then split into live and dead thallii. Thallii were deemed as being 'live' if they had more than 50% pink colouration. This step was carried out within 24 hours of taking the sample from the seabed as maerl loses its colouration as it dies.

#### 5.2.2.6. Sediment subsamples

The 150 ml sediment subsamples were used for particle size analysis and organic matter content analysis. Particle size analysis was carried out in two steps. First, a small spatula of sediment (~2 g) was taken from the subsample and washed through

a 1 mm sieve connected to a test tube to collect material <1 mm. This was repeated five times and the replicates analysed using the MALVERN HYDRO 2000G (Malvern Instruments Ltd., 2004) to obtain particle size information on sediments less than 1 mm in size. The remaining >1 mm material in the sieve was discarded. Secondly, the rest of the 150 ml sample was washed through another 1 mm sieve into a beaker to separate the >1 mm and <1 mm sediments. Both sediment size classes were transferred to beakers and left to dry overnight at 105°C. Once dry, >1 mm sediment was passed through a series of Wentworth sieves (16, 11.2, 8, 5.6, 4, 2.8, 2 and 1.4 mm), to obtain the weight of each particle size fraction. Laser diffraction and sieve data were combined to produce a dataset of 1000 µm to 0.015 µm particle sizes for each sample. Lastly, the dried <1 mm sediment samples were ground with a pestle and mortar before being transferred to pre-weighed ceramic crucibles and weighed before and after combustion in a muffle furnace overnight (at 550 °C) for four hours to calculate loss on ignition (LOI) to determine the organic matter content of the sample.

## 5.2.2.7. Organic carbon

Organic carbon was calculated from the organic matter (LOI) percentage (percentage of matter lost during combustion) using a conversion based on seagrass estimates by Fourgurean et al. (2012).

$$%C_{org} = 0.40 \times %LOI - 0.21$$

### 5.2.2.8. Particle Size Analysis (PSA)

GRADISTAT © software was used to obtain granulometric and textural groups from PSA data using the Folk and Ward (1957) method (Blott and Wye 2001). PSA was analysed separately for each habitat type (Maerl, Seagrass, Coarse Sediment).

#### 5.2.3. Data analysis

To compare the differences in epibiota (number of taxa, number of individuals and proportion of structure forming organisms), infauna (number of taxa and number of individuals) and sediment particle size and organic carbon content between treatments and locations, Generalised Linear Mixed Effects Models (GLMMs) were used in R (Wickham et al., 2019). Response variables were modelled as a function of Treatment (MPA and Open Control), Location (Ecrehous, Southeast and Minguiers), with Site (n=16 for epibiota and n=11 for infauna) as a random effect. Year was fixed for epibiota (2018, 2019 and 2020) and random for infauna due to only having two years of data (2018 and 2020). Sites were nested within Treatment and Location, with three replicates at each site. Where Treatment, Location or Year were not available, these were dropped from the model. The Southeast infauna data were modelled separately using 2020 data only due to a lack of Open Control samples in 2018. For each response variable, appropriate distributions were used: a Poisson distribution was used for count data (number of taxa and individuals), a beta distribution was used for proportional data (structure-forming organisms), and a Gaussian distribution was used for continuous data (particle size), and guasibinomial distribution was used for organic carbon proportion (Stroup, 2012). When the count data showed overdispersion a negative binomial distribution was used. For structure-forming organisms, a marginal transformation was applied to the proportion data prior to using beta models to fit the assumptions of the model where zeros and ones are not accepted (Smithson & Verkuilen, 2006). Optimal models were determined using the Akaike Information Criterion (AIC) in which an iterative process was used, where predictor variables were sequentially added and then dropped from the model, to determine variable contribution to model fit. Data

manipulation and visualisation was carried out in the statistical program R (R Core Team 2018) using the packages in the 'tidyverse' (Wickham et al. 2019). Reported values in the text are raw mean values ± standard deviation.

Grab samples were further grouped into habitat types which were modelled separately for taxa and abundance. Maerl was modelled with Location (Ecrehous and Southeast) and Treatment (MPA and Open) as fixed effects and Year (2018 and 2020) and Site (n=5) as random effects; seagrass was modelled with Location (Minquiers and Southeast) as a fixed effect with Year (2018 and 2020) and Site (n=3) as random effects; coarse sediment at the Minquiers was modelled with Treatment (MPA and Open) as a fixed effect and Year (2018 and 2020) and Site (n=4) as random effects. Sedimentary habitats characterised by *Lanice conchilega* were not analysed due to a small grab sample size.

Permutational Multivariate Analysis of Variance (PERMANOVA) in PRIMER v7 statistical software package (Clarke & Warwick, 2001), was used to test for species assemblage differences in the epibiota and infauna, between the fixed factors of Location and Treatment, with Site as a random factor. Year was a fixed factor for epibiota but random for infauna due to only having two years of data. Site was nested within Location and Treatment. There were three levels within the factor Location (Ecrehous, Minquiers, Southeast), two levels within Treatment (MPA and Open Control), and three within Year (2018, 2019, 2020) for epibiota and two within Year for infauna (2018 and 2020) and assemblages were pooled at the Site (n=15 for epibiota and n=12 for infauna) level for analysis. To test the species assemblages between Location, Treatment and Year a dummy value of 1 was added to ensure that samples that are similarly devoid of species were considered as being similar

(Clarke et al. 2006; Sheehan et al. 2013). Infauna multivariate data were square root transformed. Bray-Curtis dissimilarity index (Bray & Curtis, 1957) was used to enumerate any difference in assemblages between Location, Treatment and Year. Analyses used 9999 permutations (Anderson & Ter Braak, 2003). Non-metric Multidimensional Scaling ordinations (nMDS) were used to visualise the differences in similarity of assemblage between Locations and Treatments and SIMPER tables were generated to allow for interpretation of the taxa driving the differences in assemblage. SIMPER analyses for infauna species were grouped at the family level, and also analysed separately for each habitat type.

### 5.2.3.1. Maerl weight

Boxplots were used to show the difference in maerl weight (kg) from grab samples between Treatments (MPA and Open Control) at the Ecrehous and Southeast. Generalised Linear Mixed Effects Models (GLMMs) were used to model the response of taxa and abundance as a function of maerl weight (Total, Live and Dead), Treatment (MPA and Open Control) and Location (Ecrehous and Southeast) as fixed effects and Site (n=6) as a random effect. Optimal models were determined using the Akaike Information Criterion (AIC) in which an iterative process was used, where predictor variables were sequentially added and then dropped from the model, to determine variable contribution to model fit. Data manipulation and visualisation was carried out in the statistical program R (R Core Team 2018) using packages in the 'tidyverse' (Wickham et al. 2019).

#### 5.2.3.2. Particle Size Analysis and Organic Carbon habitat differences

Grab sample data was further split into habitat groups (maerl, seagrass, and coarse sediment) to investigate the differences in mean particle size ( $\mu$ m) and proportion of

organic Carbon. Boxplots were used to show the difference in mean particle size ( $\mu$ m) and proportion of organic Carbon from grab samples between Treatment and Location where each habitat occurred. Stacked barplots were used to show the proportion of particle size categories (Mud <63  $\mu$ m, Sand 63-2,000  $\mu$ m, and Gravel 2000-64,000  $\mu$ m) for each Location (Ecrehous, Minquiers, and Southeast) and Treatment (MPA and Open) per habitat.

## 5.3. Results

#### 5.3.1. Epibiota

#### 5.3.1.1. Structure forming organisms

Structure-forming organisms across the study were seagrass (*Zostera marina*), maerl spp. and *Lanice conchilega*. At the Ecrehous location this was maerl alone, and at Southeast the taxa were maerl and seagrass, with seagrass occurring only within the MPA. The structure-forming organisms at the Minquiers were seagrass and *Lanice conchilega*, with a higher proportion of *L. conchilega* outside and a higher proportion of seagrass inside the MPA. The proportion of cover of these combined taxa from extracted towed video images was significantly greater in the MPAs compared to the Open Controls (Figure 5.4, Table 5.1). In the MPAs the proportion of structure-forming organisms declined between 2018 and 2019 but was stable between 2019 and 2020, whereas there was a significant decline in the Open Control in 2020 compared to both 2018 and 2019. The Southeast consistently had the highest proportions of structural organisms, both inside the MPAs and in Open Controls, though this was only significant in comparison to the Ecrehous (Figure 5.4, Table 5.1).



Figure 5.4. Mean proportion of structure forming organisms for Year, Location and Treatment. Error bars show means and standard errors of mean proportions predicted using GLMMs with Treatment (MPA and Open Control), Location (Ecrehous, Southeast and Minquiers) and Year (2018, 2019 and 2020) as fixed effects and Site (n=15) as a random effect.

Table 5.1. General Linear Mixed Effects Model outputs for towed video proportions of structure forming organisms as a function of Treatment, Location, and Year with Site as a random effect. Figures in bold denote a significant result.

Terms	Estimate	Std. Error	z value	р				
Proportion of Structure Forming Species								
Intercept	-0.734	0.246	-2.98	$0.0029^{**}$				
MPA - Open	-0.678	0.267	-2.54	0.011*				
2018 - 2019	-0.714	0.079	-9.01	$< 0.0001^{***}$				
2018 - 2020	-0.630	0.085	-7.42	$< 0.0001^{***}$				
2019 - 2020	0.083	0.079	1.05	0.29				
Ecrehous - Southeast	0.884	0.360	2.45	$0.014^{*}$				
Ecrehous - Minquiers	0.413	0.282	1.47	0.14				
Minquiers - Southeast	0.470	0.336	1.40	0.16				
MPA - Open*2018 - 2019	0.226	0.136	1.67	0.096				
MPA - Open*2018 - 2020	-0.425	0.137	-3.11	$0.0019^{**}$				
MPA - Open*2019 - 2020	-0.651	0.123	-5.30	$< 0.0001^{***}$				

Table 5.1. General Linear Mixed Effects Model outputs for towed video proportions of structure forming organisms as a function of Treatment, Location, and Year with Site as a random effect. Figures in bold denote a significant result

## 5.3.1.2. Number of taxa and individuals

The number of taxa observed was significantly greater in the MPA Treatment compared to the Open Controls across all locations in 2018, except for the Southeast where there was no Open Control. In other years, the number of taxa in the MPAs was similar to the Open Controls (Figure 5.5 and Table 5.2). Across all Treatments and Locations there was a significant Year effect that showed the number of taxa in 2019 to be significantly lower compared to both 2018 and 2020. Also, there was a location effect which showed a greater diversity at the Ecrehous compared to both the Minquiers and Southeast in all years (Figure 5.5 and Table 5.2). Between 2018 and 2020, there was a significant increase in the number of taxa in the Open Controls compared to the MPA which did not change.

The number of individuals in the Open Controls was greater than in the MPAs in 2020 but showed similar values across Treatments in 2019. In 2018, there were greater numbers of individuals in the MPAs compared to Open Controls across all locations, except for the Southeast where there was no Open Control (Figure 5.5, Table 5.2).



Figure 5.5. Mean numbers of a) taxa and b) individuals per Treatment and Year at each Location. Error bars show fitted GLMM means and standard errors of individual abundance predicted using GLMMs with Treatment (MPA and Open Control), Location (Ecrehous, Southeast and Minquiers) and Year (2019 and 2020) as fixed effects and Site (n=15) as a random effect.

Table 5.2. General Linear Mixed Effects Model outputs for towed video Taxa and Abundance as a function of Treatment, Year, and Location with Site as a random effect. Figures in bold denote a significant result.

Terms	Estimate	Std. Error	z value	р
Number of Taxa				
Intercept	1.210	0.183	6.610	$< 0.0001^{***}$
MPA - Open	-0.487	0.269	-1.810	0.071
2018 - 2019	-0.473	0.175	-2.700	$0.007^{**}$
2018 - 2020	-0.026	0.161	-0.164	0.87
2019 - 2020	0.447	0.170	2.630	$0.0085^{**}$
Ecrehous - Minquiers	-0.450	0.189	-2.380	0.017*
Ecrehous - Southeast	-0.560	0.245	-2.280	$0.022^{*}$
Minquiers - Southeast	-0.110	0.236	-0.466	0.64
MPA - Open*2018 - 2019	0.473	0.304	1.560	0.12
MPA - Open*2018 - 2020	0.683	0.277	2.470	$0.014^{*}$
MPA - Open*2019 - 2020	0.210	0.254	0.829	0.41
Number of Individuals				
Intercept	2.060	0.270	7.620	$< 0.0001^{***}$
MPA - Open	-0.811	0.389	-2.080	$0.037^{*}$
2018 - 2019	-0.486	0.296	-1.640	0.1
2018 - 2020	-0.404	0.295	-1.370	0.17
2019 - 2020	0.082	0.283	0.289	0.77
Ecrehous - Minquiers	-0.514	0.226	-2.270	$0.023^{*}$
Ecrehous - Southeast	-0.837	0.288	-2.910	$0.0036^{**}$
Minquiers - Southeast	-0.322	0.275	-1.170	0.24
MPA - Open*2018 - 2019	0.581	0.476	1.220	0.22
MPA - Open*2018 - 2020	1.360	0.468	2.900	$0.0037^{**}$
MPA - Open *2019 - 2020	0.778	0.412	1.890	0.059

### 5.3.1.3. Assemblage

There was a significant Treatment x Location effect, Year x Treatment effect, and Year x Location effect on species assemblage (Table 5.3), meaning that the assemblages differed each year for both treatment and location, whilst also differing between treatments at each location. Pairwise tests revealed no significant difference in assemblage between Treatment for both the Ecrehous and Minquiers Locations in 2018 (Table 5.4). In 2020 an assemblage difference between the MPA and the Open Controls was observed for the Minquiers only, and there was a marginally non-significant difference (p = 0.055) in 2019 for the Ecrehous (Figure 5.6). SIMPER results demonstrated that the species driving the assemblage difference at the Ecrehous in 2019 was the greater average abundance of *Aplysia* spp., *Maja brachydactyla, Pecten maximus* and juvenile fish inside the MPA, compared to lower average abundances outside, and a greater average abundance of *Crepidula fornicata* outside compared to inside the MPA (Appendix Table C.1). The difference in species assemblage at the Minquiers in 2020 was driven by a greater abundance of *Cereus pedunculatus*, *M. brachydactyla* and *Pagarus* spp. outside of the MPA, compared to greater average abundance of *Actinopterygii* spp. inside. Overall, there were fewer species contributing to the average abundance for all Treatments and locations in 2019 compared to other years (Appendix Table C.1).

Table 5.3. PERMANOVA of assemblage based on Bray-Curtis dissimilarities for the interactions of Year (Yr), Treatment (Tr) and Location (Lo). No transformation was applied to the data. Figures in bold denote a significant result.

PERMANOVA							
Source	df	$\mathbf{SS}$	MS	Pseudo-F	P(perm)	perms	
Ye	2	9897.0	4948.5	2.556	0.001**	9908	
$\mathrm{Tr}$	1	2443.6	2443.6	1.262	0.242	9924	
Lo	2	12271.0	6135.7	3.170	$0.000^{***}$	9916	
YexTr	2	7005.2	3502.6	1.809	$0.026^{*}$	9914	
YexLo	4	15967.0	3991.7	2.062	$0.001^{**}$	9879	
TrxLo	2	7312.9	3656.5	1.889	$0.019^{*}$	9904	
YexTrxLo**	3	5755.5	1918.5	0.991	0.464	9888	
Pooled	161	311650.0	1935.7				
Total	177	383320.0					

Table 5.4. Pairwise tests on the interaction Year x Treatment for each Location. Note there were no Open Control replicates for the Southeast in 2018 and so no comparison has been made. Figures in bold denote a significant result.

	2018		2019			2020			
Groups	t	P(perm)	Unique perms	t	P(perm)	Unique perms	t	P(perm)	Unique perms
Minquiers MPA, Open Control	1.092	0.292	5351	1.226	0.188	9892	1.683	0.013	9943
Ecrehous MPA, Open Control	1.242	0.145	2194	1.385	0.055	4657	1.094	0.295	9585
Southeast MPA, Open Control	-	-	-	0.875	0.334	22	1.257	0.125	2159



Figure 5.6. nMDS showing the dissimilarities in assemblage composition between Treatments and Locations. Lines show the trajectory of change between years from 2018 to 2020.

# 5.3.2. Sediment properties and Infauna

# 5.3.2.1. Particle Size Analysis and Organic Carbon

The mean particle size ( $\mu$ m) was not significantly different between Treatment at any location (Figure 5.7a, Table 5.5). A Treatment effect on the proportion of organic carbon was observed at the Minquiers only, with a lower proportion of organic carbon in the MPA (0.0032 ± 0.0008) compared to the Open Control (0.007 ± 0.003) (Figure 5.7b, Table 5.5).



Figure 5.7. Mean particle size (mm) and b) Proportion of Organic Carbon for each Location and Habitat. Error bars show fitted GLMM means and standard errors predicted using GLMMs with a Poisson distribution for mean particle size and a quasibinomial distribution for Organic Carbon, with Treatment (MPA and Open) as a fixed effect and Year (2018 and 2020) as a random effect.

Table 5.5. General Linear Mixed Effects Model outputs for grab sample mean grain size  $(\mu m)$  and organic carbon proportion as a function of Location and Treatment, with Year and

Site as random effects for the Ecrehous and Minquiers and Site as a random effect for the Southeast. Figures in bold denote a significant result.

Terms	Estimate	Std. Error	z value	р			
Mean Grain Size Ecrehous and Minquiers 2018 and 2020							
Intercept	2110	469	4.490	0.01*			
MPA - Open	-645	808	-0.798	0.47			
Ecrehous - Minquiers	-1180	607	-1.940	0.12			
MPA - Open*Ecrehous - Minquiers	399	1010	0.395	0.71			
Mean Grain Size Southeast 2020							
Intercept	1040	516	2.020	0.19			
MPA - Open	1240	1020	1.220	0.36			
Terms	Estimate	Std. Error	t value	р			
Organic Carbon Proportion Ecrehous	and Minq	uiers 2018 a	nd 2020				
Intercept	-4.750	0.090	-52.900	<0.0001***			
MPA - Open	-0.231	0.168	-1.370	0.18			
Ecrehous - Minquiers	-0.984	0.149	-6.590	<0.0001***			
MPA - Open*Ecrehous - Minquiers	1.020	0.229	4.470	<0.0001***			
Organic Carbon Proportion Southeast 2020							
Intercept	-4.070	0.171	-23.800	<0.0001***			
MPA - Open	0.124	0.241	0.513	0.61			

## 5.3.3. Infauna

Overall, 271 taxa and 3,384 individuals were found across the 80 grab samples. Infaunal taxa belonged to Arthropoda (35.3%), Annelida (33.5%), Mollusca (17.8%), Nematoda (6%), and Cnidaria (2.2%). The remaining 5.2% consisted of Echinodermata, Nemertea, Sipuncula, Bryozoa, Priapulida, Chordata, Malacostraca, and Porifera. Dominant species in both Treatments included the arthropods *Apseudes latreillii* and *Apseudes talpa*, the bivalve molluscs of *Nucula* spp., Nematoda spp., and the annelid *Lumbrinereis latrielli*. The arthropod *Ampelisca brevicornis* was dominant in the MPAs only. A complete list of taxa recorded from grab samples can be found in Appendix Table C.2.

## 5.3.3.1. Number of taxa and individuals

While there were greater numbers of taxa inside the MPAs at the Ecrehous (14.3  $\pm$  4.5) and Minquiers (11.2  $\pm$  4.4), compared to Open Controls (10.8  $\pm$  5.3 and 9.3  $\pm$  5.2, respectively), this was not significant (Figure 5.8a, Table 5.6). There was no

effect of Treatment on the number of individuals at either the Ecrehous (MPA 23.8  $\pm$  11.5, Open Control 18.7  $\pm$  13.5) or Minquiers (MPA 38.4  $\pm$  36.9, Open Control 22.8  $\pm$  20.8) (Figure 5.8b, Table 5.6). At the Southeast, both the number of taxa and individuals were significantly greater in the MPAs (taxa 26.8  $\pm$  7.3, Individuals 99.2  $\pm$  55.5) than Open Controls (taxa 15.1  $\pm$  5.2, Individuals 35  $\pm$  16.2) (Figure 5.8a and 5.8b, Table 5.6). Both the number of taxa and individuals were greater at the Southeast compared to the Ecrehous and Minquiers, both inside the MPAs and in Open Controls.



Figure 5.8. Total numbers of a) infaunal taxa and b) infaunal individuals for Location and Treatment. Error bars show fitted GLMM means and standard errors of individual abundance predicted using GLMMs with Treatment and Location as fixed effects and Year and Site as random effects. There was no random effect of year included for the Southeast due to a lack of 2018 data.

Table 5.6. General Linear Mixed Effects Model outputs for infaunal grab taxa (>1 mm) as a function of Location and Treatment with Year and Site as random effects for the Ecrehous and Minquiers and Site as a random factor for the Southeast. Figures in bold denote a significant result.

Table 5.6. General Linear Mixed Effects Model outputs for infaunal grab taxa (>1 mm) as a function of Location and Treatment with Year and Site as random effects for the Ecrehous and Minquiers and Site as a random factor for the Southeast. Figures in bold denote a significant result.

Terms	Estimate	Std. Error	z value	р			
Ecrehous and Minquiers Number of Taxa 2018 and 2020							
Intercept	2.710	0.068	39.900	$< 0.0001^{***}$			
MPA - Open	-0.149	0.088	-1.690	0.091			
Ecrehous - Minquiers	-0.266	0.084	-3.180	$0.0015^{**}$			
Southeast Number of	Taxa 202	0					
Intercept	3.310	0.060	55.000	$< 0.0001^{***}$			
MPA - Open	-0.528	0.102	-5.150	$< 0.0001^{***}$			
Ecrehous and Minqui	ers Numb	er of Indivi	iduals 20	18 and 2020			
Intercept	3.290	0.402	8.190	$< 0.0001^{***}$			
MPA - Open	-0.052	0.373	-0.139	0.89			
Ecrehous - Minquiers	-0.020	0.373	-0.053	0.96			
Southeast Number of Individuals 2020							
Intercept	4.680	0.318	14.700	$< 0.0001^{***}$			
MPA - Open	-0.977	0.463	-2.110	0.035*			

# 5.3.3.2. Assemblage

Species assemblage differed between the Ecrehous and Minquiers, but not between

Treatments (Table 5.7). The species assemblage in the MPA at the Southeast was

significantly different to the Open Controls (Table 5.7, Figure 5.9).

Table 5.7. PERMANOVA of assemblage based on Bray-Curtis dissimilarities for the interactions of Location and Treatment. Year was a random factor for the Ecrehous and Minquiers Locations. The Southeast Location was only tested in 2020 due to a lack of Open
Control replicates in 2018. Data were fourth root transformed. Figures in bold denote a

significant result.

Table 5.7. PERMANOVA of assemblage based on Bray-Curtis dissimilarities for the interactions of Location and Treatment. Year was a random factor for the Ecrehous and Minquiers Locations. The Southeast Location was only tested in 2020 due to a lack of Open Control replicates in 2018. Data were fourth root transformed. Figures in bold denote a significant result.

PERMANOVA main test						
Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms
Ecrehous and Minquiers 2018 and 2020						
Lo	1	11550.0	11550.0	4.025	$< 0.0001^{***}$	9900
Tr	1	3528.2	3528.2	1.229	0.207	9902
LoxTr	1	2141.4	2141.4	0.746	0.783	9909
Pooled	44	126280.0	2869.9			
Total	47	146070.0				
Southeast 2020						
Tr	1	8340.0	8340.0	4.330	$0.001^{**}$	9428
Res	17	32742.0	1926.0			
Total	18	41082.0				



Figure 5.9. nMDS showing the dissimilarities in assemblage composition between Treatments (MPA and Open Control) at the Southeast.

5.3.4. Infaunal assemblage and sediment property responses by habitat

### 5.3.4.1. Maerl

Maerl habitat was present at the Ecrehous and Southeast only and samples contained a mixture of maerl, sand, gravel, mud, and shell material. Maerl particle size was marginally larger within the MPA (MPA =  $2032 \pm 1170$ , Open Control =  $1464 \pm 561$ ) at the Ecrehous but smaller within the MPA at the Southeast (MPA =  $1330 \pm 934$ , Open Control =  $2284 \pm 1051$ , Appendix Figure C.1a). The smaller mean particle size inside the MPA at the Southeast can be explained by the relatively higher proportions of mud (<63 µm) and lower proportions of gravel (2000-64,000 µm) compared to the Open Control (Appendix Figure C.2). Conversely, at the Ecrehous, there is a very small proportion of particles in the size range of mud in both the MPA and Open Control, with the difference in mean particle sized owed to the higher proportion of particles in the size range of gravel inside the MPA. The proportion of organic carbon was similar between treatments at both locations, but was overall greater at the Southeast (MPA 0.02 ± 0.007; Open Control 0.02 ± 0.005) compared to the Ecrehous (MPA 0.008 ± 0.003; Open Control 0.006 ± 0.002).

There was a difference in the weight of live and dead maerI at the Southeast, with a greater weight of live maerI ( $0.19 \pm 0.15$  kg) inside the MPA compared to the Open Control ( $0.08 \pm 0.06$  kg) and a greater weight of dead maerI in the Open Control ( $0.36 \pm 0.18$  kg) compared to the MPA ( $0.16 \pm 0.12$  kg; Appendix Figure C.3a-b). At the Ecrehous there were similar weights of both live (MPA =  $0.07 \pm 0.05$ , Open Control =  $0.07 \pm 0.07$ ) and dead maerI between treatments (MPA =  $0.1 \pm 0.09$ , Open Control =  $0.12 \pm 0.14$ ; Appendix Figure C.3a-b).

There was a positive relationship between live maerl weight and both the number of taxa and the number of individuals in the MPAs at both locations, although this was only significant for the number of individuals (Fig 5.10a-b, Appendix Table C.3). In the Open Control at the Ecrehous, both the number of taxa and individuals responded slightly negatively with increasing weight of live maerl, while at the Southeast there was a negative relationship with the number of taxa and a positive relationship with the number of individuals (Fig 5.10a-b, Appendix Table C.3). There was a positive relationship between the weight of dead maerl and the number of taxa at both the Southeast and the Ecrehous inside the MPAs but only a positive relationship between dead maerl weight and the number of individuals in the MPA at the Ecrehous (Fig 5.10c-d, Appendix Table C.3). A negative relationship was observed with the weight of dead maerl for both the number of taxa and individuals in the Open Controls at both Locations (Fig 5.10c-d, Appendix Table C.3).



Figure 5.10. Numbers of taxa in comparison to a) Live maerl weight, and c) Dead maerl weight in kg per litre, and number of individuals in comparison to b) Live maerl weight, and d) Dead maerl weight in kg per litre for Location and Treatment. Regression lines were predicted using GLMMs with Maerl weight (Total, Live or Dead), Treatment (MPA and Open Control) and Location (Ecrehous and Southeast) as fixed effects and Site (n=6) as a random effect. Shading around the regression line shows standard error.

At the Southeast, both the number of taxa and number of individuals on maerl was greater in the MPAs (taxa 29.4  $\pm$  8.6, Individuals 98.4  $\pm$  73.5) compared to Open Controls (taxa 15.1  $\pm$  5.2, Individuals 35  $\pm$  16.2; Figure 5.11 a-b, Appendix Table C.4). Although not significant, the number of taxa and individuals on maerl at the

Ecrehous were greater inside the MPAs (taxa  $14.3 \pm 4.5$ , Individuals  $23.8 \pm 11.5$ ) compared to the Open Controls (taxa  $10.8 \pm 5.3$ , Individuals  $18.7 \pm 13.5$ ; Fig 5.11 c-d, Appendix Table C.4). There was a significant Treatment x Location effect on the species assemblages associated with maerl habitat (Appendix Table C.5). Pairwise tests showed the assemblages to be different between the MPA and Open Controls at the Southeast, with more closely grouped assemblages in the Open Control (Appendix Table C.5, Figure C.4). This difference in species assemblage was driven by high average abundances of Veneridae, Maldanidae, Nuculidae, Terebellidae, Apseudidae, Sertulariidae, and Caprellidae in the MPA, compared to relatively high average abundances of Eunicidae, Glyceridae, Lumbrineridae, and Polynoidae in the Open Control (Appendix Table C.5).



Figure 5.11. Total numbers of a) taxa and b) individuals in maerl for Treatment at the Southeast, and c) taxa and d) individuals for Treatment at the Ecrehous. Error bars show fitted GLMM means and standard errors of individual abundance predicted using GLMMs

with Treatment (MPA and Open Control) as fixed effects and Site (n=2) as a random effect for the Southeast and Site (n=3) and Year (2018 and 2020) as random effects for the Ecrehous. Numbers of e) taxa and f) individuals on seagrass habitat within the MPAs. Error bars show fitted GLMM means and standard errors predicted using GLMMs with Location (Minquiers and Southeast) as fixed effects and Year (2018 and 2020) as a random effect. N.B. there are no seagrass beds large enough outside of the MPAs to serve as Open Controls. Numbers of g) taxa and h) individuals on sediment habitat at the Minquiers. Error bars show fitted GLMM means and standard errors predicted using GLMMs with Treatment (MPA and Open) as a fixed effect and Year (2018 and 2020) and Site (n=4) as random effects.

#### 5.3.4.2. Seagrass

Seagrass particle size differed between the two MPA locations in which seagrass occurs (Minquiers and Southeast; Appendix Figure C.1a). This was due to the higher proportion of particles in the size range of mud at the Southeast compared to the Minquiers (Appendix Figure C.5). The Minquiers also had a greater proportion of gravel than the Southeast. The proportion of organic carbon was greater at the Southeast ( $0.008 \pm 0.002$ ) where the particle size was smaller, compared to the Minquiers ( $0.003 \pm 0.0003$ ; Appendix Figure C.1b).

Both the number of taxa and individuals were significantly greater in the Southeast MPA (taxa  $24.2 \pm 5.5$ , Individuals  $100 \pm 39.1$ ) compared to the Minquiers MPA (taxa  $12.1 \pm 3.4$ , Individuals  $30.3 \pm 15.4$ ; Figure 5.11 e-f, Appendix Table C.7). Species assemblages also differed between the Southeast and the Minquiers (Appendix Table C.8). The families driving the seagrass assemblages at the Southeast were similar to the Minquiers but in greater average abundances. The Southeast had much greater average abundances of Maldanidae, Ampeliscidae and Glyceridae compared to the Minquiers. Sabellidae was present at the Southeast but absent at the Minquiers (Appendix Table C.9).

#### 5.3.4.3. Coarse sediment

Coarse sediment was of a larger particle size within the MPA (1887.81  $\pm$  417.23  $\mu$ m) at the Minquiers compared to the Open Control (846.75  $\pm$  133.45  $\mu$ m) (Appendix Figure C.1a). Overall, coarse sediment had higher proportions of particles within the size range of Gravel inside the MPA compared to the Open Control (Appendix Figure C.7), whereas there were higher proportions of sand sized particles in the Open Control. The proportion of mud was low in both Treatments. The proportion of organic carbon was marginally lower in the MPA (0.003  $\pm$  0.001) compared to the Open Control (0.005  $\pm$  0.002; Appendix Figure C.1b).

The number of taxa and individuals on coarse sediment was greater in the MPA (taxa  $11 \pm 3.96$ , Individuals  $54 \pm 44.8$ ) compared to the Open Control (taxa  $7.86 \pm 3.24$ , Individuals  $19.71 \pm 15.88$ ) at the Minquiers (Figure 5.11 g-h, Appendix Table C.10), but this was only significant for the number of individuals (Appendix Table C.10). Species assemblages did not differ between the MPA and Open Control (Appendix Table C.11). The greatest contributing family to average abundance in both the MPA and the Open Control was Apseudidae (Appendix Table C.12).

#### 5.4. Discussion

The MPAs in Jersey were designated to protect whole areas of benthic habitat from bottom-towed fishing, following a whole-site approach (Davies et al., 2021a; Solandt et al., 2020), to allow the recovery of sedimentary biota. Following the exclusion of bottom-towed fishing the response in the benthic biota varied between the MPAs, with changes noticeable in the infaunal assemblages, but with little change observed in the structure-forming organisms and epibiota assemblages. The proportional coverage of structure forming organisms was greater in the MPAs compared to the

Open Controls in all years, with a decline observed over time in the Open Controls. As bottom-towed fishing can remove structure forming ecosystem engineers from the seabed (Coleman & Williams, 2002; Thurstan et al., 2013), this was expected. Contrary to expectations, there was no increase observed in the proportional cover of structure forming organisms inside the MPAs. It may be that three years is not sufficient for a change to occur in coverage of the organisms, especially when considering maerl as it grows at a rate of 1 mm a year (Wilson et al., 2004).

#### 5.4.1. Epibiota

The presence of structure forming organisms influences the distribution and abundance of benthic species (Wright & Jones, 2004), as has been documented for Lanice conchilega (Callaway et al., 2010; Rabaut et al., 2007), maerl (Barbera et al., 2003; Riosmena-rodríguez, 2017) and seagrass beds (Githaiga et al., 2019; Jackson et al., 2002). It was expected that sites with greater coverage of these structure forming organisms would have greater numbers of epibiota taxa and individuals associated with them. However, there were only greater numbers of taxa and individuals in the MPAs compared to Open Controls in 2018, while all other years were similar. This may have been expected at the Ecrehous and Minguiers as three years of protection may not be a sufficient time period for noticeable increases in the epibiota. However, it is surprising that the Southeast MPA did not show an increase in epibiota taxa or individuals, nor any differences in species assemblage between treatments, as this is the longest established MPA (six years) and it may have been expected that a different assemblage would have begun to develop (Sheehan, Stevens, et al., 2013). Recovery trajectories are relatively unknown due to shifting baselines, and realistically, recovery may take decades (Davies et al., 2021a; Jones & Schmitz, 2009), and slow growing habitats may take longer to

recover (Hall-Spencer & Moore, 2000), such as maerl that is found in the Southeast MPA.

Where it was present, seagrass cover often reached 100%, meaning any species that may have been beneath the seagrass canopy (Hori et al., 2009) were not visible on the towed video footage, potentially resulting in an underestimation of the epibiota. Where maerl is concerned, aggregations of live and dead maerl support a high infaunal biodiversity (Riosmena-rodríguez, 2017), particularly for bivalves (Hall-Spencer et al., 2003). Therefore, epibiotic assemblages may not be the most appropriate indicators to use for seagrass and maerl habitats as both seagrass and maerl beds harbour biodiverse infaunal species assemblages that cannot be effectively assessed using towed video methods. Epifaunal biodiversity of habitats such as sediments characterised by *L. conchilega* were readily observable using towed video methods, as the structure created by the worms was not tall enough to obscure the seabed, with taxa such as anemones and benthic fish visible amongst the growth of *L. conchilega*.

