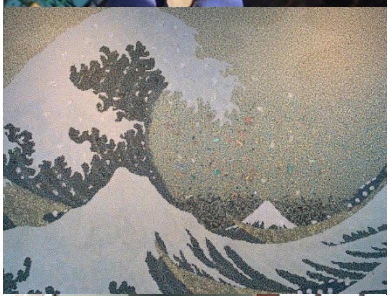

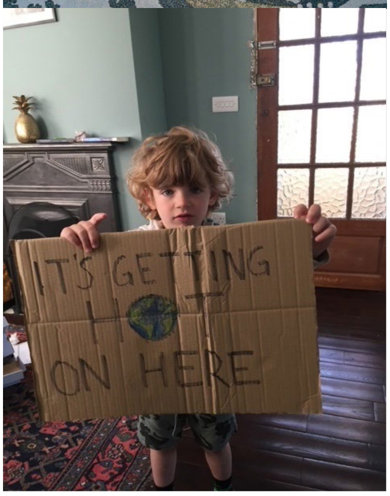


# Micro to macro and species to society – lessons learnt at the science and policy interface

Duncan Vaughan

A thesis submitted for the degree of

PhD by Publication

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University of East Anglia

Norwich, United Kingdom

31 March 2024

**Dedicated to Jane Vaughan**

**17/11/1946 – 18/04/2020**

# Abstract

The science-policy interface (SPI) comprises the processes that encompass relationships between scientists and others in policy development, which allows for discussion of knowledge and consideration of evidence with the aim of improving decision making. Development of effective responses to the climate and biodiversity crises at the scale and pace required will necessitate scientists, policy officials, politicians, and managers to effectively interact, co-design, communicate, and deliver these responses, i.e. work effectively at the SPI.

This thesis sets out the nature and impact of a body of work conducted by the author in fulfilment of a PhD by Publication. Critical analysis of the work tracks the author's career development from early-career scientist to senior government policy advisor. The work follows a trajectory of focus from the micro (species) to macro (policy development) level. The influence of the work is considered, and for the more recent research, recommendations on how the work can be progressed are set out.

The body of work completed at the SPI has principally influenced scientific understanding and policy development with regards to how marine resources, including species, habitats and space itself, should be used and managed for the benefit of wider society. Critical analysis highlighted that the prevailing SPI model could be modified to incorporate an additional component – the knowledge exchange between stakeholders considering the effects of management.

The time span of the work, varying focus of the publications considered and various positions held by the author have enabled, following reflection and consideration of the critical analysis, the identification of nine elements key to maximising effectiveness when working at the SPI: challenge, empathise, identify governance and knowledge gaps, network, collaborate with international experts, understand politics, develop trust of decision makers, communicate and persevere. These elements can help those working at the SPI influence commissioning and contribute to science that informs and influences policy decisions to deliver environmental outcomes and progress nature recovery.

# Preamble

## Author background

I am a Principal Advisor working at Natural England – the Statutory Nature Conservation Body in England. In this role I provide conservation advice to government policy officials, regulators, and developers on the pressures that activities in the marine environment exert on protected features (species and habitats). A key component of my role is to provide government with conservation advice when developing legislation that has implications for the environment. Recent advice has influenced the development of the Fisheries Act 2020, the Environment Act 2021, and the Environmental Improvement Plan – 2023, in addition to 2020 amendments to the Marine and Coastal Access Act 2009.

Over the last 20 years, I have authored publications related to the management of species and habitats for fisheries and conservation purposes. A selection of these publications is considered within this thesis for examination for a PhD by Publication.

## Purpose of the thesis

This thesis [1] provides the evidence required by the University of East Anglia (UEA) to allow me to be examined for the degree of Doctor of Philosophy. Section A is a list of the publications on which the assessment is to be based. Section B is a critical analysis of the published works. Appendix A provides confirmation of my contribution to the multi-author papers listed in Section A. Appendix B consists of the publications assessed.

[1] Suggested citation: Vaughan, D. (2024) Micro to macro and species to society – lessons learnt at the science and policy interface. PhD Thesis. University of East Anglia.

Cover images: top to bottom. The authors first fishing trip circa 1988. V-notching a berried lobster in Maine USA in 2018 – one of the oldest fisheries management measures. *Gyre* 2009 by Chris Jordan on display at the Monterey Bay Aquarium (the image is constructed of 2.4 million piece of marine plastic – the amount estimated to enter the sea each hour). Rohan Vaughan – Earth Day 2019.

## Acknowledgements

This collection of work is reflective of the societal response required to address the climate and biodiversity crisis – it is global, collaborative, and multidecadal. There has not been any personal 'grand plan' as I have weaved my way around the world working on many different topics. The work presented in this thesis draws upon the wisdom, insights, passion, and collegiate working from collaborators across the globe and from multiple disciplines, diverse cultures, nationalities, and perspectives. Without fail they all have in common a desire to make a difference for this and future generations, and for those in coastal communities at the forefront of the climate and biodiversity crises.

I take this opportunity to thank my supervisors on this PhD journey: Carol Robinson, John Pinnegar and Stefanie Nolte for their support, encouragement, and guidance. I also thank Natural England for supporting me financially to fulfil this PhD by Publication. I have had the good fortune to have worked with brilliant scientists, regulators, policy officials and marine users. There are, however, several notable individuals that have introduced me to the marine environment, and shaped my work and my outlook on life that I would like to acknowledge here: Jane Vaughan, Ken Vaughan, Kate Lazar Vaughan, Rohan Vaughan, Mark Day, Mat Mander and Mark Duffy.

# Contents

## Appendix A - confirmation of authorship

## Appendix B - published papers

Cumming, R.L., Toscano, M.A., Lovell, E.R., Carlson, B.A., Dulvy, N.K., Hughes, A, Koven, J.F., Quinn, N.J., Sykes, H.R., Taylor, O.J.S., and D. Vaughan (2000). Mass coral bleaching in the Fiji Islands, 2000. Proc 9th Int Coral Reef Symp, Bali, Indonesia. Australian Institute of Marine Science, Townsville, Australia [publication 1]

Clerveaux, W. and D. Vaughan (2003) An investigation of the effects of increasing fishing efficiency on the productivity of the queen conch (*Strombus gigas*) and Caribbean spiny lobster (*Panulirus argus*) fisheries within the Turks and Caicos Islands. Proc 54th Annu Gulf Carib Fish Inst, 12–17 Nov 2001, Providenciales, Turks and Caicos Islands, p 285–296 [publication 2]

Vaughan, D. (2004) A description of fisheries management in the Turks and Caicos Islands: an overview of the problems and suggestions for mitigation. *Proc 55th Annu Gulf Carib Fish Inst*, 11–15 Nov 2002, Xel Ha, Mexico, p 44–55 [publication 3]

Wilson, D.T., Vaughan, D., Wilson, S.K., Simon, C.N. and K. Lockhart (2008) Efficacy of a starch-iodide swab technique to detect the illegal use of bleach in a Spiny Lobster (*Panulirus argus*) fishery. Fisheries Research. 90 (1-3), pp 86-91 [publication 4]

Vaughan, D. (2017) Fishing effort displacement and the consequences of implementing Marine Protected Area management – An English perspective. Marine Policy. 84 : 228-234 [publication 5]

Vaughan, D. and T. Agardy (2019) Marine protected areas and marine spatial planning - allocation of resource use and environmental protection In: *Marine Protected Areas: Science, Policy and Practice*. (Eds. Humphreys and Clark), Elsiver [publication 6]

Vaughan, D., Shrimpton, E.A., Carpenter, G., Skerritt, D.J. and C. Williams (2021) Marinising a terrestrial concept: Public money for public goods. Ocean & Coastal Management. 213, p.105881 [publication 7]

Hague, E., Sparling, C., Morris, C., Vaughan, D., Walker, R., Culloch, R., Lyndon, R., Fernandes, T. and L. McWhinnie (2022) Same species, same space, different standards: a review of cumulative effects assessment practice for marine mammals. Frontiers in Marine Science. 25 (9) [publication 8]

Vaughan, D., Skerritt, D.J., Duckworth, J., Sumalia, U.R. and M. Duffy (2023) The use of fuel tax concessions in the UK commercial fishing fleet. Marine Policy. 155 [publication 9]

## List of figures



## List of tables

## Section A. Publications: impact factor and role.

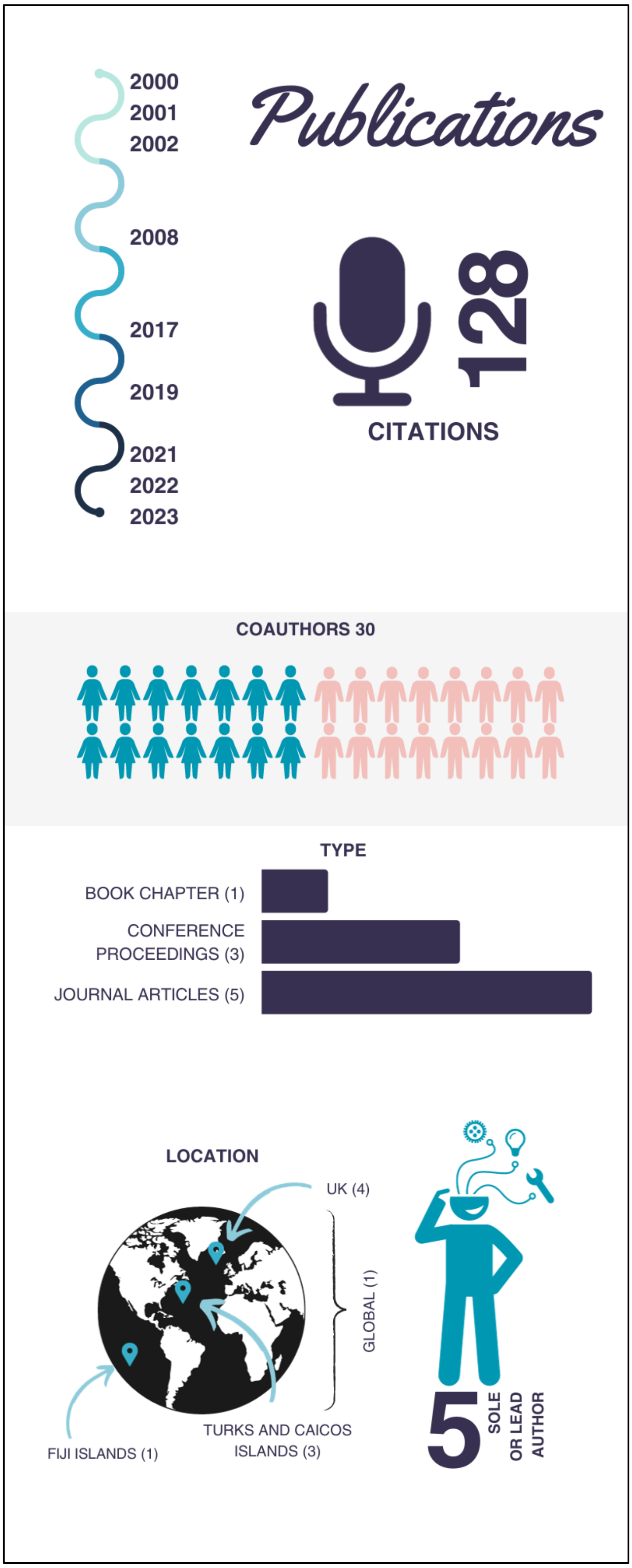


*Figure 1. Publication synopsis*

The following is a list of publications on which the thesis is based, including impact factor of the journal and my role in each of these studies.

**Publication 1:**

Cumming, R.L., Toscano, M.A., Lovell, E.R., Carlson, B.A., Dulvy, N.K., Hughes, A, Koven, J.F., Quinn, N.J., Sykes, H.R., Taylor, O.J.S., and **D. Vaughan** (2000) *Mass coral bleaching in the Fiji Islands, 2000*. Proc 9th Int Coral Reef Symp, Bali, Indonesia. Australian Institute of Marine Science, Townsville, Australia. *Cited by* 50

*Role:* Co-investigator, conception and design, development of methodology, acquisition of data, analysis and interpretation of data, writing, review, and/or revision of the manuscript.

https://coralreefwatch.noaa.gov/satellite/publications/crbpub_9thicrscumming.pdf

**Publication 2:**

Clerveaux, W. and **D. Vaughan** (2003) *An investigation of the effects of increasing fishing efficiency on the productivity of the queen conch (Strombus gigas) and Caribbean spiny lobster (Panulirus argus) fisheries within the Turks and Caicos Islands*. Proc 54th Annu Gulf Carib Fish Inst, 12–17 Nov 2001, Providenciales, Turks and Caicos Islands, p 285–296. *Cited by* 8

Role: Co-investigator, conception and design, development of methodology, acquisition of data, analysis and interpretation of data, writing, review, and revision of the manuscript.

https://aquadocs.org/bitstream/handle/1834/29501/gcfi_54-22.pdf?sequence=1

**Publication 3:**

**Vaughan, D.** (2004) *A description of fisheries management in the Turks and Caicos Islands: an overview of the problems and suggestions for mitigation*. Proc 55th Annu Gulf Carib Fish Inst, 11–15 Nov 2002, Xel Ha, Mexico, p 44–55. *Cited by* 3

*Role:* Investigator, conception and design, development of methodology, acquisition of data, analysis and interpretation of data, writing, review, and/or revision of the manuscript.

https://aquadocs.org/bitstream/handle/1834/29554/gcfi_55-5.pdf?sequence=1

**Publication 4:**

Wilson, D.T., **Vaughan, D.**, Wilson, S.K., Simon, C.N. and K. Lockhart (2008) *Efficacy of a starch-iodide swab technique to detect the illegal use of bleach in a Spiny Lobster (Panulirus argus) fishery*. Fisheries Research, 90(1-3), pp 86-91.

*Journal Impact Factor: 1.434*[2]. *Cited by* 4

*Role:* Co-investigator, conception and design, development of methodology, acquisition of data, analysis and interpretation of data, writing, review, and/or revision of the manuscript.

https://www.sciencedirect.com/science/article/abs/pii/S0165783607002494

**Publication 5:**

**Vaughan, D.** (2017) *Fishing effort displacement and the consequences of implementing Marine Protected Area management – An English perspective*. Marine Policy. 84:228-234.

*Journal Impact Factor: 2.109*. *Cited by* 32

*Role:* Investigator, conception and design, development of methodology, acquisition of data, analysis and interpretation of data, writing, review, and/or revision of the manuscript.

https://www.sciencedirect.com/science/article/abs/pii/S0308597X16307588

**Publication 6:**

**Vaughan, D.** and T. Agardy (2019) *Marine protected areas and marine spatial planning - allocation of resource use and environmental protection* In: Marine Protected Areas: Science, Policy and Practice. (Eds. Humphreys and Clark), Elsevier. *Cited by* 25

*Role:* Principal investigator, conception and design, development of methodology, acquisition of data, analysis and interpretation of data, writing, review, and/or revision of the manuscript, study supervision.

https://www.sciencedirect.com/science/article/abs/pii/B9780081026984000022

**Publication 7:**

**Vaughan, D.**, Shrimpton, E.A., Carpenter, G., Skerritt, D.J. and C. Williams (2021) *Marinising a terrestrial concept: Public money for public goods*. Ocean & Coastal Management. 213, p.105881.

*Journal Impact Factor:* 4.295. *Cited by* 2

*Role:* Principal investigator, conception and design, development of methodology, acquisition of data, analysis and interpretation of data, writing, review, and/or revision of the manuscript.

https://www.sciencedirect.com/science/article/abs/pii/S0964569121003641

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[2] Impact factor for all articles other than 2023 Marine Policy relate to the year of publication. Journal Citation Reports (JCR) accessed www.j.c.r.clarivate.com 30/10/2023.

**Publication 8:**

Hague, E., Sparling, C., Morris, C., **Vaughan, D.**, Walker, R., Culloch, R., Lyndon, R., Fernandes, T. and L. McWhinnie (2022) *Same species, same space, different standards: a review of cumulative effects assessment practice for marine mammals*. Frontiers in Marine Science. 25 (9).

*Journal Impact Factor:* 3.7. *Cited by 8*

*Role:* Co-investigator, development of methodology, analysis and interpretation of data, writing, review, and revision of the manuscript.

https://research-repository.st-andrews.ac.uk/handle/10023/25170

**Publication 9:**

**Vaughan**, **D.**, Skerritt, D.J., Duckworth, J., Sumalia, U.R. and M. Duffy (2023) *The use of fuel tax concessions in the UK commercial fishing fleet.* Marine Policy. 155.

*Journal Impact Factor:* 3.8[3]. *Cited by* 1[4]

*Role:* Principal investigator, conception and design, development of methodology, acquisition of data, analysis and interpretation of data, writing, review, and/or revision of the manuscript.

https://www.sciencedirect.com/science/article/abs/pii/S0308597X23002968

[3]Elsevier Journal Impact Factor accessed 30/10/23 https://www.sciencedirect.com/journal/marine-policy/about/insights

[4] Citation totals for each publication were determined by accessing Google Scholar and Research Gate to compile a definitive list on 27/11/2023.

# Section B. Critical analysis

## 1 Introduction

### 1.1 The Science, Policy, Management Interface

Van den Hove (2007) defines the Science Policy Interface (SPI) as "social processes which encompass relations between scientists and other actors in the policy process, and which allow for exchanges, co-evolution, and joint construction of knowledge with the aim of enriching decision-making. They are implemented to manage the intersection between science and policy". The SPI can exist at different scales, such as within an organisation, between organisations, or inter-governmentally – the Intergovernmental Panel on Climate Change (IPCC) and the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) being two of the latter most cited as examples (Kohler, 2022, Horton and Brown, 2018). Sokolovska et al. (2019) details how the communicative relationship between science and policy-making has evolved over time from that of the 'linear phase' (when science informed policy-making in a unidirectional manner) through the 'interactive phase' (when both sides found themselves in a continuous interaction) to the 'embedded phase' (when citizens' voices come to be involved within this dialogue more explicitly).

There are usually three components of the SPI: science, policy, and the interface between the two (Eroğlu and Erbil, 2022, Van den Hove, 2007). An individual can play distinct roles when operating within science or policy, as well as at the interface ('boundary spanners') between the two (Bednarek et al., 2018). Recognising the unusual role of the author in the earlier papers considered within this thesis – a manager commissioning science, developing policy, and conducting the science at a national level – a fourth component to the SPI has been introduced: management (Fig. 2). A specific management component of the SPI as presented below does not appear explicitly within scientific literature.

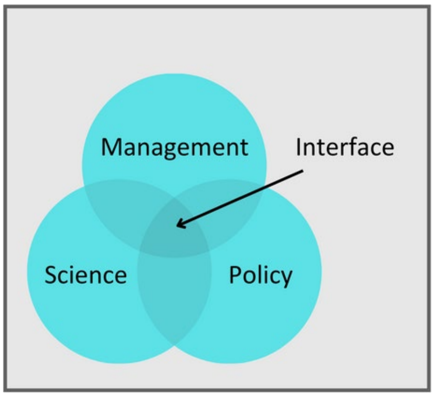



*Figure 2. The modified Science Policy Interface*

Individuals within organisations that are working effectively as boundary spanners at the interfaces modelled are in the privileged and powerful position of being able to: (1) provide advice, either publicly or privately, that is sought by politicians or policy officials within government, or (2) proffer unsought advice – again publicly or privately to government. These communication channels enable individuals and the organisations that they represent to influence policy discussions (sometimes significantly) in either a proactive (shaping the agenda) or reactive manner[5].

Opportunities to influence 'policy windows' may, at times: (1) be fleeting, and require swift interventions; (2) appear because of trust and connectedness (networking); and (3) be choreographed, opportunistic, or anticipated, i.e. knowing that a subject is to be formally consulted on (Rose et al., 2020, Kingdon and Stano, 1984, Reed et al., 2014). With these opportunities come significant responsibilities. Interventions such as the provision of advice either publicly or privately at odds to a government position or policy can result in the relationship and trust between organisations being damaged (potentially irreparably) (Lacey et al., 2018). This can have long-term and significant implications for the organisation's very existence (UK Parliament, 2018), noting that reform of Statutory Nature Conservation Bodies (SNCBs) (Defra, 2022b) and other Arms-Length Bodies (ALB) in the UK is a perennial threat (Dalton and Gill, 2022).

## 1.2 Presenting the analysis

The work presented here reflects employment and publications across different subject matter in various locations. A common thread running through the work is that it has been carried out at the SPI as a boundary spanner. My interactions at the SPI have varied over time – reflecting in part, subject matter, career progression, and experience. The publications presented within this critical analysis are covered in three phases. The phases define three distinct areas of my personal and career development rather than the phases set out by Sokolovska et al. (2019) (although each career phase loosely mirrors these). Each development phase attempts to identify how the publication has: (a) contributed to the scientific evidence base and fits within the literature and (b) influenced management and policy. For the older publications, consideration of how the topic that the work contributed to (i.e. understanding of the role that global warming plays in coral bleaching) has developed in general since the publication is provided, whereas for the more recent publications, follow-on work is suggested. Lessons learnt are summarised at the end of each phase.