There were no clear (multivariate) differences in epibiota species assemblages between treatments, but there were some species of interest that differed in their average abundance. The inconsistent but high average abundance of *Crepidula fornicata* (American slipper limpet) where it occurred in the Open Controls suggests that there is a patchy distribution of this species. The spread of *C. fornicata* is thought to be facilitated by bottom-towed fishing through dispersal of individuals and creation of furrows in the seabed that favour the settlement of *C. fornicata* (Blanchard, 2009), which may be a contributing factor to the greater average abundance in the Open Controls. *Pecten maximus* generally contributed more to

assemblages within the MPAs at the Ecrehous and Southeast compared to Open Controls, but was not a major contributor to average abundance at either treatment at the Minquiers. The Ecrehous and Southeast both have extensive maerl beds which is a primary habitat for *P. maximus* (Hall-Spencer et al., 2003), whereas no maerl was observed in the survey sites at the Minquiers. The greater average abundance within the MPAs could be related to reduced fishing pressure from bottom-towed fishing, as observed in studies that have assessed the response of scallop populations to protection measures (Beukers-Stewart et al., 2005; Howarth et al., 2011). Only in 2020 was a greater average abundance of scallops observed in the MPAs where scallop dredging is not uniform, resulting in patches of high and low scallop abundance depending on recent activity.

### 5.4.2. Sediment properties

Sediment composition is thought to influence the distribution of infaunal assemblages and the organic carbon content of sediments (Bolam et al., 2008; Somerfield et al., 2019), and the proportion of mud in the sediment was the main factor influencing organic carbon proportion both inside and outside the MPAs. Bottom-towed fishing has been shown to decrease the proportion of fine sediments (Nayar et al., 2007), with implications for organic carbon content. However, mean particle size and the proportion of organic carbon were primarily associated with Location and Habitat, with little influence of Treatment across the three MPAs. The MPAs are relatively young and the time since protection may not have been sufficient to detect a change in organic carbon resulting from the exclusion of bottom-towed fishing. Periodical sampling of the sediments in the future would be necessary to

determine whether organic carbon would eventually increase in undisturbed sediments in the MPAs relative to those in the Open Controls.

#### 5.4.3. Infauna

Infaunal communities are relatively limited in their mobility and therefore can be used as indicators of localised environmental change (Sweat et al., 2020). Differences in infaunal taxa and individuals were strongest within the older Southeast MPA, with greater numbers inside the MPA and a species assemblage that was distinct from the Open Control. While greater numbers of taxa and infauna were observed at the younger Ecrehous and Minquiers MPAs, this was not yet significant. Age is reported as a key factor in MPA success (Edgar et al., 2014) as marine habitats continue to recover for several years, if not decades, after their initial protection (Jones & Schmitz, 2009). This may explain the observed differences in diversity between old and new MPAs as the Southeast has had a longer period to develop more diverse and abundant infaunal assemblages.

The greater numbers of infaunal taxa and individuals at the Southeast Open Control compared to both treatments (MPA and Open Control) at the Ecrehous and Minquiers was unexpected. This instead may be explained by variations in fishing effort in Jersey's territorial waters, as the Southeast MPA and it's Open Controls are located within Jersey's three-mile exclusive fishing zone, while the two other MPAs and Open Controls are within the Bay of Granville that has shared fishing rights with France (Chambers et al., 2020) and consequently additional fishing effort from French bottom-towed fishing vessels (Marine Resources, 2019). However, greater accuracy of spatial fishing effort would be needed to investigate this.

#### 5.4.4. Maerl habitat

Finer sediments typically have a greater organic carbon content (Nayar et al., 2007; Somerfield et al., 2019). At the Southeast, the mean particle size of maerl habitat was smaller in the MPA compared to the Open Control, but contrary to the literature this did not equate to a greater proportion of organic carbon. However, the proportion of organic carbon within sediments at the Southeast MPA had the greatest range, suggesting that organic carbon is patchily distributed. At the Ecrehous, both the mean particle size and proportion of organic carbon were similar between treatments. The overall lower proportion of organic carbon at the Ecrehous compared to the Southeast in both treatments will in part be driven by the lower proportion of mud at the Ecrehous. Fine sediments, such as mud, are lost through bottom-towed fishing (Cooper et al., 2011) and the younger age of the MPA at the Ecrehous may mean that finer sediment has not had time to accumulate; therefore changes in organic carbon will not have been detected yet.

As highlighted earlier, maerl supports a diverse infaunal species assemblage (Riosmena-rodríguez, 2017), but the fragile nature of maerl makes it highly susceptible to damage from bottom-towed fishing and subsequent recovery is slow (Hall-Spencer & Moore, 2000). The greater number of taxa and individuals on maerl in the MPAs compared to Open Controls was only significant for the Southeast where the maerl has been protected for a longer period of time. The same was observed for the species assemblage, which was only different between treatments at the Southeast. Aggregations of maerl supports a high biomass and diversity of mollusc species (Hall-Spencer et al., 2003) and the observed greater average abundance of the bivalve families Veneridae and Nuculidae in the MPA at the Southeast supports this. The weight of live maerl was also greater in the Southeast

MPA, which may be a driving factor in the greater number of taxa and individuals as live maerl creates a biofilm that provides an environmental cue for scallop during their settlement process (Kamenos et al., 2004a). This response to biofilms has also been observed in other species and habitats (Whalan & Webster, 2014; Zhao et al., 2020), highlighting the importance of undisturbed habitat in which biofilms can develop. This is further supported by the positive relationship of live maerl weight and the number of taxa and individuals in the grab samples.

#### 5.4.5. Seagrass habitat

The particle size of seagrass sediments are indicative of the level of exposure experience by the seagrass beds (Lima et al., 2019) and results from PSA revealed the sediment of the seagrass beds at the Southeast to have a higher proportion of finer, mud sized particles compared to the Minquiers seagrass. Sediment composition plays a part in the organic carbon content of sediments characterised by seagrass species such as *Zostera* spp. where the content of mud influences the accumulation of organic carbon (Lima et al., 2019). This was observed at the Southeast where the proportion of organic carbon in seagrass sediments was greatest where the mean particle size was smallest.

A treatment effect could not be assessed for seagrass as very little seagrass exists outside of the MPAs. Instead, the time since protection has been investigated which revealed that the seagrass in the older Southeast MPA had a greater number of infaunal taxa and individuals than the seagrass within the younger Minquiers MPA. Sediment composition may be the key driver of this difference as there was a greater mud content in the Southeast. Mud is typically more biodiverse than coarse sediments because of the greater availability of organic matter that serves as a food

source for many infaunal species (Lundquist et al., 2018; Mcarthur et al., 2010). While disturbance from bottom-towed fishing reduces fine sediment content (Cooper et al., 2011), the tidal regimes at the Minquiers may be the main limiting factor in the retention of mud sized particles (Mazarrasa et al., 2018). However, the seagrass grab samples at the Minquiers had a greater number of taxa and individuals than the coarse sediment samples from the Minquiers, highlighting the localised importance of seagrass in promoting biodiversity within the MPA.

#### 5.4.6. Coarse sediment habitat

The lower proportion of organic carbon of coarse sediments in the MPA at the Minquiers will primarily have been driven by the larger particle size in the MPA compared to the Open Control. Bare sediments are thought to be less diverse than those characterised by structure forming organisms (Bruschetti, 2019; Githaiga et al., 2019) and while this was the case at the Minquiers, the number of taxa and individuals on coarse sediment were greater in the MPA compared to Open Controls, highlighting the potential of these habitats to contribute to the recovery of biodiversity within MPAs.

The results from this study suggest that whole-site approach MPAs will be able to support the recovery of biodiversity to a greater extent than feature based MPAs. While habitats such as seagrass and maerl would typically be included as protected features within a MPA, sand mason worm communities and bare sediment would not. This study provides evidence of greater infaunal diversity on protected bare sediments compared to those in areas open to bottom-towed fishing. By protecting sedimentary habitats in addition to reef, the overall species richness can be expected to increase (Davies et al., 2021a). Protecting whole-areas of seabed in an

ecosystem based approach has also been shown to increase the functional diversity of benthic communities (Davies et al., 2021b), which may lead to improved ecosystem services.

The timeframe for recovery of biodiversity is poorly understood due to a disparity of whole-site approach MPAs having been established or monitored over sufficient timeframes to determine when benthic communities will reach a climax community. This is supported by research from Lyme Bay where species richness and abundance have continued to increase 11 years after protection (Davies et al., 2021a), with a plateau in biodiversity yet to be reached. The type of sedimentary habitats within a MPA may be related to the expected recovery time. After just three years, the number of taxa and individuals was greater on protected bare sediments as opposed to those that were open to bottom-towed fishing, whereas greater numbers of taxa and individuals were only observed on maerl habitat that had been protected for six years. This current study has shown that some recovery can be observed over short time-scales (three years) following the removal of anthropogenic pressure, as observed in early years of Lyme Bay's protection (Sheehan et al., 2013), but further years of monitoring are needed to improve understanding of time scales for full recovery (a plateau of diversity, abundance and coverage of structure forming organisms). Expansion of the MPAs in Jersey to include larger areas of sedimentary habitat may allow for further recovery of benthic communities. Sedimentary habitats should be considered in the global network of MPAs in a move away from the current, fragmented, feature-based approach to MPA management.

#### 5.5. Conclusion

This study has shown that the response of benthic communities associated with marine sediments following protection from bottom-towed fishing is varied, with habitat type bearing a significant influence on recovery. There was little change in the proportion of organic carbon in the sediments following the exclusion of bottomtowed fishing and it may take several more years of monitoring to determine if and when organic carbon can be expected to accumulate. Structure forming organisms create their own habitats that harbour more diverse species assemblages than bare sediment and compared to habitats that are subject to pressure from bottom-towed fishing. However, protected bare sediments also showed greater numbers of taxa and individuals than Open controls, highlighting the importance of protecting these habitats to improve the ecological effectiveness of MPAs. Response to treatment was strongest in the infaunal taxa and abundance, particularly in the older MPA. Through the protection of these food web supporting species, it can be expected that other, higher trophic level species, will benefit from the greater availability of food. This has important implications for future management of MPAs and evidences the need for whole areas of seabed to be protected from bottom-towed fishing to enable the recovery of benthic communities that support the functioning of ecosystems.

 Chapter 6 – The response of commercial crustacean populations to the removal of bottom-towed fishing within Marine Protected Areas in Jersey, Channel Islands, UK

### Author Contributions

Samantha Blampied was responsible for the conception and design of this research. She organised the fieldwork, undertook the analysis, interpretation of data and writing of the chapter. Francis Binney assisted in fieldwork logistics and acquisition of data from government sources. All supervisors contributed to the critical revision of manuscripts and have given their approval for inclusion in this thesis.

Author	Institution	Contribution
Samantha Blampied	University of Plymouth	70%
Sian Rees	University of Plymouth	5%
Martin Attrill	University of Plymouth	5%
Francis Binney	Government of Jersey	10%
Emma Sheehan	University of Plymouth	10%

#### Abstract

Marine Protected Areas (MPAs) are advocated as tools to support sustainable fisheries and biodiversity by excluding the most destructive fishing practices. Some MPAs protect the whole-site of habitat mosaics from bottom-towed fishing while allowing other, low-impact, fishing metiers to continue. This form of MPA management has shown benefits in terms of increased biodiversity and biomass. As MPAs are often advocated as having fishery benefits it is important to monitor key commercial species in the years following MPA designation. In Jersey, Channel Islands, two offshore MPAs that are managed following the whole-site approach were designated in 2017. This provided an opportunity to assess changes in abundance of economically important crustacean species (lobster (Homarus gammarus), spider crab (Maja squinado), and brown crab (Cancer pagurus)) following the exclusion of bottom-towed fishing. Results varied for each of the three species considered but there was little observed benefit of the MPAs three years since their designation. The exception was a greater abundance of lobsters in the Ecrehous MPA compared to the Open Control, particularly for below Minimum Landing Size (MLS) lobsters. There was an observed drop off in size class frequencies above MLS for lobster and brown crab in all locations and treatments that is indicative of selection pressure from fishing. Assessment of historic landings showed both lobster and brown crab to be in decline locally. This decline in lobster was reflected in the results from the potting surveys at the Minguiers MPA. While brown crab was recorded in generally low abundance across locations and treatments. Historic landings of spider crab have been increasing in recent years and spider crab abundance from the potting surveys was generally observed to be

greater in Open Controls. The impact of selective fishing pressure on crustaceans from potting in all areas of the reefs (both inside and outside the MPAs) may reduce any benefits to crustaceans from the removal of mobile gear. Securing a sustainable pot fisheries in Jersey's waters will require the implementation of gear or effort limits within the MPAs and beyond.

### 6.1. Introduction

Commercial fisheries depend on healthy, functioning ecosystems to support fishery species (Dobson et al., 2006; Staples & Funge-Smith, 2009). Overfishing is one of the most significant threats to this resource (Pauly et al., 2005). In 2020, 34.2% of stocks were reported to be unsustainably fished, and another 59.6% maximally sustainably fished (FAO, 2020). Destructive fishing practices, such as bottom-towed fishing, can cause damage to the seabed and have detrimental effects to the benthic biota, reducing the overall health of the seabed (De Grave & Whitaker, 1999; Stewart & Howarth, 2016; Thrush & Dayton, 2002), potentially leading to altered food webs and trophic collapse (Dobson et al., 2006). Further, this fishing practice has the potential to create conflict with other fishing sectors through spatial overlap of fishing grounds and decreasing the abundance of other commercially exploited species through direct damage to individuals (Beukers-Stewart & Beukers-Stewart, 2009a; Kaiser & Spencer, 1995).

Historically, fisheries management has been species-specific, where the focus is on regulating fishing pressure to maximise yield of targeted species (Berkeley et al., 2004; Leary et al., 2012; Sundelöf et al., 2015; Ulrich et al., 2017). This may involve: spatial closures; minimum and maximum landing sizes; exclusions of highly selective

gear types; or seasonal closures to reduce fishing pressure on species during key stages in their lifecycle, such as breeding or migration. More recently, fisheries management legislation is evolving with stipulations for fisheries regulations to not only consider the target species, but also the wider ecosystem (Long et al., 2015; Pikitch et al., 2004). The ecosystem-based approach is supported by international policy (Food and Agriculture Organization of the United Nations (Staples and Funge-Smith, 2009) and more specifically in European policy (Marine Strategy Framework Directive (European Parliament and Council, 2008); Common Fisheries Policy (European Parliament, 2013), Maritime Spatial Planning Directive (European parliament, 2014). This approach aims to improve ecological sustainability while also supporting societal values (Thrush et al., 2016).

Marine Protected Areas (MPAs) are geographically defined areas that are managed to achieve the long-term conservation of nature (Day et al., 2012). However, the management of marine systems using MPAs has been fragmented as policy has been focussed on the conservation of singular features of conservation interest with little consideration for the wider ecosystem. Further, compromises between protection of the marine environment and human activities have often been made in management decisions regarding natural resources (McShane et al., 2011). Recently, there has been a shift in UK policy to move towards a 'whole-site' approach (HM Government, 2018), and in the marine environment this can be achieved through the use of MPAs that exclude destructive fishing practices from all habitats within the MPA boundary (Sheehan et al., 2013b; Solandt et al., 2020).

Although the primary goals of MPAs are cemented in ecology to protect or enhance biodiversity (Lester et al., 2009; Zupan et al., 2018), they are also advocated as

tools to improve fishery management and secure socio-economic benefits (Di Franco et al., 2016; Rees et al., 2020). Whole-site approach MPAs can be multiuse, and activities that are deemed to be low impact and unlikely to negatively impact the ecological goals of the MPA, are permitted to continue (Defra, 2013). These activities tend to be static-gear fisheries, such as pots, static nets and diving. The potential of these MPAs to meet fisheries objectives will be subject to the management of permitted fishing practices within the MPA boundary; while bottomtowed fishing is considered one of the most destructive forms of fishing (Eno et al., 2013), other fishing practices are not without their impacts on both the benthic communities and target species. The designation of areas that exclude bottomtowed fishing may also result in an increase in fishing gears that are permitted to continue, impacting target species through the removal of adults, altering populations structures and reducing the reproductive potential of the stock (Anderson et al., 2008; Rochet & Benoit, 2012). Following the exclusion of bottomtowed fishing in Lyme Bay (SW England, UK), there was an increase in effort from the static pot fishery (Mangi et al., 2011) and a similar response was seen in the Isle of Skomer Marine Conservation Zone, UK, where potting increased in the years after designation (Burton et al., 2016). In these scenarios, between-sector conflict was reduced but within-sector conflict was increased (Blyth et al., 2000).

High potting levels within the Lyme Bay MPA, have been shown to negatively impact commercial catch of crustacean populations (Rees et al., 2021a), while a complete absence of potting within the Lundy No Take Zone (NTZ), UK, resulted in an increase in lobster abundance and overspill into adjacent fishing areas (Hoskin et al., 2011). Additionally, highly selective fishing pressure on adult populations typically results in truncated populations (Froese et al., 2008), and may be a driving factor in the

variability of abundance of exploited species (Anderson et al., 2008). The size frequency distribution of a population can be used as an indication of stock structure and is an important consideration in stock assessments (Rochet and Trenkel, 2003). It is therefore important to consider other fishing practices, such as static pot fisheries, that are typically permitted to continue within 'whole-site' MPAs, when assessing their effectiveness in supporting sustainable fisheries. The response of crustacean populations to other management measures, such as the exclusion of bottom-towed fishing, while allowing potting to continue, needs further exploration. Additionally, data concerning below Minimum Landing Size (MLS) crustaceans is not available through landings data as it is illegal to land crustaceans of this size. Therefore, targeted studies are required to better understand population structures of these stocks.

### 6.2. Case study

Jersey's marine environment supports mobile, static and diving fisheries, targeting a range of benthic, demersal and pelagic fish, but primarily benthic shellfish comprising of lobster (*Homarus gammarus*), brown crab (*Cancer pagurus*), spider crab (*Maja brachydactyla*), scallop (*Pecten maximus*), and whelk (*Buccinum undatum*). To protect areas of fragile habitat and high biodiversity associated with two offshore reefs (the Minquiers and the Ecrehous), two Marine Protected Areas, which excluded bottom-towed fishing, were designated in Jersey's territorial waters in 2017 (Chambers et al. 2020) (Figure 6.1). The majority of the static fleet are made up of potting fishers, targeting either crab and lobster, or whelk; the highest value fishery in Jersey is the static crab and lobster fishery, with a value worth approximately 70% of all fisheries landings (Marine Resources, 2019). Crab and

lobster are caught using a mixture of open pots (inkwells, creels and D-pots) and closed pots (parlour pots).





The intensity of static fisheries has historically been difficult to analyse spatially in British waters, and Jersey is no exception, as static fishing vessels under the 12 m threshold length are not required to transmit their fishing location via Vessel Monitoring Systems (VMS). This has meant that spatial fishing activity of static vessels, which are all under 12 m, is largely unknown. The only exception being information regarding Jersey's six reporting zones that must be recorded in logbooks. In the UK, the Marine Management Organisation (MMO) reports that the quantity and value of key shellfish species landed has increased over the last 25 years (MMO, 2019). Additionally, shellfish are higher in value per tonne than other sea fish, and just under half of all shellfish landings are caught by static potting fisheries (MMO, 2019). In Jersey, shellfish make up ~95% of the fishing economy, with 70% attributed to lobster and brown crab. For such an important fishery there is relatively little information regarding effort, distribution and frequency. The potting industry is lightly regulated, both in the UK and Jersey, with no quotas or restrictions on time at sea. However, Jersey does have specific regulations across its territorial waters, such as pot limits per vessel and mandatory escape gaps (Table 6.1).

Table 6.1. Key management measures in place for the major commercial	
crustacean species in Jersey.	

Species	Minimum landing size (mm)	Maximum landing size (mm)	Pot limit per vessel	Closed season	Quota	Escape gaps
Lobster	87	N/A	1000 pots	None	None	Yes
Brown crab	150	N/A	1000 pots	None	None	Yes
Spider crab	120	N/A	1000 pots	6 weeks in Autumn	None	Yes

Static fishing occurs both inside and outside of the MPAs in Jersey, so it is prudent to monitor target species for signs of decline following MPA designation, as previously there have been increased potting levels following the exclusion of bottom-towed fishing gear from a MPA (Burton et al., 2016; Mangi et al., 2011). As a result of the MPAs being fished locally by small vessels without VMS, and by French vessels whose landings data are not readily available or attributable to the MPAs, it is not possible to directly measure landings that have come from the MPAs. As the static potting fishery constitutes ~80% of the fleet, a commercial potting study was conducted inside the MPAs to investigate commercial crustacean stocks in comparison to Open Controls.

Due to concerns of depleting crustacean stocks in Jersey waters, standardised potting surveys were repeated annually by the Government of Jersey Marine Resources team from 2014. In addition, two MPAs were designated in 2017 that excluded bottom-towed fishing. To complement the wider monitoring and assess the benefits of the MPA on the crustacean populations the newly designated MPAs and adjacent control sites were monitored over three years. The aim of this study is to: 1) Assess the overall trends of crustacean populations in Jersey since 2014, and 2) Test whether this change in spatial management has increased the abundance of the crustacean populations and changed their overall size frequency distribution. As individuals above MLS continue to be targeted by potting fisheries inside the MPAs, it is expected that any initial benefit of the protection would initially be observed in below MLS individuals and so these two size groups have been investigated separately. The following hypotheses are tested for three target commercial crustacean species (European lobster *H. gammarus*, brown crab *C. pagurus*, and spider crab *M. brachydactyla*):

- 1) Abundance of target species have been declining over time (2014 to 2020).
- 2) Abundance of target species increase overtime (2018-2020) in each MPA compared to Open Controls.

- 3) Inside MPAs the abundance of target species below MLS increases over time while the abundance of target species above MLS does not change. Outside MPAs no differences between these size classes are detected.
- 4) The size frequency distribution of target species is statistically significantly different, and has a greater range, in each MPA compared to Open Controls.

### 6.3. Study design

Two MPAs (Minquiers and Ecrehous) that exclude bottom-towed fishing, designated in 2017, were selected as locations to survey commercial crustacean abundances. Surveys were carried out in September and October in 2018, 2019 and 2020. Treatments consisted of MPA (where bottom-towed fishing is excluded) and Open Control (where there are no restrictions on fishing gears). Due to resource logistics, at the Minquiers there was one MPA site and two Open Control sites, whereas at the Ecrehous there were two MPA sites and one Open Control site. Open controls were located between 0.5 and 5.4 km from the MPAs.

### 6.4. Methodology

### 6.4.1. Site selection

Locations of sites were selected based on the availability of suitable rocky reef habitat with depth ranges of 5-15 m (Below Chart Datum) within each of the treatments (MPA and Open Control) at both the Ecrehous and Minquiers (Figure 6.2a and 6.2b). Areas were preliminary surveyed using sonar to verify the presence of reef habitat based on uneven benthic topography, and adjusted accordingly so that the 500 x 500 m<sup>2</sup> areas contained a high coverage of reef habitat.



Figure 6.2. Management measure boundaries and 500 x 500 m potting sites within each treatment at a) the Ecrehous and b) the Minquiers.

### 6.4.2. Data collection

Parlour pots (Figure 6.3) were used to sample the commercial crustacean species in 2018, 2019 and 2020. With permission of the Government of Jersey Marine Resources team, pots modified with a meshed base and closed escape gaps were used for this study as undersize crustaceans were also of interest to obtain a representative sample of the entire population in order to understand size frequency distributions. Pots were deployed and left to 'soak' for 24 hours at each treatment site then recovered to collect abundance data on commercially targeted species. To account for seasonal variation, sampling was conducted at the same time of year, September and October, each year. In each treatment, three strings of ten pots (30 pots total) were deployed on areas of reef habitat within a 500 x 500 m predetermined area. Pots were baited with 'Scad' (*Trachurus trachurus*) based on the methods used by Rees et al. (2017) and its local availability. All crabs and lobsters were held in separate tubs until they could be measured and weighed for each string. For brown crab and spider crab; carapace length (CL), width (CW) and depth (CD) were measured using Vernier callipers (± 1 mm). For lobsters, carapace

length and abdomen width were measured. Sex, presence of eggs and general condition, such as missing appendages and disease, were also noted for all species.

Additional data from the Government of Jersey Marine Resources team were available for open fishing areas on the west coast of Jersey. This data has been collected using the same 30 modified parlour pots each year from three locations on the west coast of Jersey. Due to discrepancies in the survey design prior to 2014, only data from 2014 onwards were used. There were no survey data available for 2015. Different soak times (48 hours) were used, so this data is included as a reference for localised changes in crab and lobster abundance over time.



Figure 6.3. Diagram of a Parlour Pot showing A) the first chamber with the mouth entrance at the top, this is also the chamber which holds the bait, B) the second chamber, which is linked to chamber one via an opening (illustrated by a blue oval), and C) the hinged released door, which is secured by a hook on a bungee cord. Chamber two would normally have an escape gap via which juvenile crab and lobster may exit, but these were closed for the purpose of this study.

A Parlour Pot, as defined by Jersey Sea Fisheries Regulations, is "any pot, container or item whatsoever which is designed, adapted or intended to be used, or which is capable of being used, for the taking of crustaceans and which is a doublechambered pot, container or item, with the second chamber known as the parlour from which escape is restricted".

#### 6.2.3. Data analysis

### 6.2.3.1. Historic potting surveys

Data from annual potting surveys, conducted by the Government of Jersey Marine Resources team, were used to calculate abundance (individuals caught per 10 pots) across three locations each year. Results were split into two size classes of individuals above MLS and below MLS.

Change in above and below MLS crustacean abundance (lobster, brown crab and spider crab) overtime (2014 to 2020) was tested using General Linear Mixed Effect Models (GLMMs). Counts of individuals per string were modelled as a function of Year (n=7), with Location (n=3) as a random effect, using a Poisson distribution.

### 6.2.3.2. Ecrehous and Minquiers MPA potting survey

To calculate mean abundance (individuals caught per 10 pots), values were averaged across the three string replicates in each treatment for each year surveyed. Crustacean abundances were compared between Treatments and Size Classes (above MLS and below MLS) and across locations (Ecrehous and Minquiers) using General Linear Models (GLMs). Counts of individuals per string were modelled as a function of year (2018, 2019 and 2020), treatment (MPA and Open Control) and size class (above MLS and below MLS and below MLS), using a Poisson distribution. Optimal models were determined using the Akaike Information Criterion

(AIC) iteratively, where predictor variables were sequentially added and then dropped from models to determine variable contribution to model fit. Optimal models were then applied and evaluated. Reported values in the text are raw mean values ± standard deviation. All analyses and data visualisations were carried out in R using packages from the 'tidyverse' (R Core Team, 2021; Wickham et al., 2019).

Size frequency distribution for the total catch was assessed in 10 mm carapace width intervals for Brown Crab, 10 mm carapace length intervals for Spider Crab, and in 5 mm carapace length intervals for Lobsters. Kolmogorov-Smirnov tests (Daniel, 1990) were used to test the null hypothesis of no difference in size frequency distributions between Treatments (MPA and Open Control), and between Years (2018, 2019 and 2020) within each Treatment at both Locations (Ecrehous and Minquiers).

#### 6.5. Results

#### 6.5.1. Historic potting surveys

Abundance of below MLS lobsters did not change significantly over time, while above MLS lobster abundance declined significantly from 2014 to 2020 (Figure 6.4, Table 6.2). Abundance of both above and below MLS brown crabs declined over time since 2014, but this was only significant for above MLS brown crabs. Both above and below MLS spider crab abundance increased over the same period, but this was only significant for above MLS spider crabs (Figure 6.4, Table 6.2). The abundance of below MLS was greater than the above MLS for both lobster and brown crab while the opposite is observed for spider crab, with greater numbers of above MLS individuals caught in most years (Figure 6.4).



Figure 6.4. Historic data from annual potting surveys conducted by the Government of Jersey Marine Resources team between 2014 and 2020. Lines show mean number of a) lobster, b) brown crab and c) spider crab per 10 pots and shading shows standard error, split for above MLS (dark blue) and below MLS (green).

Table 6.2. General Linear Mixed Effects Model outputs for change in Above MLS and Below MLS lobster, brown crab and spider crab abundance over time (2014-2020). Results are modelled as a function of Year with Location as a random effect. Bold denotes a significant result.

	Estimate	Std. Error	z value	р
Lobster				
(Intercept)	4620.000	1240.000	3.730	$0.0018^{**}$
Above MLS	-2.280	0.614	-3.720	$0.0019^{**}$
(Intercept)	1640.000	6010.000	0.272	0.79
Below MLS	-0.771	2.980	-0.259	0.8
Brown crab				
(Intercept)	7670.000	1410.000	5.450	$< 0.0001^{***}$
Above MLS	-3.800	0.698	-5.440	$< 0.0001^{***}$
(Intercept)	7310.000	3780.000	1.930	0.07
Below MLS	-3.610	1.870	-1.930	0.071
Spider crab				
(Intercept)	-7220.000	1930.000	-3.740	$0.0018^{**}$
Above MLS	3.590	0.958	3.750	$0.0018^{**}$
(Intercept)	-2630.000	1380.000	-1.910	0.074
Below MLS	1.310	0.683	1.920	0.073

## 6.5.2.1. Lobster

### 6.5.2.1.1. Abundance

At the Minquiers, there was no effect of treatment on abundance of lobsters but abundance was observed to decrease from 2018 to 2020 in both treatments (Figure 6.5, Table 6.3). Abundance of below MLS lobsters was significantly greater than above MLS lobsters in both treatments (Figure 6.5, Table 6.3). Between 2018 and 2019 the number of above MLS lobsters decreased from  $1.7 \pm 1.5$  to  $1 \pm 0$  in the MPA and from  $3 \pm 1.6$  to  $1.3 \pm 1$  in the Open Control. Over the same period, below MLS lobsters decreased from  $12.3 \pm 2.1$  to  $7 \pm 3.5$  in the MPA and from  $10.7 \pm 4.2$  to  $6.7 \pm 3.4$  in the Open Control (Figure 6.5, Table 6.3).

At the Ecrehous, the abundance of lobsters was greater in the MPA compared to the Open Control (Table. 6.3), and this was primarily driven by the abundance of below MLS lobsters inside the MPA (Figure 6.5). There was a significant Year x Treatment interaction in 2020 where lobster abundance increased significantly in the MPA compared to the Open Control which decreased (Table 6.3, Figure 6.5). The abundance of below MLS individuals was greater than above MLS in both treatments and both size classes were in greater abundance in the MPA compared to the Open Control (Figure 6.5, Table 6.3). Above MLS lobster abundance at the Ecrehous was greater in the MPA compared to the Open Control and this difference in abundance was greatest in 2020 (MPA =  $1.6 \pm 1.3$ , Open Control =  $0.7 \pm 0.6$ ). Below MLS lobster abundance in the MPA in all years, with the greatest abundance in the MPA observed in 2020 ( $8.3 \pm 6.2$ ), whereas abundance was lowest in the Open Control in 2020 ( $0.3 \pm 0.6$ ). Figure 6.5).



Figure 6.5. Modelled mean estimates of lobsters per 10 pots (string) at the Minquiers and Ecrehous per year and treatment. Grouped by Total (all lobsters), above MLS lobsters and below MLS lobsters.

Table 6.3. General linear model outputs for Lobster abundance at the Minquiers and Ecrehous. Total results are modelled as a function of Year, Treatment. Size Class results

	Estimate	Std. Error	z value	р
Minquiers Total				
(Intercept)	1.973	0.128	15.400	$< 0.0001^{***}$
2018-2019	-0.078	0.139	-0.559	0.58
2018-2020	-0.491	0.151	-3.250	$0.0012^{**}$
2019-2020	-0.413	0.160	-2.580	$0.0098^{**}$
MPA-Open	0.072	0.129	0.561	0.58
Minquiers Size Class				
(Intercept)	0.349	0.324	1.080	0.28
2018-2019	-0.263	0.139	-1.900	0.058
2018-2020	-0.540	0.151	-3.580	$< 0.0001^{***}$
2019-2020	-0.277	0.160	-1.730	0.083
MPA-Open	0.605	0.370	1.630	0.1
Size	2.067	0.336	6.160	$< 0.0001^{***}$
MPA-Open x Size	-0.561	0.395	-1.420	0.16
Ecrehous Total				
(Intercept)	1.587	0.151	10.500	$< 0.0001^{***}$
2018-2019	0.087	0.209	0.417	0.68
2018-2020	0.205	0.195	1.050	0.29
2019-2020	0.118	0.190	0.621	0.53
MPA-Open	-0.894	0.326	-2.740	$0.0061^{**}$
2018-2019 x MPA-Open	0.231	0.467	0.496	0.62
2018-2020 x MPA-Open	-1.186	0.674	-1.760	0.079
2019-2020 x MPA-Open	-1.420	0.678	-2.090	$0.037^{*}$
Ecrehous Size Class	•			
(Intercept)	0.348	0.249	1.400	0.16
2018-2019	-0.074	0.209	-0.353	0.72
2018-2020	0.083	0.195	0.426	0.67
2019-2020	0.157	0.190	0.825	0.41
MPA-Open	-1.267	0.326	-3.890	$< 0.0001^{***}$
Size	1.689	0.227	7.430	$< 0.0001^{***}$
2018-2019 x MPA-Open	0.433	0.467	0.927	0.35
$2018-2020 \ge MPA-Open$	-1.023	0.674	-1.520	0.13
$2019-2020 \ge MPA-Open$	-1.460	0.678	-2.150	$0.032^{*}$

are modelled as a function of Year, Treatment and Size Class. Bold denotes a significant result.

# 6.5.2.1.2. Size frequency distribution

At both the Ecrehous and Minquiers, there were considerably more lobsters caught

that were below MLS than above, with a distinct drop in frequencies for size classes

above MLS (87 mm CL: Figure 6.6a and 6.6b). This trend was observed across all treatments, locations and years. The range of size classes at the Ecrehous was less than the Minquiers, with the minimum (52-56 mm) and maximum (102-106 mm) size classes missing from most years at the Ecrehous (Figure 6.6a and 6.6b). The size frequency distribution of lobsters caught in both treatments exhibited bell-shaped distributions except for in the Open Control at the Ecrehous, which had a comparatively uniform distribution in all years. Size frequency distributions were not significantly different between treatments in any year at either Location (Appendix Table D.1 and D.2).

At the Minquiers, the modal size class in the MPA was smaller in most years compared to the Open Control. In 2018 the modal size class in the MPA was 77-81mm, whereas in the Open Control it was 82-86mm. In 2020 in the MPA there was no distinct modal class with equal frequencies across four size classes from 67-71 to 82-86 mm, while in the Open Control the modal class was 72-76 (Figure 6.6). At the Ecrehous, the modal size class in the MPA was greater in all years (2018: split across 72-76 and 82-86 mm; 2019 and 2020: 77-81 mm) compared to the Open Control (2018: 77-81 mm; 2019: split across 62-66 and 72-76 mm; 2020: no distinct modal size class) (Figure 6.6).