At the start of my career (professional establishment), my focus was on the use of science to describe or solve specific management problems (phase one). As my career and experience progressed, my

[5] This is the case in the UK where these individuals (such as the author) represent arm's length bodies (ALB). In situations such as this as, government may a) be legally obligated to seek and consider the views of these organisations and set out how it has considered the advice or b) consider unsolicited, but proffered advice.

professional network expanded, providing opportunities to collaborate. This period was also formative as it provided opportunities to work across scientific disciplines (phase two). More recently (phase three), opportunities have arisen to influence policy and management decisions because of the experiences and skills that I have been able to develop over two decades – often in a resource management role. This has enabled me to identify policy or legislative gaps that I have been able to address through advice provision in various forms, drawing on the scientific evidence base that I have contributed to through the production of the publications presented within this thesis. These opportunities also coincide with an increasing desire to influence policy and, ultimately, environmental outcomes in a positive manner at scale and pace to contribute to addressing the biodiversity and climate crises.

Phase one (*Baselining and knowledge transfer – Section 2*) reflects: (a) a focus on the impacts of anthropogenic activities on species, (b) the importance of quantifying the status of the marine environment, and (c) the level of pressures exerted upon it. In this phase, the author is predominantly engaged in the science and management components of the modified SPI model (Fig. 2). This phase highlights the importance of collaborating and sharing knowledge in the most appropriate manner to ensure that scientific, management, and enforcement advancements are discoverable and can be replicated.

The second phase (*Protecting the wider seas – Section 3*) recognises that both species and habitat protection through designation using marine protected areas (MPAs), and subsequent introduction of appropriate management, will confer a degree of protection. However, this protection is unlikely to be a sufficient response to the existing and future types and levels of pressures (such as benthic disturbance) that the marine environment is exposed to. Therefore, this second phase explores how MPAs can provide wider seas environmental benefits and, conversely, how improved wider seas marine management can augment protection conferred by MPAs. This second phase recognises the challenge of sectoral management and a propensity for solving problems in isolation, which can merely move pressures resulting from activities (i.e. abrasion caused by commercial fishing) around the marine environment that may inadvertently result in perverse environmental outcomes.

The third phase (*Influencing policy at scale – Section 4*) is a recognition that to affect effective marine management at scale and pace, there is a need to seek a paradigm shift in approach and attitudes of marine users, regulators, and policy officials. To achieve this, peer-reviewed academic papers were developed highlighting societal and financial levers that can be used in support of the transformational reform of marine use which will be required to support marine nature recovery – a goal of the UK government (Defra, 2023a).

For each publication analysed within this thesis, an infographic is provided (e.g. Fig. 4) detailing the: (1) publication title; (2) authors of the publication; (3) country of institutes associated with authors to illustrate international collaboration; (4) synopsis of the paper; (5) identification of the target audience for the work (primary = green, secondary = yellow, not targeted = grey); (6) maturity of the work when published/now; (7) identification of where the paper fits with the timeline for this body of work; (8) identification of elements key to success (see Discussion for detail) when working at the modified SPI interface (coloured green in the infographic); and (9) dominant modified SPI interaction(s) – arrows indicate interaction direction (Fig. 3).

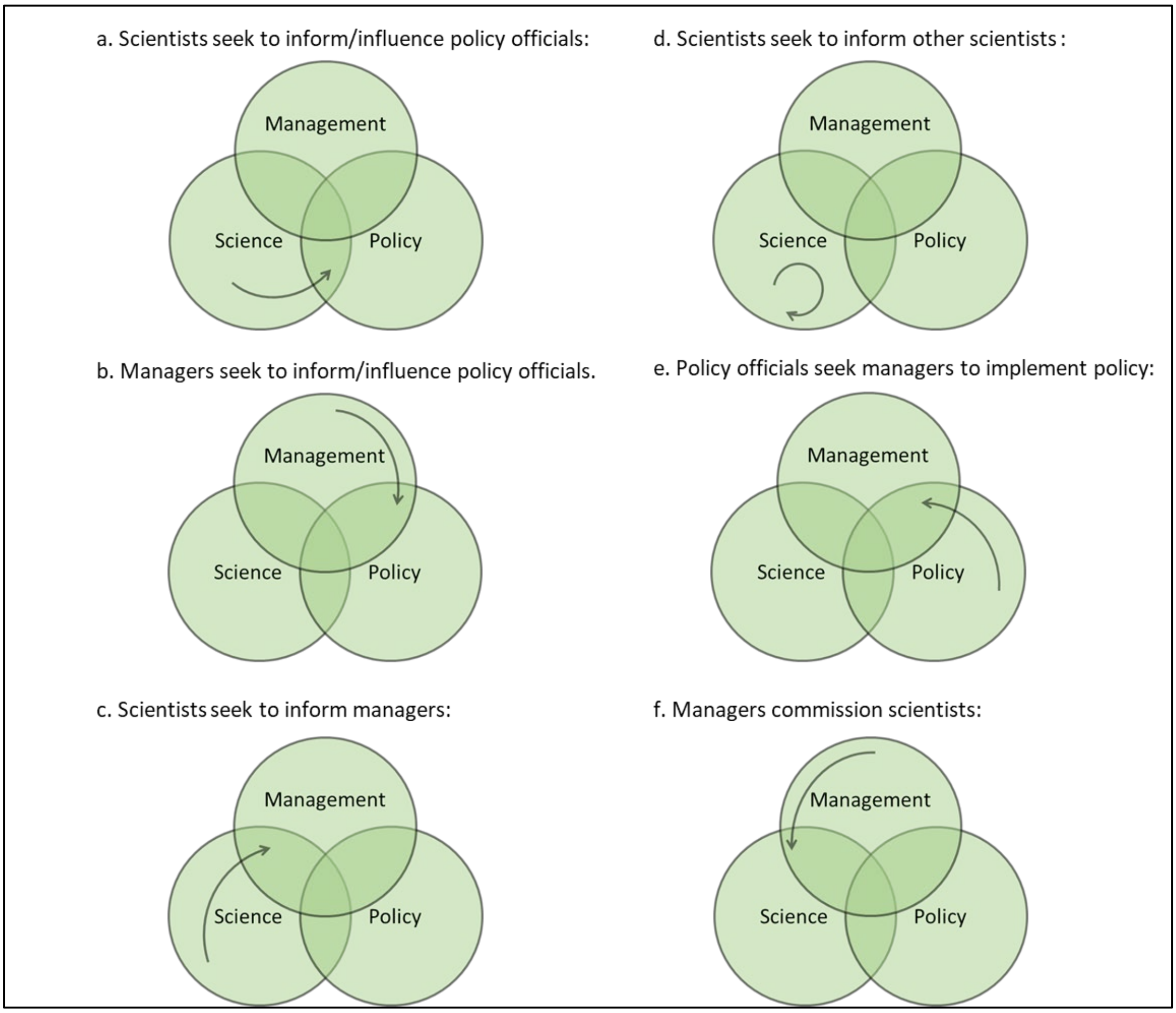



*Figure 3. The modified Science Policy Interface interactions*

# 2 Phase one: Baselining and knowledge transfer

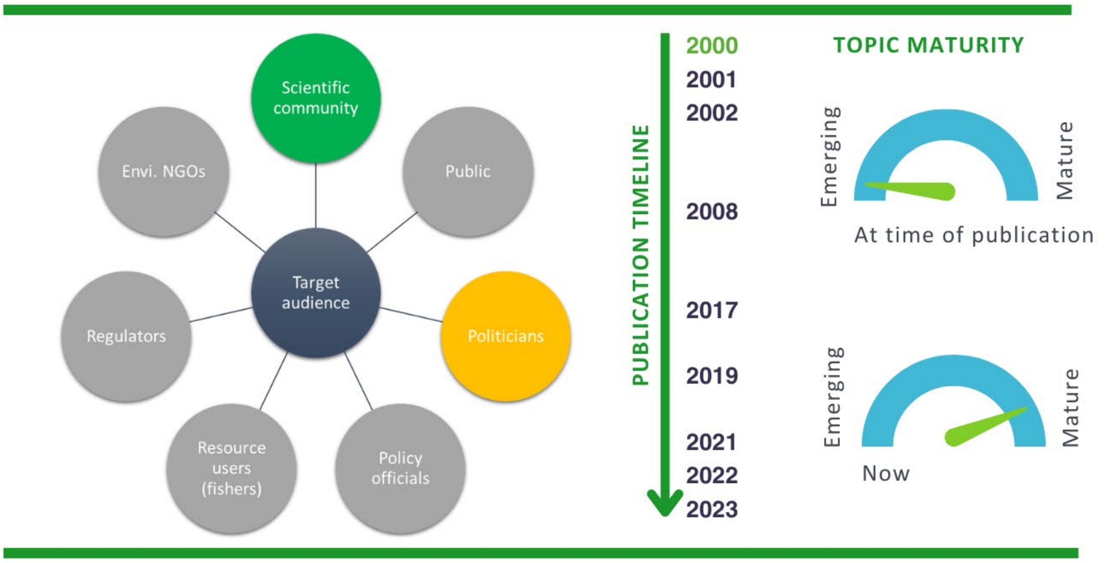
The papers considered within phase one focus on the provision of information of the biological state of species and habitats. High-quality, understandable, transparent, and accessible information enables legislation, evidence-based policy, and important management measures such as stock assessments to be developed.

The first paper in this section (Cumming et al., 2000) [publication 1] considered the first mass coral bleaching event in the Fiji Islands, whereas the second and third papers (Clerveaux and Vaughan, 2003, Vaughan, 2004) [publications 2 and 3] consider fisheries management within the Turks and Caicos Islands (TCI) – the latter two papers are considered together within this phase. The fourth paper (Wilson et al., 2008) [publication 4] considers the development of a test to determine whether bleach has been used to catch lobsters illegally within the TCI.