### 6.5.2.2. Spider crab

### 6.5.2.2.1. Abundance

Spider crab abundance did not show an overall increase over time in either Treatment at either location (Figure 6.7, Table 6.4). This is inconsistent with historical potting data which showed a steady increase in spider crab abundance since 2014 (Figure 6.4). At the Minquiers, there was no significant difference in total
abundance between treatments nor a change in abundance over time in either treatment (Figure 6.7, Table 6.4). A lower abundance of above MLS spider crabs was observed in the MPA ( $0.2 \pm 0.4$ ) compared to the Open Control ( $1.7 \pm 1.7$ ) but this was not significant (Table 6.4). Abundance of below MLS spider crabs was not different between treatments (MPA:  $1.9 \pm 1.9$ , Open Control:  $1.6 \pm 1.1$ : Figure 6.7, Table 6.4).

At the Ecrehous, there was a significant Year x Treatment interaction for total spider crab abundance between 2019 and 2020, with abundance steady in the MPA compared to an overall decline in the Open Control (Table 6.4, Figure 6.7). This interaction was also observed in both the above MLS and below MLS size classes. Abundance of below MLS spider crabs was generally lower in the MPA compared to the Open Control and was highest in both treatments in 2019 (MPA =  $2.7 \pm 1.5$ ; Open Control =  $7 \pm 8.2$ ) and lowest in 2020 (MPA =  $0.5 \pm 0.6$ ; Open Control =  $0.3 \pm 0.6$ ). The abundance of above MLS individuals was lower in the MPA in 2018 (2.8  $\pm 1.5$ ) and 2019 ( $1.3 \pm 2.3$ ) compared to the Open Control ( $2018 = 14 \pm 8.2$ ; 2019 =  $18.7 \pm 13.6$ ). Abundance of above MLS individuals in the Open Control in 2020 ( $3.7 \pm 3.2$ ) was similar to the MPA ( $2.4 \pm 2.2$ ) due to a considerable decrease in the number of above MLS spider crabs caught in the Open Control (Figure 6.7, Table 6.4).



Figure 6.7. Modelled mean estimates of spider crabs per 10 pots (string) at the Minquiers and Ecrehous per year and treatment. Grouped by Total (all spider crabs), above MLS spider crabs and below MLS spider crabs.

Table 6.4. General linear model outputs for spider crab abundance at the Minquiers and Ecrehous. Total results are modelled as a function of Year, Treatment. Size Class results

are modelled as a function of Year, Treatment and Size Class. Bold denotes a significant result.

	Estimate	Std. Error	z value	р
Minquiers Total				
(Intercept)	0.765	0.292	2.620	$0.0088^{**}$
2018-2019	-0.290	0.330	-0.878	0.38
2018-2020	0.182	0.303	0.602	0.55
2019-2020	0.472	0.317	1.490	0.14
MPA-Open	0.233	0.277	0.842	0.4
Minquiers Size Class				
(Intercept)	-1.100	1.000	-1.100	0.27
2018-2019	0.000	1.410	0.000	1
2018-2020	-17.200	3300.000	-0.005	1
2019-2020	-17.200	3300.000	-0.005	1
MPA-Open	1.440	1.070	1.340	0.18
Size	1.390	1.120	1.240	0.21
2018-2019 x MPA-Open	-0.336	1.530	-0.220	0.83
2018-2020 x MPA-Open	17.400	3300.000	0.005	1
2019-2020 x MPA-Open	17.800	3300.000	0.005	1
2018-2019 x Size	-0.064	1.570	-0.041	0.97
2018-2020 x Size	17.400	3300.000	0.005	1
2019-2020 x Size	17.500	3300.000	0.005	1
MPA-Open x Size	-1.250	1.230	-1.020	0.31
Ecrehous Total				
(Intercept)	1.500	0.236	6.380	$< 0.0001^{***}$
2018-2019	-0.118	0.373	-0.316	0.75
2018-2020	-0.474	0.356	-1.330	0.18
2019-2020	-0.357	0.393	-0.907	0.36
MPA-Open	1.090	0.265	4.110	$< 0.0001^{***}$
2018-2019 x MPA-Open	0.768	0.408	1.880	0.06
2018-2020 x MPA-Open	-0.735	0.475	-1.550	0.12
2019-2020 x MPA-Open	-1.500	0.501	-3.000	$0.0027^{**}$
Ecrehous Size Class				
(Intercept)	0.760	0.284	2.680	$0.0074^{**}$
2018-2019	-0.378	0.430	-0.879	0.38
2018-2020	-0.229	0.396	-0.578	0.56
2019-2020	0.149	0.476	0.314	0.75
MPA-Open	1.190	0.305	3.900	$< 0.0001^{***}$
Size	-0.563	0.391	-1.440	0.15
2018-2019 x MPA-Open	0.653	0.435	1.500	0.13
2018-2020 x MPA-Open	-0.926	0.491	-1.890	0.059
2019-2020 x MPA-Open	-1.580	0.547	-2.890	0.0039**
2018-2019 x Size	0.824	0.367	2.240	0.025*
2018-2020 x Size	-0.617	0.692	-0.892	0.37
2019-2020 x Size	-1.440	0.696	-2.070	0.038*
MPA-Open x Size	-0.810	0.413	-1.960	$0.05^{*}$

#### 6.5.2.2.2. Size frequency distribution

At the Minguiers, there were more spider crabs caught that were below MLS than above in both treatments in all years, with a drop in frequencies of size classes above MLS (120 mm CL). At the Ecrehous the reverse was observed, with fewer numbers of individuals caught below MLS compared to above MLS in both treatments in all years. The range of sizes at the Ecrehous was less than that of the Minguiers, with the minimum (40-49 mm) and maximum (160-169 mm) size classes missing from most years at the Ecrehous (Figure 6.8a and 6.8b). There was a relatively uniform size frequency distribution at the Minguiers with few individuals caught across the size classes in all years. Likewise, at the Ecrehous in the Open Control there was a uniform size frequency distribution in all years, whereas in the MPA at the Ecrehous in 2018 and 2019 the size frequency distribution was bell-shaped. Size frequency distributions were significantly different between Treatments in 2019 at both Locations (Appendix Table D.1), with size frequency distributions at the Minquiers skewed to the right in the MPA compared to the Open Control, and skewed to the left in the MPA compared to the Open Control at the Ecrehous (Figure 6.8a and 6.8b). Size frequency distributions did not differ significantly with Year in either Treatment at the Minguiers. Whereas, size frequency distributions were significantly different at the Ecrehous between 2019 and 2020 in the MPA, with fewer individuals caught in smaller size classes in 2020, and between 2018 and 2019 in the Open Control, with fewer individuals caught in larger size classes in 2019 (Figure 6.8a and 6.8b, Appendix Table D.2).

At the Minquiers, too few spider crabs were caught to assess the modal size (Figure 6.8). At the Ecrehous, the modal size class in the MPA was typically smaller (2018: 120-129 mm; 2019: split across 110-119 and 120-129 mm, 2020: 150-159 mm)

compared to the Open Control (2018: 130-139 mm; 2019: 120-129 mm; 2020: split across 140-149 and 150-159 mm: Figure 6.8).



Figure 6.8. Carapace width frequency distribution of spider crab per Treatment (MPA and Open Control) and Year (2018, 2019 and 2020) in 10 mm classes for a) the Minquiers and b) the Ecrehous. The dashed line indicates the minimum landing size (MLS) of 120 mm.

#### 6.5.2.3. Brown crab

## 6.5.2.3.1. Abundance

At the Minquiers, the total abundance of brown crab was significantly greater in the MPA compared to the Open Control in 2019 only (Figure 6.9, Table 6.5). The same trend was observed in both size classes. The abundance of below MLS brown crabs in the MPA in 2019 was 7.3  $\pm$  4 compared to 1.8  $\pm$  1.7 in the Open Control. Abundance of below MLS brown crabs was similar in both treatments in 2018 (MPA = 3.7  $\pm$  1.5, Open Control = 2  $\pm$  1.7) and 2020 (MPA = 2.7  $\pm$  2.1, Open Control 2.7  $\pm$  1.6: Figure 6.9, Table 6.5). Above MLS brown crab abundance was relatively low compared to below MLS brown crabs in both Treatments. The abundance of above MLS brown crabs in the MPA in 2019 was 1  $\pm$  1.7 compared to 0  $\pm$  0 in the Open Control. Abundance of below MLS brown crabs abundance was similarly low in both treatments in 2018 (MPA = 0  $\pm$  0, Open Control = 1.8  $\pm$  1.3) and 2020 (MPA = 0.3  $\pm$  0.6, Open Control 1.7  $\pm$  2.1: Figure 6.9, Table 6.5).

At the Ecrehous, the number of brown crabs caught was very low, with less than two per 10 pots in all years and treatments (Figure 6.9). Total brown crab abundance was not significantly different between treatment in any year, nor was there a significant change in abundance over time (Figure 6.9, Table 6.5). There were significantly more brown crabs below MLS than above but neither size class significantly differed in abundance between treatments or years (Figure 6.9, Table 6.5).



Figure 6.9. Modelled mean estimates of brown crabs per 10 pots (string) at the Minquiers and Ecrehous per year and treatment. Grouped by Total (all brown crabs), above MLS spider crabs and below MLS brown crabs.

Table 6.5. General linear model outputs for brown crab abundance at the Minquiers and Ecrehous. Total results are modelled as a function of Year, Treatment. Size Class results

are modelled as a function of Year, Treatment and Size Class. Bold denotes a significant result.

	Estimate	Std. Error	z value	р	
Minquiers Total					
(Intercept)	1.300	0.302	4.310	< 0.0001***	
2018-2019	0.821	0.362	2.270	0.023*	
2018-2020	-0.201	0.449	-0.446	0.66	
2019-2020	-1.020	0.389	-2.630	0.0086**	
MPA-Open	-0.383	0.397	-0.965	0.33	
2018-2019 x MPA-Open	-1.180	0.583	-2.020	0.044*	
2018-2020 x MPA-Open	0.537	0.562	0.955	0.34	
2019-2020 x MPA-Open	1.710	0.584	2.930	0.0033**	
Minquiers Size Class					
(Intercept)1	-0.243	0.368	-0.662	0.51	
2018-2019	0.821	0.362	2.270	0.023*	
2018-2020	-0.201	0.449	-0.446	0.66	
2019-2020	-1.020	0.389	-2.630	0.0086**	
MPA-Open	0.177	0.398	0.445	0.66	
Size	1.300	0.267	4.870	$< 0.0001^{***}$	
2018-2019 x MPA-Open	-1.680	0.585	-2.880	0.004**	
2018-2020 x MPA-Open	0.090	0.565	0.160	0.87	
2019-2020 x MPA-Open	1.770	0.585	3.030	$0.0024^{**}$	
Ecrehous Total					
(Intercept)	0.560	0.378	1.480	0.14	
2018-2019	-0.965	0.802	-1.200	0.23	
2018-2020	-0.108	0.483	-0.223	0.82	
2019-2020	0.857	0.769	1.120	0.26	
MPA-Open	-0.560	0.690	-0.811	0.42	
2018-2019 x MPA-Open	0.560	1.210	0.461	0.65	
2018-2020 x MPA-Open	-19.200	5440.000	-0.004	1	
2019-2020 x MPA-Open	-19.800	5440.000	-0.004	1	
Ecrehous Size Class					
(Intercept)	-1.620	0.775	-2.090	0.036*	
2018-2019	-0.993	0.802	-1.240	0.22	
2018-2020	0.075	0.484	0.155	0.88	
2019-2020	1.070	0.770	1.390	0.17	
MPA-Open	-0.587	0.690	-0.851	0.39	
Size	2.090	0.740	2.830	0.0047**	
2018-2019 x MPA-Open	0.587	1.220	0.483	0.63	
2018-2020 x MPA-Open	-18.400	3310.000	-0.006	1	
0010 0000 1001 0	10.000	0010.000	0.000	-	

# 6.5.2.3.2. Size frequency distribution

At both the Ecrehous and Minquiers, there were considerably more brown crabs caught that were below MLS than above MLS, with a distinct drop in frequencies of size classes above MLS (150 mm CW). This trend was observed across all

treatments, locations, and years. The size frequency distribution of brown crab (*C. pagurus*) catch was relatively truncated at the Ecrehous compared to the Minquiers, with very few size classes recorded above MLS (Figure 6.10). Across all years, there were very few individuals caught above MLS in the MPA at the Minquiers. The range of sizes at the Ecrehous was less than that of the Minquiers, with the smaller (80-89 and 90-99 mm) and larger (160-169 to 190-199 mm) size classes missing from most years at the Ecrehous (Figure 6.10a and 6.10b). The size frequency of brown crabs caught in both treatments at both locations did not exhibit a bell-shaped distribution except for in the Open Control at the Minquiers in 2020. At the Minquiers there was a significant difference in size frequency distributions between the MPA and Open Control in years 2018 and 2019, with a greater range of sizes in the Open Control in 2018 and a greater range in the MPA in 2019 (Figure 6.10, Appendix Table D.2). Size frequency distributions were not significantly different between treatments in any year at the Ecrehous (Appendix Table D.1).

At the Minquiers, the modal size class in the MPA was smaller in all years compared to the Open Control. In 2018 the modal size class in the MPA was split across 110-119 and 140-149 mm, whereas in the Open Control it was 150-159mm, and in 2020 the modal size class in the MPA was 100-109 mm while in the Open Control the modal size class was 130-139 mm (Figure 6.10). It was not appropriate to assess modal size classes at the Ecrehous due to the low number of brown crabs caught (Figure 6.10).



Figure 6.10. Carapace width frequency distribution of brown crab per Treatment (MPA and Open Control) and Year (2018, 2019 and 2020) in 10 mm classes for a) the Minquiers and b) the Ecrehous. The dashed line indicates the minimum landing size (MLS). Note, MLS increased from 140 mm to 150 mm in 2019.

# 6.6. Discussion

In this study, the response of commercial crustacean abundance to the exclusion of bottom-towed fishing within two offshore MPAs has been assessed. The results varied for each of the three species considered (lobster, spider crab and brown crab) but there was little observed benefit of the MPAs three years since their designation. The exception was a greater abundance of lobsters in the MPA at the Ecrehous compared to the Open Control, particularly for below MLS lobsters. Size frequency distribution can be used as an indication of stock structure changes and often forms an important component of stock assessment (Rochet and Trenkel, 2003). The expected effect of fishing is a shift towards smaller individuals and a steeper slope of the size–abundance relationship. Here there was an observed drop off in size class frequencies above MLS for lobster and brown crab that is indicative of selection pressure from fishing. The results of each species are discussed in context of other research and regional stock assessments below.

## 6.6.1. Lobster

The MPA had not yet had a significant effect on lobster abundance at the Minquiers and decreases were observed in both size classes and treatments over the survey period. The decline in lobsters at the Minguiers is reflected in the overall landings data for Jersey (Marine Resources, 2019), and the historic potting surveys conducted by Marine Resources, but does not reflect stock assessments of lobster on the southwest coast of the UK, which show lobster landings are relatively stable in the southwest English Channel (Cefas, 2020b; Cornwall IFCA, 2018b; Southern IFCA, 2020). This suggests that the decline of both lobster size classes at the Minquiers, found here, is primarily a result of overexploitation from static pot fisheries, and this removal of larger, more fecund, individuals may have implications for recruitment (Costa et al., 2020; Fenberg & Roy, 2008). Conversely, at the Ecrehous, lobster was in greater abundance in the MPA compared to the Open Control, particularly for the below MLS. Nursery areas are characterised by high densities of juvenile species and support increased growth and survival compared to surrounding habitat (Beck et al. 2001). The high abundance of below MLS

individuals alongside low abundances in above MLS individuals in the MPA treatment at the Ecrehous indicates that this is a nursery area (Beck et al., 2001) rather than an area that experiences lower levels of fishing. At present it is not possible to accurately estimate potting density in Jersey waters as most potting vessels are under 12 m in length and therefore are not required to transmit their location while fishing. To better understand changes in lobster abundance at the MPAs, more detailed spatial fishing information is needed to determine levels of potting in relation to population abundance and size frequency distributions. This is particularly true for the Minquiers, which has historically been an important fishing ground for both Jersey and French fishers targeting lobster (Chambers et al., 2016) and is therefore a priority for management.

Currently there are few restrictions on the level of potting that can occur in Jersey's waters (a maximum of 1000 pots per vessel: Table 1), meaning there is a potentially significant pool of static gear in use and concentration of static fishing effort on specific fishing grounds may have undesirable effects on target species and benthic diversity (Rees et al., 2021a). This was the case in Lyme Bay, where static pot fishing effort increased following the exclusion of bottom-towed fishing (Mangi et al., 2011; Rees et al., 2021a) and a similar response was seen in the Isle of Skomer Marine Conservation Zone, where potting also increased in the years after designation (Burton et al., 2016). In these scenarios, between-sector conflict was reduced but within-sector conflict was increased (Blyth et al., 2000). This increase in fishing intensity has the potential to negatively impact both the stock levels of the target species and the benthic environment (Rees et al., 2021a). As static fisheries, and in particular potting fisheries, are normally permitted within multi-use MPAs, due to their perceived low impact on benthic habitats and high selectivity for target species,

it is important that the ecological impacts on target species are better understood to inform best practice for a sustainable fishery.

## 6.6.2. Spider crab

Spider crabs were generally observed in greater abundance in Open Controls than inside the MPAs. As suggested earlier, if potting effort has increased within the MPAs this may be resulting in reduced commercial crustacean abundance, and as static pot fishers typically target all three species at the same time, it is possible that all three species would be affected by increased potting levels. One clear observation between the two locations was the greater abundance of spider crabs overall at the Ecrehous compared to the Minguiers. The reason for this was unclear as the habitat composition and depth of the survey areas were consistent. Additionally, at the Ecrehous, the size frequency distribution was weighted towards above MLS individuals. Smaller individuals may have differing habitat preferences, or the presence of larger individuals may deter smaller spider crabs from entering the pot, resulting in fewer small individuals being caught. Spider crab stock assessments are not as readily available from regional fisheries management authorities as the more highly valuable brown crab and lobster but stock assessments from Cornwall have shown an increase in landings from spider crab pot fisheries between 2016 and 2018 (Cornwall IFCA, 2018c), which reflects the trends observed in landings data (Marine Resources, 2019) and historic potting survey data from Jersey, but not the results from the current study, which showed an overall decline in above MLS and below MLS spider crabs at the Ecrehous and a stable but lower abundance at the Minquiers. Further years of surveys are needed to determine whether spider crab abundance is beginning to decline.

#### 6.6.3. Brown crab

The low numbers of brown crab caught in comparison to lobster and spider crab may have been an artifact of the habitats surveyed. All pots were deployed in areas of rocky reef that are primarily targeted by potting fishers. While brown crab use rocky reef, it is also associated with sedimentary habitats where individuals will dig pits to forage or rest in (Hall et al., 1991) and therefore it may be appropriate to also target brown crab on sediments. However, the Landings Per Unit Effort (LPUE) or brown crab has been declining locally since 2012 (Marine Resources, 2019), and the low abundance observed across locations and treatments in the current study is most likely a reflection of a significantly reduced population. This reduced population will in part be due to fishing pressure but may be further confounded by wider environmental effects that are causing declines in brown crab populations at a regional scale. Stock assessment information is available from various fisheries management authorities in the UK that suggest that brown crab is declining in the southwest of the English Channel (Cefas, 2020a; Cornwall IFCA, 2018a). Reasons for this are not fully understood and with no such decline seen in the southeast English Channel this may be a result of regional shifts in brown crab populations. Emerging evidence of a novel disease (paramoebiasis) suggests there is a significant proportion of the brown crab population affected in the English Channel (Bateman et al., 2020). While the prevalence of this disease is not yet understood, it may be a contributing factor in brown crab decline. The low abundance of brown crab caught in all years and locations, combined with the declines observed in landings both locally and in the southwest UK and northern France (ICES CIEM, 2019), are cause for concern and it is prudent to monitor this stock closely to inform management. The majority of marine sedimentary habitats in Jersey's waters are not

protected from mobile gear and this too may be exacerbating the decline in brown crab if they are subject to mechanical damage from dredges when utilising sedimentary habitats (Veale et al., 2000).

While all three commercial crustacean species overlap in their habitat use, they have different preferences that may in part explain the observed variation in response to management measures. Brown crab spend their juvenile stage in the intertidal and migrate to subtidal habitats once they have developed (Bakke, 2019; Bennett, 1995; Neal & Wilson, 2008). Lobster juvenile stage habitat preference remains debated in the literature but they are thought to seek shelter amongst cobbles or burrow in soft sediments (Linnane et al., 2000a, 2000b), including in the intertidal zone, whereas adults live primarily on subtidal reef and boulder habitats where they seek shelter in crevices (Galparsoro et al., 2009; Lawton & Lavalli, 1995). Prior to the designation of the MPAs, mobile gear fishers will have been deterred from many areas of the reefs as there are numerous reef heads that have the potential to damage towed fishing gears. Therefore, noticeable recovery is more likely to be observed on fringe habitats where sediments meet the reefs. For lobster, this may be beneficial to the early benthic stage which are thought to seek shelter in cobble habitat or burrow in soft sediments. Spider crab is the most ubiquitous of the three commercial crustacean species, and does not have a strong preference of habitat in its adult life stage but is thought to favour high algal cover areas, both intertidally and subtidally in their juvenile stages to seek refuge from predators (Corgos et al., 2011; Gonzalez-Gurriaran & Freire, 1994; Hosie, 2009). The MPAs are primary comprised of reef habitat and so may not cover all of the essential habitats needed for the crustacean species considered here.

Both brown crab and lobster take a minimum of four years to reach maturity (Bakke, 2019; Neal & Wilson, 2008; Schmalenbach et al., 2011), meaning there will not have been another generation since the MPAs were established. Spider crabs mature within one to two years (Durán et al., 2013; MarLIN, 2006), but this still is unlikely to be detected within three years. Gestation periods also differ between the species, with brown crabs carrying eggs for 6 to 9 months which may make them vulnerable when crossing areas targeted by mobile gear areas as they migrate to spawning grounds (Brown & Bennett, 1980; Eaton et al., 2003; Tonk & Rozemeijer, 2019). Future MPAs should take the life history of these species into account in order to maximise the benefits to local fisheries. However, the life cycles of the three species will influence the time taken to see a response to fishing restrictions.

# 6.7. Conclusion

MPAs are advocated as tools to improve sustainability of fisheries, but it may take many years for benefits to be realised depending on the life history traits of the target species. These results have shown that the response of commercial crustaceans to marine spatial management is complex and varies between species. While the removal of mobile gear can improve the integrity of the seabed, enhancing diversity and supporting greater biomasses of commercial species, this alone is unlikely to be enough to create a sustainable fishery. Both lobster and brown crab were either in low or declining abundance overall; with lobster at the Minquiers showing a reduction in size class frequencies, regardless of treatment, that is indicative of size selective fishing pressure. However, the MPA at the Ecrehous showed greater abundance of below MLS lobsters highlighting its potential value as a nursey habitat. The young age of the MPAs means there is unlikely to have been a measurable effect on commercial crustacean abundance, but for fishery benefits to be realised in the

future, the level of static pot fisheries must also be managed. As potting occurs in all areas of the reefs, any benefits to crustaceans from the removal of mobile gear may be overshadowed by the impact of selective fishing pressure on crustaceans. The current priority should be to reduce the fishing pressure from pots to enable these populations to recover to a biologically and economically sustainable level. These results provide a baseline for future monitoring of commercial crustacean populations in Jersey across spatial management measures at the offshore reefs. However, improved spatial fishing information is needed to help inform sustainable pot limits and continued monitoring into the future to determine the timeframe for recovery of commercial stocks.

# 7. Discussion

## 7.1. Overview

Marine habitats span the maritime boundaries of national jurisdictions, meaning the effective and coherent management of shared resources can be complex. In jointly managed waters where management measures such as closed seasons or quota systems may not be possible to enforce on all jurisdictions equally, Marine Protected Areas (MPAs) that exclude specific fishing gears from all nations equally can provide a solution (Chambers et al., 2020). MPAs that adopt a whole-site approach and exclude bottom-towed fishing from a mosaic of habitats are rare (Solandt et al., 2020). This is despite the evidence base surrounding their performance in terms of recovering biodiversity, improving resilience to climatic events, and supporting fisheries (Davies et al., 2021a; Rees et al., 2010; Sheehan et al., 2021; Sheehan et al., 2013a). Ecological research on MPAs has typically focused on the recovery of reef habitat (Goetze et al., 2021; Sheehan, Stevens, et al., 2013), with a lack of evidence surrounding the biological importance of non-reef, sedimentary habitats inside MPAs. The MPAs in Jersey have provided the first opportunity to investigate the effects of excluding bottom-towed fishing from MPAs on a range of sedimentary habitats. Examined were the differences in biodiversity metrics (richness and abundance of mobile species, epibiota and infaunal species, and coverage of structure-forming species) on sedimentary habitats within MPAs compared to Open Controls and their role in supporting fisheries in an area of international fishing access.

Several methods were employed to investigate both the ecological and socioeconomic effects of MPAs. To understand the response of commercial crustacean populations and the consequences for static fisheries, potting surveys were

employed, with 540 pot lifts over three years and 741 crabs and lobsters measured. Complementary studies were conducted to examine the socio-economic impacts of the MPAs, including interviews with 21 fishers to understand impacts to their livelihoods; in addition to the development of a habitat valuation model that highlighted how multiple habitats contribute to the local fishing economy. To survey changes in the ecology of the sedimentary habitats inside three different MPAs and in openly fished control sites, over 400 hours were spent at sea using a combination of towed videos, baited videos and grab surveys. This resulted in 222 hours of footage from towed and baited videos from which 61 mobile and sessile taxa were recorded from 8 different phyla, and a further 263 taxa were identified from the sediment grab samples across 11 phyla.

This thesis has highlighted the reliance of fishing fleets on the range of marine habitats in Jersey's waters and the greater biodiversity supported by sedimentary habitats in the MPAs compared to near-by fished Open Controls. In chapter 1, the current literature on MPAs was explored and knowledge gaps identified that could be addressed through research of MPAs that follow the whole-site approach in Jersey. In chapter 2, Jersey's marine habitats were assessed for their value to Jersey and French fisheries that highlighted the importance of protecting multiple habitats, not just features of conservation concern, within MPAs to safeguard fishing economy. Then in chapter 3, the impact of the MPAs on Jersey fisher livelihoods was examined using interviews that indicated the MPAs had yet had an impact on fishing areas, catch or fisher wellbeing. In fact, wider concerns relating to issues outside of the MPAs were raised, such as local declines in high value target species and conflict with French fishing gear. Identified in chapters 2 and 3 was the value of shellfish to local fishers, particularly crab and lobster as the fleet is primarily

comprised of static potting vessels. The MPAs, whilst not established as a fisheries management tool, do confer an advantage to static gear fishers as the ecology of the sites support crab and lobster during essential life history stages. The removal of bottom-towed fishing gear provides potential for recovery of the ecosystem.

Chapters 4 and 5 assessed the ecological changes following the exclusion of bottom-towed fishing. Mixed sediment habitats, such as those targeted by bottom-towed fishing practices, were found to have greater biodiversity (greater numbers of mobile and infaunal taxa, and greater cover of structure forming species) within the MPAs compared to Open Controls. In chapter 4, greater numbers of mobile taxa were recorded within all three MPAs compared to Open Controls. Sediment properties showed mixed results between treatments in chapter 5, with little difference observed in particle size and carbon content between treatments. While there was greater biodiversity of infaunal assemblages in the MPAs, this was only statistically significant for the older Southeast MPA highlighting the importance of age when considering the ecological success of a MPA.

Finally, potting surveys were used in chapter 6 to investigate the response of key crustacean species to the exclusion of bottom-towed fishing. No positive effect was observed three years following the designation of the MPAs; with declines observed over time for lobster that reflected local landings. Combined, the findings from this thesis highlight the following key areas for discussion.

7.2. Towards an ecosystem based approach to fisheries management Chapter 2 highlighted the importance of an ecosystem-based approach to fisheries management as fishery species utilise multiple habitats across their life cycle and therefore multiple habitats contribute to the combined annual economic value (~£14

million) of Jersey and French fisheries. The five key commercial species (lobster, brown crab, Cancer pagurus; spider crab, Maja Homarus gammarus; brachydactyla; scallop, Pecten maximus and whelk Buccinum undatum) are all benthic species and therefore the management and protection of benthic habitats is fundamental to securing sustainable fisheries in Jersey. The results from this chapter highlighted the role of subtidal sediments in supporting both Jersey and French fisheries as all five key commercial species utilise this habitat in at least one stage of their life histories. Identified were large areas of valuable sedimentary habitats (subtidal sediment and maerl) outside of the MPAs. Subtidal sediment may be supporting the highly valuable lobster in their juvenile stage and brown crab during spawning or migration to spawning grounds (Bakke, 2019; Howard & Bennett, 1979b), in addition to supporting various life stages of scallop, whelk and spider crab (chapter 2). This association with sedimentary habitats may be particularly important for consideration in brown crab management as brown crabs co-occur in the same habitat as scallops (Jenkins et al., 2004) and are therefore often reported as bycatch in scallop dredges (Veale 2000, Öndes, Kaiser and Murray, 2016).

Also worthy of consideration in ecosystem based fisheries management is the evidence that berried brown crab move into deeper water to spawn (Öndes, Kaiser and Murray, 2016). This is an important concern for management as bottom-towed fishing beyond Jersey's territorial waters, on sediments where berried females are sheltering during gestation, may have implications for recruitment. Therefore, internationally agreed temporal and spatial restrictions are needed during spawning seasons to allow berried (egg carrying) females to reach spawning grounds.

The sustainability of commercial fisheries is intrinsically linked to ecosystem health as the species they target depend on various benthic habitats for nursery, foraging or spawning grounds (Kritzer et al., 2016; Seitz et al., 2014; Stewart & Howarth, 2016). As Jersey fisheries are dependent on benthic species, seabed integrity and habitat connectivity are essential to maintaining populations at a viable level (Kritzer et al., 2016; Seitz et al., 2014). Therefore, the inclusion of multiple habitats within MPAs will be fundamental in supporting life stages of fish and shellfish, including those of commercial interest (Kritzer et al., 2016; Stewart & Howarth, 2016). Life histories need to be considered to maximise fishery benefits, such as nursery habitat (Seitz et al., 2014) and larval dispersal (Krueck et al., 2017). This research demonstrates that the type of habitats that underpin static fisheries are not fully represented within the MPAs in Jersey (chapter 2).

## 7.3. Indicators of biodiversity recovery

In addition to supporting commercial fisheries, sedimentary habitats were shown to have a greater number of mobile taxa recorded inside all three MPAs compared to those open to bottom-towed fishing (Chapter 4). Also observed was a greater abundance of juvenile bream (*Spondyliosoma cantharus*) inside the MPAs, evidencing the nursery role of the MPAs for this species. Nursery areas tend to be highly productive (Beck et al., 2001; Heck et al., 1995) and complex habitat structures improve the survivorship of juvenile species (Bradshaw et al., 2003).

Structural complexity was investigated in chapter 5 through assessing the coverage of structure forming taxa (seagrass, maerl and *Lanice conchilega*) in addition to surveying biodiversity metrics (species richness, abundance and assemblage composition) of epifauna and infauna. Differences in biodiversity of the epibiota and infaunal assemblages between treatments were variable across MPA locations and

habitat types. The number of taxa and individuals of epibiota assemblages were not greater within the MPAs compared to open controls after three years of protection at the Ecrehous and Minguiers, nor after six years of protection at the Southeast, but the coverage of structure forming taxa was greater overall inside the MPAs, which, in time, may allow for the development of more diverse epibiota communities within the MPAs (Davies et al., 2021a; Sheehan et al., 2013a). Timescales for full recovery (where a plateau of species richness and abundance with a distinct species assemblage is reached) is unknown, but the recovery of benthic communities and fish biomass may continue for decades (Babcock et al., 2010; Davies et al., 2021a; Russ & Alcala, 2004). The greater coverage of structure building taxa, combined with the visual survey method of the towed video, may have led to an underestimation of the epibiota diversity, especially where high seagrass cover occurred. Therefore, other methods may be more appropriate for assessment of these habitats where the ability to record species is less affected by the habitat type, such as baited videos to survey mobile species diversity or grab surveys to assess infaunal diversity.

The species inhabiting sediments are a major food source for higher trophic levels, including commercial species (Snelgrove, 1998). Protection of sedimentary habitats from bottom-towed fishing may allow for the development of greater biodiversity, which in turn may benefit four of the five key commercial species in Jersey that are primarily foraging and scavenging species (lobster, brown crab, spider crab and whelk). The diversity and abundance of infaunal taxa were greater in the MPAs compared to Open Controls, however this difference was only statistically significant between the oldest MPA and the controls. This again highlights the importance of considering time frames when assessing the ecological effectiveness of a MPA and

suggests local timescales for initial recovery are between three and six years. It is important to note that some benthic assemblages may take longer to recover than others (Babcock et al., 2010; Duarte et al., 2020). This may be particularly true for biogenic habitats that form over long time periods, such as maerl that grows at a rate of 1 mm per year (Wilson et al., 2004). The proportion of live maerl may provide an indication of the associated diversity and abundance (Hall-Spencer, 1998). Live maerl creates a biofilm that is thought to act as an environmental cue that improves the settlement of certain species (Kamenos et al., 2004a) and healthy, 'pristine' maerl beds are thought to harbour a greater biodiversity than degraded beds (Hall-Spencer et al., 2003). The results in chapter 5 support this as infaunal diversity (number of taxa and individuals) was positively correlated with the weight of live maerl in each sample. As maerl can be fragmented and smothered by bottom-towed fishing, maerl beds, and their associated biodiversity, are particularly vulnerable in unprotected areas. There are large areas of maerl beds currently outside of the MPAs (chapter 2) that have the potential to increase the biodiversity on the seafloor if protected, but it may take many years, if not decades, for this to be realised due to the slow growth rate of maerl.

#### 7.4. Shortfalls in MPA placement for fisheries

Results from chapter 3 highlighted the importance of shellfish to local fishery economy and that landings of both lobster and brown crab were in decline and should therefore be a management concern for local fisheries managers. Landings may indicate a change in the stock but do not provide insights into the factors causing change. The results from chapter 3 suggest that the MPAs have not displaced many Jersey mobile vessels but that there had been an increase in potting effort within the MPAs, as indicated by interview respondents. One of the points raised during interviews by several static fishers was the small size of the Ecrehous MPA. Size is a key factor in the ecological success of MPAs (Edgar et al., 2014), reinforcing concerns over the potential for this MPA to contribute to sustainable fisheries. However, the smallest MPA (Ecrehous) may be providing a nursery ground for juvenile lobster as the number of below MLS individuals in the MPA was far greater than the Open Control. Based on the results of the potting study it would appear that the MPAs have not yet had a significant effect on crustacean populations, but due to the generation time of crab and lobster species, an effect is unlikely to have been noticed after three years of protection from bottom-towed fishing. Results from the potting study (chapter 6) showed lobster landings to be in decline locally, while landings of brown crab were in decline across the region (Cefas, 2020a; Cornwall IFCA, 2018a) which was reflected in the low number of brown crab caught across the treatments and locations (chapter 6). Brown crab and lobster are associated with sedimentary and intertidal habitats in the juvenile stages (chapter 2), which may make them more vulnerable to bottom-towed fishing impacts.

#### 7.5. MPAs alone are not enough

Where fisheries management is concerned, MPAs can be used to enhance the ecological condition of the habitats supporting fisheries species but should not be used in place of other management measures, rather to compliment them (Hilborn *et al.*, 2004). MPAs will also be more effective for multi-species or sedentary stocks, in Jersey's case, scallop and whelk would be most likely to benefit from appropriately placed MPAs more than crab and lobster as, once settled, they have limited mobility. While the MPAs in Jersey protect all features within the boundary, and are therefore likely to have a more meaningful impact on conservation than feature based MPAs

(Solandt et al., 2020), they alone will not improve the sustainability of key fish stocks if target species continue to be exploited at unsustainable levels. As potting is permitted in all areas of the reefs, it is questionable whether the exclusion of bottomtowed fishing would be enough to improve these particular stocks as the selective fishing pressure on crustaceans may mask any benefits from the removal of bottomtowed fishing. The level of potting sustained by the offshore reefs needs to be quantified and sustainable fishing limits determined to create a sustainable future for potting fisheries.

## 7.6. Promoting sustainable fisheries

There is little monetary benefit for local fishers using sustainable practices in Jersey and, historically, the majority of scallop landings were from dredging vessels (chapter 3). However, the landings from scallop divers are beginning to increase. The development of a reserve seafood label in Lyme Bay has allowed those fishing in the MPA to sell their catch for a premium (Blue Marine Foundation, 2016; Rees et al., 2021b), further incentivising the use of sustainable fishing practices. A similar label that could be used by Jersey fishers, whose effort for sustainable, low impact fishing would be rewarded though achieving a higher market price point than species caught using unsustainable methods. There is currently a hand-dived scallop label being developed by the Blue Marine Foundation that scallop divers can use to help them sell their catch for a premium but there is yet to be a label that promotes sustainable catch from other metiers or from within the MPAs.

The main blockage for sustainable fisheries in Jersey is the persistent issue for fisheries managers of not knowing where the smaller vessels (which make up the majority of the Jersey fleet) undertake the most fishing effort. The roll out of iVMS units (a form of Vessel Monitoring System) on board all vessels in Jersey is in its

early phases (pers. comm. Government of Jersey Marine Resources) which will allow fisheries managers to understand fishing effort and location in relation to MPAs. If there were a form of 'traceability' for sustainably caught fish, this would allow alternative sales strategies to be explored using a local sustainability brand designed to support Jersey's small-scale fishers.

## 7.7. Global context

In the midst of global biodiversity decline (Dasgupta, 2021; Worm et al., 2006), it is essential to maximise recovery of biodiversity wherever possible. While NTZs are advocated as the best option for conserving biodiversity (Sala & Giakoumi, 2018), MPAs that follow the whole-site approach offer a compromise that allows the recovery of biodiversity while also allowing fisheries to benefit. As evidenced in Lyme Bay, UK, were the exclusion of bottom-towed fishing has been shown to increase commercial fish species abundance by 370% (Davies et al., 2021a), while also allowing low-impact fishing to continue. The results of this current study have implications for global MPA management, especially in the UK where many are feature-based, as it is likely that these MPAs are not reaching their full potential to recover biodiversity if sediment areas between the conservation features are not afforded the same protection. The spatial extent of bottom-towed fishing is beginning to be managed in areas outside of national jurisdiction, typically in the deep sea, to protect vulnerable marine ecosystems (VMEs; (FAO, 2009)). The concept of VMEs has emerged from discussions at the United Nations General Assembly, with criteria for classification relating to: the functional importance of an area (supporting life history stages of commercial or threatened species); association with long-lived or slow growing species; the fragility of the ecosystem; and the structural complexity of the system. However, there is little reference to the importance of habitat

connectivity or merit of protecting mosaics of habitats in areas outside of national jurisdiction, which should also be an important consideration for maintaining biodiversity and sustainable fisheries for many nations.

Marine ecosystems are further threated by the effects of climate change (Brown et al., 2010; Duarte et al., 2013; Euskirchen et al., 2013). Protecting the seabed from bottom-towed fishing can improve resilience of benthic communities to climatic events (Sheehan et al., 2021). Additionally, there is emerging research on the release of carbon stored in sediments from bottom-towed fishing (Epstein & Roberts, 2022; Sala et al., 2021), and there are calls to exclude this form of fishing as a way of combatting climate change. In chapter 5, there was no significant difference in organic carbon content of the sediments between the MPAs and Open Controls, with location and habitat type being the main drivers of organic carbon content. As this is an emerging area of research and there are typically few areas that have been protected from bottom-towed fishing globally, it is relatively unknown under what time scales organic carbon can be expected to accumulate in the sediments of MPAs (Epstein & Roberts, 2022).

In the face of data paucity, it is recommended to take a precautionary approach and prioritise the inclusion of habitats with high organic carbon content within MPAs, such as seagrass and maerl, and continue monitoring to allow for temporal estimations of organic carbon accumulation. It is also recommended to expand this research to assess the inorganic carbon alongside the organic to understand the total carbon associated with marine sediments. With many industries and companies turning to carbon credits to offset their carbon emissions (Wylie et al., 2016), it is

extremely important to improve the accuracy of estimated carbon content in habitats to ensure the credibility of carbon credits.

## 7.8. Implications and recommendations for management

To support effective marine management, the socio-economic benefits from marine systems need to be considered not just in conservation policy but also within social and economic policy (Pittman et al., 2019; Rees et al., 2020a). Within the coastal community context, National Marine Parks have been proposed as a new tool to connect social, economic and health benefits that flow from marine ecosystems to the more terrestrially focused economic growth and productivity agenda of local councils (Pittman et al., 2019; Rees et al., 2020a). Stakeholders and NGOs in Jersey have discussed the possibility of incorporating the conservation and development of the marine sector in the Island Plan through plans for a Marine Park that would cover 30% of Jersey's territorial waters. Current aspirations for a Marine Park, whilst not a legal designation, have been recommended by the Blue Marine Foundation as a vehicle to facilitate the expansion of the current MPAs to cover all shallow coastal habitats, from which mobile gear would be prohibited (Blue Marine Foundation, 2022). Jersey currently has less than the 10% MPA coverage target as set by the UN under Sustainable Development Goal (SDG) 14 (UN General Assembly, 2015). However, if a suggested expansion of the MPAs is taken forward, through the drive to create a Jersey Marine Park, this would result in roughly 30% of seabed protected from mobile gear, in line with current EU recommendations (European Commission, 2020). This would make Jersey an ideal case study to determine differences in ecosystem service supply under 30% coverage of MPAs, with baseline data available from previous research to compare against. The outcomes of this could inform both national and international MPA targets.

Based on the findings from this research there are several recommendations to improve the current management of marine resources in Jersey to secure both conservation goals and fisheries objectives. These recommendations are listed in Table 7.1. along with the requirements necessary to implement each one. For each proposed management change, it is vital to involve stakeholders at each stage of the implementation process to improve both understanding and acceptance as well as tailoring management to minimise social impacts where it does not detract from the conservation goals. Adaptive management could be achieved through the continued monitoring of ecosystem condition with *a priori* thresholds that trigger intervention measures. A feedback loop in which all of these elements are taken into account is illustrated in Figure 7.1.

Table 7.1. Recommendations and requirements for best practice in marine management to meet conservation goals and fisheries objectives.

Recommendation	Requirement		
Fisheries management and	A network of Marine Protected Areas that		
conservation efforts should be	exclude bottom-towed fishing across a		
ecosystem based and balanced	mosaic of habitats. With all habitats		
for long term sustainability.	supporting the life stages of commercial		
	species adequately represented within these		
	MPAs.		
Fisheries management should be	Thresholds of 'good status' that, through		
adaptive.	monitoring, trigger adaptive management if		
	exceeded. To mitigate the impacts of a		
	changing climate and changing fisheries.		
Sustainable gear limits should be	Limits on static gear to reduce pressure on		
set.	commercial species. In particular, stricter		
	pot limits per vessel to allow for recovery of		
	the lobster stock that is in decline locally.		

Seasonal closures based on life	Seasonal restrictions on bottom-towed
history requirements.	fishing in migration corridors of brown crab
	and spider crab to protect them from
	mechanical damage from dredges.
Stakeholder engagement should	Stakeholders to be considered and
be incorporated into all levels of	consulted at every stage when deciding on
management.	new management measures such as MPAs.
	To improve understanding of measures and
	to increase compliance and self-policing.
Improved value for sustainably	Introduce VMS systems on all vessels to
caught fish and shellfish.	improve traceability of catch. This will allow
	for promotion of sustainably caught fish
	within MPAs, adding value to catch.



Figure 7.1. A decision-making flow diagram for implementing new management measures as outlined in table 7.1.

# 7.9. Ongoing monitoring and further research

The challenging field conditions presented by Jersey's waters are problematic for efficient survey work. Methods that capture the key information needed to inform management decisions should be continued with an emphasis placed on greater replication in favour of continuing all methods in their current design. To continue monitoring the recovery of biodiversity of sedimentary habitats within the MPAs, it is recommended that baited video and grab surveys are maintained. Further potting surveys that include the older Southeast MPA would help to improve understanding of how MPAs that follow the whole-site approach effect commercial crustacean abundance and whether abundance will increase over time despite continued static fishing within the MPA. This could be achieved in collaboration with local fishers to increase monitoring of commercial species abundance and improve engagement with the local fishing fleet. As French fleets will continue to fish in Jersey waters for the foreseeable, it is also recommended that the interviews are repeated to capture the impacts experienced by French fishers to tell a more complete story of how the MPAs affect fisher livelihoods. Lastly, there is a need to understand the habitat associations of commercial species across their life stages in fuller detail, especially for lobster that contributes significantly to local fishing economy, and ensure essential habitats are adequately represented within MPAs.

## 7.10. Concluding remarks

This thesis has investigated both the ecological and socio-economic aspects of MPAs in Jersey that follow a whole-site approach. The overarching findings of this thesis have demonstrated the importance of protecting multiple habitats from destructive fishing practices to support both conservation and fisheries objectives. Socio-economic studies identified the economic value of multiple habitats to Jersey fisheries and, in particular, highlighted the role of subtidal sediment in supporting all five of the key commercial species (lobster, brown crab, spider crab, scallop and whelk). Ecological surveys focused on sedimentary habitats and showed the number of mobile species to be greater within the MPAs compared to Open Controls. Infaunal species richness was also greater in the MPAs but this was only significant in the older MPA, and habitat type played a significant role in the diversity and

abundance of infaunal species. Continuation of surveys to monitor ongoing change in the MPAs is recommended to further understand the time frames of recovery across different habitat types. The MPAs appear to be supporting the commercial species bream during juvenile stages, and the MPA at the Ecrehous may be supporting lobsters that are below MLS, but within the time frame of the study there was no measurable benefit of the MPAs to commercial crustacean species abundance. Currently lobster is the backbone of the Jersey fishing fleet and declines in this species are a concern for local fishers. Ongoing monitoring of economically valuable species, and improved understanding of their habitat requirements, is a necessary step in improving management to secure sustainable fisheries. The research in this thesis adds to the evidence base available to decision makers that could be used to help guide the designation and management of MPAs both in Jersey and further afield to support SDG goal 14 to conserve and sustainably use the oceans.

# 8. Appendix A

Table A.8.1. Literature identified during data collection on life history and habitat associations.

Ager, O. E. D. 2008. *Buccinum Undatum* Common Whelk. In Tyler-Walters H. and Hiscock K. (Eds) *Marine Life Information Network: Biology and Sensitivity Key Information Reviews*, [on-Line]. Plymouth: Marine Biological Association of the United Kingdom. [Cited 27-11-2020].

Blunden, G., W. F. Farnham, N. Jephson, R. H. Fenn, and B. A. Plunkett. 1977.

The Composition of Maërl from the Glenan Islands of Southern Brittany.

Botanica Marina 20(2):121–26. doi: 10.1515/botm.1977.20.2.121.

Brown, C. G., and D. B. Bennett. 1980. Population and Catch Structure of the Edible Crab (*Cancer pagurus*) in the English Channel. *ICES Journal of Marine Science* 39(1):88–100. doi: 10.1093/icesjms/39.1.88.

Corgos, Antonio, Cristina Bernárdez, Paz Sampedro, Patricia Verísimo, and Juan Freire. 2011. "Spatial Structure of the Spider Crab, Maja Brachydactyla Population: Evidence of Metapopulation Structure." Journal of Sea Research 66(1):9–19. doi: 10.1016/j.seares.2011.04.011.

Galparsoro, Ibon, Ángel Borja, Juan Bald, Pedro Liria, and Guillem Chust. 2009. Predicting Suitable Habitat for the European Lobster (*Homarus Gammarus*), on the Basque Continental Shelf (Bay of Biscay), Using Ecological-Niche Factor Analysis. *Ecological Monitoring* 220:556–67. doi:

10.1016/j.ecolmodel.2008.11.003.

Gonzalez-Gurriaran, E., and J. Freire. 1994. Movement Patterns and Habitat Utilization In The Spider Crab *Maja-Squinado* (Herbst) (Decapoda, Majidae) Measured By Ultrasonic Telemetry. *Journal of Experimental Marine Biology and Ecology* 184(2):269–91. doi: 10.1016/0022-0981(94)90009-4.

Hall, S. J., Basford, D. J., Robertson, M. R., Raffaelli, D. G., & Tuck, I. (1991).

Patterns of recolonisation and the importance of pit-digging by the crab Cancer

pagurus in a subtidal sand habitat. Marine Ecology Progress Series, 72(1–2),

93–102. https://doi.org/10.3354/meps072093

Hall, S. J., M. R. Robertson, D. J. Basford, and R. Fryer. 1993. "Pit-Digging by the Crab Cancer Pagurus: A Test for Long-Term, Large-Scale Effects on Infaunal Community Structure." The Journal of Animal Ecology 62(1):59–66. doi: 10.2307/5482.

Hauton, C., J. M. Hall-Spencer, and P. G. Moore. 2003. An Experimental Study of the Ecological Impacts of Hydraulic Bivalve Dredging on Maerl. *ICES Journal of Marine Science* 60(2):381–92. doi: 10.1016/s1054-3139(03)00015-8.

Hosie, A. M. 2009. *Maja Brachydactyla* Common Spider Crab. In Tyler-Walters

H. and Hiscock K. (Eds) Marine Life Information Network: Biology and Sensitivity

Key Information Reviews, [on-Line]. Plymouth: Marine Biological Association of

the United Kingdom. [Cited 27-11-2020]. Retrieved

(https://www.marlin.ac.uk/species/detail/2225).

Howard, A. E., and D. B. Bennett. 1979. The Substrate Preference and Burrowing Behaviour of Juvenile Lobsters (*Homarus Gammarus* (L.)). *Journal of Natural History* 13(4).

Howarth, L. M., H. L. Wood, A. P. Turner, and B. D. Beukers-Stewart. 2011. "Complex Habitat Boosts Scallop Recruitment in a Fully Protected Marine Reserve." Marine Biology 158(8):1767–80. doi: 10.1007/s00227-011-1690-y.
Irlandi, E. A., B. A. Orlando, and W. G. Ambrose. 1999. Influence of Seagrass Habitat Patch Size on Growth and Survival of Juvenile Bay Scallops, *Argopecten Irradians Concentricus* (Say). *Journal of Experimental Marine Biology and Ecology* 235(1):21–43. doi: 10.1016/s0022-0981(98)00185-3.

Kamenos, N. A., P. G. Moore, and J. M. Hall-Spencer. 2004. Nursery-Area

Function of Maerl Grounds for Juvenile Queen Scallops Aequipecten Opercularis

and Other Invertebrates. Marine Ecology Progress Series 274:183-89. doi:

10.3354/meps274183.

Lawton, Peter, and Kari L. Lavalli. 1995. Chapter 4. Postlarval, Juvenile,

Adolescent, and Adult Ecology. *Biology of the Lobster* 47-88

Linnane, Adrian, Brendan Ball, Brian Munday, and John P. Mercer. 2000. On the

Occurrence of Juvenile Lobster *Homarus Gammarus* in Intertidal Habitat.

Journal of the Marine Biological Association of the United Kingdom 80(2):375-

76. doi: 10.1017/S0025315499002039.

Marshall, Charlotte, and Emily Wilson. 2008. Great Scallop, (Pecten Maximus) -

MarLIN – Marine Life Information Network Biology and Sensitivity Key

Information Review. Aquaculture International 305–18.

Neal, K. J., and E. Wilson. 2008. Cancer pagurus Edible Crab. In Tyler-Walters

H. and Hiscock K. (Eds) Marine Life Information Network: Biology and Sensitivity

Key Information Reviews, [on-Line]. Plymouth: Marine Biological Association of

the United Kingdom. [Cited 27-11-2020]. Retrieved

(https://www.marlin.ac.uk/species/detail/1179).

Seitz, R. D., Wennhage, H. kan, Bergstrom, U., Lipcius, R. N., & Ysebaert, T.

(2014). Ecological value of coastal habitats for commercially and ecologically

important species. ICES Journal of Marine Science, 71(3), 648–665.

Sheeny, Matt R. J., and Andrew E. Prior. 2008. Progress on an Old Question for Stock Assessment of the Edible Crab *Cancer pagurus*. *Marine Ecology Progress Series* 353:191–202. doi: 10.3354/meps07229.

Unsworth, Richard K. F., and Leanne C. Cullen-Unsworth. 2015. Pen Llŷn a'r

Sarnau Special Area of Conservation (SAC) Porthdinllaen Seagrass Project : A Review of Current Knowledge. 2–19.

Valentinsson, D., F. Sjodin, P. R. Jonsson, P. Nilsson, and C. Wheatley. 1999. Appraisal of the Potential for a Future Fishery on Whelks (*Buccinum undatum*) in Swedish Waters: CPUE and Biological Aspects. *Fisheries Research* 42(3):215– 27. doi: 10.1016/s0165-7836(99)00050-8.

Wahle, Richard A., and Robert S. Steneck. 1991. Recruitment Habitats and

Nursery Grounds of the American Lobster Homarus americanus: A

Demographic Bottleneck? *Marine Ecology Progress Series* 69(3):231–43.

Wilson, E. 2008. Homarus gammarus Common Lobster. In Tyler-Walters H. and

Hiscock K. (Eds) Marine Life Information Network: Biology and Sensitivity Key

Information Reviews, [on-Line]. Plymouth: Marine Biological Association of the

United Kingdom. [Cited 27-11-2020]. Retrieved

(https://www.marlin.ac.uk/species/detail/1171).

# 9. Appendix B – Fisher Questionnaire

Please make the interviewee aware of the following and provide copies of the information sheet and a reference copy of the consent form:

You are invited to participate in research being conducted by Samantha Blampied as part of a project concerning the impact of No Mobile Gear Zones (NMGZs). This interview forms part of a study being carried out by University of Plymouth and the Blue Marine Foundation to evaluate the impact of the NMGZs on fisheries stocks and fisher wellbeing. This work is funded by the Blue Marine Foundation.

For the purpose of this project the No Mobile Gear Zones refer to the zones designated at the Ecrehous and the Minquiers in October 2017. Please see map below.



The interview should last no longer than 45min -1hr. The interview will be **recorded** and notes taken. Answers given will **remain confidential** and only anonymised and grouped data will be used in the analysis and reporting. The name of your vessel will be asked for but will not be published, this information will be matched to vessel size records and log book data. By taking part in this interview you are consenting to your data being used as part of this study. You have the right to withdraw from this interview at any time or to request your data is removed from the project up to two weeks after the interview. You do not have to answer any individual question that you do not wish to answer.

Ticking the following box indicates that you understand the information provided above, that you willingly agree to participate and that you may withdraw your consent at any time and discontinue participation. You can withdraw or amend your responses to this survey by contacting Samantha Blampied

Email: <a href="mailto:samantha.blampied@plymouth.ac.uk">samantha.blampied@plymouth.ac.uk</a>

Date: \_\_\_\_\_ No. \_\_\_\_\_

Fishing Activity

- 1. Vessel name: \_\_\_\_\_
- 2. Home port (if not St. Helier) \_\_\_\_\_
- 3. Are you a member of the JFA/JIFA/Other?
- 4. Would you say that your catch has increased/decreased/stayed the same compared with 2017? *Please circle one*

5. Have you fished at the Ecrehous or Minquiers in the last 5 years? If yes, which years and for what species? Was it a good or bad year?

	2014	2015	2016	2017	2018
Ecrehous					
Minquiers					

No Mobile Gear Zones (please ask for answers whether the interviewee fishes there or not)

Interview

6. On a scale of 0-10 *where 1 = 'Not at all' and 10 = 'Completely'*, to what extent do you support the NMGZ at the **Minquiers**?



7. On a scale of 0-10 *where 1 = 'Not at all' and 10 = 'Completely'*, to what extent do you support the NMGZ at the **Ecrehous**?

1	2	3	4	5	6	7	8	9	10
Not	at						Comple	etely	
all									

8. On a scale of 0-10 *where 1 = 'Not at all' and 10 = 'Completely'*, to what extent have the NMGZs changed **where** you fish?

1		2	3	4	5	6	7	8	9	10
Not	a	t						Complet	ely	
all										

9. On a scale of 0-10 *where 1 = 'Not at all' and 10 = 'Completely'*, to what extent have the NMGZs changed **how often** you fish?

1	2	3	4	5	6	7	8	9	10
Not a	ıt					(	Complet	ely	
all									

10. Have there been any changes in the quality of the catch as a result of the NMGZs? Yes/No (please circle one)

Answers to these questions will remain strictly confidential. At no point will economic details be made available other than in an aggregated form.

Job satisfaction

11. On a scale of 1-10 where 1 is 'Completely dissatisfied' and 10 is 'Completely satisfied' how satisfied were you with your fishing in 2017 and 2018? (i.e. using the gear you wanted, where you wanted and catching plenty of fish).

2017										
2018										
	1	2	3	4	5	6	7	8	9	10
	Compl	etely						Comple	tely sati	sfied
	dissati	sfied								

12. If there has been a change between 2017 and 2018 please indicate why. Ask for details

# <u>Conflict</u>

13. On a scale of 1-10 where 1 is 'No conflict' and 10 is 'High levels of conflict' what level of conflict did you experience in 2017 and 2018? (such as disagreements with other fishermen, damage to gear, loss of gear, or other instances of conflict)

2017										
2018										
	1	2	3	4	5	6	7	8	9	10
	No cor	nflict						High lev	els of c	onflict

14. If there has been a change between 2017 and 2018 please indicate why. Ask for details

# Health & Wellbeing

15. On the scale of 1-10 where 1 is 'No stress' and 10 is 'High levels of stress', how would you rank your level of stress in 2017 and 2018? (prompts: resilience to highly stressful circumstances or actual experiences of stress, anxiety, anger, frustration).

2017										
2018										
	1	2	3	4	5	6	7	8	9	10
	No stre	ess						High lev	els of st	ress

16. If there has been a change between 2017 and 2018 please indicate why. *Ask for details* 



## **Compliance**

17. On the scale of 1-10, where 1 = complete compliance and 10 = complete noncompliance, how would you rank general levels of compliance with the NMGZs. (non-compliance is when you witnessed or knew personally fishermen or fishing vessels undertaking activities prohibited by the fisheries bye-laws).

1	2	3	4	5	6	7	8	9	10
Comple	te						Comple	te	non-
complia	nce						complia	nce	

18. Please add any details. What sort of non-compliance and how often?

### **Economics**

Answers given will **remain confidential** and only anonymised and grouped data will be used in the analysis and reporting. This data will be used to help determine the value of the fishing industry in Jersey.

19. What is your approximate annual turnover from fishing currently? (*income before deduct costs*)

£0-£10,000	£10001-£20,000	£20,001-£30,000	£30,001-£40,000				
£40,001-£50,000	£50,001-£60,000	£60,001-£70,000	£70,001-£80,000				
£80,001-£90,000	£90,001-£100,000	£100,001-£110,000	£110,001-£120,000				
£120,001-£130,000	£130,001-£140,000	£140,001-£150,000	£150,001-£160,000				
£160,001-£170,000	£170,001-£180,000	£180,001-£190,000	£191,000-£200,000				
£200,000 +	Please specify within £10,000 or exact if known:						

### Income and income satisfaction

20. On the scale of 1-10, where 1 = completely dissatisfied and 10 = completely satisfied, how satisfied were you with your net fishing income in 2018? (income minus expenses). January-January? Financial year??

2017										
2018										
	1	2	3	4	5	6	7	8	9	10
	No at a	all						Comple	tely	

21. If there has been a change between 2017 and 2018 please indicate why. Ask for details

22. Please could you indicate what % of your annual turnover is dedicated to:

Fuel	Harbour	Gear	Crew	Capital	Quota	Profit	Other
	dues	(maintenance	No.	payments	licence		(please
		and	Crew:	e.g. Boat	(towed		name)
		replacement)		and	gear)		
				licence			
				payments			

23. What % of your landings did you sell to the following in 2018? And approx. price per kilo/time of year?

	Fish	Restaurant	France	Private	Other/export?
	vendor			buyer	
Lobster					
Brown Crab					
Spider Crab					
Scallop					
Whelk					
Wet fish					
Other?					

24. How, if at all, has your sales strategy changed from 2017?





## **Demographics**

The following questions are required to validate the study. Your cooperation in answering these questions is greatly appreciated. Please remember that the answers are anonymous and confidential, and only aggregated data will be used for the project.

Gender:	Male	Female			
Age:	a) 18-24	b) 25-34	c) 35-44	d) 45-54	e) 55-

### 10. Appendix C

Table C.1. SIMPER analysis results showing the differences in average abundance (Av.Abund) and dissimilarity (Av.Diss) between MPA and Open Control treatments at each Location (Ecrehous, Southeast and Minquiers) and Year (2018, 2019 and 2020). Note there were no Open Control replicates for the Southeast in 2018 and so no comparison has been made. Species contribution cut-off was set at 80%.

Species	MPA Av.Abund	Open Control Av.Abund	Av.Diss	$\mathrm{Diss}/\mathrm{SD}$	$\operatorname{Contrib}\%$	Cum.%
Minquiers 2018						
Ammodytidae sp.	0.63	0.70	16.47	0.60	17.41	17.41
Shoaling fish	0.00	0.30	12.63	0.45	13.35	30.76
Maja brachydactyla	0.50	0.50	11.55	0.58	12.21	42.97
Cereus pedunculatus	0.00	1.00	11.28	0.50	11.93	54.90
Pagurus sp.	0.88	0.00	8.61	0.71	9.10	64.00
Actinontorugii co	0.25	0.20	8.39	0.44	8.87	72.86
Actinopterygii sp.	0.50	0.00	5.24	0.00	5.98	(0.04 84.40
Minamiona 2010	0.10	0.10	0.04	0.40	0.00	04.40
Maja brachydactyla	0.76	0.77	38.68	1.06	43.53	43 53
Cereus pedunculatus	0.00	0.73	17.76	0.63	19.99	43.53 63.52
Aplysia punctata	0.76	0.05	11.81	0.42	13.29	76.80
Scyliorhinus canicula	0.12	0.23	7.79	0.54	8.76	85.57
Minquiers 2020						
Cereus pedunculatus	0.25	1.31	19.94	0.78	21.47	21.47
Maja brachydactyla	0.13	0.88	16.41	0.82	17.68	39.15
Actinopterygii sp.	0.69	0.31	13.26	0.73	14.28	53.43
Pagurus sp.	0.06	0.50	6.38	0.52	6.87	60.31
Labrus bergylta	0.31	0.19	5.54	0.45	5.96	66.27
Aplysia punctata	0.19	0.06	4.23	0.35	4.55	70.82
Wrasse sp.	0.13	0.19	3.92	0.53	4.22	75.04
Snoaling fish	0.13	0.06	3.50	0.37	3.11	78.81
Seynorminus canicula	0.25	0.00	0.07	0.29	0.00	02.44
Ecrehous 2018	0.70	0.00	11.07	0.40	11.00	11.00
Vracco en	0.70	0.00	11.25	0.49	11.90	22.07
Sevijorhinus canicula	0.40	0.17	9.73	0.33	10.30	33.37
Stylla clava	0.80	0.33	9.56	0.72	10.11	43.48
Maja brachydactyla	0.10	0.50	7.92	0.76	8.37	51.85
Symphodus melops	0.50	0.33	7.33	0.72	7.76	59.61
Acquipecten opercularis	0.10	0.00	6.49	0.28	6.87	66.48
Actinopterygii sp.	0.50	0.00	4.70	0.56	4.97	71.45
Mullus sp.	0.80	0.00	4.08	0.34	4.32	75.76
Trisopterus luscus	0.00	0.17	3.54	0.37	3.74	79.51
Labrus bergylta	0.30	0.00	2.52	0.42	2.67	82.17
Ecrehous 2019						
Aplysia punctata	2.70	0.00	22.97	0.83	23.69	23.69
Maja brachydactyla	0.90	0.17	11.19	0.58	11.54	35.23
Juv. Actinopterygn Cromidule formioste	3.00	0.17	9.88	0.45	10.19	45.42
Poston maximus	0.10	2.33	9.10 7.18	0.52	9.36	54.80 62.21
Tethya citrina	0.00	0.55	6.98	0.45	7.91	69.41
Neopentadactyla mixta	1.00	0.00	6.77	0.42	6.99	76.39
Wrasse sp.	0.00	0.17	4.29	0.28	4.43	80.82
Ecrebous 2020						
Scyliorhinus canicula	1.20	0.73	15.02	0.97	18.13	18.13
Maja brachydactyla	1.20	0.82	14.26	1.02	17.21	35.34
Pecten maximus	0.70	0.82	13.74	0.83	16.59	51.93
Aplysia punctata	0.30	0.00	5.27	0.40	6.36	58.29
Styela clava	0.30	0.18	3.89	0.52	4.70	62.99
Anemonia viridis	0.20	0.00	3.67	0.30	4.43	67.42
Centrolabrus exoletus	0.20	0.00	3.14	0.45	3.79	71.21
Sepia officinalis	0.20	0.18	3.07	0.61	3.71	74.92
Cereus pedunculatus	0.30	0.00	3.05	0.42	3.68	78.60
Acthopterygn sp.	0.00	0.09	2.02	0.20	2.44	81.04
Southeast 2019	0.07	0.50	00.04	0.00	00.07	20.07
Scyliorhinus canicula	0.27	0.50	38.84	0.86	39.97	39.97
Pecten maximus	0.00	0.00	37.02	0.81	17.08	16.07
Fourthoost 2000	0.02	0.00	11.41	0.07	11.90	30.00
Southeast 2020	0.95	1.00	14 00	0.50	16.91	16.91
Pooton maximus	0.25	1.00	14.89	0.50	14.87	21.10
Crepidula fornicata	0.25	0.07	12.00	0.62	13.14	44.33
Maja brachydaetyla	0.25	0.50	11.89	0.54	13.03	57.36
Sepia officinalis	0.50	0.00	6.73	0.78	7.37	64.73
Symphodus melops	0.13	0.17	4.63	0.43	5.07	69.80
Gobius sp.	0.25	0.00	4.62	0.42	5.07	74.87
Carangidae sp.	0.00	0.33	4.31	0.43	4.72	79.59
Scyliorhinus canicula	0.13	0.33	3.51	0.57	3.85	83.44

phylum	class	order	family	Taxa
Annelida	-	-	-	Annelida spp.
	Clitellata	Haplotaxida	Naididae	Chaetogaster spp.
	Clitellata	Haplotaxida	Naididae	Naididae
	Clitellata	Haplotaxida	Tubificidae	Tubificidae
	Clitellata	Haplotaxida	Tubificidae	Tubificidae
	Clitellata	-	-	Oligochaeta
	Polychaeta	Eunicida	Dorvilleidae	Protodorvillea atlantica
	Polychaeta	Eunicida	Dorvilleidae	Dorvilleidae
	Polychaeta	Eunicida	Eunicidae	Eunicidae
	Polychaeta	Eunicida	Eunicidae	Lysidice ninetta
	Polychaeta	Eunicida	Eunicidae	Marphysa bellii
	Polychaeta	Eunicida	Eunicidae	Marphysa sanguinea
	Polychaeta	Eunicida	Eunicidae	Nematonereis spp.
	Polychaeta	Eunicida	Lumbrineridae	Lumbrineris latreilli
	Polychaeta	Eunicida	Lumbrineridae	Lumbrineris spp.
	Polychaeta	Eunicida	Oenonidae	Arabella iricolor
	Polychaeta	Eunicida	Oenonidae	Drilonereis filum
	Polychaeta	-	Arenicolidae	Arenicolidae
	Polychaeta	-	Capitellidae	Capitellidae
	Polychaeta	-	Capitellidae	Notomastus mossambicu
	Polychaeta		Chaetopteridae	Chaetopteridae
	Polychaeta		Chaetopteridae	Chaetopterus varionedat
	Polychaeta		Maldanidae	Fuclymene lumbricoides
	Polychaeta	_	Maldanidae	Euclymene robusta
	Polychaeta	-	Maldanidae	Euclymene
	Polychaeta		Maldanidae	Bhodine loveni
	Polychaeta	-	Maidandae	Polyohaotao
	Polychaeta	-	- Opholiidaa	Orbolio
	Polychaeta	-	Ophemidae	Ophena
	Polychaeta	-	Orbiniidae	Coolonidae Coolonida ampiron
	Polychaeta	-	Oroniidae	Organia fugiformia
	Polychaeta	-	Oweniidae	Owenia fusiformis
	Polychaeta	-	Demonside	Cimera harris
	Polychaeta	-	Paraonidae	Cirrophorus spp.
	Polycnaeta	Phyllodocida	Aphroditidae	Aphroditidae
	Polychaeta	Phyllodocida	Glyceridae	Glycera alba
	Polychaeta	Phyllodocida	Glyceridae	Glycera tridactyla
	Polychaeta	Phyllodocida	Glyceridae	Glyceridae
	Polychaeta	Phyllodocida	Goniadidae	Goniadidae
	Polychaeta	Phyllodocida	Hesionidae	Syllidia armarta
	Polychaeta	Phyllodocida	Nephtyidae	Nephtyidae
	Polychaeta	Phyllodocida	Nephtyidae	Nephtys caeca
	Polychaeta	Phyllodocida	Nephtyidae	Nephtys cirrosa
	Polychaeta	Phyllodocida	Nephtyidae	Nephtys hombergii
	Polychaeta	Phyllodocida	Nereidae	Nereidae
	Polychaeta	Phyllodocida	Nereididae	Nereis pelagica
	Polychaeta	Phyllodocida	Nereididae	Nereis diversicolor
	Polychaeta	Phyllodocida	Nereididae	Nereis fucata
	Polychaeta	Phyllodocida	Nereididae	Nereis spp.
	Polychaeta	Phyllodocida	Nereididae	Perinereis macropus
	Polychaeta	Phyllodocida	Nereididae	Platynereis dumerilii
	Polychaeta	Phyllodocida	Phyllodocidae	Anaitides groenlandica
	Polychaeta	Phyllodocida	Phyllodocidae	Anaitides spp.
	Polychaeta	Phyllodocida	Phyllodocidae	Eulalia bilineata
	Polychaeta	Phyllodocida	Phyllodocidae	Eumida punctifera
	Polychaeta	Phyllodocida	Phyllodocidae	Mysta picta
	Polychaeta	Phyllodocida	Phyllodocidae	Phyllodocidae