## 2.1 Mass coral bleaching in the Fiji Islands [publication 1]

**CUMMING, R.L., TOSCANO, M.A., LOVELL, E.R., CARLSON, B.A., DULVY, N.K., HUGHES, A, KOVEN, J.F., QUINN, N.J., SYKES, H.R., TAYLOR, O.J.S., AND D. VAUGHAN (2000). MASS CORAL BLEACHING IN THE FIJI ISLANDS, 2000.**

- 1st recorded mass coral bleaching event in Fiji
- 19 reef locations in 8 geographic regions surveyed April-July 2000
- Seawater temperatures exceeded expected summertime maxima for 5 months peaking at 30-30.5°C March-April 2000
- Bleaching threshold for Fiji appears to be in the range of 29.5-30°C
- Estimated 10-40% of coral colonies had died from bleaching within four months of the onset of bleaching



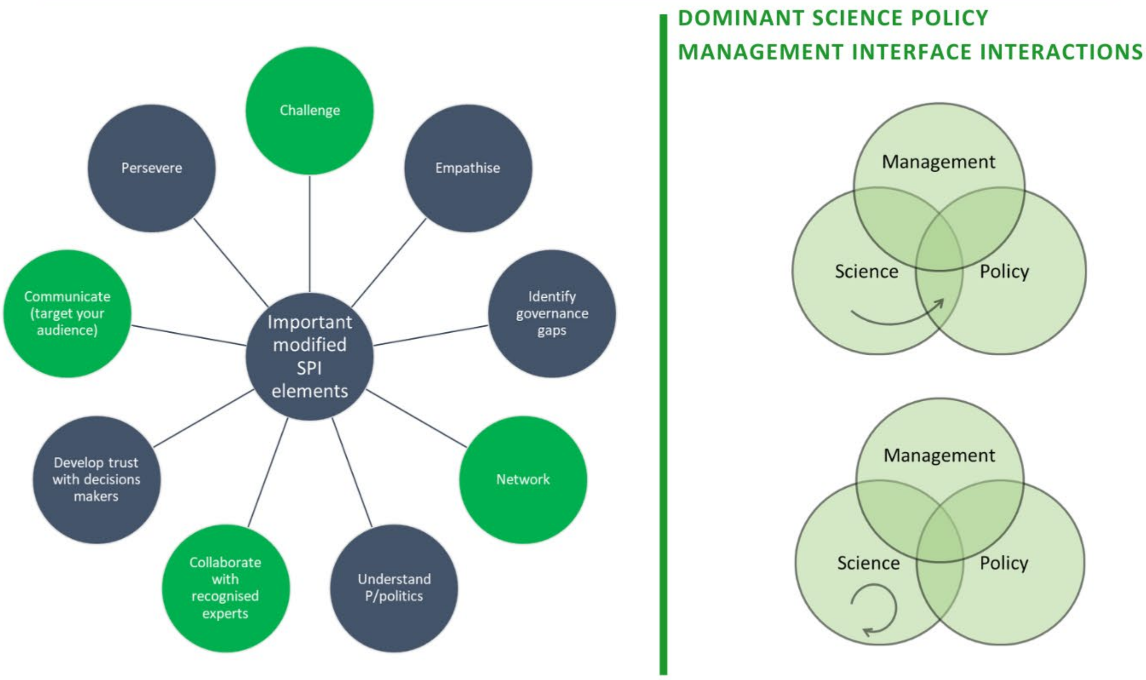




*Figure 4. Cumming et al. 2000 [publication 1] infographic*

### 2.1.1 How has the work contributed to the scientific evidence base?

Publication 1 not only provided the baseline scientific data within the Fiji Islands (29.5–30°C), but it also formed one of the initial temperature threshold datasets on the water temperatures that mass coral bleaching occurs globally. Publication 1 investigated temperature as a forcing factor in coral colony die off by analysing both *in situ* and satellite-derived sea surface temperature (SST) data during the 2000 La Nińa event and coral reef survey data.

Veron et al. (2009) wrote 'Temperature-related effects of global warming on coral reefs are highly visible, well-defined, and extensively documented. Correlations between rising $CO_2$ levels, rising ocean temperature and the biological responses of reefs are therefore known in detail, providing a particularly well-grounded basis for future prediction'. This paper and many others that followed (Sully et al., 2019, Virgen-Urcelay and Donner, 2023), as well as the IPCC report (IPCC 2022), drew upon the initial work on coral bleaching conducted in the 1970s when bleaching occurred in isolated locations, and research conducted during mass bleaching events in the early 2000s, such as the event in Fiji [publication 1] (see also (Hoegh-Guldberg, 1999)).

The research that led to publication 1 highlighted the importance of data collection using repeatable methodologies, networking, and collaboration (sharing data and co-authoring publications) between scientists – the co-authors of publication 1 were all working independently in Fiji. The *Status of coral reefs in the Fiji Islands 2006* (Sykes, 2007) referred to publication 1, stating that it established the need for standardised and regular surveys of representative sites across the Fijian archipelago, in order to measure long-term changes affecting the entire country.

### 2.1.2 How has the work influence management and policy?

The research conducted in the early 2000s (including publication 1) was critical in establishing the causal link between sea surface temperatures and coral bleaching – indeed, before the research in Fiji was published, the IPCC in 2000 noted disagreement on the cause–effect relationship between global climate change and an increase in coral bleaching (McCarthy, 2001). The mass bleaching events and the collection of research data that were disseminated were critical in securing the attention of politicians of Small Islands Developing States. These politicians were then able to advocate successfully for climate change action on the global scale.

Concerningly, models incorporating bleaching thresholds are now predicting the large-scale loss of coral reefs by mid-century under even low-emission scenarios. Even achieving emission reduction targets consistent with the ambitious goal of 1.5°C of global warming under the Paris Agreement will result in the further loss of 70–90% of reef-building corals compared with today, with 99% of corals being lost under warming of 2°C or more above the pre-industrial period (IPCC, 2022, Hoegh-Guldberg, 1999).

### 2.1.3 How has the work been updated?

The temperature threshold for mass coral bleaching events identified in publication 1 has been refined. A global analysis of coral bleaching over the past two decades by Sully et al. (2019) noted that not only has coral bleaching increased in frequency and intensity, but in the last decade (2007–2017), the onset of coral bleaching has occurred at significantly higher SSTs (~0.5°C) than in the previous decade (1998–2006: 28.7°C). The increase in threshold temperature for bleaching suggests that past bleaching events may have culled the thermally susceptible individuals, resulting in a recent adjustment of the remaining coral populations to higher thresholds of bleaching temperatures and/or coral communities have acclimatised to increasing SSTs (Sully et al., 2019).

The value of a national coral reef monitoring network highlighted by publication 1 led to the establishment of the Fiji Coral Reef Monitoring Network (FCRMN), a branch of the Global Coral Reef Monitoring Network (GCRMN). The FCRMN established 12 survey sites across Fiji, placed temperature loggers on some of them, and published a report focusing on the recovery of Fiji's reefs between 2000 and 2004 (Lovell and Sykes, 2004) in Sykes (2007). Publication 1 informed the first biannual publication of the Status of Coral Reefs of the World in 2000 (Wilkinson, 2000). Coral bleaching events continue to occur throughout Fijian waters. The FCRMN survey sites established following publication 1 form part of the long-term monitoring datasets within the global dataset (Souter et al., 2021, Delaval, 2021).

## 2.2a An investigation of the effects of increasing fishing efficiency on the productivity of the queen conch (*Strombus gigas*) and Caribbean spiny lobster (*Panulirus argus*) fisheries within the Turks and Caicos Islands [publication 2]

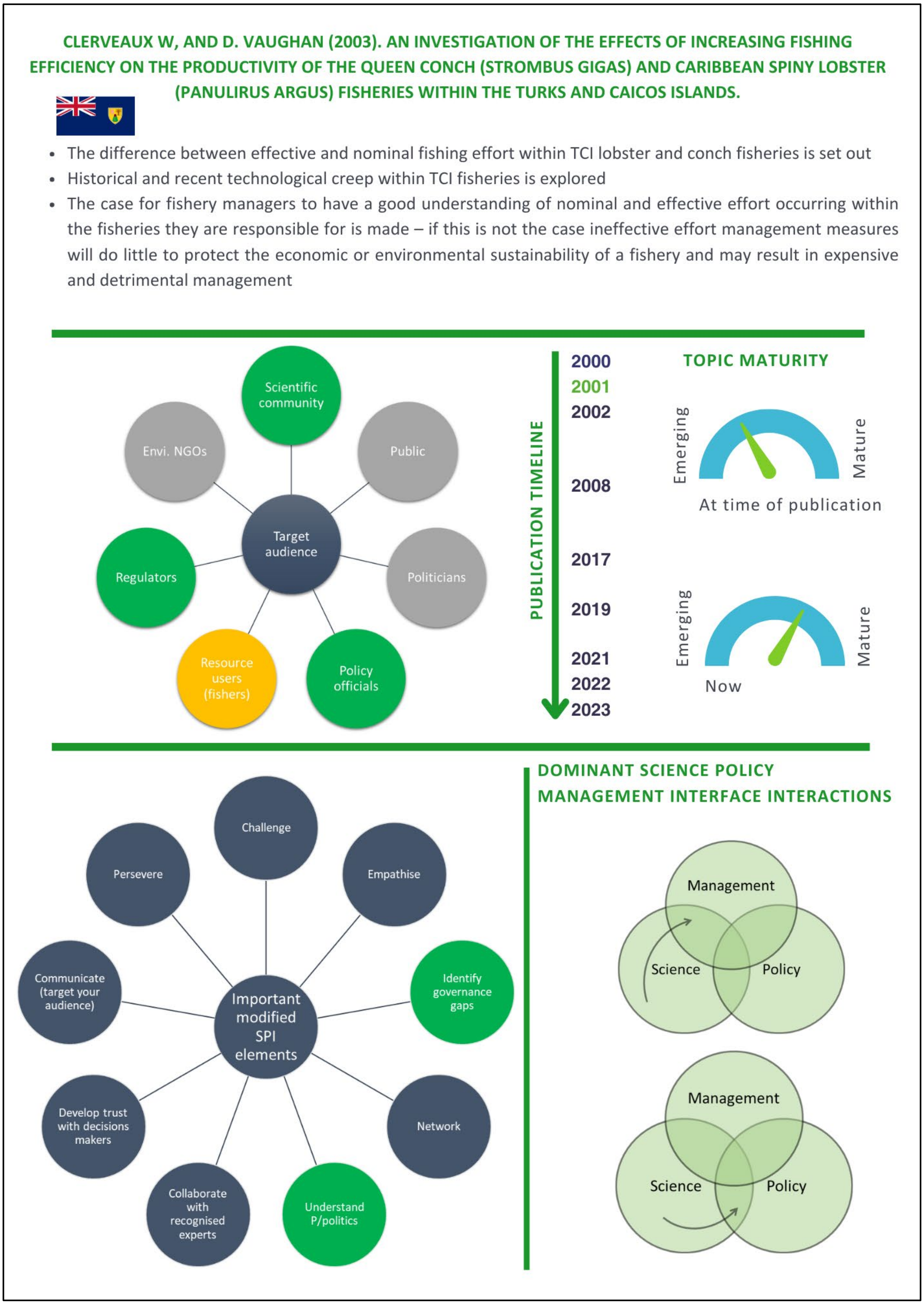
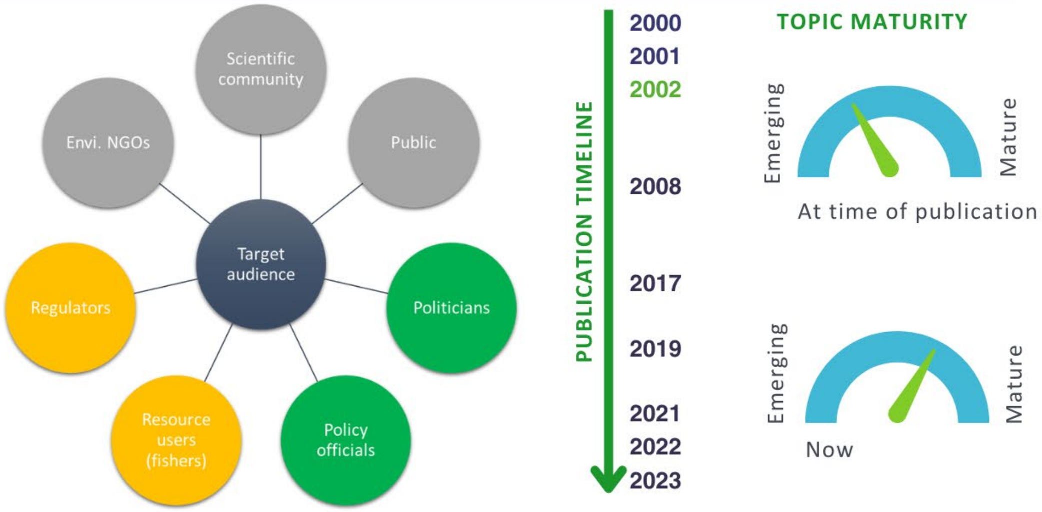



*Figure 5. Clerveaux and Vaughan 2003 [publication 2] infographic*

## 2.2b A description of fisheries management in the Turks and Caicos Islands: an overview of the problems and suggestions for mitigation [publication 3]

**VAUGHAN, D (2004). A DESCRIPTION OF FISHERIES MANAGEMENT IN THE TURKS AND CAICOS ISLANDS: AN OVERVIEW OF THE PROBLEMS AND SUGGESTIONS FOR MITIGATION.**

- TCI fisheries are used as a case study to highlight fisheries management challenges within the region
- Key challenges to delivering effective fisheries management are set out as well as recommendations for addressing these challenges
- Technological creep is recognised as problems and the imperative to manage fishing effort accordingly is set out
- The importance of societal engagement and including fishers in the management of fishery resources is stressed



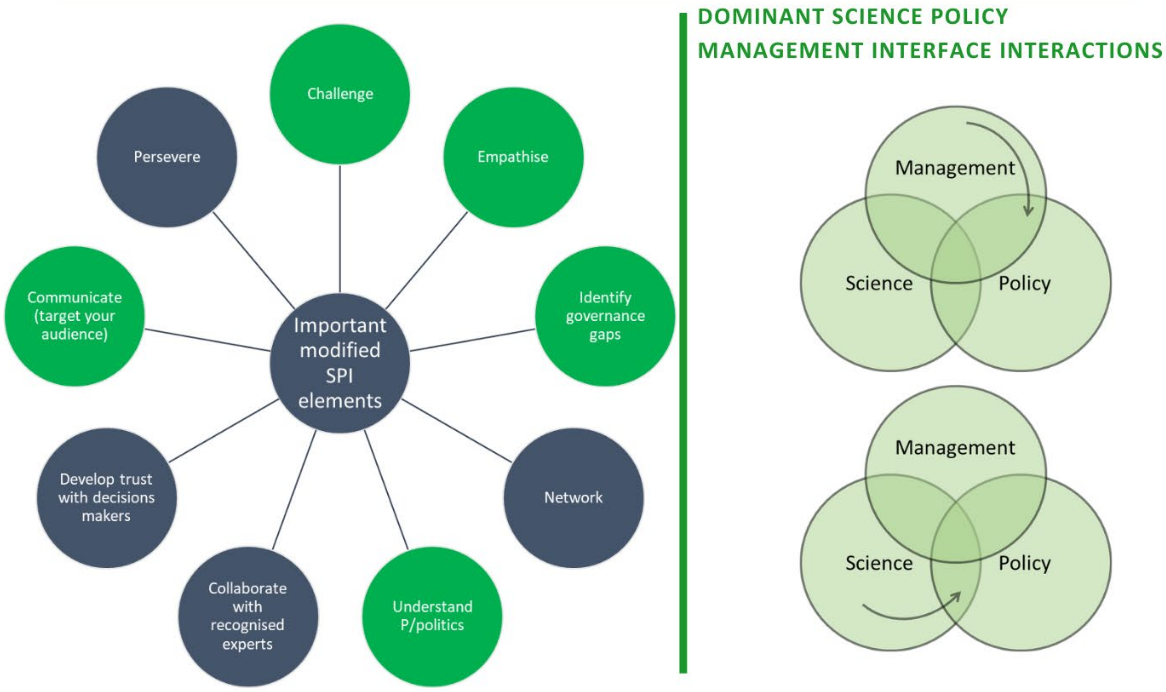




*Figure 6. Vaughan 2004 [publication 3] infographic*

### 2.2.1 How has the work contributed to the scientific evidence base?

The two papers considered here (Clerveaux and Vaughan, 2003, Vaughan, 2004) [publications 2 and 3, respectively] provide important information about the state of the two key fishery stocks of the TCI. The target audience for the papers was fishery managers working within the Caribbean. The work was presented at two Gulf and Caribbean Fisheries Institute Conferences, with the papers later being published in the conference proceedings. Although this approach reached the local target audience at the time, it has become clear in the subsequent academic literature and with conversations with practitioners that these papers have been overlooked. In early 2023, the Joint Nature Conservation Committee (JNCC) had been commissioned to conduct research in support of conch (*Strombus gigas*) fishery management improvements in the TCI (JNCC, 2023). On contacting the JNCC manager for this programme, it became apparent that JNCC were aware of publication 2 but not of publication 3. The latter publication was subsequently shared to provide important historical data relating to the conch fishery, thus strengthening the evidence base. This collaboration highlighted: (1) the importance of ensuring that completed work is published in widely accessible, if not open access, peer-reviewed journals to ensure the longevity of the research rather than conference proceedings, and (2) the value of networking and offering to share information on subjects of common interest. In 2023, a report exploring visual survey methodology for deep water surveying of conch in the TCI was published by JNCC (van Rijn et al., 2023).

### 2.2.2 How has the work influenced management and policy?

The influence of the work at the time of presentation and publication is difficult to discern as the target audience for the work comprised regional fishery manager counterparts rather than academics writing papers. Despite the apparent relative lack of influence the work had at global and regional levels, as evidenced through a lack of citations, the work was used to establish fishing quotas within the TCI. It also informed my work as Chief Fisheries and Conservation Officer (CFCO) for the TCI Government.

Publication 2, which investigated fishing efficiency, was formative in developing my understanding of technological creep within fisheries and the significant implications that this has for management of fisheries and other marine activities. This, coupled with the requirement to introduce fisheries management as the CFCO within the extensive TCI MPA network, first led me to consider the importance of managing fisheries displacement.

Publication 3 highlights the use of a Fishery Advisory Council (FAC) (comprising active and retired fishers, fisheries scientists, fisheries managers, and processors) within the TCI fisheries management framework. Personal experience of the TCI FAC demonstrated to me the importance of Fisheries

Science Partnerships (FSPs). This personal experience is in line with a substantial and growing body of work on this topic internationally extoling the importance of FSPs that provide bottom-up stakeholder engagement and participation within fisheries management and decision-making frameworks (Karr et al., 2017, Lomonico et al., 2021, Gray and Catchpole, 2021, Hipwell, 1998). This experience within the TCI prompted interventions with Defra policy officials recommending the inclusion of an FAC within the nascent fisheries management framework that was developing in the UK post EU exit.

### 2.2.3 How has the work been updated?

Two publications comprise the extent of detailed TCI fisheries management updates since Publication 2 and 3, respectively. The first (Lockhart et al., 2005) considers the status and threats to the fisheries of the TCI. The second (Ulman et al., 2016) uses a catch reconstruction approach to estimate catches for 1950–2012, estimating all removals, including reported catch destined for export, and unreported domestic artisanal and subsistence catches[6]. The results from the catch reconstructions incorporate estimates for poached stock by foreign fishing fleets and improved subsistence and recreational catch data. The reconstructions indicate that incomplete catch totals have been used for decades to calculate sustainable catch limits for the islands' marine resources. The implication of Ulman et al. (2016) is that the stock assessments and landing data presented in publications 2 and 3 are incomplete and the levels of exploitation of the fisheries set out within these publications are underestimates. Recognising the implications of the catch reconstruction work, the TCI government, advised by DEMA[7], reduced subsequent quotas and recommended an export cessation of up to five years (Ulman et al., 2016).

[6] Catch reconstruction for small-scale fisheries is a relatively recent scientific development (Pauly and Zeller, 2016, Zeller and Pauly, 2007) providing important insights into the historical state of stocks – notwithstanding the challenges recognised by pioneers in this approach (Pauly and Palomares, 2019)

[7] DEMA was the Department for Environment and Maritime Affairs in the TCI. A previous (and current) incarnation of DEMA is the Department of Environment and Coastal Resources. (DECR).