1

Grabs Taxa list

	class	order	family	Taxa
	Polychaeta	Phyllodocida	Polynoidae	Gattyana cirrhosa
	Polychaeta	Phyllodocida	Polynoidae	Harmothoe spp.
	Polychaeta	Phyllodocida	Polynoidae	Lepidonotus clava
	Polychaeta	Phyllodocida	Polynoidae	Lepidonotus pilosus
	Polychaeta	Phyllodocida	Polynoidae	Lepidonotus
	Polychaeta	Phyllodocida	Polynoidae	Malmgrenia castanea
	Polychaeta	Phyllodocida	Polynoidae	Malmgrenia lunulata
	Polychaeta	Phyllodocida	Polynoidae	Malmgrenia spp.
	Polychaeta	Phyllodocida	Sigalionidae	Pholoe minuta
	Polychaeta	Phyllodocida	Sphaerodoridae	Sphaerodoridae
	Polychaeta	Phyllodocida	Syllidae	Eusyllis blomstrandi
	Polychaeta	Phyllodocida	Syllidae	Exogone spp.
	Polychaeta	Phyllodocida	Syllidae	Odontosyllis gibba
	Polychaeta	Phyllodocida	Syllidae	Pionosyllis lamelligera
	Polychaeta	Phyllodocida	Syllidae	Sphaerosyllis bulbosa
	Polychaeta	Phyllodocida	Syllidae	Sphaerosvllis hystrix
	Polychaeta	Phyllodocida	Syllidae	Syllidae
	Polychaeta	Phyllodocida	Syllidae	Syllis gracilis
	Polychaeta	Phyllodocida	Syllidae	Typosyllis prolifera
	Polychaeta	Sabellida	Sabellidae	Jasmineira elegans
	Polychaeta	Sabellida	Sabellidae	Sabellidae
	Polychaeta	Sabellida	Serpulidae	Spirorbis
	Polychaeta	Scolecida	Maldanidae	Maldanidae
	Polychaeta	Terebellida	Ampharetidae	Ampharete
	Polychaeta	Terebellida	Cirratulidae	Chaetozone setosa
	Polychaeta	Terebellida	Cirratulidae	Chaetozone spp
	Polychaeta	Terebellida	Cirratulidae	Cirretulideo
	Polychaeta	Terebellida	Cirratulidae	Cirratulus cirratus
	Deluchaeta	Terebellida	Cimatulidae	Themme menioni
	Polychaeta	Terebellida	Cirratulidae	Themas Themas
	Polychaeta	Terebellida	Destinentidae	Destinanie heleise
	Polychaeta	Terebellida	Pectinariidae	Pectinaria beigica
	Polychaeta	Terebellida	Terebellidae	Amphitrite spp.
	Polychaeta	Terebellida	Terebellidae	Lanice conchilega
	Polychaeta	Terebellida	Terebellidae	Terebellidae
	Polychaeta	Terebellida	Trichobranchidae	Terebellides stroemi
	l'Olychaeta			
Arthropoda	Malacostraca	Amphipoda	Ampeliscidae	Ampelisca brevicornis
Arthropoda	Malacostraca Malacostraca	Amphipoda Amphipoda	Ampeliscidae Ampeliscidae	Ampelisca brevicornis Ampelisca gibba
Arthropoda	Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi
Arthropoda	Malacostraca Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis
Arthropoda	Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes
Arthropoda	Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus anomalus
Arthropoda	Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus anomalus Microdeutopus chelifer
Arthropoda	Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae Aoridae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus anomalus Microdeutopus versiculat
Arthropoda	Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae Aoridae Aoridae Atvlidae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus anomalus Microdeutopus chelifer Microdeutopus versiculat Atylus spp.
Arthropoda	Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae Aoridae Atylidae Bathyporeiidae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus anomalus Microdeutopus chelifer Microdeutopus versiculat Atylus spp. Bathyporeia pelagica
Arthropoda	Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae Aoridae Atylidae Bathyporeiidae Caprellidae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus anomalus Microdeutopus versiculat: Atylus spp. Bathyporeia pelagica Caprella acanthífera
Arthropoda	Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae Aoridae Atylidae Bathyporeiidae Caprellidae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus anomalus Microdeutopus versiculat Atylus spp. Bathyporeia pelagica Caprella acanthifera Caprella spp.
Arthropoda	Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae Aoridae Aoridae Aoridae Bathyporeiidae Caprellidae Caprellidae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus anomalus Microdeutopus versiculat Atylus spp. Bathyporeia pelagica Caprella acanthifera Caprella spp. Canrellidae
Arthropoda	Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae Aoridae Atylidae Bathyporeiidae Caprellidae Caprellidae Caprellidae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus anomalus Microdeutopus chelifer Microdeutopus versiculat Atylus spp. Bathyporeia pelagica Caprella acanthifera Caprella agp. Caprellidae Pariambus typicus
Arthropoda	Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae Aoridae Atylidae Bathyporeiidae Caprellidae Caprellidae Caprellidae Caprellidae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus anomalus Microdeutopus versiculat Atylus spp. Bathyporeia pelagica Caprella acanthifera Caprella app. Caprellidae Pariambus typicus Phtisica marina
Arthropoda	Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae Aoridae Autylidae Bathyporeiidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus anomalus Microdeutopus versiculati Atylus spp. Bathyporeia pelagica Caprella acanthifera Caprella spp. Caprella dep Pariambus typicus Phtisica marina Pesudoportofalu phesma
Arthropoda	Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae Aoridae Atylidae Bathyporeiidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus chelifer Microdeutopus versiculat Atylus spp. Bathyporeia pelagica Caprella acanthifera Caprella acanthifera Pariambus typicus Phtisica marina Pseudoprotella phasma Cheirocratus
Arthropoda	Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae Aoridae Atylidae Bathyporeiidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus anomalus Microdeutopus versiculat Atylus spp. Bathyporeia pelagica Caprella acanthifera Caprella agn. Caprella pp. Caprella pp. Caprella pp. Pariambus typicus Phtisica marina Pseudoprotella phasma Cheirocratus Computation
Arthropoda	Malacostraca Malacostraca	Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae Aoridae Aoridae Atylidae Bathyporeiidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus anomalus Microdeutopus chelifer Microdeutopus versiculati Atylus spp. Bathyporeia pelagica Caprella canthifera Caprella app. Caprellidae Pariambus typicus Phtisica marina Pseudoprotella phasma Cheirocratus Corophildae
Arthropoda	Malacostraca         Malacostraca	Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae Aoridae Aoridae Aoridae Atylidae Bathyporeiidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Comphildae Comphildae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus chelifer Microdeutopus versiculat Atylus spp. Bathyporeia pelagica Caprella acanthifera Caprella acanthifera Pariambus typicus Phtisica marina Pseudoprotella phasma Cheirocratus Corophildae Corophildae
Arthropoda	Malacostraca         Malacostraca	Amphipoda Amphipoda	Ampeliscidae Ampeliscidae Amphilochidae Aoridae Aoridae Aoridae Aoridae Aoridae Atylidae Bathyporeiidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Caprellidae Corpellidae Corphildae Corophildae	Ampelisca brevicornis Ampelisca gibba Gitana sarsi Aora gracilis Lembos longipes Microdeutopus anomalus Microdeutopus chelifer Microdeutopus versiculat Atylus spp. Bathyporeia pelagica Caprella acanthifera Caprella acanthifera Pariambus typicus Phtisica marina Pseudoprotella phasma Cheirocratus Corophidae Corophium volutator Leptocheirus pilosus

Grabs Taxa list (continued)

phylum	class	order	family	Taxa
	Malacostraca	Amphipoda	Dexaminidae	Dexamine spinosa
	Malacostraca	Amphipoda	Eusiridae	Eusirus longipes
	Malacostraca	Amphipoda	Gammaridae	Chaetogammarus marinus
	Malacostraca	Amphipoda	Hyalidae	Hyale spp.
	Malacostraca	Amphipoda	Ischyroceridae	Ericthonius brasiliensis
	Malacostraca	Amphipoda	Ischyroceridae	Ericthonius punctatus
	Malacostraca	Amphipoda	Leucothoidae	Leucothoe lilljeborgi
	Malacostraca	Amphipoda	Leucothoidae	Leucothoe spinicarpa
	Malacostraca	Amphipoda	Liljeborgiidae	Liljeborgia spp.
	Malacostraca	Amphipoda	Lysianassidae	Lysianassa ceratina
	Malacostraca	Amphipoda	Lysianassidae	Lysianassidae
	Malacostraca	Amphipoda	Lysianassidae	Nannonyx goesi
	Malacostraca	Amphipoda	Maeridae	Maera grossimana
	Malacostraca	Amphipoda	Maeridae	Maera othonis
	Malacostraca	Amphipoda	Melitidae	Abludomelita obtusa
	Malacostraca	Amphipoda	Melitidae	Melita gladiosa
	Malacostraca	Amphipoda	Melitidae	Melitidae
	Malacostraca	Amphipoda	Melphidippidae	Melphidippa spp.
	Malacostraca	Amphipoda	-	Amphipoda
	Malacostraca	Amphipoda	Photidae	Gammaropsis maculata
	Malacostraca	Amphipoda	Phoxocephalidae	Harpinia antennaria
	Malacostraca	Amphipoda	Phoxocephalidae	Phoxocephalidae
	Malacostraca	Amphipoda	Phoxocephalinae	Metaphoxus fultoni
	Malacostraca	Amphipoda	Stenothoidae	Metopa borealis
	Malacostraca	Amphipoda	Tryphosidae	Lepidepecreum longicorne
	Malacostraca	Amphipoda	Tryphosidae	Tryphosites longipes
	Malacostraca	Amphipoda	Tryphosidae	Tryphosites spp.
	Malacostraca	Amphipoda	Unciolidae	Unciola crenatipalma
	Malacostraca	Amphipoda	Urothoidae	Urothoe marina
	Malacostraca	Cumacea	Bodotriidae	Bodotria arenosa
	Malacostraca	Cumacea	Bodotriidae	Cumopsis fagei
	Malacostraca	Cumacea	Bodotriidae	Cumopsis longipes
	Malacostraca	Cumacea	Bodotriidae	Iphinoe trispinosa
	Malacostraca	Cumacea	Diastylidae	Diastylis spp.
	Malacostraca	Cumacea	-	Cumacea
	Malacostraca	Cumacea	Psuedocuma	Pseudocuma similis
	Malacostraca	Decapoda	Acanthephyridae	Hymenodora
	Malacostraca	Decapoda	Alpheidae	Athanas nitescens
	Malacostraca	Decapoda	Atelecyclidae	Atelecyclus rotundatus
	Malacostraca	Decapoda	Cancridae	Cancer pagurus
	Malacostraca	Decapoda	Crangonidae	Philocheras fasciatus
	Malacostraca	Decapoda	Galatheidae	Galathea intermedia
	Malacostraca	Decapoda	Galatheidae	Galatheidae
	Malacostraca	Decapoda	Inachidae	Macropodia rostrata
	Malacostraca	Decapoda	Leucosiidae	Ebalia tuberosa
	Malacostraca	Decapoda	Leucosiidae	Ebalia tumefacta
	Malacostraca	Decapoda	Majidae	Eurynome
	Malacostraca	Decapoda	Majidae	Majidae
	Malacostraca	Decapoda	-	Crustacea
	Malacostraca	Decapoda	-	Decapoda
	Malacostraca	Decapoda	Paguridae	Anapagurus
	Malacostraca	Decapoda	Paguridae	Paguridae
	Malacostraca	Decapoda	Paguridae	Pagurus bernhardus
	Malacostraca	Decapoda	Paguridae	Pagurus spp.
		Deceneda	Pasiphaeidae	Pasiphaea spp.
	Malacostraca	Decapoda	* doipitoroidae	r conbucco oppi
	Malacostraca Malacostraca	Decapoda	Polybiidae	Liocarcinus pusillus

Grabs Taxa list (continued)

# Table C.2. continued. Infaunal taxa recorded in the grab samples.

phylum	class	order	family	Taxa
	Malacostraca	Decapoda	Polybiidae	Liocarcinus spp.
	Malacostraca	Decapoda	Porcellanidae	Pisidia longicornis
	Malacostraca	Decapoda	Thoridae	Thoralus cranchii
	Malacostraca	Decapoda	-	Brachyura
	Malacostraca	Isopoda	Cirolanidae	Cirolana cranchii
	Malacostraca	Isopoda	Cirolanidae	Eurydice pulchra
	Malacostraca	Isopoda	-	Isopoda
	Malacostraca	Isopoda	Sphaeromatidae	Cymodoce truncata
	Malacostraca	Isopoda	Sphaeromatidae	Dynamene bidentata
	Malacostraca	Leptostraca	Nebaliidae	Nebalia bipes
	Malacostraca	Tanaidacea	Apseudidae	Apseudes latreillii
	Malacostraca	Tanaidacea	Apseudidae	Apseudes talpa
	Malacostraca	Tanaidacea	Leptocheliidae	Leptochelia savignyi
	Malacostraca	Tanaidacea	Paratanaoidea incertae sedis	Pseudoparatanais batei
	Malacostraca	Tanaidacea	Tanaissuidae	Tanaissus lilljeborgi
	Ostracoda	-	-	Ostracoda
	Pycnogonida	Pantopoda	Nymphonidae	Nymphon
	Pycnogonida	Pantopoda	Phoxichilidiidae	Anoplodactylus
Brvozoa	Gymnolaemata	Cheilostomatida	Candidae	Scrupocellaria reptans
	Gymnolaemata	Cheilostomatida	Cellariidae	Cellaria spp.
	Gymnolaemata	Cheilostomatida	Celleporidae	Celleporina hassallii
	Gymnolaemata	Cheilostomatida	Electridae	Electra pilosa
	Stenolaemata	Cyclostomatida	Crisiidae	Crisia denticulata
	Stenolaemata	Cyclostomatida	Crisiidae	Filicrisia spp.
	Stenolaemata	Cyclostomatida	Tubuliporidae	Tubulipora
Chordata	Actinoptervgii	-	-	Actinontervaji
	Ascidiacea	-	-	Tunicata
	Ascidiacea	Phlebobranchia	Ascidiidae	Ascidia mentula
	Ascidiacea	Stolidobranchia	Molgulidae	Molgula spp.
	Leptocardii		Branchiostomatidae	Branchiostoma lanceolatum
	Leptocardii	-	Branchiostomatidae	Branchiostoma spp.
Cnidaria	Anthozoa	-	-	Cnidaria
	Hydrozoa	Leptothecata	Campanulinidae	Opercularella lacerata
	Hydrozoa	Leptothecata	Plumulariidae	Nemertesia antennina
	Hydrozoa	Leptothecata	Sertularellidae	Sertularella distans
	Hydrozoa	Leptothecata	Sertularellidae	Sertularella spp.
	Hydrozoa	Leptothecata	Sertulariidae	Diphasia attenuata
	Hydrozoa	Leptothecata	Sertulariidae	Diphasia pinaster
	Hydrozoa	Leptothecata	Sertulariidae	Diphasia rosacea
	Hydrozoa	Leptothecata	Sertulariidae	Hydrallmania falcata
	Hydrozoa	Leptothecata	Sertulariidae	Sertularia cupressina
	Hydrozoa	Leptothecata	Sertulariidae	Sertulariidae
	Hydrozoa	-	-	Hydrozoa
Echinodermata	Echinoidea	Clyneasteroide	Fibulariidae	Echinocyamus pusillus
	Echinoidea	Clypeasteroida		Clypeasteroida
	Holothuroidea	Dendrochirotida	Phyllophoridae	Neopentadactyla mixta
	Ophiuroidea	Amphilepidide	Amphiuridae	Amphipholis squamata
	Ophiuroidea	Amphilepidida	Amphiuridae	Amphijira
	Ophiuroidea	Ophiurida	Ophiuridae	Ophiura affinis
	Ophiuroidea	Ophiurida	Ophiuridae	Ophiura spp.
Mollusca	Bivalvia	Adapedonta	Pharidae	Ensis magnus
monusca	Bivalvia	Adapedonta	Pharidae	Encie enn
	Bivalvia	Arcido	Chronmorididao	Chearmorie glucarmorie
	Divalvia	Arciua	Giyeymeriuldae	Giveymens giveymens

Grabs Taxa list (continued)

4

phylum	class	order	family	Taxa
	Bivalvia	Cardiida	Cardiidae	Cerastoderma edule
	Bivalvia	Cardiida	Cardiidae	Parvicardium minimum
	Bivalvia	Cardiida	Cardiidae	Parvicardium ovale
	Bivalvia	Cardiida	Cardiidae	Parvicardium scabrum
	Bivalvia	Cardiida	Semelidae	Abra alba
	Bivalvia	Cardiida	Tellinidae	Macoma balthica
	Bivalvia	Cardiida	Tellinidae	Moerella pygmaea
	Bivalvia	Cardiida	Tellinidae	Tellina pygmaea
	Bivalvia	Galeommatida	Lasaeidae	Mysella bidentata
	Bivalvia	Limida	Limidae	Limaria loscombi
	Bivalvia	Mvtilida	Mytilidae	Modiolula phaseolina
	Bivalvia	Mvtilida	Mytilidae	Modiolus modiolus
	Bivalvia	-	-	Bivalvia
	Bivalvia	Nuculida	Nuculidae	Nucula nucleus
	Bivalvia	Nuculida	Nuculidae	Nucula spp.
	Bivalvia	Pectinida	Anomiidae	Anomia ephippium
	Bivalvia	Pectinida	Pectinidae	Pecten maximus
	Bivalvia	Venerida	Mactridae	Mactra corallina
	Bivalvia	Venerida	Veneridae	Chamelea gallina
	Bivalvia	Venerida	Veneridae	Clausinella fasciata
	Bivalvia	Venerida	Veneridae	Dosinia lupinus
	Bivalvia	Venerida	Veneridae	Couldia minima
	Bivalvia	Vonorida	Veneridae	Tapos rhomboidos
	Bivalvia	Vonorida	Veneridae	Timocloa ovata
	Bivalvia	Venerida	Veneridae	Vonus vorrucosa
	Castropoda	Lonotollido	Figurellidae	Emorginulo fissuro
	Gastropoda	Lepetenida	Columtracidae	Caluptraga abiponcia
	Gastropoda	Littorinimorpha	Calyptracidae	Cropidula formicata
	Gastropoda	Littorinimorpha	Littorinidaa	Littoring opp
	Gastropoda	Littorinimorpha	Littorinidae	Tratum similar
	Gastropoda	-	Lottildae D-t-llidae	Detelle mellucide
	Gastropoda	-	Patellidae	Patella pellucida
	Gastropoda	Neogastropoda	Muricidae	Ocenebra erinaceus
	Gastropoda	Neogastropoda	Nassariidae	Hima reticulata
	Gastropoda	Neogastropoda	Nassariidae	Tritia reticulata
	Gastropoda	Neogastropoda	Nassariidae	Tritia reticulata
	Gastropoda	Trochida	Trochidae	Gibbula magus
	Gastropoda	Trochida	Trochidae	Gibbula umbilicalis
	Monoplacophora	-		Monoplacophora
	Polyplacophora	Chitonida	Acanthochitonidae	Acanthochitona discrepan
	Polyplacophora	Chitonida	Tonicellidae	Tonicella rubra
	Polyplacophora	-	-	Polyplacophora
	Scaphopoda	-	-	Scaphopoda
Nematoda	Nematoda	-	-	Nematoda
	Nemertea	-	-	Nemertea
Porifera	Demospongiae	Dictyoceratida	Dysideidae	Dysidea fragilis
Priapulida	Priapulida	-	-	Priapulida
Sipuncula	Sipunculidea	Golfingiida	Golfingiidae	Golfingia vulgaris
	Sipunculidea	-	-	Sipuncula
	-			

Grabs Taxa list (continued)

5





Figure C.10.1. a) the mean particle size ( $\mu$ m), and b) the proportion of organic carbon, for each Habitat, Location and Treatment. Note that not all Habitats occur in all Treatments or Locations.



Figure C.10.2. Proportion of particle size categories (Mud, Sand, and Gravel) for each Location and treatment where maerl occurred. Each bar represents an individual sample.



Figure C.10.3. Weight of a) Live maerl, and b) Dead maerl in kg per litre of sediment for Treatment (MPA and Open Control) and Location (Ecrehous and Southeast). Filled points represent outliers.

Table C.10.1. General Linear Mixed Effects Model outputs for infaunal taxa and individuals (>1 mm) on maerl habitat as a function of Location (Ecrehous and Southeast), Treatment

(MPA and Open Control) and maerl weight (live or dead), with Site as a random effect. Figures in bold denote a significant result.

Terms	Estimate	Std. Error	z value	р
Live Maerl - Number of Taxa				
Intercept	2.68	0.0753	35.5	$< 0.0001^{***}$
Live maerl weight	0.645	0.371	1.74	0.082
MPA - Open	-0.145	0.137	-1.06	0.29
Ecrehous - Southeast	0.259	0.0863	3	$0.0027^{**}$
Live maerl weight*MPA - Open	-0.69	1.25	-0.554	0.58
Maerl weight*Ecrehous - Southeast	-	-	-	-
MPA - Open*Ecrehous - Southeast	-	-	-	-
Maerl weight*MPA - Open*Ecrehous - Southeast	-	-	-	-
Live Maerl - Number of Individuals				
Intercept	2.89	0.116	24.8	$< 0.0001^{***}$
Live maerl weight	5.86	1.09	5.35	$< 0.0001^{***}$
MPA - Open	0.268	0.186	1.44	0.15
Ecrehous - Southeast	0.996	0.146	6.84	$< 0.0001^{***}$
Live maerl weight*MPA - Open	-6.45	1.79	-3.6	$< 0.0001^{***}$
Live maerl weight*Ecrehous - Southeast	-4.69	1.13	-4.16	$< 0.0001^{***}$
MPA - Open*Ecrehous - Southeast	-0.513	0.234	-2.19	0.028*
Live maerl weight*MPA - Open*Ecrehous - Southeast	6.02	2.05	2.93	$0.0034^{**}$
Dead Maerl - Number of Taxa				
Intercept	2.58	0.085	30.3	$< 0.0001^{***}$
Dead maerl weight	1.18	0.42	2.81	$0.0049^{**}$
MPA - Open	0.0434	0.139	0.312	0.75
Ecrehous - Southeast	0.3	0.0868	3.46	$< 0.0001^{***}$
Dead maerl weight*MPA - Open	-1.63	0.539	-3.02	$0.0025^{**}$
Maerl weight*Ecrehous - Southeast	-	-	-	-
MPA - Open*Ecrehous - Southeast	-	-	-	-
Maerl weight*MPA - Open*Ecrehous - Southeast	-	-	-	-
Dead Maerl - Number of Individuals				
Intercept	3.11	0.175	17.8	$< 0.0001^{***}$
Dead maerl weight	2.12	0.921	2.31	$0.021^{*}$
MPA - Open	0.0383	0.266	0.144	0.89
Ecrehous - Southeast	0.943	0.218	4.33	$< 0.0001^{***}$
Dead maerl weight*MPA - Open	-2.46	1.16	-2.12	$0.034^{*}$
Dead maerl weight*Ecrehous - Southeast	-2.08	0.96	-2.17	$0.03^{*}$
MPA - Open*Ecrehous - Southeast	-0.123	0.36	-0.343	0.73
Dead maerl weight*MPA - Open*Ecrehous - Southeast	1.65	1.23	1.34	0.18

Table C.10.2. General Linear Mixed Effects Model outputs for infaunal taxa and individuals (>1 mm) on maerl habitat at the Southeast and Ecrehous as a function of Treatment with

Site as a random effect for both the Southeast and Ecrehous, and Year as a random effect for the Ecrehous only. Figures in bold denote a significant result.

Terms	Estimate	Std. Error	z value	р					
Southeast Nu	Southeast Number of Taxa								
Intercept	3.430	0.081	42.500	$< 0.0001^{***}$					
MPA - Open	-0.641	0.116	-5.550	$< 0.0001^{***}$					
Ecrehous Nun	nber of Ta	xa							
Intercept	2.720	0.074	36.700	$< 0.0001^{***}$					
MPA - Open	-0.193	0.137	-1.410	0.16					
Southeast Nu	mber of In	dividuals							
Intercept	4.940	0.277	17.800	$< 0.0001^{***}$					
MPA - Open	-1.240	0.479	-2.590	$0.0096^{**}$					
Ecrehous Nun	Ecrehous Number of Individuals								
Intercept	3.380	0.166	20.300	$< 0.0001^{***}$					
MPA - Open	-0.268	0.290	-0.924	0.36					

Table C.10.3. PERMANOVA of maerl habitat assemblage based on Bray-Curtis dissimilarities for the interactions of Location and Treatment. Data were fourth root transformed. Figures in bold denote a significant result.

PERMANOVA main test					Pairw	vise				
Source	df	SS	MS	$\operatorname{Pseudo-F}$	P(perm)	Unique perms	Groups	t	р	Unique perms
Tr	1	2914.4	2914.4	1.168	0.244	9905	Ecrehous: MPA - Open	0.903	0.69	7734
Lo	1	5291.9	5291.9	2.121	$0.004^{**}$	9896	Southeast: MPA - Open	1.709	$< 0.0001^{***}$	9869
TrxLo	1	5383.4	5383.4	2.158	$0.003^{**}$	9898				
Pooled	37	92317.0	2495.1							
Total	40	106620.0								



Figure C.10.4. nMDS illustrating the dissimilarities in assemblage composition between Locations and Treatments.

Table C.10.4. SIMPER analysis results showing the differences in average abundance (Av.Abund) and dissimilarity (Av.Diss) between MPA and Open Control treatments on maerl habitat at each Location (Ecrehous, Southeast). Species contribution cut-off was set at 50%.

Family	Av.Abund MPA	Av.Abund Open	$\operatorname{Av.Diss}$	$\mathrm{Diss}/\mathrm{SD}$	$\operatorname{Contrib}\%$	$\operatorname{Cum.\%}$
Ecrehous						
Lumbrineridae	1.06	0.84	3.09	1.34	4.30	4.30
Syllidae	1.13	0.86	2.97	1.10	4.14	8.44
Glyceridae	0.90	0.39	2.84	1.08	3.95	12.38
Polynoidae	0.80	0.53	2.70	1.13	3.76	16.14
Eunicidae	0.70	0.76	2.58	1.13	3.59	19.73
Nuculidae	1.15	1.07	2.53	0.95	3.52	23.25
Nematoda	0.72	0.82	2.37	1.08	3.30	26.55
Clitellata	0.59	0.78	2.21	1.09	3.08	29.63
Terebellidae	0.33	0.70	2.14	1.03	2.98	32.61
Ophiuridae	0.55	0.50	2.10	1.04	2.92	35.54
Nereidae	0.55	0.00	2.08	0.84	2.90	38.44
Cardiidae	0.56	0.00	2.02	0.90	2.82	41.26
Annelida	0.53	0.39	1.92	0.97	2.68	43.93
Polyplacophora	0.46	0.39	1.71	0.96	2.38	46.31
Veneridae	0.35	0.26	1.66	0.74	2.31	48.62
Sertularellidae	0.42	0.00	1.50	0.79	2.08	50.71
Southeast						
Veneridae	1.17	0.35	2.89	0.81	4.17	4.17
Syllidae	1.25	1.39	2.88	1.38	4.15	8.32
Eunicidae	0.63	1.17	2.30	1.26	3.31	11.63
Maldanidae	1.02	0.33	2.21	0.97	3.19	14.81
Glyceridae	0.47	1.07	2.13	1.29	3.07	17.88
Lumbrineridae	0.99	1.60	2.06	1.14	2.96	20.84
Nuculidae	1.68	1.20	2.04	0.99	2.94	23.79
Terebellidae	0.72	0.13	1.95	0.60	2.81	26.59
Cirratulidae	0.42	0.78	1.76	1.09	2.54	29.14
Annelida	0.71	1.29	1.71	1.08	2.47	31.60
Apseudidae	0.73	0.13	1.70	0.85	2.45	34.06
Polynoidae	0.31	0.83	1.66	1.14	2.39	36.44
Nematoda	0.93	0.74	1.65	1.06	2.37	38.82
Paraonidae	0.44	0.79	1.63	1.14	2.34	41.16
Clitellata	0.80	0.57	1.62	1.09	2.34	43.50
Sertulariidae	0.71	0.00	1.54	0.87	2.22	45.72
Caprellidae	0.76	0.11	1.52	0.69	2.19	47.91
Ophiuridae	0.45	0.55	1.50	0.92	2.16	50.07





Table C.10.5. General Linear Mixed Effects Model outputs for infaunal taxa and individuals (>1 mm) on seagrass habitat as a function of Location with Year and Site as random effects for the Minquiers and Southeast. Figures in bold denote a significant result.

Terms	Estimate	Std. Error	z value	р
Number of Taxa				
Intercept	2.50	0.108	23.00	$< 0.0001^{***}$
Minquiers - Southeast	0.69	0.142	4.87	$< 0.0001^{***}$
Number of Individual	s			
Intercept	3.41	0.069	49.70	$< 0.0001^{***}$
Minquiers - Southeast	1.19	0.082	14.60	$< 0.0001^{***}$

Table C.10.6. PERMANOVA of seagrass habitat assemblage based on Bray-Curtis dissimilarities for Location within the Treatment MPA. Data were square root transformed. Figures in bold denote a significant result.

Seagrass main test								
Source df SS MS Pseudo-F P(perm) Unique perms								
Lo	1	8026.5	8026.5	3.699	$0.01^{*}$	792		
Pooled	10	21697.0	2169.7					
Total	11	29723.0						

Table C.10.7. SIMPER analysis results showing the differences in average abundance (Av.Abund) and dissimilarity (Av.Diss) between the Minquiers and Southeast MPA Locations on seagrass habitat. Species contribution cut-off was set at 50%.

Species	Av.Abund Minquiers	Av.Abund Southeast	Av.Diss	Diss/SD	Contrib%	Cum.%
Maldanidae	0.55	5.50	8.51	3.25	11.75	11.75
Ampeliscidae	0.66	4.31	6.22	4.38	8.58	20.33
Apseudidae	3.33	3.45	4.20	1.40	5.79	26.12
Glyceridae	0.69	2.65	3.58	1.60	4.94	31.06
Nematoda	1.01	2.76	3.31	1.54	4.57	35.63
Nemertea	0.14	1.99	3.19	2.46	4.40	40.03
Aoridae	0.34	1.97	3.12	1.38	4.30	44.33
Corophiidae	0.20	1.29	2.06	1.33	2.85	47.18
Clitellata	0.71	1.17	1.72	1.43	2.37	49.55
Sabellidae	0.00	0.97	1.72	1.55	2.37	51.92
1	1					





Table C.10.8. General Linear Mixed Effects Model outputs for infaunal taxa and individuals (>1 mm) on coarse sediment habitat as a function of Treatment with Year and Site as random effects for the Location Minquiers. Figures in bold denote a significant result.

	Terms	Estimate	Std. Error	z value	р
Number of Taxa					
Intercept	2.370	0.143	16.60	$< 0.0001^{***}$	
MPA - Open Control	-0.327	0.172	-1.90	0.057	
Number of Individual	ls				
Intercept	3.580	0.537	6.67	$< 0.0001^{***}$	
MPA - Open	-0.947	0.098	-9.69	$< 0.0001^{***}$	

Table C.10.9. PERMANOVA of sediment habitat assemblage based on Bray-Curtis dissimilarities for Treatment within the Location Minquiers. Data were fourth root transformed. Figures in bold denote a significant result.

Sediment main test								
Source df SS MS Pseudo-F P(perm) Unique perms								
$\operatorname{Tr}$	1	2355.1	2355.1	0.849	0.56	5097		
Pooled	13	36063	2774.1					
Total	14	38418						

Table C.10.10. SIMPER analysis results showing the differences in average abundance (Av.Abund) and dissimilarity (Av.Diss) between the Treatment (MPA and Open Control) on coarse sediment at the Minquiers. Species contribution cut-off was set at 50%.

Species	Av.Abund MPA	Av.Abund Open	Av.Diss	Diss/SD	Contrib%	Cum.%
Apseudidae	1.18	2.06	4.54	1.37	6.53	6.53
Nematoda	0.60	1.42	4.34	1.25	6.24	12.77
Lysianassidae	0.45	0.94	3.41	1.37	4.89	17.66
Syllidae	0.99	0.88	3.31	1.18	4.76	22.42
Maldanidae	0.88	0.27	3.06	1.19	4.40	26.81
Veneridae	0.14	0.77	2.73	0.73	3.92	30.73
Glyceridae	0.43	0.38	2.67	0.77	3.84	34.57
Lumbrineridae	0.60	0.31	2.50	0.91	3.58	38.16
Tubificidae	0.63	0.57	2.45	0.96	3.52	41.67
Eunicidae	0.43	0.50	2.07	0.90	2.97	44.64
Orbiniidae	0.43	0.38	1.96	0.88	2.82	47.47
Sertularellidae	0.29	0.38	1.89	0.82	2.72	50.18

#### 11. Appendix D

Table D.11.1. Kolmogorov-smirnov tests of differences in size distribution between Treatment (MPA and Open Control) each Year (2018, 2019 and 2020) at each Location (Minquiers and Ecrehous) for Lobster, Brown crab and Spider crab. Note there was no test for Brown crab at the Ecrehous in 2020 due to no individuals of brown crab caught in the Open Control. Bold denotes a significant result.

	Lobster		Spider crab		Brown crab	
	D	p	D	p	D	p
Minqui	iers					
2018	0.111	0.887	0.412	0.231	0.600	0.021*
2019	0.118	0.971	0.697	$0.010^{*}$	0.640	$0.023^{*}$
2020	0.153	0.858	0.349	0.426	0.349	0.426
Ecreho	us					
2018	0.242	0.637	0.302	0.151	0.857	0.067
2019	0.407	0.103	0.422	$0.049^{*}$	0.500	1.000
2020	0.621	0.218	0.202	0.954	-	-

Table D.11.2. Kolmogorov-smirnov tests of differences in size distribution between Years (2018, 2019 and 2020) for each Treatment (MPA and Open Control) and Location (Minquiers and Ecrehous) for Lobster, Brown crab and Spider crab. Bold denotes a significant result.