## 2.3 Efficacy of a starch-iodide swab technique to detect the illegal use of bleach in a spiny lobster (*Panulirus argus*) fishery [publication 4]

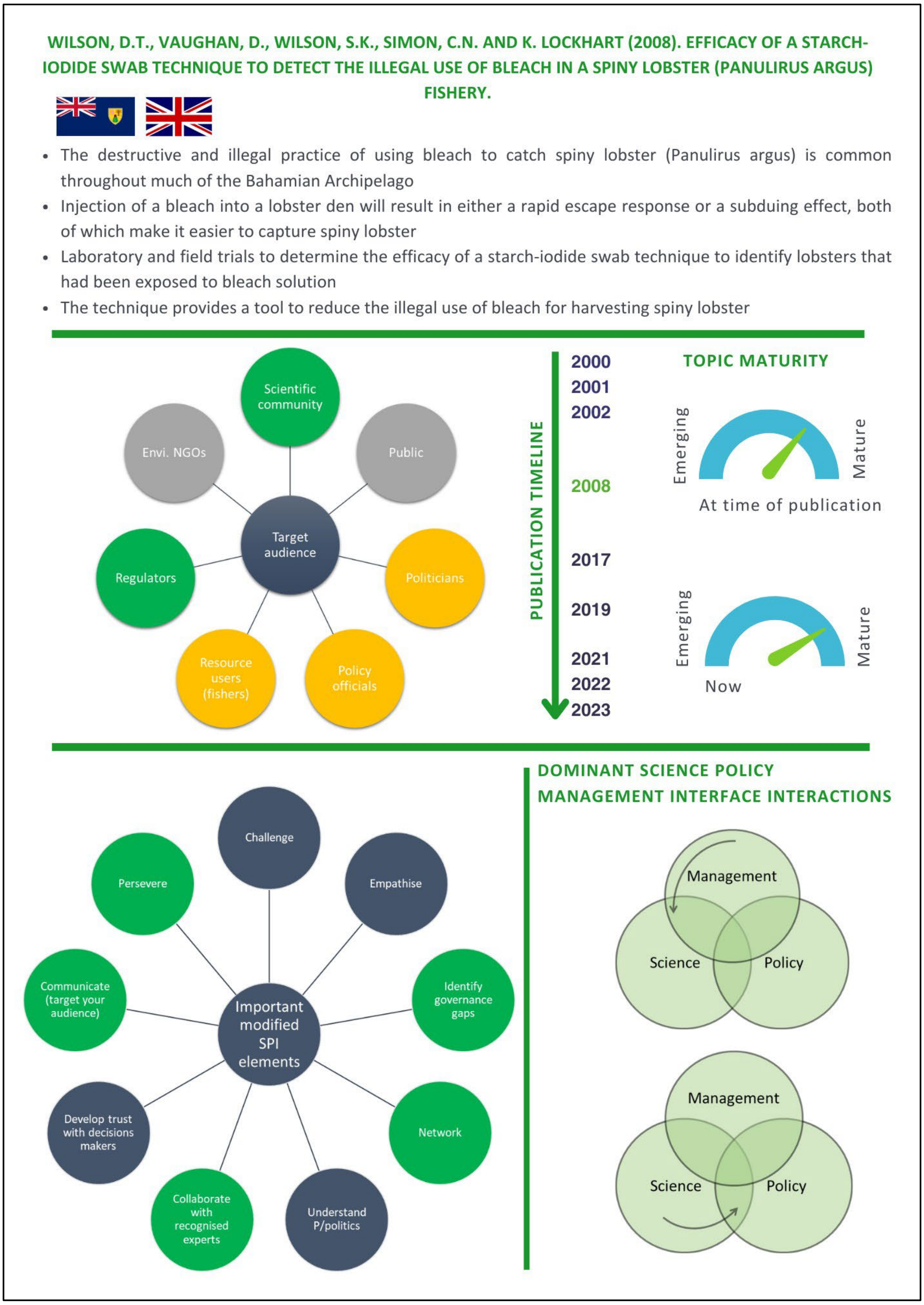



*Figure 7. Wilson et al. 2008 [publication 4] infographic*

### 2.3.1 How has the work contributed to the scientific evidence base?

Wilson et al. (2008) [publication 4] has enhanced the scientific evidence base detailing the illegal use of noxious substances (such as household bleach) to both catch marine species (e.g. lobsters, fish for food and the aquarium trade, and octopi) and strip eggs from female lobsters (known as berried hens). These practices are conducted in fisheries throughout the world (Wright and Esmonde, 2001, Tissot and Hallacher, 2003, Center, 2017).

Developing effective methodologies that can detect these illegal acts that are simple to deploy in the field and can withstand scrutiny in the courtroom have long been sought by enforcement personnel (publication 3). Publication 4 built upon preliminary work by Wilson et al. (2004) conducted in 2000–2002 and published only in the Conference Proceedings of the Gulf and Caribbean Fisheries Institute. The work was subsequently published in the journal *Fisheries Research* (impact factor 1.434) in 2008 to enable the data to be more widely findable than the 2004 conference proceedings.

### 2.3.2 How has the work influenced management and policy?

The swabbing technique developed in publication 4 has influenced policy and management as it has been successfully rolled out and is in use within the TCI for fisheries compliance purposes (Lockhart et al., 2005).

In my role as CFCO for the Department of Environment and Coastal Resources of the TCI Government, I had the challenge of managing the spiny lobster (*Panulirus argus*) fishery – one of the country's most valuable natural resources. Illegal fishing undermined fisheries management measures and the credibility of the fishery officers who would regularly come across fishing vessels with empty bleach bottles in the vessels, fishers of those vessels with bleach-stained clothing, and lobsters smelling strongly of bleach, yet were limited in how they could respond.

Recognizing the need to address this problem, I collaborated with the Centre Director of the School for Field Studies. Techniques were identified that had been developed to address this type of illegal activity elsewhere, albeit for a different species being targeted by fishers. A key aim of the work was to demonstrate that an existing fisheries enforcement technique used to identify illegally obtained species could be replicated within the TCI for use in identifying illegally caught spiny lobster. The decision to publish in a specialist topic scientific journal was taken to lend credibility to the research so that it would have greater standing in court.

### 2.3.3 How has the work been updated?

The scientific literature base does not reflect any update to the lobster bleaching swab test since publication 4. However, working on the test was a formative experience for me as this led to the development of a further enforcement tool in a different lobster fishery. In my role as Deputy Clerk and Fishery Officer at Eastern Sea Fisheries Joint Committee[8] (ESFJC), I drew upon the lessons learnt in the TCI to address the illegal removal of eggs from lobsters in the UK through scrubbing by fishers. The legislation being contravened by fishers was a bylaw of the ESFJC. National legislation prohibiting the landing of berried lobsters had not been introduced, partly because of the lack of a UK test to identify berried lobsters that had been scrubbed. However, a test (Karlsson and Sisson, 1973) deployed by the Commonwealth of Massachusetts in the USA to identify scrubbed American lobsters (*Homarus americanus*) did exist. The Karlsson and Sisson test was subsequently researched and replicated for the European lobster (*Homarus gammarus*) (Jessop et al., 2010). The research was successful, with the test being deployed in 2007 – this test resulted in a successful prosecution (Eastern Sea Fisheries Joint Committee, date unknown).

In March 2017, Defra launched the Consultation on the Prohibition on Landing Egg-Bearing ('Berried') Lobsters and Crawfish in England (Defra, 2017a). In September 2017, Defra published the Government response and summary of responses to the consultation (Defra, 2017b). National legislation in the form of a Statutory Instrument (SI) (UK Government, 2017a) was subsequently introduced prohibiting the landing of berried lobsters. The introduction of this legislation was a direct result of the research that resulted in a UK test for European lobsters, and work undertaken to influence policy officials. The explanatory note that accompanied the SI (Section 9.1) (UK Government, 2017b) set out: *The Marine Management Organisation and IFCA[9] enforcement officers will be required to train staff to use testing kits that can demonstrate if eggs have been recently removed from a berried lobster or crawfish*.

## 2.4 Phase one: Science, Policy, and Management Interface summary

The publications included within this phase focused on the science component of the modified SPI model but include links from science to management and policy. Publication 1 saw the collection of biological data following a physical event resulting from climate change. The key output from this work was the communication of the results to other scientists, with subsequent dissemination to politicians. The importance of scientific curiosity and collaboration were vital in ensuring that the evidence of the bleaching impact could be captured and disseminated.

[8] Eastern Sea Fisheries Joint Committee was replaced with the Eastern Inshore Fisheries and Conservation Authority (EIFCA) in 2011 following the introduction of the Marine and Coastal Access Act in 2009.

[9] Inshore Fisheries and Conservation Authorities (IFCA), of which there are ten, are public authorities responsible for inshore fisheries and conservation management within English waters 0-6nm.

Publications 2 and 3 both describe one of the key challenges of fisheries management – that of fishing technology and effort creep. These publications build the evidence base for this topic and seek to secure wider attention of the challenge and the identification of measures to address it by managers and policy officials. In contrast, publication 4 provides a clear example whereby a resource manager may seek a scientific solution to a particular problem. Publication 4 demonstrates the value of scientists and managers looking to adapt existing solutions to problems they face.

Key elements for working effectively at the SPI identified within this phase were: (a) the importance of collaborating, (b) the value of networking and (c) communicating these gaps to the appropriate audiences. Figs. 5, 6 and 7 highlight three interactions with the missing component (management) from the classic SPI model: science → management, management → policy, and management → science.

# 3 Phase two: Protecting the wider seas

From a marine conservation perspective, a country's Economic Exclusive Zone can either be designated as an MPA or be considered 'wider seas'. The papers (Vaughan, 2017, Vaughan and Agardy, 2019) [publications 5 and 6] within phase two developed from a growing realisation within the wider scientific community of the role that an ecosystem approach[10] to management should play in addressing marine management challenges. Management measures such as fishing effort control (e.g. as quotas), enforcement tools (e.g. berried lobster scrubbing identification kits), and the designation of MPAs have often been advocated within research, conservation, management, and enforcement silos as responses to solve a specific local issue. The downside of this is often unintended consequences in the wider seas, i.e. the underlying problem – such as (a) shifting stocks, (b) too much effort in the system, or (c) a system under increasing pressures from different sectors – is often not addressed. As a result, fishing effort and the pressures exerted by this activity are directed elsewhere within the marine system when management measures are introduced.

For several decades, commercial fishing effort displacement had received limited attention in the UK and elsewhere in both the scientific literature and from policy officials, the consensus being that displacement was too difficult a topic to be addressed with no clear method of assessment or palatable management options. More recently, the increasing demand for marine space (principally through the growth of offshore wind farms) (Gourvenec et al., 2022) and the introduction of management measures within MPAs (Marine Management Organisation, 2014), in conjunction with the recognition that the wider marine environment is in poor condition (Defra, 2019), required the topic of fishing effort displacement to be revisited.

Publications 5 and 6 have tried to bring the pressing need to holistically address demands on marine resources to the attention of various stakeholder groups.

[10] The ecosystem approach is defined by within the Convention of Biological Diversity as a *strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way* SECRETARIAT OF THE CONVENTION ON BIOLOGICAL DIVERSITY 2004. The Ecosystem Approach. Montreal.

## 3.1 Fishing effort displacement and the consequences of implementing Marine Protected Area management – An English perspective [publication 5]

**VAUGHAN, D. (2017). FISHING EFFORT DISPLACEMENT AND THE CONSEQUENCES OF IMPLEMENTING MARINE PROTECTED AREA MANAGEMENT – AN ENGLISH PERSPECTIVE.**

- MPA network creation can displace activities (primarily fisheries) that are thought to be incompatible with the habitats and species of conservation importance that the network has been established to protect
- The designation of MPAs and the subsequent introduction of management measures that displace activities may result in unintended impacts/consequences on protected benthic habitats or species within (a) the MPA where management measures have been introduced, (b) other MPAs or (c) wider UK or international waters
- To aid the better description and understanding of the various facets of fisheries effort displacement a structure to differentiate the types of fisheries displacement is proposed.

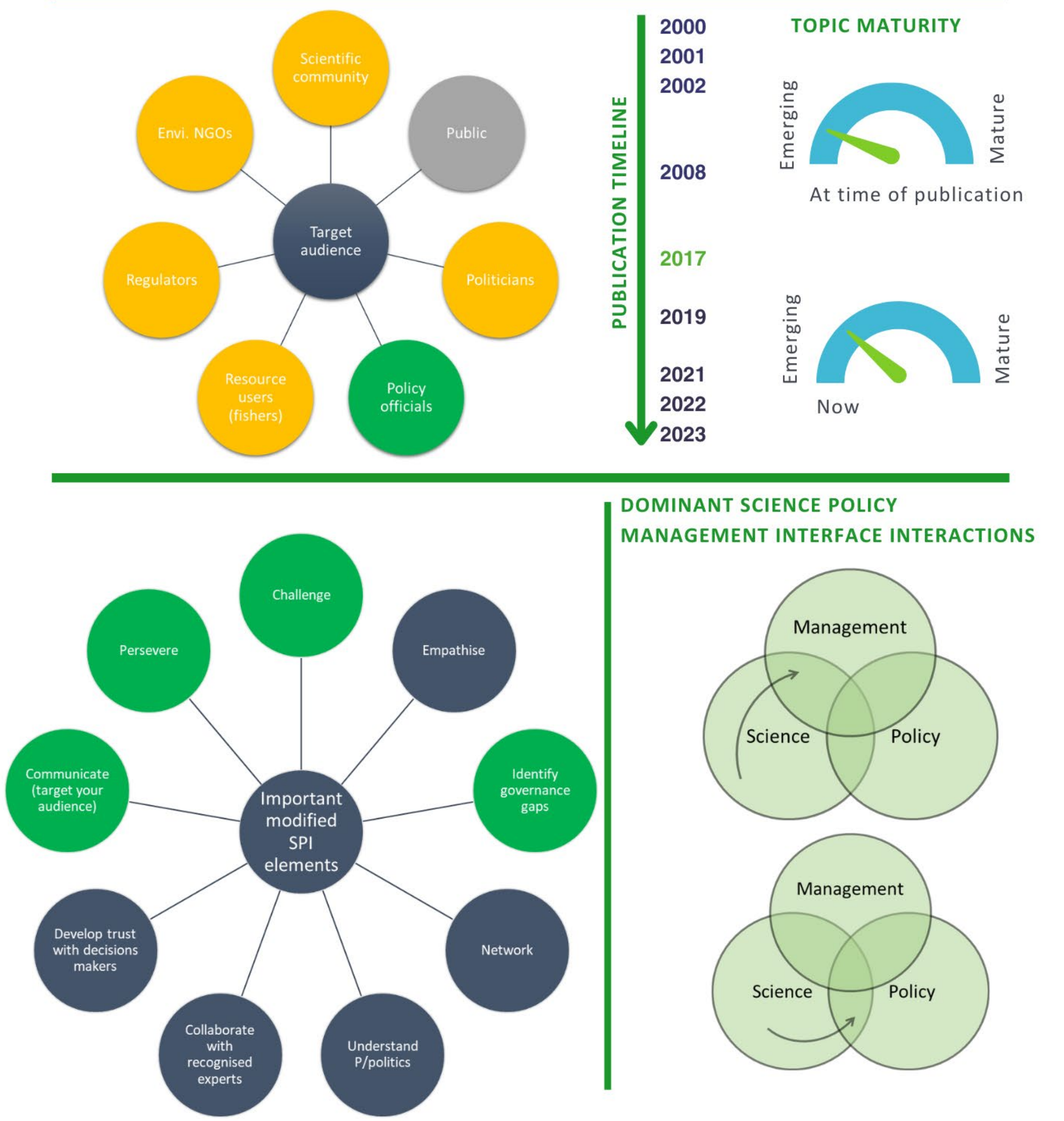



*Figure 8. Vaughan 2017 [publication 5] infographic*

### 3.1.1 How has the work contributed to the scientific evidence base?

A general definition of fishing displacement was proposed in McLeod (2014); however, a typology for fishing effort displacement had not been proposed until publication 5. Identification of fishing effort displacement as subject matter within the scientific literature occurred in the early 2000s with a focus on changes in fisher behaviour (Dinmore et al., 2003, Wilen et al., 2002). Later papers focused on the recovery of species, habitats, and ecosystem services (Epstein and Roberts, 2023) within MPAs because of displacement of fishing effort from those areas. Several papers citing publication 5 have identified fishing effort displacement as an important consideration when conducting marine spatial planning (Chollett et al., 2022, Iwona et al., 2021, Trouillet et al., 2019).

### 3.1.2 How has the work influenced management and policy?

Publication 5 significantly influenced the Beynon Report into Highly Protected Marine Areas (HPMAs) (Benyon, 2020), with fisheries displacement being mentioned as an important topic (19 times in 130 pages) to be addressed during the identification of HPMAs, with *Recommendation 7* of the report setting out: *Government should acknowledge displacement in its decision making during HPMA designation. It should put strategies in place to support marine uses and avoid creating new problems from moving pressures to other parts of the marine environment.*

The Government's response to this report (UK Government, 2022a) addressing *Recommendation 7* pledged to: (a) undertake further research to increase understanding of displacement; (b) consider the potential social, economic, and environmental impact of displacement from HPMA designation; and (c) consider whether the current provisions considering displacement within marine plans are adequate.

Publication 5 led to a resulting increased interest in the topic and an acknowledgement that this was a problem that government was required to address. This provided the impetus for Natural England (NE) to commission a report setting out how fisheries displacement could be assessed; this report (ABPmer, 2017) was managed by the author of this thesis. The displacement assessment report was guided by a steering group that included a representative of the National Federation of Fishermen's Organisations (NFFO). Vaughan (2017) and ABPMer (2017) influenced the Marine Management Organisation (MMO) in 2020 to publish *Evidence Requirement R141* (Marine Management Organisation, 2020a), thus indicating the recognition and acceptance of the MMO as to their role in addressing displacement.

In 2018, Defra published the White Paper[11] *Sustainable fisheries for future generations* consultation document (Defra, 2018a). The consultation sought feedback on various aspects of fisheries

[11] White papers are policy documents produced by the Government that set out their proposals for future legislation.

management, with feedback intended to inform the development of the draft Fisheries Bill and subsequent Fisheries Act 2020 (UK Government, 2020). NE submitted a response (Natural England, 2018) to this consultation setting out the pressing need to address fishing effort displacement within a new fisheries management framework – the response included reference to publication 5. Displacement concerns were identified as being raised within the S*ummary of consultation responses and government response* (Defra, 2018b).

A deeper understanding of the threat that displacement of fishing activities posed led the NFFO in collaboration with the Scottish Fishermen's Federation (SFF) to commission ABPMer to produce the report: *Spatial Squeeze in Fisheries* (ABPmer, 2022). This report, along with dissemination of the paper on displacement (publication 5) and the report on assessing displacement (ABPmer, 2017), played a critical role in in securing the *Displacement Policy* (section 4.2.9) within the Joint Fisheries Statement (JFS) (UK Government, 2022b)[12].

In 2022, the Scottish Government drew heavily on publication 5 and the ABPMer report in their *Good Practice Guide for Assessing Fishing Displacement by Other Licenced Marine Activities* (Marine Science Scotland, 2022).

### 3.1.3 How could the work be updated?

Section 3 (*Management Solutions*) of publication 5 sets out not only the need to understand the magnitude of the problem, but also the need to consider and develop appropriate management responses that address displacement. This element, addressing displacement, is the most challenging facet of the displacement management problem, yet is fundamental to resolving excess effort.

The challenge of addressing displacement in England remains even if suitable mechanisms to do so are not identified because there is a lack of a clear vision for: (1) how English waters should be used, including the extent to which fishing activity fits within this (something that the MSPri work being undertaken by government hopes to inform[13]), and (2) what the desired fishing fleet composition should be in terms of numbers and sizes of vessels, as well as economic and environmental performance.

Although a clear vision for how English waters should be used is lacking, there is a desire to co-locate and co-exist different marine activities (including fishing) (Christie et al., 2014). However, these options are not anticipated to be adequate as the blue growth agenda in the form of offshore windfarm development is still in its infancy (13.9 GW out of a proposed 50 GW by 2030 are in

[12] The UK Government is required to publish a JFS that sets out how the eight fisheries objectives of the Fisheries Act 2020 are to be delivered.
[13] MSPri is considered later in this section of the thesis.

operation) (Department for Business and Trade, 2023) and the marine environment in terms of both the MPA network (Defra, 2023a) and the wider seas is in poor condition (McQuatters-Gollop et al., 2022) – trade-offs are therefore required. Although clear legal targets and commitment to deliver carbon net zero/offshore windfarm generating capacity and to improve the state of the marine environment exist (UK Government, 2019, UK Government, 2023a, Defra, 2023a), similar targets do not exist for fishing. As action is taken to deliver both carbon net zero and state of the marine environment targets, it is anticipated that fishing may be the sector that will be displaced, with fishing effort requiring removal from the marine system. The political unpalatability of this (even when support is provided for a just transition) is likely to be significant. This brings to the fore the urgent need to identify suitable displacement management mechanisms.

The first step towards a coherent approach to better managing displacement is to clearly articulate the current approach(s) within a jurisdiction – in England this has not been done. The current approach, however, could be considered one of an accidental *laissez faire*, where capitalist market-forces will address fishing effort displacement with uneconomic fishing operations existing within the fishery ceasing. If this is the de facto approach, it is one undermined by the provision of tax concessions and subsidies to economically support the sector (see publication 9), thus enabling those operations that are marginally economically viable to remain within the fishery.

To date, there has not been a comprehensive collation and structured analysis (the STEEPLE[14] framework is suggested) of the various mechanisms used internationally that could be implemented by policy officials and marine managers to address fishing effort displacement. It is recommended that this research is conducted, and presented in such a manner that the options identified are globally relevant. In doing so, this will enable policy officials to identify the option(s) that would be most suitable in that jurisdiction for more in-depth consideration, thus increasing the global relevance of the work. This approach would also enable options that may otherwise be considered politically challenging (such as decommissioning schemes) to be presented to policy officials. Officials could then form a view as to the options that could be taken forward for more in-depth country-relevant review.