	Lobster		Spider crab		Brown crab	
	D	p	D	p	D	p
Minquiers N	MPA					
2018-2019	0.229	0.418	0.295	0.521	0.182	0.962
2019-2020	0.159	0.926	0.276	0.696	0.324	0.489
2018-2020	0.208	0.521	0.273	0.855	0.182	0.997
Minquiers 0	Open		-		-	
2018-2019	0.153	0.359	0.533	0.132	0.600	0.064
2019-2020	0.103	0.936	0.333	0.604	0.286	0.785
2018-2020	0.139	0.640	0.429	0.080	0.381	0.158
Ecrehous M	IPA					
2018-2019	0.159	0.607	0.361	0.305	0.571	0.556
2019-2020	0.076	0.997	0.643	0.010*	0.636	0.385
2018-2020	0.152	0.579	0.405	0.151	0.221	0.985
Ecrehous O	pen					
2018-2019	0.470	0.159	0.369	$< 0.0001^{***}$	1.000	0.200
2019-2020	0.485	0.637	0.333	0.604	0.286	0.785
2018 - 2020	0.583	0.388	0.429	0.080	0.381	0.158

#### References

- Abecasis, D., Bentes, L., & Erzini, K. (2009). Home range, residency and movements of Diplodus sargus and Diplodus vulgaris in a coastal lagoon: Connectivity between nursery and adult habitats. *Estuarine, Coastal and Shelf Science*, *85*(4), 525–529. https://doi.org/10.1016/j.ecss.2009.09.001
- Agardy, T. (2000). Information Needs For Marine Protected Areas: Scientific And Societal. *Bulletin of Marine Science*, *66*(3), 875–888.
- Agardy, T., di Sciara, G. N., & Christie, P. (2011). Mind the gap Addressing the shortcomings of marine protected areas through large scale marine spatial planning. *Marine Policy*, *35*(2), 226–232. https://doi.org/10.1016/j.marpol.2010.10.006
- Anderson, C. N. K., Hsieh, C., Sandin, S. A., Hewitt, R., Hollowed, A., Beddington, J., May, R. M., & Sugihara, G. (2008). Why fishing magnifies fluctuations in fish abundance. *Nature*, 452(1), 835–839. https://doi.org/10.1038/nature06851
- Babcock, R. C., Shears, N. T., Alcala, A. C., Barrett, N. S., Edgar, G. J., Lafferty,
  K. D., McClanahan, T. R., & Russ, G. R. (2010). Decadal trends in marine
  reserves reveal differential rates of change in direct and indirect effects. *Proceedings of the National Academy of Sciences of the United States of America*, 107(43), 18256–18261. https://doi.org/10.1073/pnas.0908012107
- Bakke, S. (2019). *Life history and distribution of the edible crab (Cancer pagurus) in Norway*. The Arctic University of Norway.

Balestri, E., Vallerini, F., & Lardicci, C. (2017). Recruitment and patch establishment by seed in the seagrass Posidonia oceanica: Importance and conservation implications. *Frontiers in Plant Science*, 8. https://doi.org/10.3389/fpls.2017.01067

- Ban, N. C., Gurney, G. G., Marshall, N. A., Whitney, C. K., Mills, M., Gelcich, S.,
  Bennett, N. J., Meehan, M. C., Butler, C., Ban, S., Tran, T. C., Cox, M. E., &
  Breslow, S. J. (2019). Well-being outcomes of marine protected areas. *Nature Sustainability*, *2*, 524–532.
- Barbera, C., Bordehore, C., Borg, J. A., Glemarec, M., Grall, J., Hall-Spencer, J.
  M., la Huz, C., Lanfranco, E., Lastra, M., Moore, P. G., Mora, J., Pita, M. E.,
  Ramos-Espla, A. A., Rizzo, M., Sanchez-Mata, A., Seva, A., Schembri, P. J., &
  Valle, C. (2003). Conservation and management of northeast Atlantic and
  Mediterranean maerl beds. *Aquatic Conservation-Marine and Freshwater Ecosystems*, *13*, S65–S76. https://doi.org/10.1002/agc.569
- Barneche, D. R., White, C. R., & Marshall, D. J. (2018). Fish reproductive-energy output increases disproportionately with body size. *Science*, *360*, 642–645.
- Bastardie, F., Nielsen, J. R., Ulrich, C., Egekvist, J., & Degel, H. (2010). Detailed mapping of fishing effort and landings by coupling fishing logbooks with satellite-recorded vessel geo-location. *Fisheries Research*, *106*(1), 41–53. https://doi.org/10.1016/j.fishres.2010.06.016
- Bateman, K. S., Stentiford, G. D., Stone, D., Feist, S. W., P., W., Edwards, M., Kerr, R., Green, M. J., Ross, S., Evans, C., Bass, D., & OIE. (2020).
  Emergence of paramoebiasis in edible crabs (Cancer pagurus) from UK waters. *Report of the 11 Th Annual Workshop of the National Reference* 281

Baudron, A. R., Brunel, T., Blanchet, M. A., Hidalgo, M., Chust, G., Brown, E. J.,
Kleisner, K. M., Millar, C., MacKenzie, B. R., Nikolioudakis, N., Fernandes, J.
A., & Fernandes, P. G. (2020). Changing fish distributions challenge the
effective management of European fisheries. *Ecography*, 43(4), 494–505.
https://doi.org/10.1111/ecog.04864

Beaumont, N. J., Austen, M. C., Atkins, J. P., Burdon, D., Degraer, S., Dentinho, T.
P., Derous, S., Holm, P., Horton, T., van Ierland, E., Marboe, A. H., Starkey,
D. J., Townsend, M., & Zarzycki, T. (2007). Identification, definition and
quantification of goods and services provided by marine biodiversity:
Implications for the ecosystem approach. *Marine Pollution Bulletin*, *54*(3),
253–265. https://doi.org/10.1016/j.marpolbul.2006.12.003

Beck, M. W., Heck, K. L., Able, K. W., Childers, D. L., Eggleston, D. B., Gillanders, B. M., Halpern, B. S., Hays, C. G., Hoshino, K., Minello, T. J., Orth, R. J., Sheridan, P. F., & Weinstein, M. R. (2001). The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates. *Bioscience*, *51*(8), 633–641. https://doi.org/10.1641/0006-3568(2001)051[0633:ticamo]2.0.co;2

Bennett, D. B. (1995). Factors in the life history of the edible crab (Cancer pagurus
L .) that influence modelling and management. *ICES Marine Science* Symposium, 199, 89–98.

Bennett, N. J., & Dearden, P. (2014). From measuring outcomes to providing inputs: Governance, management, and local development for more effective marine protected areas. *Marine Policy*, 50, 96–110. https://doi.org/10.1016/j.marpol.2014.05.005

- Berger, M. (2021). Accepting the loss of habitat specialists in a changing world. *Nature Ecology and Evolution*, *5*(556–557).
- Berkeley, S. A., Hixon, M. A., Larson, R. J., & Love, M. S. (2004). Fisheries
  Sustainability via Protection of Age Structure and Spatial Distribution of Fish
  Populations. *Fisheries*, *29*(8), 23–32. https://doi.org/10.1577/15488446(2004)29[23:fsvpoa]2.0.co;2
- Beukers-Stewart, B. D., & Beukers-Stewart, J. (2009a). Principles for the Management of Inshore Scallop Fisheries around the United Kingdom. Report to Natural England, Scottish Natural Heritage and Countryside Council for Wales. Marine Ecosystem Management Report no. 1.
- Beukers-Stewart, B. D., & Beukers-Stewart, J. S. (2009b). Principles for the Management of Inshore Scallop Fisheries around the United Kingdom. Report to Natural England, Scottish Natural Heritage and Countryside Council for Wales.
- Beukers-Stewart, B. D., Vause, B. J., Mosley, M. W. J., Rossetti, H. L., & Brand, A. R. (2005). Benefits of closed area protection for a population of scallops. *Marine Ecology Progress Series*, *298*(November 2016), 189–204.
  https://doi.org/10.3354/meps298189
- Bicknell, A. W. J., Godley, B. J., Sheehan, E. V, Votier, S. C., & Witt, M. J. (2016).
  Camera technology for monitoring marine biodiversity and human impact. *Frontiers in Ecology and the Environment*, *14*(8), 424–432.
  https://doi.org/10.1002/fee.1322

Blanchard, M. (2009). Recent expansion of the slipper limpet population
(Crepidula fornicata) in the Bay of Mont-Saint-Michel (Western Channel, France). *Aquatic Living Resources*, *22*(1), 11–19.
https://doi.org/10.1051/alr/2009004

- Blue Marine Foundation. (2016). *Lyme Bay Fisheries and Conservation Reserve Ecological and Fisheries Data*.
- Blue Marine Foundation. (2022). *How to Make Jersey a Marine Sustainability Leader in Europe*.
- Blyth, R. E., Kaiser, M. J., Hart, P. J. B., & Edwards-Jones, G. (2000). An Example of Conservation and Exploitation Achieved Through a Voluntary Fishery
  Management System. *Putting Fishers' Knowledge to Work Conference Proceedings*, 409–425.
- Bolam, S. G., Eggleton, J., Smith, R., Mason, C., Vanstaen, K., & Rees, H. (2008).
  Spatial distribution of macrofaunal assemblages along the English Channel. *Journal of the Marine Biological Association of the United Kingdom*, 88(4),
  675–687. https://doi.org/10.1017/S0025315408001276
- Bonin, M. C., Barrier, G., & Foundation, R. (2015). Specializing on vulnerable habitat: Acropora selectivity among damselfish recruits and the risk of bleaching-induced habitat loss. *Coral Reefs*, *March 2011*. https://doi.org/10.1007/s00338-011-0843-2
- Borum, J., Duarte, C. M., Krause-Jensen, D., & Greve, T. M. (2004). European seagrasses: An introduction to monitoring and management.

Boulcott, P., & Howell, T. R. W. (2011). The impact of scallop dredging on rocky-

reef substrata. *Fisheries Research*, *110*(3), 415–420. https://doi.org/10.1016/j.fishres.2011.05.006

- Bouma, T. J., Olenin, S., & Reise, K. (2009). Ecosystem engineering and biodiversity in coastal sediments: posing hypotheses. *Helgoland Marine Research*, 63, 95–106. https://doi.org/10.1007/s10152-009-0146-y
- Bradley, M., Baker, R., & Sheaves, M. (2017). Hidden Components in Tropical Seascapes: Deep-Estuary Habitats Support Unique Fish Assemblages. *Estuaries and Coasts*, 40, 1195–1206. https://doi.org/10.1007/s12237-016-0192-z
- Bradshaw, C., Collins, P., & Brand, A. R. (2003). To what extent does upright sessile epifauna affect benthic biodiversity and community composition? *Marine Biology*, 143(4), 783–791. https://doi.org/10.1007/s00227-003-1115-7
- Bradshaw, C., Veale, L. O., Hill, A. S., & Brand, A. R. (2001). The effect of scallop dredging on Irish Sea benthos: Experiments using a closed area. *Hydrobiologia*, *465*, 129–138. https://doi.org/10.1023/A
- Braeckman, U., Rabaut, M., Vanaverbeke, J., Degraer, S., & Vincx, M. (2014).
  Protecting the Commons: the use of Subtidal Ecosystem Engineers in Marine. *Aquatic Conservation: Marine and Freshwater Ecosystems*.
  https://doi.org/10.1002/agc.2448
- Brown, B., Soule, E., & Kaufman, L. (2010). Effects of excluding bottom-disturbing mobile fishing gear on abundance and biomass of groundfishes in the Stellwagen Bank National Marine Sanctuary, USA. *Current Zoology*, *56*(1),

- Brown, C. G., & Bennett, D. B. (1980). Population and catch structure of the edible crab (Cancer pagurus) in the English Channel. *ICES Journal of Marine Science*, *39*(1), 88–100. https://doi.org/10.1093/icesjms/39.1.88
- Brown, C. J., Broadley, A., Adame, M. F., Branch, T. A., Turschwell, M. P., & Connolly, R. M. (2019). The assessment of fishery status depends on fish habitats. *Wiley, Fish and Fisheries*, 20, 1–14. https://doi.org/10.1111/faf.12318
- Brown, C. J., Fulton, E. A., Hobday, A. J., Matear, R. J., Possingham, H. P.,
  Bulman, C., Christensen, V., Forrest, R. E., Gehrke, P. C., Gribble, N. A.,
  Griffiths, S. P., Lozano-Montes, H., Martin, J. M., Metcalf, S., Okey, T. A.,
  Watson, R., & Richardson, A. J. (2010). Effects of climate-driven primary
  production change on marine food webs: Implications for fisheries and
  conservation. *Global Change Biology*, *16*(4), 1194–1212.
  https://doi.org/10.1111/j.1365-2486.2009.02046.x
- Bruschetti, M. (2019). Role of Reef-Building, Ecosystem Engineering Polychaetes in Shallow Water Ecosystems. *Diversity-Basel*, *11*(9), 19. https://doi.org/10.3390/d11090168
- Bullimore, B. A., Newman, P. B., Kaiser, M. J., Gilbert, S. E., & Lock, K. M. (2001).
  A study of catches in a fleet of "ghost-fishing" pots. *Fisheries Bulletin*, *99*, 247–253.
- Bunce, M., Rodwell, L. D., Gibb, R., & Mee, L. (2008). Shifting baselines in fishers' perceptions of island reef fishery degradation. *Ocean & Coastal Management*,

51, 285-302. https://doi.org/10.1016/j.ocecoaman.2007.09.006

Burton, M., Lock, K., Newman, P., & Jones, J. (2016). *Skomer MCZ Scallop Survey 2016* (Issue 196).

Cabanellas-Reboredo, M., Díaz, D., Mallol, S., Barberá, C., & Goñi, R. (2018). Morpho - demographic traits of two maërl - forming algae in beds with different depths and fishing histories. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *June 2017*, 133–145. https://doi.org/10.1002/agc.2827

Callaway, R., Desroy, N., Dubois, S. F., Fournier, J., Frost, M., Godet, L., Hendrick, V. J., & Rabaut, M. (2010). Ephemeral bio-engineers or reef-building polychaetes: How stable are aggregations of the tube worm lanice conchilega (Pallas, 1766)? *Integrative and Comparative Biology*, *50*(2), 237–250. https://doi.org/10.1093/icb/icg060

Cappo, M., Harvey, E. S., & Shortis, M. R. (2006). Counting and measuring fish with baited video techniques - an Overview. *Australian Society for Fish Biology Workshop Proceedings*, 101–114.

Cappo, M., Speare, P., & De'ath, G. (2004). Comparison of baited remote underwater video stations (BRUVS) and prawn (shrimp) trawls for assessments of fish biodiversity in inter-reefal areas of the Great Barrier Reef Marine Park. *Journal of Experimental Marine Biology and Ecology*, *302*, 123– 152. https://doi.org/10.1016/j.jembe.2003.10.006

Carruthers, T. J. B., Dennison, W. C., Longstaff, B. J., Waycott, M., Abal, E. G., McKenzie, L. J., & Lee Long, W. J. (2002). Seagrass habitats of northeast
Australia: Models of key processes and controls. *Bulletin of Marine Science*, *71*(3), 1153–1169.

- Cefas. (2012). Proposed UK Targets for achieving GES and Cost-Benefit Analysis for the MSFD: Final Report.
- Cefas. (2020a). Edible crab (Cancer pagurus) Cefas Stock Status Report 2019. Report to Defra.
- Cefas. (2020b). Lobster (Homarus gammarus) Cefas Stock Status Report 2019. Report to Defra.
- Cesar, H. S. J. (2002). Coral Reefs: Their Functions, Threats and Economic Value. In H. S. J. Cesar (Ed.), *Collected essays on the economics of coral reefs* (pp. 14–39).
- Chambers, P., & Binney, F. (2015). South-east Inshore Seabed Survey Initial Habitat Assessment. Societe Jersiaise Marine Biology Section.
- Chambers, P., & Binney, F. (2016). *Les Écréhous Seabed Survey Initial Habitat Assessment*. Societe Jersiaise Marine Biology Section.
- Chambers, P., Binney, F., & Jeffreys, G. (2016). Les Minquiers: A Natural History.
- Chambers, P., Morel, G., Binney, F., Jeffreys, G., & Blampied, S. (2020). Chapter 21 - Crossing jurisdictions: The implementation of offshore marine protected areas in an international fishery. Marine Protected Areas. In *Marine Protected Areas: Science, Policy and Management* (pp. 411–436).
- Chapman, M. G. (2012). Restoring Intertidal Boulder-Fields as Habitat for "Specialist" and "Generalist" Animals. *Restoration Ecology*, *2*0(2), 277–285.

https://doi.org/10.1111/j.1526-100X.2011.00789.x

- Charles, A., & Wilson, L. (2009). Human dimensions of Marine Protected Areas. *ICES Journal of Marine Science*, *66*, 6–15.
- Clarke, R., & Warwick, R. (2001). *Change in marine communities: an approach to statistical analysis and interpretation* (First). PRIMER-E, Plymouth.
- Claudet, J. (2018). Six conditions under which MPAs might not appear effective (when they are). *ICES Journal of Marine Science*, *75*(3), 1172–1174. https://doi.org/10.1093/icesjms/fsx074
- Claudet, J., Loiseau, C., Sostres, M., Zupan, M., Claudet, J., Loiseau, C., Sostres, M., & Zupan, M. (2020). Underprotected Marine Protected Areas in a Global Biodiversity Hotspot. *One Earth*, *2*(4), 380–384.
  https://doi.org/10.1016/j.oneear.2020.03.008
- Clavel, J., Julliard, R., & Devictor, V. (2011). Worldwide decline of specialist species: toward a global functional homogenization? *Frontiers in Ecology and the Environment*, *9*(4), 222–228. https://doi.org/10.1890/080216
- Coleman, E. C., Koenig, C. C., Huntsman, G. R., Musick, J. A., Eklund, A. M.,
  Mcgovern, J. C., Chapman, R. W., Sedberry, G. R., & Grimes, C. B. (2000).
  Long-lived Reef Fishes: The Grouper-Snapper Complex. *Fisheries*, *25*(3), 14–21.
- Coleman, F., & Williams, S. L. (2002). Overexploiting marine ecosystem engineers: potential consequences for biodiversity. *Trends in Ecology & Evolution*, *17*(1), 40–44. https://doi.org/10.1080/03155986.1999.11732391

Coleman, R. A., Hoskin, M. G., Carlshausen, E. Von, & Davis, C. M. (2013). Using 289

a no-take zone to assess the impacts of fishing: Sessile epifauna appear insensitive to environmental disturbances from commercial potting. *Journal of Experimental Marine Biology and Ecology*, *440*, 100–107. https://doi.org/10.1016/j.jembe.2012.12.005

- Coll, J., Garcia-Rubies, A., Morey, G., & Grau, A. M. (2012). The carrying capacity and the effects of protection level in three marine protected areas in the Balearic Islands (NW Mediterranean). *Scientia Marina*, *76*(4), 809–826. https://doi.org/10.3989/scimar.03531.02H
- Collie, J. S., Hall, S. J., Kaiser, M. J., & Poiner, I. R. (2000). A quantitative analysis of Fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, *69*, 785–798.
- Colton, M. A., & Swearer, S. E. (2010). A comparison of two survey methods:
  differences between underwater visual census and baited remote underwater
  video. *Marine Ecology Progress Series*, 400, 19–36.
  https://doi.org/10.3354/meps08377
- Cook, R., Farina-Franco, J. M., Gell, F. R., Holt, R. H. F., Holt, T., Lindenbaum, C., Porter, J. S., Seed, R., Skates, L. R., Stringell, T. B., & Sanderson, W. G. (2013). The Substantial First Impact of Bottom Fishing on Rare Biodiversity Hotspots: A Dilemma for Evidence-Based Conservation. *Plos One*, *8*(8), 1–10. https://doi.org/10.1371/journal.pone.0069904
- Cooper, K. M., Curtis, M., Hussin, W. M. R. W., Froján, C. R. S. B., Defew, E. C., Nye, V., & Paterson, D. M. (2011). Implications of dredging induced changes in sediment particle size composition for the structure and function of marine benthic macrofaunal communities. *Marine Pollution Bulletin*, *62*(10), 2087– 290

2094. https://doi.org/10.1016/j.marpolbul.2011.07.021

- Corgos, A., Bernárdez, C., Sampedro, P., Verísimo, P., & Freire, J. (2011). Spatial structure of the spider crab, Maja brachydactyla population: Evidence of metapopulation structure. *Journal of Sea Research*, 66(1), 9–19. https://doi.org/10.1016/j.seares.2011.04.011
- Cornwall IFCA. (2018a). Cornwall IFCA Monthly Shellfish Permit Statistics Analysis Edible crab ( Cancer pagurus ) Pot and Net Fisheries Summary Statistics 2016-2018. 1, 2–7.
- Cornwall IFCA. (2018b). Cornwall IFCA Monthly Shellfish Permit Statistics Analysis Lobster ( Homarus gammarus ) Pot Fishery Summary Statistics 2016-2018 Lobster ( Homarus gammarus ) Pot Fishery Summary Statistics 2016-2018. 1, 2016–2019.
- Cornwall IFCA. (2018c). Cornwall IFCA Monthly Shellfish Permit Statistics Analysis Spider crab (Maja spp .) Pot and Net Fisheries Summary Statistics 2016-2018 Cornwall IFCA Monthly Shellfish Permit Statistics Analysis Spider crab ( Maja spp .) Pot Fishery Summary Statistics 2.1, 2–7.
- Costa, L. B., Marinho, N. C. M., Gomes, P. B., Santos, P. J. P., Carvalho, P. V. V.
  C., & Botter-Carvalho, M. L. (2020). Interpopulation differences of the burrowing shrimp Callichirus major on urban beaches under different levels of fishing pressure. *Ocean and Coastal Management*, *197*(October 2019). https://doi.org/10.1016/j.ocecoaman.2020.105310
- Costanza, R., Costanza, R., Arge, R., Groot, R. De, Farber, S., Grasso, M., & Hannon, B. (1996). The value of the world's ecosystem services and natural

capital. Nature, 387.

- Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S. J., Kubiszewski, I., Farber, S., & Turner, R. K. (2014). Changes in the global value of ecosystem services. *Global Environmental Change*, *26*(1), 152–158. https://doi.org/10.1016/j.gloenvcha.2014.04.002
- Cullen-Unsworth, L., & Unsworth, R. (2013). Seagrass meadows, ecosystem services, and sustainability. *Environment*, 55(3), 14–28. https://doi.org/10.1080/00139157.2013.785864
- Daniel, W. W. (1990). *Applied Nonparametric Statistics* (2nd Edn. (ed.)). PWS-KENT Publishing.
- Dasgupta, P. (2021). *The Economics of Biodiversity : The Dasgupta Review*. London: HM Treasury.
- Davies, B. F. R., Attrill, M. J., Holmes, L., Rees, A., Witt, M. J., & Sheehan, E. V. (2020). Acoustic Complexity Index to assess benthic biodiversity of a partially protected area in the southwest of the UK. *Ecological Indicators*, *111*(106019), 1–10. https://doi.org/10.1016/j.ecolind.2019.106019
- Davies, B. F. R., Holmes, L., Bicknell, A., Attrill, M. J., & Sheehan, E. V. (2021). A
  Decade Implementing Ecosystem Approach to Fisheries Management
  Improves Diversity and Ecosystem Function Within a Marine Protected Area in
  the UK. *Diversity and Distributions*, 00, 1–16.
  https://doi.org/10.1111/ddi.13451
- Davies, B. F. R., Holmes, L., Rees, A., Attrill, M. J., Cartwright, A., & Sheehan, E. V. (2021). Ecosystem Approach to Fisheries Management works—How

switching from mobile to static fishing gear improves populations of fished and non-fished species inside a marine-protected area. *Journal of Applied Ecology*, *58*(11), 2463–2478.

- Davies, C., Moss, D., & Hill, M. (2004). *EUNIS Habitat Classification Revised 2004* (Issue October).
- Day, J., Dudley, N., Hockings, M., Holmes, G., Laffoley, D., Stolton, S., & Wells, S.
  (2012). *Guidelines for Applying the IUCN Protected Area Management Categories to Marine Protected Areas. Page 2.*
- Day, L., Saulnier, E., Pinsivy, L., & Brind, A. (2020). Benthic prey production index estimated from trawl survey supports the food limitation hypothesis in coastal fish nurseries. *Estuarine, Coastal and Shelf Science*, *235*.
  https://doi.org/10.1016/j.ecss.2020.106594
- De Best, R. (2021). Euro (EUR) to British pound sterling (GBP) exchange rate from January 1999 to February 10, 2021. Statista.Com. https://www.statista.com/statistics/412806/euro-to-gbp-average-annualexchange-rate/
- De Grave, S., & Whitaker, A. (1999). Benthic Community Re-adjustment following Dredging of a Muddy-Maerl Matrix. *Marine Pollution Bulletin*, *38*(2), 102–108.
- Revised Approach To The Management Of Commercial Fisheries In European Marine Sites - Overarching Policy And Delivery Document, (2013).
- Devillers, R., Pressey, R. L., Grech, A., Kittinger, J. N., Edgar, G. J., Ward, T., & Watson, R. (2015). *Reinventing residual reserves in the sea: are we favouring ease of establishment over need for protection?* 504(February 2014), 480–

504. https://doi.org/10.1002/aqc.2445

Devillers, R., Pressey, R. L., Ward, T. J., Grech, A., Kittinger, J. N., Edgar, G. J., & Watson, R. A. (2020). Residual marine protected areas five years on: Are we still favouring ease of establishment over need for protection? *Aquatic Conservation: Marine and Freshwater Ecosystems*, *March*, 1758–1764. https://doi.org/10.1002/aqc.3374

- Di Franco, A., Thiriet, P., Di Carlo, G., Dimitriadis, C., Francour, P., Gutiérrez, N.
  L., Jeudy De Grissac, A., Koutsoubas, D., Milazzo, M., Otero, M. D. M.,
  Piante, C., Plass-Johnson, J., Sainz-Trapaga, S., Santarossa, L., Tudela, S., &
  Guidetti, P. (2016). Five key attributes can increase marine protected areas
  performance for small-scale fisheries management. *Scientific Reports*,
  6(November), 1–9. https://doi.org/10.1038/srep38135
- Dobson, A., Lodge, D., Alder, J., Cumming, G. S., Keymer, J., McGlade, J.,
  Mooney, H., Rusak, J. A., Sala, O., Wolters, V., Wall, D., Winfree, R., &
  Xenopoulos, M. A. (2006). Habitat loss, trophic collapse, and the decline of
  ecosystem services. *Ecology*, *87*(8), 1915–1924.
  https://doi.org/10.1890/0012-9658(2006)87[1915:HLTCAT]2.0.CO;2
- Duarte, C. M. (2000). Marine biodiversity and ecosystem services: an elusive link. Journal of Experimental Marine Biology and Ecology, 250, 117–131.

Duarte, C. M., Agusti, S., Barbier, E., Britten, G. L., Castilla, J. C., Gattuso, J. P., Fulweiler, R. W., Hughes, T. P., Knowlton, N., Lovelock, C. E., Lotze, H. K., Predragovic, M., Poloczanska, E., Roberts, C., & Worm, B. (2020). Rebuilding marine life. *Nature*, *580*(7801), 39–51. https://doi.org/10.1038/s41586-020-2146-7

- Duarte, C. M., Losada, I. J., Hendriks, I., & Mazarrasa, I. (2013). The role of coastal plant communities for climate change mitigation and adaptation. *Nature Climate Change*, *3*. https://doi.org/10.1038/nclimate1970
- Duffy, J. E. (2006). Biodiversity and the functioning of seagrass ecosystems. *Marine Ecology Progress Series*, *311*, 233–250. https://doi.org/10.3354/meps311233
- Dunn, D. C., Boustany, A. M., & Halpin, P. N. (2011). Spatio-temporal management of fisheries to reduce by-catch and increase fishing selectivity. *Fish and Fisheries*, *12*, 110–119. https://doi.org/10.1111/j.1467-2979.2010.00388.x
- Durán, J., Palmer, M., & Pastor, E. (2013). Growing reared spider crabs (Maja squinado) to sexual maturity: The first empirical data and a predictive growth model. *Aquaculture*, 408–409, 78–87.
  https://doi.org/10.1016/j.aguaculture.2013.05.031
- Eaton, D. R., Brown, J., Addison, J. T., Milligan, S. P., & Fernand, L. J. (2003).
  Edible crab (Cancer pagurus) larvae surveys off the east coast of England: implications for stock structure. *Fisheries Research*, *65*, 191–199. https://doi.org/10.1016/j.fishres.2003.09.036
- Edgar, G. J., Barrett, N. S., & Morton, A. J. (2004). Biases associated with the use of underwater visual census techniques to quantify the density and sizestructure of fish populations. *Journal of Experimental Marine Biology and Ecology*, *308*, 269–290. https://doi.org/10.1016/j.jembe.2004.03.004

Edgar, G. J., & Samson, C. R. (2004). Catastrophic decline in mollusc diversity in

eastern Tasmania and its concurrence with shellfish fisheries. *Conservation Biology*, *18*(6), 1579–1588. https://doi.org/10.1111/j.1523-1739.2004.00191.x

- Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., Barrett, N. S., Becerro, M. A., Bernard, A. T. F., Berkhout, J., Buxton, C. D., Campbell, S. J., Cooper, A. T., Davey, M., Edgar, S. C., Försterra, G., Galván, D. E., Irigoyen, A. J., Kushner, D. J., ... Thomson, R. J. (2014). Global conservation outcomes depend on marine protected areas with five key features. *Nature*, *506*(7487), 216–220. https://doi.org/10.1038/nature13022
- Elliott, S. A. M., Milligan, R. J., Heath, M. R., Turrell, W. R., & Bailey, D. M. (2016).
  Disentangling habitat concepts for demersal marine fish management.
  Oceanography and Marine Biology: An Annual Review , volume 54 July 2016
  Abstra. Oceanography and Marine Biology An Annual Review, 54, 173–192.
- Elliott, S. A. M., Sabatino, A. D., Heath, M. R., Turrell, W. R., & Bailey, D. M. (2017). Landscape effects on demersal fish revealed by field observations and predictive seabed modelling. *Plos One*, *12*(12), 13. https://doi.org/10.1371/journal.pone.0189011
- Elliott, S. A. M., Turrell, W. R., Heath, M. R., & Bailey, D. M. (2017). Juvenile gadoid habitat and ontogenetic shift observations using stereo-video baited cameras. *Marine Ecology Progress Series*, *568*, 123–135. https://doi.org/10.3354/meps12068
- Eno, N. C., Frid, C. L. J., Hall, K., Ramsay, K., & Sharp, R. A. M. (2013). Assessing the sensitivity of habitats to fishing: from seabed maps to sensitivity maps. *Journal of Fish Biology*, 83, 826–846. https://doi.org/10.1111/jfb.12132 296

- Eno, N. C., Macdonald, D. S., Jim, A. M., Amos, S. C., Chapman, C. J., Robin, A., Bunker, F. S. P. D., Munro, C., St, F. P. D., & E, C. (2001). Effects of crustacean traps on benthic fauna. *ICES Journal of Marine Science*, *58*, 11– 20. https://doi.org/10.1006/jmsc.2000.0984
- Epstein, G., & Roberts, C. M. (2022). Protecting seabed sediment carbon for climate mitigation: a UK case study. *BioRxiv*, 1–23.
- European Commission. (2020). EU Biodiversity Strategy for 2030 Bringing nature back into our lives.
- European parliament. (2014). Directive 2014/89/EU Maritime Spatial Planning Directive 2014/89/EU of the European parliament and of the council of 23 July 2014 establishing a framework for maritime spatial planning.
- European Parliament. (2013). *Regulation (EU) No 1380/2013 of the European Parliament and of the Council of 11 December 2013 on the Common Fisheries Policy, amending Council Regulations (EC) No 1954/2003 and (EC) No 1224/2009 and repealing Council Regulations (EC) No 2371/2002 and (EC.)*
- European Parliament and Council. (2008). *Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive)*.
- Euskirchen, E. S., Goodstein, E. S., & Huntington, H. P. (2013). An estimated cost of lost climate regulation services caused by thawing of the Arctic cryosphere. *Ecological Applications*, *23*(8), 1869–1880.

- Falle, B., & Humphreys, G. E. (1886). Les Ecrehous. *The National Review.*, 8(44), 242.
- FAO. (2009). International Guidelines for the Management of Deep-Sea Fisheries in the High Seas.
- FAO. (2020). The state of world fisheries and aquaculture. Sustainability in action.
- Fenberg, P. B., & Roy, K. (2008). Ecological and evolutionary consequences of size-selective harvesting: How much do we know? *Molecular Ecology*, 17(1), 209–220. https://doi.org/10.1111/j.1365-294X.2007.03522.x
- Fletcher, S., Saunders, J., & Herbert, R. J. H. (2011). A review of the ecosystem services provided by broad-scale marine habitats in England's MPA network. *Journal of Coastal Research*, SPEC. ISSUE 64, 378–383.
- Fletcher, S., Saunders, J., Herbert, R. J. H., Roberts, C., & Dawson, K. (2012). Marine ecosystem services. Description of the ecosystem services provided by broad-scale habitats and features of conservation importance that are likely to be protected by Marine Protected Areas in the Marine Conservation Zone Project area.
- Fletcher, W. J., Shaw, J., Metcalf, S. J., & Gaughan, D. J. (2010). An Ecosystem Based Fisheries Management framework : the efficient , regional-level planning tool for management agencies. *Marine Policy*, *34*, 1226–1238. https://doi.org/10.1016/j.marpol.2010.04.007
- Fleury, C. (2011). Jersey and Guernsey: Two Distinct Approaches to Cross-Border Fishery Management. *Shima: The International Journal of Research into Island Cultures*, *5*(1), 24–43.

- Fleury, C., & Johnson, H. (2015). The Minquiers and Écréhous in spatial context: Contemporary issues and cross perspectives on border islands, reefs and rocks. *Island Studies Journal*, *10*(2), 163–180.
- Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., &
  Holling, C. S. (2004). Regime Shifts, Resilience, and Biodiversity in Ecosystem
  Management. *Annual Review of Ecology, Evolution, and Systematics*, *35*, 557–581. https://doi.org/10.1146/annurev.ecolsys.35.021103.105711
- Fourqurean, J. W., Duarte, C. M., Kennedy, H., Marbà, N., Holmer, M., Mateo, M. A., Apostolaki, E. T., Kendrick, G. A., & Krause-jensen, D. (2012). Seagrass ecosystems as a globally significant carbon stock. *Nature Geoscience*, 5(May), 505–510. https://doi.org/10.1038/NGEO1477
- Froese, R., Stern-pirlot, A., Winker, H., & Gascuel, D. (2008). Size matters: How single-species management can contribute to ecosystem-based fisheries management. *Fisheries Research*, *92*, 231–241. https://doi.org/10.1016/j.fishres.2008.01.005
- Frojan, C. R. S. B., Kendall, M. A., Paterson, G. L. J., Hawkins, L. E., Nimsantijaroen, S., & Aryuthaka, C. (2009). The importance of bare marine sedimentary habitats for maintaining high polychaete diversity and the implications for the design of marine protected areas. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *19*, 748–757. https://doi.org/10.1002/agc
- Gaines, S. D., White, C., Carr, M. H., & Palumbi, S. R. (2010a). Designing marine reserve networks for both conservation and fisheries management.
   Proceedings of the National Academy of Sciences of the United States of

America, 107(43), 18286–18293. https://doi.org/10.1073/pnas.0906473107

Gaines, S. D., White, C., Carr, M. H., & Palumbi, S. R. (2010b). Designing marine reserve networks for both conservation and fisheries management.
 Proceedings of the National Academy of Sciences of the United States of America, 107(43), 18286–18293. https://doi.org/10.1073/pnas.0906473107

Gall, S. C., & Rodwell, L. D. (2016). Evaluating the social acceptability of Marine Protected Areas. *Marine Policy*, 65, 30–38. https://doi.org/10.1016/j.marpol.2015.12.004

- Gall, S. C., Rodwell, L. D., Clark, S., Robbins, T., Attrill, M. J., Holmes, L. A., & Sheehan, E. V. (2020). The impact of potting for crustaceans on temperate rocky reef habitats: Implications for management. *Marine Environmental Research*, 162.
- Galparsoro, I., Borja, Á., Bald, J., Liria, P., & Chust, G. (2009). Predicting suitable habitat for the European lobster (Homarus gammarus), on the Basque continental shelf (Bay of Biscay), using Ecological-Niche Factor Analysis. *Ecological Monitoring*, *220*, 556–567.

https://doi.org/10.1016/j.ecolmodel.2008.11.003

- Gell, F. R., & Roberts, C. M. (2003). Benefits beyond boundaries: the fishery effects of marine reserves. *Trends in Ecology & Evolution*, 18(9), 448–455. https://doi.org/10.1016/s0169-5347(03)00189-7
- Gemert, R. Van, & Andersen, K. H. (2018). Implications of late-in-life densitydependent growth for fishery size-at-entry leading to maximum sustainable yield. *ICES Journal of Marine Science*, *75*(4), 1296–1305.

https://doi.org/10.1093/icesjms/fsx236

Giakoumi, S., McGowan, J., Mills, M., Beger, M., Bustamante, R. H., Charles, A., Christie, P., Fox, M., Garcia-Borboroglu, P., Gelcich, S., Guidetti, P., Mackelworth, P., Maina, J. M., McCook, L., Micheli, F., Morgan, L. E., Mumby, P. J., Reyes, L. M., White, A., ... Possingham, H. P. (2018). Revisiting "success" and "failure" of marine protected areas: A conservation scientist perspective. *Frontiers in Marine Science*, *5*(JUN), 1–5. https://doi.org/10.3389/fmars.2018.00223

- Githaiga, M. N., Frouws, A. M., Kairo, J. G., & Huxham, M. (2019). Seagrass Removal Leads to Rapid Changes in Fauna and Loss of Carbon. *Frontiers in Ecology and Evolution*, 7, 12. https://doi.org/10.3389/fevo.2019.00062
- Gnanalingam, G., Subritzky, P., Pritchard, D. W., Richards, D. K., Flack, B., & Hepburn, C. D. (2021). Local management to support local fisheries: Rahui (temporary closure) and bag limits for blackfoot abalone (Haliotis iris) in southern New Zealand. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *31*, 2320–2333. https://doi.org/10.1002/agc.3662
- Godet, L., Harmange, C., Marquet, M., Joyeux, E., & Fournier, J. (2018).
  Differences in home-range sizes of a bird species in its original, refuge and substitution habitats: challenges to conservation in anthropogenic habitats. *Biodiversity and Conservation*, *27*(3), 719–732.
  https://doi.org/10.1007/s10531-017-1460-3
- Goetze, J. S., Wilson, S., Radford, B., Fisher, R., Langlois, T. J., Monk, J., Knott, N. A., Malcolm, H., Currey-Randall, L. M., lerodiaconou, D., Harasti, D., Barrett,

N., Babcock, R. C., Bosch, N. E., Brock, D., Claudet, J., Clough, J.,

Fairclough, D. V., Heupel, M. R., ... Harvey, E. S. (2021). Increased connectivity and depth improve the effectiveness of marine reserves. *Global Change Biology*, *27*(15), 3432–3447. https://doi.org/10.1111/gcb.15635

- Gonzalez-Gurriaran, E., & Freire, J. (1994). Movement Patterns and Habitat
  Utilization in the Spider Crab Maja-Squinado (Herbst) (Decapoda, Majidae)
  Measured By Ultrasonic Telemetry. *Journal of Experimental Marine Biology and Ecology*, *184*(2), 269–291. https://doi.org/10.1016/0022-0981(94)90009-4
- Government of Jersey. (2008). Integrated Coastal Zone Management Strategy -Making the Most of Jersey's Coast.

Government of Jersey. (2013). Marine Resources Strategy Consultation.

- Graham, N. A. J., & Nash, K. L. (2013). The importance of structural complexity in coral reef ecosystems. *Coral Reefs*, *32*(2), 315–326. https://doi.org/10.1007/s00338-012-0984-y
- Gray, J. S. (1997). Marine biodiversity: patterns, threats and conservation needs. *Biodiversity & Conservation*, *6*, 153–175.
- Greathead, C., Buhl-mortensen, L., & Dannheim, J. (2020). A generic framework to assess the representation and protection of benthic ecosystems in European marine protected areas. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 30, 1253–1275. https://doi.org/10.1002/aqc.3401
- Green, A. E., Chadwick, M. A., Jones, P. J. S., & Green, A. E. (2021). Historical Analysis Exposes Catastrophic Seagrass Loss for the United Kingdom. *Frontiers in Plant Science*, *12*. https://doi.org/10.3389/fpls.2021.629962

- Groot, R. S. De, Wilson, M. A., & Boumans, R. M. J. (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*, *41*, 393–408.
- Haddad, N. M., Brudvig, L. A., Clobert, J., Davies, K. F., Gonzalez, A., Holt, R. D., Lovejoy, T. E., Sexton, J. O., Austin, M. P., Collins, C. D., Cook, W. M., Damschen, E. I., Ewers, R. M., Foster, B. L., Jenkins, C. N., King, A. J., Laurance, W. F., Levey, D. J., Margules, C. R., ... Townshend, J. R. (2015).
  Habitat fragmentation and its lasting impact on Earth's ecosystems. *Science Advances*, *1*(2), 1–10. https://doi.org/10.1126/sciadv.1500052
- Hall-Spencer, J. M. (1998). Conservation issues relating to maerl beds as habitats for molluscs. *Journal of Conchology*, 271–285. %3CGo
- Hall-Spencer, J. M., Grall, J., Moore, P. G., & Atkinson, R. J. A. (2003). Bivalve fishing and maerl-bed conservation in France and the UK retrospect and prospect. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *41*(April 2001). https://doi.org/10.1002/aqc.566
- Hall-Spencer, J. M., & Moore, P. G. (2000). Scallop dredging has profound, long-term impacts on maerl habitats. *Ices Journal of Marine Science*, *57*(5), 1407–1415. https://doi.org/10.1006/jmsc.2000.0918
- Hall-Spencer, J. M., Tasker, M., Soffker, M., Christiansen, S., Rogers, S., Campbell, M., & Hoydal, K. (2009). Design of Marine Protected Areas on high seas and territorial waters of rockall bank. *Marine Ecology Progress Series*, 397, 305–308. https://doi.org/10.3354/meps08235

Hall, S. J., Basford, D. J., Robertson, M. R., Raffaelli, D. G., & Tuck, I. (1991).

Patterns of recolonisation and the importance of pit-digging by the crab Cancer pagurus in a subtidal sand habitat. *Marine Ecology Progress Series*, 72(1–2), 93–102. https://doi.org/10.3354/meps072093

- Halpern, B. S. (2004). Habitat bottlenecks in stage-structured species: hermit crabs as a model system. *Marine Ecology Progress Series*, *276*, 197–207.
- Halpern, B. S., Frazier, M., Afflerbach, J., Lowndes, J. S., Micheli, F., Hara, C. O., Scarborough, C., & Selkoe, K. A. (2019). Recent pace of change in human impact on the world's ocean. *Scientific Reports*, *July 2019*, 1–9. https://doi.org/10.1038/s41598-019-47201-9
- Halpern, B. S., Frazier, M., Afflerbach, J., Lowndes, J. S., Micheli, F., Hara, C. O., Scarborough, C., & Selkoe, K. A. (2020). Recent pace of change in human impact on the world's ocean. *Scientific Reports Nature Research*, *9*(11609), 1–8. https://doi.org/10.1038/s41598-019-47201-9
- Halpern, B. S., Lester, S. E., & McLeod, K. L. (2010). Placing marine protected areas onto the ecosystem-based management seascape. *Proceedings of the National Academy of Sciences of the United States of America*, 107(43), 18312–18317. https://doi.org/10.1073/pnas.0908503107
- Halpern, B. S., Selkoe, K. A., Micheli, F., & Kappel, C. V. (2007). Evaluating and Ranking the Vulnerability of Global Marine Ecosystems to Anthropogenic Threats. *Conservation Biology*, *21*(5), 1301–1315.
  https://doi.org/10.1111/j.1523-1739.2007.00752.x
- Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V, Micheli, F., D'Agrosa, C., Bruno, J. F., Casey, K. S., Ebert, C., Fox, H. E., Fujita, R., Heinemann, D.,

Lenihan, H. S., Madin, E. M. P., Perry, M. T., Selig, E. R., Spalding, M., Steneck, R., & Watson, R. (2008). A global map of human impact on marine ecosystems. *Science*, *319*(5865), 948–952. https://doi.org/10.1126/science.1149345

- Harvey, E., Gounand, I., Ward, C. L., & Altermatt, F. (2017). Bridging ecology and conservation: from ecological networks to ecosystem function. *Journal of Applied Ecology*, *54*, 371–379. https://doi.org/10.1111/1365-2664.12769
- Hattam, C. E., Mangi, S. C., Gall, S. C., & Rodwell, L. D. (2014). Social impacts of a temperate fisheries closure: Understanding stakeholders' views. *Marine Policy*, 45, 269–278. https://doi.org/10.1016/j.marpol.2013.09.005
- Heck, K. L., Able, K. W., Roman, C. T., & Fahay, M. P. (1995). Composition, abundance, biomass, and production of macrofauna in a new-england estuary comparisons among eelgrass meadows and other nursery habitats. *Estuaries*, *18*(2), 379–389. https://doi.org/10.2307/1352320
- Heery, E. C., Bishop, M. J., Critchley, L. P., Bugnot, A. B., Airoldi, L., Mayer-pinto, M., Sheehan, E. V, Coleman, R. A., Loke, L. H. L., Johnston, E. L., Komyakova, V., Morris, R. L., Strain, E. M. A., Naylor, L. A., & Dafforn, K. A. (2017). Identifying the consequences of ocean sprawl for sedimentary habitats. *Journal of Experimental Marine Biology and Ecology*, *492*, 31–48. https://doi.org/10.1016/j.jembe.2017.01.020
- Hiddink, J. G., Jennings, S., Sciberras, M., Bolam, S. G., Cambiè, G.,
  McConnaughey, R. A., Mazor, T., Hilborn, R., Collie, J. S., Pitcher, C. R.,
  Parma, A. M., Suuronen, P., Kaiser, M. J., & Rijnsdorp, A. D. (2019).
  Assessing bottom trawling impacts based on the longevity of benthic

invertebrates. *Journal of Applied Ecology*, *56*(5), 1075–1084. https://doi.org/10.1111/1365-2664.13278

- Hiddink, J. G., Jennings, S., Sciberras, M., Szostek, C. L., Hughes, K. M., Ellis, N., Rijnsdorp, A. D., McConnaughey, R. A., Mazor, T., Hilborn, R., Collie, J. S., Pitcher, C. R., Amoroso, R. O., Parma, A. M., Suuronen, P., & Kaiser, M. J. (2017). Global analysis of depletion and recovery of seabed biota after bottom trawling disturbance. *Proceedings of the National Academy of Sciences of the United States of America*, *114*(31), 8301–8306. https://doi.org/10.1073/pnas.1618858114
- Hilborn, R., Oscar, R., Anderson, C. M., Baum, J. K., & Branch, T. A. (2020).
  Effective fisheries management instrumental in improving fish stock status. *PNAS*, *117*(4), 2218–2224. https://doi.org/10.1073/pnas.1909726116

Hilborn, R., Stokes, K., Maguire, J. J., Smith, T., Botsford, L. W., Mangel, M.,
Orensanz, J., Parma, A., Rice, J., Bell, J., Cochrane, K. L., Garcia, S., Hall, S.
J., Kirkwood, G. P., Sainsbury, K., Stefansson, G., & Walters, C. (2004). When
can marine reserves improve fisheries management? *Ocean & Coastal Management*, 47(3–4), 197–205.

https://doi.org/10.1016/j.oceoamann.2004.04.001

- Hines, A. H., Whitlatch, R. B., Thrush, S. F., Hewitt, J. E., Cummings, V. J., Dayton,
  P. K., & Legendre, P. (1997). Nonlinear foraging response of a large marine predator to benthic prey: Eagle ray pits and bivalves in a New Zealand sandflat. *Journal of Experimental Marine Biology and Ecology*, *216*(1–2), 191–210. https://doi.org/10.1016/S0022-0981(97)00096-8
- Hislop, J. R. G. (1988). The influence of maternal length and age on the size and 306

weight of the eggs and the relative fecundity of the haddock, Melanogrammus aeglefinus, in British waters. *Journal of Fish Biology*, *32*, 923–930.

HM Government. (2018). A Green Future: Our 25 Year Plan to Improve the Environment. In *Annex 1: Supplementary Evidence Report*. https://www.gov.uk/government/publications/25-year-environment-plan

Hooper, T., Beaumont, N., Griffiths, C., Langmead, O., & Somerfield, P. J. (2017).
Assessing the sensitivity of ecosystem services to changing pressures. *Ecosystem Services*, 24, 160–169.

https://doi.org/10.1016/j.ecoser.2017.02.016

Hori, M., Suzuki, T., Monthum, Y., Srisombat, T., Tanaka, Y., Nakaoka, M., &
Mukai, H. (2009). High seagrass diversity and canopy-height increase
associated fish diversity and abundance. *Marine Biology*, *156*, 1447–1458.
https://doi.org/10.1007/s00227-009-1184-3

Hosie, A. M. (2009). Maja brachydactyla Common spider crab. In Tyler-Walters H. and Hiscock K. (eds) Marine Life Information Network: Biology and Sensitivity Key Information Reviews, [on-line]. Plymouth: Marine Biological Association of the United Kingdom. [Cited 27-11-2020]. https://www.marlin.ac.uk/species/detail/2225

Hoskin, M. G., Coleman, R. A., Carlshausen, E. Von, & Davis, C. M. (2011).
Variable population responses by large decapod crustaceans to the establishment of a temperate marine no-take zone. *Canadian Journal of Fisheries and Aquatic Sciences*, *68*, 185–200. https://doi.org/10.1139/F10-143

- Hosokawa, S., Nakaoka, M., Miyoshi, E., & Kuwae, T. (2015). Seed dispersal in the seagrass Zostera marina is mostly within the parent bed in a protected bay. *Marine Ecology Progress Series*, *523*, 41–56. https://doi.org/10.3354/meps11146
- Howard, A. E., & Bennett, D. B. (1979a). The substrate preference and burrowing behaviour of juvenile lobsters (Homarus gammarus (L.)). *Journal of Natural History*, *13*(4).
- Howard, A. E., & Bennett, D. B. (1979b). The substrate preference and burrowing behaviour of juvenile lobsters (Homarus gammarus (L.)). *Journal of Natural History*, *13*(4), 433–438. https://doi.org/10.1080/00222937900770341
- Howarth, L. M., Roberts, C. M., Hawkins, J. P., Steadman, D. J., & Beukers-Stewart, B. D. (2015). Effects of ecosystem protection on scallop populations within a community-led temperate marine reserve. *Marine Biology*, *162*(4), 823–840. https://doi.org/10.1007/s00227-015-2627-7
- Howarth, L. M., Wood, H. L., Turner, A. P., & Beukers-Stewart, B. D. (2011). Complex habitat boosts scallop recruitment in a fully protected marine reserve. *Marine Biology*, *158*(8), 1767–1780. https://doi.org/10.1007/s00227-011-1690-y
- Hughes, T. P., Baird, A. H., Bellwood, D. R., Card, M., Connolly, S. R., Folke, C.,
  Grosberg, R., Hoegh-Guldberg, O., Jackson, J. B. C., Kleypas, J., Lough, J.
  M., Marshall, P., Nystrom, M., Palumbi, S. R., Pandolfi, J. M., Rosen, B., &
  Roughgarden, J. (2003). Climate change, human impacts, and the resilience
  of coral reefs. *Science*, *301*(5635), 929–933.

https://doi.org/10.1126/science.1085046

- Hunt, G. L., & McKinnell, S. (2006). Interplay between top-down, bottom-up, and wasp-waist control in marine ecosystems. *Progress in Oceanography*, 68(2–4), 115–124. https://doi.org/10.1016/j.pocean.2006.02.008
- Ibrahim, A., David, S., Bruno, G., Mark, C., Mohammed, A., & Vay, L. (2018).
  Decline in oyster populations in traditional fishing grounds; is habitat damage by static fishing gear a contributory factor in ecosystem degradation? *Journal of Sea Research*, *140*, 40–51. https://doi.org/10.1016/j.seares.2018.07.006
- ICES. (2018). Report of the Benchmark Workshop on Seabass (WKBASS).
- ICES CIEM. (2019). Working Group on the Biology and Life History of Crabs (WGCRAB; outputs from 2019 meeting) (Vol. 3, Issue 32).
- Jackson, E., Rees, S. E., Wilding, C., & Attrill, M. J. (2015). Use of a seagrass residency index to apportion commercial fishery landing values and recreation fisheries expenditure to seagrass habitat service. *Conservation Biology*, *29*(3), 899–909. https://doi.org/10.1111/cobi.12436
- Jackson, E., Rowden, A. A., Attrill, M. J., Bossy, S. F., & Jones, M. B. (2002). Comparison of fish and mobile macroinvertebrates associated with seagrass and adjacent sand at St. Catherine Bay, Jersey (English Channel): Emphasis on commercial species. *Bulletin of Marine Science*, *71*(3), 1333–1341.
- Jackson, J. B. C., Alexander, K. E., & Sala, E. (2011). *Shifting Baselines: The Past and Future of Ocean Fisheries*. Island Press.
- Jones, C. G., Lawton, J. H., & Shachak, M. (1994). Organisms as Ecosystem Engineers. *Oikos*, *69*(3), 373–386. https://doi.org/10.2307/3545850

Jones, H. P., & Schmitz, O. J. (2009). Rapid Recovery of Damaged Ecosystems.

- Juan, S. De, Thrush, S. F., & Demestre, M. (2007). Functional changes as indicators of trawling disturbance on a benthic community located in a fishing ground (NW Mediterranean Sea). *Marine Ecology Progress Series*, *334*, 117– 129.
- Kaiser, M. J., & Spencer, B. E. (1995). Survival of by-catch from a beam trawl. *Marine Ecology Progress Series*, *126*(1–3), 31–38.
  https://doi.org/10.3354/meps126031
- Kamenos, N. A., Moore, P. G., & Hall-Spencer, J. M. (2003). Substratum
  heterogeneity of dredged vs un-dredged maerl grounds. *Journal of the Marine Biological Association of the United Kingdom*, 83(2), 411–413.
  https://doi.org/10.1017/S0025315403007264h
- Kamenos, N. A., Moore, P. G., & Hall-Spencer, J. M. (2004a). Attachment of the juvenile queen scallop (Aequipecten opercularis (L.)) to Maerl in mesocosm conditions; juvenile habitat selection. *Journal of Experimental Marine Biology and Ecology*, 306(2), 139–155. https://doi.org/10.1016/j.jembe.2003.10.013
- Kamenos, N. A., Moore, P. G., & Hall-Spencer, J. M. (2004b). Maerl grounds provide both refuge and high growth potential for juvenile queen scallops (Aequipecten opercularis L.). *Journal of Experimental Marine Biology and Ecology*, 313(2), 241–254. https://doi.org/10.1016/j.jembe.2004.08.007
- Kaplan, D. M. (2009). Fish life histories and marine protected areas: an odd couple? *Marine Ecology Progress Series*, 377, 213–225. https://doi.org/10.3354/meps07825

- Kaplan, D. M., Botsford, L. W., O'Farrell, M. R., Gaines, S. D., & Jorgensen, S. (2009). Model-based assessment of persistence in proposed marine protected area designs. *Ecological Applications*, *19*(2), 433–448.
- Karnofsky, E. B., Atema, J., & Elgin, R. H. (1989). Field Observations of Social Behavior, Shelter Use, and Foraging in the Lobster, Homarus americanus. *The Biological Bulletin*, *176*(3), 239–246. https://doi.org/10.2307/1541982
- Klein, C. J., Brown, C. J., Halpern, B. S., Segan, D. B., Mcgowan, J., Beger, M., & Watson, J. E. M. (2015). Shortfalls in the global protected area network at representing marine biodiversity. *Nature Publishing Group, Scientific Reports*, 1–7. https://doi.org/10.1038/srep17539
- Klein, C. J., Tulloch, V. J., Halpern, B. S., Selkoe, K. A., Watts, M. E., Steinback,
  C., Scholz, A., & Possingham, H. P. (2013). Tradeoffs in marine reserve
  design: Habitat condition, representation, and socioeconomic costs. *Conservation Letters*, 6(5), 324–332. https://doi.org/10.1111/conl.12005
- Komyakova, V., Chamberlain, D., Jones, G. P., & Swearer, S. E. (2019). Assessing the performance of artificial reefs as substitute habitat for temperate reef fishes: Implications for reef design and placement. *Science of the Total Environment*, 668, 139–152. https://doi.org/10.1016/j.scitotenv.2019.02.357
- Kopp, D., Lefebvre, S., Cachera, M., Villanueva, M. C., & Ernande, B. (2015).
  Reorganization of a marine trophic network along an inshore-offshore gradient due to stronger pelagic-benthic coupling in coastal areas. *Progress in Oceanography*, *130*, 157–171. https://doi.org/10.1016/j.pocean.2014.11.001

Kritzer, J. P., Delucia, M. B., Greene, E., Shumway, C., Topolski, M. F., Thomas-

Blate, J., Chiarella, L. A., Davy, K. B., & Smith, K. (2016). The Importance of Benthic Habitats for Coastal Fisheries. *Bioscience*, *66*(4), 274–284. https://doi.org/10.1093/biosci/biw014

- Krueck, N. C., Ahmadia, G. N., Green, A., Jones, G. P., Possingham, H. P., Riginos, C., Treml, E. A., & Mumby, P. J. (2017). Incorporating larval dispersal into MPA design for both conservation and fisheries. *Ecological Applications*, 27(3), 925–941. https://doi.org/10.1002/eap.1495
- Kuempel, C. D., Jones, K. R., Watson, J. E. M., & Possingham, H. P. (2019).
  Quantifying biases in marine-protected-area placement relative to abatable threats. *Conservation Biology*, *33*(6), 1350–1359.
  https://doi.org/10.1111/cobi.13340
- Laffoley, D. (2021). The ocean and climate: why the call to protect at least 30 % of the ocean by 2030 must go hand in hand with ambitious climate action.
- Laffoley, D., Maltby, E., Vincent, M., Mee, L., Dunn, E., Gilliland, P., Hamer, J., Mortimer, D., & Pound, D. (2004). The Ecosystem Approach. Coherent actions for marine and coastal environments. A Report to the UK Government. *Report: English Nature*.
- Langlois, T. J., Harvey, E. S., & Meeuwig, J. J. (2012). Strong direct and inconsistent indirect effects of fishing found using stereo-video: Testing indicators from fisheries closures. *Ecological Indicators*, *23*, 524–534. https://doi.org/10.1016/j.ecolind.2012.04.030
- Laurel, B. J., Ryer, C. H., Knoth, B., & Stoner, A. W. (2009). Temporal and ontogenetic shifts in habitat use of juvenile Pacific cod (Gadus

macrocephalus). Journal of Experimental Marine Biology and Ecology, 377(1), 28–35. https://doi.org/10.1016/j.jembe.2009.06.010

Lawton, P., & Lavalli, K. L. (1995). Postlarval, Juvenile, Adolescent, and Adult Ecology. In *Biology of the Lobster* (pp. 47–88). https://doi.org/10.1016/b978-012247570-2/50026-8

Le Hir, P. (1986). Présentation de l'Etude, Hydrodynamique et Sédimentologie.

- Leary, B. C. O., Smart, J. C. R., Neale, F. C., Hawkins, J. P., Newman, S., Milman, A. C., & Roberts, C. M. (2012). Fisheries mismanagement. *Marine Pollution Bulletin*, *62*(12), 2642–2648. https://doi.org/10.1016/j.marpolbul.2011.09.032
- Lee, J., South, A. B., & Jennings, S. (2010). Developing reliable , repeatable , and accessible methods to provide high-resolution estimates of fishing-effort distributions from vessel monitoring system (VMS) data. *ICES Journal of Marine Science*, *67*(6), 1260–1271.
- Leenhardt, P., Low, N., Pascal, N., Micheli, F., & Claudet, J. (2015). The Role of Marine Protected Areas in Providing Ecosystem Services. *Book: Aquatic Funcitonal Biodiversity - An Ecological and Evolutionary Perspective*.
- Leleu, K., Alban, F., Pelletier, D., Charbonnel, E., Letourneur, Y., & Boudouresque,
  C. F. (2012). Fishers' perceptions as indicators of the performance of Marine
  Protected Areas (MPAs). *Marine Policy*, *36*(2), 414–422.
  https://doi.org/10.1016/j.marpol.2011.06.002
- Lester, S. E., & Halpern, B. S. (2008). Biological responses in marine no-take reserves versus partially protected areas. *Marine Ecology Progress Series*, 367, 49–56. https://doi.org/10.3354/meps07599

Lester, S. E., Halpern, B. S., Grorud-Colvert, K., Lubchenco, J., Ruttenberg, B. I., Gaines, S. D., Airame, S., & Warner, R. R. (2009). Biological effects within notake marine reserves: a global synthesis. *Marine Ecology Progress Series*, *384*, 33–46. https://doi.org/10.3354/meps08029

Lilley, R. J., & Unsworth, R. K. F. (2014). Atlantic Cod (Gadus morhua) benefits from the availability of seagrass (Zostera marina) nursery habitat. *Global Ecology and Conservation*, *2*, 367–377.
https://doi.org/10.1016/j.gecco.2014.10.002

- Lima, M. do A. C., Ward, R. D., & Joyce, C. B. (2019). Environmental drivers of sediment carbon storage in temperate seagrass meadows. *Hydrobiologia*, *847*, 1773–1792. https://doi.org/10.1007/s10750-019-04153-5
- Linnane, A., Ball, B., Munday, B., & Mercer, J. P. (2000). On the occurrence of juvenile lobster Homarus gammarus in intertidal habitat. *Journal of the Marine Biological Association of the United Kingdom*, 80(2), 375–376. https://doi.org/10.1017/S0025315499002039
- Linnane, A., Mazzoni, D., & Mercer, J. P. (2000). A long-term mesocosm study on the settlement and survival of juvenile European lobster Homarus gammarus
  L. in four natural substrata. *Journal of Experimental Marine Biology and Ecology*, 249(1), 51–64. https://doi.org/10.1016/s0022-0981(00)00190-8
- Lipcius, R. N., Schreiber, S. J., Seitz, R. D., & Shen, J. (2008). Importance of Metapopulation Connectivity to Restocking and Restoration of Marine Species. *Reviews in Fisheries Science*, *16*(1–3), 2008. https://doi.org/10.1080/10641260701812574

- Little, S., Spencer, K. L., Schuttelaars, H. M., Millward, G. E., & Elliot, M. (2017).
  Estuarine, Coastal and Shelf Science Unbounded boundaries and shifting baselines: Estuaries and coastal seas in a rapidly changing world. *Estuarine, Coastal and Shelf Science*, *198*, 311–319.
  https://doi.org/10.1016/j.ecss.2017.10.010
- Long, R. D., Charles, A., & Stephenson, R. L. (2015). Key principles of marine ecosystem-based management. *Marine Policy*, *57*, 53–60. https://doi.org/10.1016/j.marpol.2015.01.013
- Lotze, H. K., Bourque, B. J., Bradbury, R. H., & Cooke, R. G. (2006). Depletion, Degradation, and Recovery Potential of Estuaries and Coastal Seas. *Science*, *312*, 1806–1809. https://doi.org/10.1126/science.1128035
- Lotze, H. K., Coll, M., Magera, A. M., Ward-Paige, C., & Airoldi, L. (2011). Recovery of marine animal populations and ecosystems. *Trends in Ecology and Evolution*, *26*(11), 595–605. https://doi.org/10.1016/j.tree.2011.07.008
- Lubchenco, J., & Grorud-Colvert, K. (2015). Making waves: The science and politics of ocean protection. *Science*, *350*(6259), 382–383. https://doi.org/10.1126/science.aad5443
- Lundquist, C. J., Jones, T. C., Parkes, S. M., & Bulmer, R. H. (2018). Changes in benthic community structure and sediment characteristics after natural recolonisation of the seagrass Zostera muelleri. May, 1–9. https://doi.org/10.1038/s41598-018-31398-2
- Macreadie, P. I., Anton, A., Raven, J. A., Beaumont, N., Connolly, R. M., Friess, D. A., Kelleway, J. J., Kennedy, H., Kuwae, T., Lavery, P. S., Lovelock, C. E.,

Smale, D. A., Apostolaki, E. T., Atwood, T. B., Baldock, J., Bianchi, T. S., Chmura, G. L., Eyre, B. D., Fourqurean, J. W., ... Duarte, C. M. (2019). The future of Blue Carbon science. *Nature Communications*, *10*(1), 1–13. https://doi.org/10.1038/s41467-019-11693-w

- Malvern Instruments Ltd. (2004). *Malvern Instruments (2004) Mastersizer 2000E* operators guide.
- Mangi, S. C., Rodwell, L. D., & Hattam, C. (2011). Assessing the impacts of establishing MPAs on fishermen and fish merchants: The case of Lyme Bay, UK. *Ambio*, 40(5), 457–468. https://doi.org/10.1007/s13280-011-0154-4

Marine Conservation Institute. (2022). *The Marine Protection Atlas*. http://mpatlas.org

- Marine Resources. (2019). *Government of Jersey Marine Resources Annual Report 2019*.
- MarLIN. (2006). BIOTIC Biological Traits Information Catalogue. Marine Life Information Network. Plymouth: Marine Biological Association of the United Kingdom. %3Cwww.marlin.ac.uk/biotic%3E
- Marshall, C., & Wilson, E. (2008). Great scallop, (Pecten maximus) MarLIN Marine Life Information Network Biology and Sensitivity Key Information Review. *Aquaculture International*, 305–318.
- Marshall, K. N., Jensen, O. P., Koehn, L. E., Levin, P. S., & Essington, T. E. (2019). Inclusion of ecosystem information in US fish stock assessments suggests progress toward ecosystem-based fisheries management. *ICES Journal of Marine Science*, *76*(1), 1–9. https://doi.org/10.1093/icesjms/fsy152

- Maxwell, S. M., Hazen, E. L., Lewison, R. L., Dunn, D. C., Bailey, H., Bograd, S. J., Briscoe, D. K., Fossette, S., Hobday, A. J., Bennett, M., Benson, S., Caldwell, M. R., Costa, D. P., Dewar, H., Eguchi, T., Hazen, L., Kohin, S., Sippel, T., & Crowder, L. B. (2015). Dynamic ocean management: Defining and conceptualizing real-time management of the ocean. *Marine Policy*, *58*, 42– 50. https://doi.org/10.1016/j.marpol.2015.03.014
- Mazaris, A. D., Almpanidou, V., Giakoumi, S., & Katsanevakis, S. (2018). Gaps and challenges of the European network of protected sites in the marine realm. *ICES Journal of Marine Science*, 75(1), 190–198. https://doi.org/10.1093/icesjms/fsx125
- Mazarrasa, I., Samper-villarreal, J., Serrano, O., Lavery, P. S., Lovelock, C. E., Marbà, N., Duarte, C. M., & Cortés, J. (2018). Habitat characteristics provide insights of carbon storage in seagrass meadows. *Marine Pollution Bulletin*, *134*, 106–117. https://doi.org/10.1016/j.marpolbul.2018.01.059
- Mcarthur, M. A., Brooke, B. P., Przeslawski, R., Ryan, D. A., Lucieer, V. L., Nichol, S., Mccallum, A. W., Mellin, C., Cresswell, I. D., & Radke, L. C. (2010). On the use of abiotic surrogates to describe marine benthic biodiversity. *Estuarine, Coastal and Shelf Science*, *88*(1), 21–32.
  https://doi.org/10.1016/j.ecss.2010.03.003
- McClanahan, T. R., & Mangi, S. (2000). Spillover of Exploitable Fishes from a Marine Park and its Effect on the Adjacent Fishery. *Ecological Applications*, *10*(6), 1792–1805.
- McIntyre, F. D., Neat, F., Collie, N., Stewart, M., & Fernandes, P. G. (2015). Visual surveys can reveal rather different "pictures" of fish densities: Comparison of 317

trawl and video camera surveys in the Rockall Bank, NE Atlantic Ocean. Deep-Sea Research Part I-Oceanographic Research Papers, 95, 67–74. https://doi.org/10.1016/j.dsr.2014.09.005

- Mclean, D. L., Green, M., Harvey, E. S., Williams, A., Daley, R., & Graham, K. J. (2015). Comparison of baited longlines and baited underwater cameras for assessing the composition of continental slope deepwater fish assemblages off southeast Australia. *Deep-Sea Research Part I*, 98, 10–20. https://doi.org/10.1016/j.dsr.2014.11.013
- McShane, T., Hirsch, P. D., & Tran, T. (2011). Hard choices: Making trade-offs between biodiversity conservation and human well-being. *Biological Conservation*, 144(3), 966–972. https://doi.org/10.1016/j.biocon.2010.04.038
- Meadows, P. S., Meadows, A., & Murray, J. M. H. (2012). Biological modifiers of marine benthic seascapes: Their role as ecosystem engineers. *Geomorphology*, *157–158*, 31–48.

https://doi.org/10.1016/j.geomorph.2011.07.007

- Millennium Ecosystem Assessment. (2005). *Ecosystems and Human Well-Being: Synthesis*.
- MMO. (2019). UK Sea Fisheries Statistics 2019. A report produced for the Marine Management Organisation. First published: 24 September 2020.
   https://www.gov.uk/government/statistics/uk-sea-fisheries-annual-statisticsreport-2019
- Moran, D., Hussain, S., Fofana, A., Frid, C., Paramour, O., Robinson, L., & Winrow-Griffin, A. (2008). *Marine Bill Marine Nature Conservation Proposals*

- Valuing the Benefits. Final Report (Issue October).

- MPA Reality Check. (2019). MPA Reality Check Statistics. Mpa-Reality-Check.Org. https://map.mpa-reality-check.org/statistics/
- Mucientes, G., Irisarri, J., & Villegas-Ríos, D. (2019). Interannual fine-scale site fidelity of male ballan wrasse Labrus bergylta revealed by photo-identification and tagging. *Journal of Fish Biology*, *95*(4), 1151–1155. https://doi.org/10.1111/jfb.14111
- Nagelkerken, I., Sheaves, M., Baker, R., & Connolly, R. M. (2015). The seascape nursery: a novel spatial approach to identify and manage nurseries for coastal marine fauna. *Fish and Fisheries*, *16*(2), 362–371. https://doi.org/10.1111/faf.12057
- Nayar, S., Miller, D. J., Hunt, A., Goh, B. P. L., & Chou, L. M. (2007). Environmental effects of dredging on sediment nutrients, carbon and granulometry in a tropical estuary. *Environmental Monitoring and Assessment*, *127*, 1–13. https://doi.org/10.1007/s10661-006-9253-2
- Neal, K. J., & Wilson, E. (2008). Cancer pagurus Edible crab. In Tyler-Walters H. and Hiscock K. (eds) Marine Life Information Network: Biology and Sensitivity Key Information Reviews, [on-line]. Plymouth: Marine Biological Association of the United Kingdom. [Cited 27-11-2020].

https://www.marlin.ac.uk/species/detail/1179

Newell, R. C., Seiderer, L. J., & Hitchcock, D. R. (1998). The Impact of Dredging Works in Coastal Waters: A Review of the Sensitivity to Disturbance and Subsequent Recovery of Biological. *Oceanography and Marine Biology: An*  Annual Review, 36, 127–178.

Nieto, A., Ralph, G. M., Comeros-Raynal, T., M., Kemp, J., Criado, M. G., Allen, D.
J., Dulvy, N. K., Walls, R. H. L., Russell, B., Pollard, D., García, S., Craig, M.,
Collette, B. B., Pollom, R., Biscoito, M., Chao, N. L., Abella, A., Afonso, P., ...
Williams, J. T. (2015). *European Red List of Marine Fishes*.
http://edepot.wur.nl/345883

- Oberle, F. K. J., Puig, P., & Martín, J. (2017). Fishing Activities. In *Submarine Geomorphology* (pp. 503–534).
- OSPAR. (2002). Convention for the Protection of the Marine Environment of the North-East Atlantic.
- Ovando, D., Dougherty, D., & Wilson, J. R. (2016). Market and design solutions to the short-term economic impacts of marine reserves. *Fish and Fisheries*, *17*, 939–954. https://doi.org/10.1111/faf.12153
- Palma, A. T., Wahle, R. A., & Steneck, R. S. (1998). Different early post-settlement strategies between American lobsters Homarus americanus and rock crabs
  Cancer irroratus in the Gulf of Maine. *Marine Ecology Progress Series*, *162*, 215–225. https://doi.org/10.3354/meps162215
- Parrish, F. A., & Polovina, J. J. (1994). Habitat thresholds and bottlenecks in production of the spiny lobster (Panulirus marginatus) in the northwestern Hawaiian Islands. *Bulletin of Marine Science*, *54*(1), 151–163.
- Pauly, D, Christensen, V., Dalsgaard, J., Froese, R., & Torres, F. (1998). Fishing down marine food webs. *Science*, *279*(5352), 860–863. https://doi.org/10.1126/science.279.5352.860

- Pauly, D, Watson, R., & Alder, J. (2005). Global trends in world fisheries: impacts on marine ecosystems and food security. *Philosophical Transactions of the Royal Society B-Biological Sciences*, 360(1453), 5–12. https://doi.org/10.1098/rstb.2004.1574
- Pauly, Daniel. (2009). Beyond duplicity and ignorance in global fisheries\*. *Scientia Marina*, 73(2), 215–224. https://doi.org/10.3989/scimar.2009.73n2215
- Pendleton, L. H., Ahmadia, G. N., Browman, H. I., Thurstan, R. H., Kaplan, D. M.,
  & Bartolino, V. (2018). Debating the effectiveness of marine protected areas. *ICES Journal of Marine Science*, 75(3), 1156–1159.
  https://doi.org/10.1093/icesjms/fsx154
- Pérez Roda, M. A., Gilman, E., Huntington, T., Kennelly, S. J., Suuronen, P., Chaloupka, M., & Medley, P. (2019). A third assessment of global marine fisheries discards.
- Pieraccini, M., Coppa, S., & De Lucia, G. A. (2017). Beyond marine paper parks? Regulation theory to assess and address environmental non-compliance. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *27*(1), 177–196. https://doi.org/10.1002/aqc.2632
- Pikesley, S. K., Solandt, J.-L., Trundle, C., & Witt, M. J. (2021). Benefits beyond 'features ': Cooperative monitoring highlights MPA value for enhanced seabed integrity. *Marine Policy*, *134*, 1–10. https://doi.org/10.1016/j.marpol.2021.104801
- Pikitch, E. K., Santora, C., Babcock, E. A., Bakun, A., Bonfil, R., Conover, D. O., Dayton, P., Doukakis, P., Fluharty, D., Heneman, B., Houde, E. D., Link, J.,

Livingston, P. A., Mangel, M., McAllister, M. K., Pope, J., & Sainsbury, K. J. (2004). Ecosystem-Based Fishery Management. *Science*, *305*, 14–16.

- Pitcher, T. J., Kalikoski, D., Short, K., Varkey, D., & Pramod, G. (2009). An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries. *Marine Policy*, *33*, 223–232. https://doi.org/10.1016/j.marpol.2008.06.002
- Pittman, S. J., Rodwell, L. D., Shellock, R. J., Williams, M., Attrill, M. J., Bedford, J., Curry, K., Fletcher, S., Gall, S. C., Lowther, J., McQuatters-Gollop, A., Moseley, K. L., & Rees, S. E. (2019). Marine parks for coastal cities: A concept for enhanced community well-being, prosperity and sustainable city living. *Marine Policy*, *103*, 160–171. https://doi.org/10.1016/j.marpol.2019.02.012
- Plaster, A. (2017). The use of Baited Remote Underwater Video Stations (BRUVs) to study the relationship between habitat type and fish assemblages. Cardiff University.
- Pomeroy, R. S., Watson, L. M., Parks, J. E., & Cid, G. A. (2005). How is your MPA doing? A methodology for evaluating the management effectiveness of marine protected areas. *Ocean & Coastal Management*, 48, 485–502. https://doi.org/10.1016/j.ocecoaman.2005.05.004

Putten, I. van, Koopman, M., Fleming, A., Hobday, A. J., & Knuckey, I. (2019).
Fresh eyes on an old issue: Demand-side barriers to a discard problem. *Fisheries Research*, 209, 14–23. https://doi.org/10.1016/j.fishres.2018.09.007

R Core Team. (2021). R: a Language and Environment for Statistical Computing. R

Foundation for Statistical Computing.

- Rabaut, M., Guilini, K., Van Hoey, G., Vincx, M., & Degraer, S. (2007). A bioengineered soft-bottom environment: The impact of Lanice conchilega on the benthic species-specific densities and community structure. *Estuarine Coastal and Shelf Science*, *75*, 525–536.
- Ramirez, E. (2002). Fecundity and Life-history Strategies in Marine Invertebrates. Advances in Marine Biology, 43.
- Read, A. D., West, R. J., Haste, M., & Jordan, A. (2011). Optimizing voluntary compliance in marine protected areas: A comparison of recreational fisher and enforcement officer perspectives using multi-criteria analysis. *Journal of Environmental Management*, *92*(10), 2558–2567.
  https://doi.org/10.1016/j.jenvman.2011.05.022
- Rees, A. (2017). The Ecological Effects of Increasing Potting Density in the Lyme Bay Marine Protected Area.
- Rees, A., Sheehan, E. V, & Attrill, M. J. (2021a). Optimal fishing effort benefits fisheries and conservation. *Scientific Reports*, *11*(3784), 1–15. https://doi.org/10.1038/s41598-021-82847-4
- Rees, S. E., Ashley, M., Evans, L., Mangi, S., Sheehan, E. V, Mullier, T., Rees, A., & Attrill, M. J. (2021b). An evaluation of the social and economic impact of a Marine Protected Area on commercial fisheries. *Fisheries Research*, 235(August 2020). https://doi.org/10.1016/j.fishres.2020.105819
- Rees, S. E., Attrill, M. J., Austen, M. C., Mangi, S. C., Richards, J. P., & Rodwell, L. D. (2010). Is there a win-win scenario for marine nature conservation? A case
study of Lyme Bay, England. Ocean and Coastal Management, 53(3), 135– 145. https://doi.org/10.1016/j.ocecoaman.2010.01.011

- Rees, S. E., Attrill, M. J., Austen, M. C., Mangi, S. C., & Rodwell, L. D. (2013). A thematic cost-benefit analysis of a marine protected area. *Journal of Environmental Management*, *114*, 476–485.
  https://doi.org/10.1016/j.jenvman.2012.10.048
- Rees, S. E., Pittman, S. J., Foster, N., Langmead, O., Griffiths, C., Fletcher, S., Johnson, D. E., & Attrill, M. J. (2018). Bridging the divide: Social-ecological coherence in Marine Protected Area network design. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *28*(3), 754–763. https://doi.org/10.1002/aqc.2885
- Rees, S. E., Rodwell, L. D., Searle, S., & Bell, A. (2013). Identifying the issues and options for managing the social impacts of Marine Protected Areas on a small fishing community. In *Fisheries Research* (Vol. 146). Elsevier B.V. https://doi.org/10.1016/j.fishres.2013.04.003
- Rees, S. E., Sheehan, E. V, Jackson, E., Gall, S. C., Cousens, S. L., Solandt, J.,
  Boyer, M., & Attrill, M. J. (2013). A legal and ecological perspective of 'site integrity' to inform policy development and management of Special Areas of Conservation in Europe. *Marine Pollution Bulletin*, *72*, 14–21.
  https://doi.org/10.1016/j.marpolbul.2013.03.036
- Rees, S. E., Sheehan, E. V, Stewart, B. D., Clark, R., Appleby, T., Attrill, M. J.,
  Jones, P. J. S., Johnson, D., Bradshaw, N., Pittman, S., Oates, J., & Solandt,
  J.-L. (2020). Emerging themes to support ambitious UK marine biodiversity
  conservation. *Marine Policy*, *117* (March 2019).

https://doi.org/10.1016/j.marpol.2020.103864

- Reimer, J., & Devillers, R. (2021). Benefits and gaps in area-based management tools for the ocean Sustainable Development Goal. *Nature Sustainability*, 4(4), 1–9. https://doi.org/10.1038/s41893-020-00659-2
- Retière, C. (1979). Contribution à l'étude des peuplements ben-thiques du golfe normano-breton.
- Rhodes, N., Wilms, T., Baktoft, H., Ramm, G., Bertelsen, J. L., Flávio, H., Støttrup, J. G., Kruse, B. M., & Svendsen, J. C. (2020). Comparing methodologies in marine habitat monitoring research: An assessment of species-habitat relationships as revealed by baited and unbaited remote underwater video systems. *Journal of Experimental Marine Biology and Ecology*, *526*. https://doi.org/10.1016/j.jembe.2020.151315
- Riosmena-rodríguez, R. (2017). Chapter 1 Natural History of Rhodolith/Maërl Beds: Their Role in Near-Shore Biodiversity and Management. In *Rhodolith/Maërl Beds: A Global Perspective* (pp. 3–26). https://doi.org/10.1007/978-3-319-29315-8
- Roberts, C. M., Hawkins, J. P., & Gell, F. R. (2005). The role of marine reserves in achieving sustainable fisheries. *Philosophical Transactions of the Royal Society B-Biological Sciences*, *360*(1453), 123–132.
  https://doi.org/10.1098/rstb.2004.1578
- Robinson, M., & Tully, O. (2000). Spatial variability in decapod community structure and recruitment in sub-tidal habitats. *Marine Ecology Progress Series*, *194*(April), 133–141. https://doi.org/10.3354/meps194133

- Rochet, M. J., & Benoit, E. (2012). Fishing destabilizes the biomass flow in the marine size spectrum. *Proceedings of the Royal Society*, *279*, 284–292. https://doi.org/10.1098/rspb.2011.0893
- Rosenberg, A., Bigford, T. E., Leathery, S., Hill, R. L., & Bickers, K. (2000). Ecosystem approaches to fishery management through essential fish habitat. *Bulletin of Marine Science*, *66*(3), 535–542.
- Rosenberg, A., & McLeod, K. L. (2005). Implementing ecosystem-based approaches to management for the conservation of ecosystem services. *Marine Ecology Progress Series*, 300, 270–274. %3CGo
- Russ, G. R., & Alcala, A. C. (2004). Marine reserves: Long-term protection is required for full recovery of predatory fish populations. *Oecologia*, 138(4), 622–627. https://doi.org/10.1007/s00442-003-1456-4
- Russ, G. R., Alcala, A. C., Maypa, A. P., Calumpong, H. P., & White, A. T. (2004).
  Marine reserve benefits local fisheries. *Ecological Applications*, *14*(2), 597–606.
- Sala, E., & Giakoumi, S. (2018). No-take marine reserves are the most effective protected areas in the ocean. *ICES Journal of Marine Science*, *75*(3), 1166–1168. https://doi.org/10.1093/icesjms/fsx059
- Sala, E., Mayorga, J., Bradley, D., Cabral, R. B., Atwood, T. B., Auber, A., Cheung,
  W., Costello, C., Ferretti, F., Friedlander, A. M., Gaines, S. D., Garilao, C.,
  Goodell, W., Halpern, B. S., Hinson, A., Kaschner, K., Kesner-reyes, K.,
  Leprieur, F., Mcgowan, J., ... Mouillot, D. (2021). Protecting the global ocean
  for biodiversity, food and climate. *Nature*. https://doi.org/10.1038/s41586-

- Scheffer, M., Carpenter, S., & de Young, B. (2005). Cascading effects of overfishing marine systems. *Trends in Ecology & Evolution*, 20, 18–21. https://doi.org/10.1016/j.tree.2005.08.018
- Schmalenbach, I., Mehrtens, F., Janke, M., & Buchholz, F. (2011). A markrecapture study of hatchery-reared juvenile European lobsters, Homarus gammarus, released at the rocky island of Helgoland (German Bight, North Sea) from 2000 to 2009. *Fisheries Research*, *108*(1), 22–30. https://doi.org/10.1016/j.fishres.2010.11.016
- Seitz, R. D., Wennhage, H. kan, Bergstrom, U., Lipcius, R. N., & Ysebaert, T. (2014). Ecological value of coastal habitats for commercially and ecologically important species. *ICES Journal of Marine Science*, *71*(3), 648–665.
- Sheaves, M., Johnston, R., & Baker, R. (2016). Use of mangroves by fish: new insights from in-forest videos. *Marine Ecology Progress Series*, 549, 167–182. https://doi.org/10.3354/meps11690
- Sheehan, E. V, Bridger, D., & Attrill, M. J. (2015). The ecosystem service value of living versus dead biogenic reef. *Estuarine Coastal and Shelf Science*, 154, 248–254. https://doi.org/10.1016/j.ecss.2014.12.042
- Sheehan, E. V, Bridger, D., Cousens, S. L., & Attrill, M. J. (2015). Testing the resilience of dead maerl infaunal assemblages to the experimental removal and re-lay of habitat. *Marine Ecology Progress Series*, *535*, 117–128. https://doi.org/10.3354/meps11400

Sheehan, E. V, Cousens, S. L., Nancollas, S. J., Stauss, C., Royle, J., & Attrill, M.

J. (2013). Drawing lines at the sand: Evidence for functional vs. visual reef boundaries in temperate Marine Protected Areas. *Marine Pollution Bulletin*, *76*(1–2), 194–202. https://doi.org/10.1016/j.marpolbul.2013.09.004

Sheehan, E. V, Holmes, L. A., Davies, B. F. R., Cartwright, A., Rees, A., Attrill, M. J., Devlin, M. J., & Barrett, N. S. (2021). Rewilding of Protected Areas
Enhances Resilience of Marine Ecosystems to Extreme Climatic Events. *Frontiers in Marine Science*, 8(671427).
https://doi.org/10.3389/fmars.2021.671427

- Sheehan, E. V, Stevens, T. F., & Attrill, M. J. (2010). A quantitative, nondestructive methodology for habitat characterisation and benthic monitoring at offshore renewable energy developments. *PLoS ONE*, *5*(12). https://doi.org/10.1371/journal.pone.0014461
- Sheehan, E. V, Stevens, T. F., Gall, S. C., Cousens, S. L., & Attrill, M. J. (2013). Recovery of a Temperate Reef Assemblage in a Marine Protected Area following the Exclusion of Towed Demersal Fishing. *Plos One*, 8(12). https://doi.org/10.1371/journal.pone.0083883
- Shephard, S., Minto, C., Zolck, M., Jennings, S., Brophy, D., & Reid, D. (2014). Marine Science. *Encyclopedia of Environment and Society*, 71, 398–405. https://doi.org/10.4135/9781412953924.n678
- Silva, J. F., & Ellis, J. R. (2019). Bycatch and discarding patterns of dogfish and sharks taken in English and Welsh commercial fisheries. *Journal of Fish Biology*, 94, 966–980. https://doi.org/10.1111/jfb.13899

Skajaa, K., Ferno, A., Lokkeborg, S., & Haugland, E. K. (1998). Basic movement

pattern and chemo-oriented search towards baited pots in edible crab (Cancer pagurus L.). *Hydrobiologia*, *372*, 143–144.

- Skiftesvik, A. B., Blom, G., Agnalt, A. L., Durif, C. M. F., Browman, H. I., Bjelland,
  R. M., Harkestad, L. S., Farestveit, E., Paulsen, O. I., Fauske, M., Havelin, T.,
  Johnsen, K., & Mortensen, S. (2014). Wrasse (Labridae) as cleaner fish in
  salmonid aquaculture The Hardangerfjord as a case study. *Marine Biology Research*, *10*(3), 289–300. https://doi.org/10.1080/17451000.2013.810760
- Smithson, M., & Verkuilen, J. (2006). A better lemon squeezer? Maximumlikelihood regression with beta-distributed dependent variables. *Psychological Methods*, *11*(1), 54–71. https://doi.org/https://doi.org/10.1037/1082-989X.11.1.54
- Snelgrove, P. V. R. (1998). The biodiversity of macrofaunal organisms in marine sediments. *Biodiversity and Conservation*, 7(9), 1123–1132. https://doi.org/10.1023/A:1008867313340
- Snelgrove, P. V. R., Thrush, S. F., Wall, D. H., & Norkko, A. (2014). Real world biodiversity – ecosystem functioning: a seafloor perspective. *Trends in Ecology & Evolution*, *29*(7), 398–405. https://doi.org/10.1016/j.tree.2014.05.002
- Solandt, J.-L., Jones, P. J. S., Duval-Diop, D., Kleiven, A. R., & Frangoudes, K. (2014). Governance challenges in scaling up from individual MPAs to MPA networks. *Aquatic Conservation-Marine and Freshwater Ecosystems*, 24, 145–152. https://doi.org/10.1002/aqc.2504

Solandt, J.-L., Mullier, T., Elliott, S. A. M., & Sheehan, E. V. (2020). Managing

marine protected areas in Europe: moving from "feature-based" to "wholesite" management of sites. *Marine Protected Areas: Science, Policy and Management*, 157–181. https://doi.org/10.1016/B978-0-08-102698-4.00009-5

Somerfield, P. J., Mcclelland, I. L., Mcneill, C. L., Bolam, S. G., & Widdicombe, S. (2019). Environmental and sediment conditions, infaunal benthic communities and biodiversity in the Celtic Sea. *Continental Shelf Research*, *185*, 23–30. https://doi.org/10.1016/j.csr.2018.09.002

Southern IFCA. (2020). SIFCA Lobster Assessments 2010-2018 (unpublished).

- Staples, D., & Funge-Smith, S. (2009). Ecosystem approach to fisheries and aquaculture: Implementing the FAO Code of Conduct for Responsible Fisheries.
- Steneck, R. S., & Pauly, D. (2019). Fishing through the Anthropocene. *Current Biology*, *29*(19), R987–R992. https://doi.org/10.1016/j.cub.2019.07.081
- Stephenson, F., Mill, A. C., Scott, C. L., Polunin, N. V. C., & Fitzsimmons, C. (2017). Experimental potting impacts on common UK reef habitats in areas of high and low fishing pressure. *ICES Journal of Marine Science*, 74(6), 1648– 1659. https://doi.org/10.1093/icesjms/fsx013

Stevens, B. G., & Kittaka, J. (1998). Postlarval settling behavior, substrate preference, and time to metamorphosis for red king crab Paralithodes camtschaticus. *Marine Ecology Progress Series*, *167*, 197–206. https://doi.org/10.3354/meps167197

Stevens, T. F. (2003). Mapping Benthic Habitats for Representation in Marine

- Stewart, B. D., & Howarth, L. M. (2016). Quantifying and Managing the Ecosystem Effects of Scallop Dredge Fisheries. *Developments in Aquaculture and Fisheries Science*, 40, 585–609.
- Stratoudakis, Y., Fryer, R. J., Cook, R. M., Pierce, G. J., & Coull, K. A. (2001). Fish bycatch and discarding in Nephrops trawlers in the Firth of Clyde (west of Scotland) Fish bycatch and discarding in Nephrops trawlers in the Firth of Clyde (west of Scotland). *Aquatic Living Resources*, 14, 283–291.
- Stroup, W. W. (2012). Generalized Linear Mixed Models: Modern Concepts, Methods and Applications. CRC Press.
- Sumaila, R., Bellmann, C., & Tipping, A. (2016). Fishing for the future: An overview of challenges and opportunities. *Marine Policy*, *69*, 173–180. https://doi.org/10.1016/j.marpol.2016.01.003
- Sumaila, R., Cheung, W., Dyck, A., Gueye, K., Huang, L., Lam, V., Pauly, D.,
  Srinivasan, T., Swartz, W., Watson, R., & Zeller, D. (2012). Benefits of
  Rebuilding Global Marine Fisheries Outweigh Costs. *Plos One*, *7*(7).
  https://doi.org/10.1371/journal.pone.0040542
- Sundelöf, A., Grimm, V., Ulmestrand, M., & Fiksen, Ø. (2015). Modelling harvesting strategies for the lobster fishery in northern Europe: the importance of protecting egg-bearing females. *Population Ecology*, *57*(1), 237–251. https://doi.org/10.1007/s10144-014-0460-3
- Suuronen, P., Tschernij, V., Jounela, P., Valentinsson, D., & Larsson, P. (2007). Factors affecting rule compliance with mesh size regulations in the Baltic cod

trawl fishery. ICES Journal of Marine Science, 64(8), 1603–1606.

- Sweat, L. H., Stephens, M., & Reed, S. A. (2020). Insights from 15 years of benthic infaunal monitoring in a coastal lagoon system. *Proceedings of Indian River Lagoon Symposium 2020 Insights*, 147–161.
- Taylor, M. D., Baker, J., & Suthers, I. M. (2013). Tidal currents, sampling effort and baited remote underwater video (BRUV) surveys: Are we drawing the right conclusions? *Fisheries Research*, *140*, 96–104. https://doi.org/10.1016/j.fishres.2012.12.013
- TEEB. (2010). The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations.
- Thrush, S. F., & Dayton, P. K. (2002). Disturbance to marine benthic habitats by trawling and dredging: implications for marine biodiversity. *Annual Review of Ecology and Systematics*, *33*, 449–473.

https://doi.org/10.1146/annurev.ecolsys.33.010802.150515

- Thrush, S. F., Ellingsen, K. E., & Davis, K. (2016). Implications of fisheries impacts to seabed biodiversity and ecosystem-based management. *ICES Journal of Marine Science*, 73, 44–50.
- Thrush, S. F., Hewitt, J. E., Funnell, G. A., Cummings, V. J., Ellis, J., Schultz, D.,
  Talley, D., & Norkko, A. (2001). Fishing disturbance and marine biodiversity:
  role of habitat structure in simple soft-sediment systems. *Marine Ecology Progress Series*, 221, 255–264. https://doi.org/10.3354/meps221255
- Thrush, S. F., Hewitt, J. E., Kraan, C., Lohrer, A. M., Pilditch, C. A., & Douglas, E. (2017). Changes in the location of biodiversity– ecosystem function hot spots

across the seafloor landscape with increasing sediment nutrient loading. *Proceedings of the Royal Society B: Biological Sciences*, 284(1852). https://doi.org/10.1098/rspb.2016.2861

- Thurstan, R. H., Brockington, S., & Roberts, C. M. (2010). The effects of 118 years of industrial fishing on UK bottom trawl fisheries. *Nature Communications*, 1–6. https://doi.org/10.1038/ncomms1013
- Thurstan, R. H., Hawkins, J. P., Raby, L., & Roberts, C. M. (2013). Oyster (Ostrea edulis) extirpation and ecosystem transformation in the Firth of Forth,
  Scotland. *Journal for Nature Conservation*, *21*(5), 253–261.
  https://doi.org/10.1016/j.jnc.2013.01.004
- Thurstan, R. H., & Roberts, C. M. (2010). Ecological meltdown in the firth of clyde, Scotland: Two centuries of change in a coastal marine ecosystem. *PLoS ONE*, 5(7). https://doi.org/10.1371/journal.pone.0011767
- Tillin, H. M., Hiddink, J. G., Jennings, S., & Kaiser, M. J. (2006). Chronic bottom trawling alters the functional composition of benthic invertebrate communities on a sea-basin scale. *Marine Ecology Progress Series*, *318*, 31–45. https://doi.org/10.3354/meps318031
- Tonk, L., & Rozemeijer, M. (2019). *Ecology of the brown crab (Cancer pagurus) and production potential for passive fisheries in Dutch offshore wind farms*. http://library.wur.nl/WebQuery/wurpubs/553352
- Tracey, S. R., & Lyle, J. M. (2011). Linking scallop distribution and abundance with fisher behaviour: implication for management to avoid repeated stock collapse in a recreational fishery. *Fisheries Management and Ecology*, *18*, 221–232.

https://doi.org/10.1111/j.1365-2400.2010.00775.x

- Treasurer, J. W. (1994). The distribution, age and growth of wrasse (Labridae) in inshore waters of west Scotland. In *Journal of Fish Biology* (Vol. 44, Issue 5, pp. 905–918). https://doi.org/10.1111/j.1095-8649.1994.tb01263.x
- Turnbull, J., Shah Esmaeili, Y., & Clark, G. (2018). Key drivers of effectiveness in small marine protected areas. *Biodiversity and Conservation*, 27(9), 2217– 2242.
- Ulman, A., & Pauly, D. (2016). Making history count: The shifting baselines of Turkish fisheries. *Fisheries Research*, 183, 74–79. https://doi.org/10.1016/j.fishres.2016.05.013
- Ulrich, C., Vermard, Y., Dolder, P. J., Brunel, T., Jardim, E., Holmes, S. J., Kempf,
  A., Mortensen, L. O., Poos, J., & Rindorf, A. (2017). Achieving maximum sustainable yield in mixed fisheries: a management approach for the North
  Sea demersal fisheries. *ICES Journal of Marine Science*, *74*(2), 566–575.
  https://doi.org/10.1093/icesjms/fsw126
- UN General Assembly. (2015). *Transforming our world: the 2030 agenda for sustainable development*.
- UNEP. (2006a). Marine and coastal ecosystems and human well-being : a synthesis report based on the findings of the Millennium Ecosystem Assessment.
- UNEP. (2006b). Marine and Coastal Ecosystems and Human Well-Being: a synthesis report based on the findings of the Millennium Ecosystem Assessment.

Unsworth, R. K. F., Peters, J. R., McCloskey, R. M., & Hinder, S. L. (2014). Optimising stereo baited underwater video for sampling fish and invertebrates in temperate coastal habitats. *Estuarine Coastal and Shelf Science*, *150*, 281– 287. https://doi.org/10.1016/j.ecss.2014.03.020

- Van Hoey, G., Guilini, K., Rabaut, M., Vincx, M., & Degraer, S. (2008). Ecological implications of the presence of the tube-building polychaete Lanice conchilega on soft-bottom benthic ecosystems. *Marine Biology*, *154*(6), 1009–1019. https://doi.org/10.1007/s00227-008-0992-1
- Vause, B. J., Beukers-Stewart, B. D., & Brand, A. R. (2007). Fluctuations and forecasts on the fishery for queen scllops (Aequipecten opercularis) around the isle of man. *ICES Journal of Marine Science*, 64(6), 1124–1135. https://doi.org/10.1093/icesjms/fsm089
- Veale, L. O., Hill, A. S., Hawkins, S. J., & Brand, A. R. (2000). Effects of long-term physical disturbance by commercial scallop fishing on subtidal epifaunal assemblages and habitats. *Marine Biology*, *137*(2), 325–337. https://doi.org/10.1007/s002270000357
- Villnas, A., & Norkko, A. (2011). Benthic diversity gradients and shifting baselines: implications for assessing environmental status. *Ecological Applications*, 21(6), 2172–2186.
- Wahle, R. A., & Steneck, R. S. (1991). Recruitment habitats and nursery grounds of the American lobster Homarus americanus: a demographic bottleneck? *Marine Ecology Progress Series*, 69(3), 231–243.

Waldbusser, G. G., Marinelli, R. L., Whitlatch, R. B., & Visscher, P. T. (2004). The

effects of infaunal biodiversity on biogeochemistry of coastal marine sediments. *Limnology and Oceanography*, *49*(5), 1482–1492. https://doi.org/10.4319/lo.2004.49.5.1482

- Walker, T. I., Rigby, C. L., Pacoureau, N., Ellis, J., Kulka, D. W., Chiaramonte, G.
  E., & Herman, K. (2020). *Galeorhinus galeus. The IUCN Red List of Threatened Species 2020: e.T39352A2907336.*https://doi.org/10.2305/IUCN.UK.2020-2.RLTS.T39352A2907336.en
- Walmsley, S. F., Bowles, A., Eno, N. C., & West, N. (2015). Evidence for Management of Potting Impacts on Designated Features.
- Watling, L. E. S., & Norse, E. A. (1998). Disturbance of the Seabed by Mobile Fishing Gear: A Comparison to Forest Clearcutting. *Conservation Biology*, *12*(6), 1180–1197.
- Watson, D. L., Harvey, E. S., Anderson, M. J., & Kendrick, G. A. (2005). A comparison of temperate reef fish assemblages recorded by three underwater stereo-video techniques. *Marine Biology*, *148*(2), 415–425.
  https://doi.org/10.1007/s00227-005-0090-6
- Watson, H. V., McKeown, N. J., Coscia, I., Wootton, E., & Ironside, J. E. (2016).
  Population genetic structure of the European lobster (Homarus gammarus) in the Irish Sea and implications for the effectiveness of the first British marine protected area. *Fisheries Research*, *183*, 287–293.
  https://doi.org/10.1016/j.fishres.2016.06.015
- Whalan, S., & Webster, N. S. (2014). Sponge larval settlement cues: the role of microbial biofilms in a warming ocean. *Scientific Reports*, *4*(4072).

White, J., Simpfendorfer, C. A., Tobin, A. J., & Heupel, M. R. (2013). Application of baited remote underwater video surveys to quantify spatial distribution of elasmobranchs at an ecosystem scale. *Journal of Experimental Marine Biology and Ecology*, 448, 281–288.

https://doi.org/10.1016/j.jembe.2013.08.004

- Whomersley, P. (2014). Technical Guideline No . 02 Collection of benthic grab samples and impact assessment associated with subsea oil releases and dispersant use in UK waters. 02.
- Wickham, H., Averick, M., Bryan, J., Chang, W., McGowan, L., François, R.,
  Grolemund, G., Hayes, A., Henry, L., Hester, J., Kuhn, M., Pedersen, T.,
  Miller, E., Bache, S., Müller, K., Ooms, J., Robinson, D., Seidel, D., Spinu, V.,
  ... Yutani, H. (2019). Welcome to the {tidyverse}. *Journal of Open Source Software*, 4(43), 1686. https://doi.org/10.21105/joss.01686
- Wilcox, C., & Donlan, C. J. (2007). Compensatory mitigation as a solution to fisheries bycatch – biodiversity conservation conflicts. *Frontiers in Ecology and the Environment*, 5(6), 325–331.
- Willis, T. J., Millar, R. B., & Babcock, R. C. (2000). Detection of spatial variability in relative density of fishes: comparison of visual census, angling, and baited underwater video. *Marine Ecology Progress Series*, 198, 249–260.
- Willis, T. J., Millar, R. B., & Babcock, R. C. (2003). Protection of exploited fish in temperate regions: high density and biomass of snapper Pagrus auratus (Sparidae) in northern New Zealand marine reserves. *Journal of Applied*

Wilson, S., Blake, C., Berges, J. A., & Maggs, C. A. (2004). Environmental tolerances of free-living coralline algae (maerl): implications for European marine conservation. *Biological Conservation*, *120*(2), 279–289. https://doi.org/10.1016/j.biocon.2004.03.001

Wood, H. (2018). Image of Dredging Impact on Seabed. In COAST.

- Woodin, S. A., Volkenborn, N., Pilditch, C. A., Lohrer, A. M., Wethey, D. S., Hewitt, J. E., & Thrush, S. F. (2016). Same pattern, different mechanism: Locking onto the role of key species in seafloor ecosystem process. *Scientific Reports*, 6, 1–11. https://doi.org/10.1038/srep26678
- Worm, B., Barbier, E. B., Beaumont, N., Duffy, J. E., Folke, C., Halpern, B. S.,
  Jackson, J. B. C., Lotze, H. K., Micheli, F., Palumbi, S. R., Sala, E., & Selkoe,
  K. A. (2006). Impacts of Biodiversity Loss on Ocean Ecosystem Services. *Science*, *314*, 787–790. https://doi.org/10.1126/science.1132294
- Worm, B., & Duffy, J. E. (2003). Biodiversity, productivity and stability in real food webs. *Trends in Ecology and Evolution*, *18*(12), 628–632.
  https://doi.org/10.1016/j.tree.2003.09.003
- Wright, J. P., & Jones, C. G. (2004). Predicting effects of ecosystem engineers on patch-scale species richness from primary productivity. *Ecology*, 85(8), 2071– 2081. https://doi.org/10.1890/02-8018
- Wylie, L., Sutton-grier, A. E., & Moore, A. (2016). Keys to successful blue carbon projects: Lessons learned from global case studies. *Marine Policy*, 65, 76–84. https://doi.org/10.1016/j.marpol.2015.12.020

- Yagnesh, M., Durga, F., Raj, D., Rehanavaz, M., & Poojaben, T. (2020). Importance of sharks in ocean ecosystem. *Journal of Entomology and Zoology Studies*, 8(1). http://www.entomoljournal.com
- Zhao, B., Zhang, S., & Qian, P. (2020). Larval settlement of the silver- or goldlip pearl oyster Pinctada maxima (Jameson) in response to natural biofilms and chemical cues. *Aquaculture*, *220*, 883–901. https://doi.org/10.1016/S0044-8486(02)00567-7
- Zupan, M., Bulleri, F., Evans, J., Fraschetti, S., Guidetti, P., Garcia-Rubies, A.,
  Sostres, M., Asnaghi, V., Caro, A., Deudero, S., Goni, R., Guarnieri, G.,
  Guilhaumon, F., Kersting, D., Kokkali, A., Kruschel, C., Macic, V., Mangialajo,
  L., Mallol, S., ... Claudet, J. (2018). How good is your marine protected area
  at curbing threats? *Biological Conservation*, *221*, 237–245.
  https://doi.org/10.1016/j.biocon.2018.03.013