[14] STEEPLE – Social, Technical, Economic, Environmental, Political, Legal, Ethical.

## 3.2 Marine protected areas and marine spatial planning – allocation of resource use and environmental protection [publication 6]

**VAUGHAN, D. AND T. AGARDY (2019) MARINE PROTECTED AREAS AND MARINE SPATIAL PLANNING - ALLOCATION OF RESOURCE USE AND ENVIRONMENTAL PROTECTION.**

- The strong link between Marine Spatial Planning and Marine Protected Areas is explicitly recognised
- Marine areas important for marine conservation form common data layers within marine plans
- Where MPAs do not exist, marine plans can aid in the identification of areas where they could be sited optimally (taking into account environmental, social, economic and political considerations)
- MPAs and marine plans are generally based on current information, be it habitat/species distribution or marine uses; however, both marine plans and MPAs may be used as tools to drive future sustainable use of the marine environment

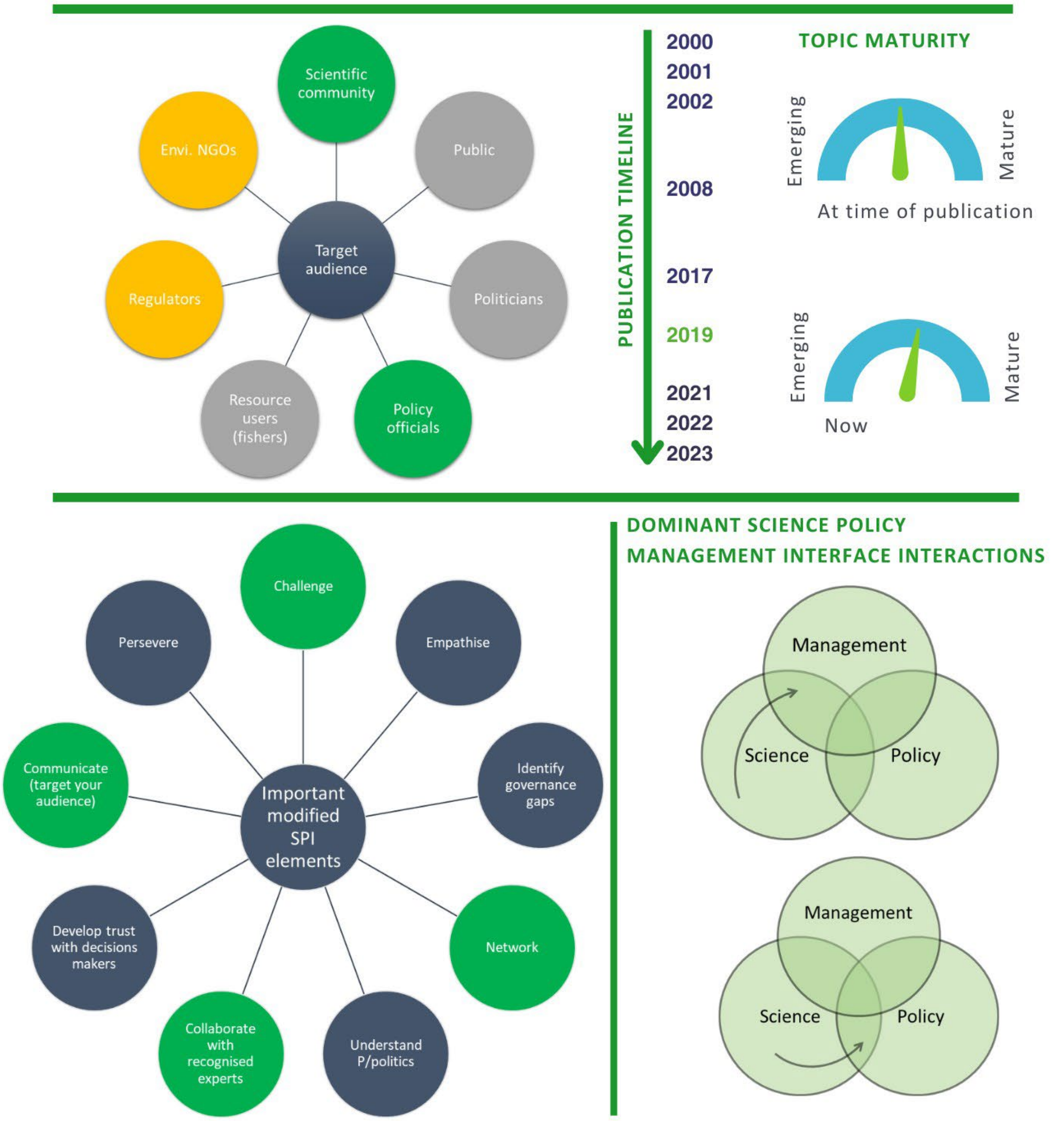



*Figure 9. Vaughan and Agardy 2019 [publication 6] infographic*

### 3.2.1 How has the work contributed to the scientific evidence base?

Publication 6 is the first attempt to comprehensively set out how marine planning can aid the delivery of MPA objectives and vice versa. Marine planning and MPA designation and management are two distinct but interconnected areas of marine management. Although countries have been designating MPAs for several decades, the first International Marine Protected Areas Congress (IMPAC I) was only held in 2005. This was only a short time before UNESCO held the first International Workshop in 2006 on marine spatial planning (MSP)[15], which is thought of as the birthplace of marine planning internationally.

Publication 6 is already being cited by others building the argument for a more holistic approach when implementing marine protection. von Thenen et al. (2021) and Manea et al. (2023) both cite publication 6 when setting out their views that MSP anchored to ecosystem-based management can aid the delivery of marine sustainable development and marine nature recovery.

### 3.2.2 How has the work influenced management and policy?

*Marine Protected Areas: Science, Policy and Management* (Humphreys and Clark, 2019) was commissioned with the aspiration that it would become the reference text for academics, policy officials and university students. Prior to inclusion of publication 6 as a chapter, the book focused narrowly on MPAs, neglecting the contribution of MPAs to the improvement in the state of the wider seas and vice versa – how protection and management of the wider seas can improve the condition of species and habitats afforded protection through MPA designation. It is hoped that the inclusion of publication 6 will highlight to readers the interconnectedness of the marine environment, and an imperative to ensure that protection is effective and should look beyond the protection of individual species or habitats to an ecosystem-based approach.

Publication 6 is being used by NE to inform its strategic approach to marine conservation with the MPA network ostensibly complete, and has been used to influence wider seas management to aid the delivery of the UK Marine Strategy (Natural England, 2024).

### 3.2.3 How could the work be updated?

There are two elements in publication 6 that could benefit from further work. The first is a more in-depth consideration of what a transition to a whole-site approach (WSA) to MPA management might entail. In 2019, there was little in the public domain on this topic other than (a) a reference to a transition to a WSA in the UK Government's 25 Year Plan (UK Government, 2018) and (b) the

[15] Marine Planning (MP), Marine Spatial Planning (MSP) and Marine Spatial Prioritisation (MSPri) can be considered as three iterations of marine planning with ever increasing levels of spatial specificity regarding the sighting and prioritisation of marine activities occurring within a country's maritime jurisdiction. Confusingly marine planning and marine spatial planning have been used interchangeably in the scientific literature and by practitioners.

statement by the UK government that the WSA is reflected in the designation of three HPMAs (Defra, 2023c). However, there has been little in the way of detail or debate (both prior to and since 2019) regarding an accepted definition for, or wider application of, a WSA. This has led to academics and environmental non-governmental organisations forming their own subtly different interpretations (Solandt et al., 2020, Davies et al., 2022, Wildlife and Countryside Link, 2020). Developing an articulate description of a progressive interpretation of what a WSA could entail from within government provides the opportunity to shape the future narrative on the progression of a WSA within English waters and the future design of the English MPA network. This was a topic Defra raised in the Green Paper[16] on Nature Recovery consultation (Defra, 2022b).

The second element that may benefit from further consideration is the second and most recent evolution of Marine Planning (MP), that of Marine Spatial Prioritisation (MSPri). In MSPri, marine resources (physical and space itself) are considered and allocated according to the prioritisation that government seeks in order to strategically deliver specific objectives (UK Government, 2022c). MSPri has developed in England and English waters rapidly as it has become clear that the ambitious blue growth agenda (principally offshore power generation) and existing uses are increasingly challenging to reconcile with: (a) the current state of English seas, (b) the status of protected species and habitats within the English MPA network, and (c) marine protection ambitions and commitments (UK Parliament, 2022a) (Slater and Claydon, 2020). That this second evolution is occurring within English waters is unsurprising as these are some of the busiest marine areas in the world, and areas that have key resources, i.e. biologically productive shallow seas (Marine Management Organisation, 2020b, Kröger et al., 2018).

## 3.3 Phase two: Science, Policy, and Management Interface summary

The publications included within this phase focus on directly and indirectly influencing policy through communication at the SPI. Directly, this is achieved through publication 5, by drawing the attention of policy officials to the topic of fisheries displacement. Indirectly, this was sought through publication 6, where the case for managing marine space holistically was made; the intended audience for the book incorporating publication 6 included marine conservation and management postgraduates and researchers, as well as policy professionals.

Publication 5 was successful in securing UK policy attention to the topic of displacement, as the publication provided credibility to and underpinned interventions by: (a) me in my role of conservation adviser to government during the drafting of the JFS and (b) colleagues in relation to HPMA

[16] Green Papers are consultation documents produced by the Government. The aim of this document is to allow people both inside and outside Parliament to give the department feedback on its policy or legislative proposals.

designation conservation advice provision. Publication 5 provided the opportunity to commission further fisheries displacement work – namely, the report *Displacement of fishing effort from Marine Protected Areas* (ABPmer, 2017). To deliver this report, a steering group was established which included the NFFO. NFFO involvement was sought to ensure that the outputs of the work were credible in the eyes of an organisation that may otherwise have been wary of engaging this topic. Empathy and a desire to understand the fishing industry's perspective regarding fishing effort displacement were key to ensuring the outputs were useful and progressed understanding of this topic by interested parties (ABPmer, 2022).

Key elements for working effectively at the SPI identified within this phase were: (a) identification of knowledge gaps and (b) communicating these gaps to the appropriate audiences. The impact of publication 5 is exemplified through the inclusion of displacement policy within the JFS. Figs. 8 and 9 highlight one interaction with the missing component (management) from the classic SPI model: science → management.

# 4 Phase three: Influencing policy at scale

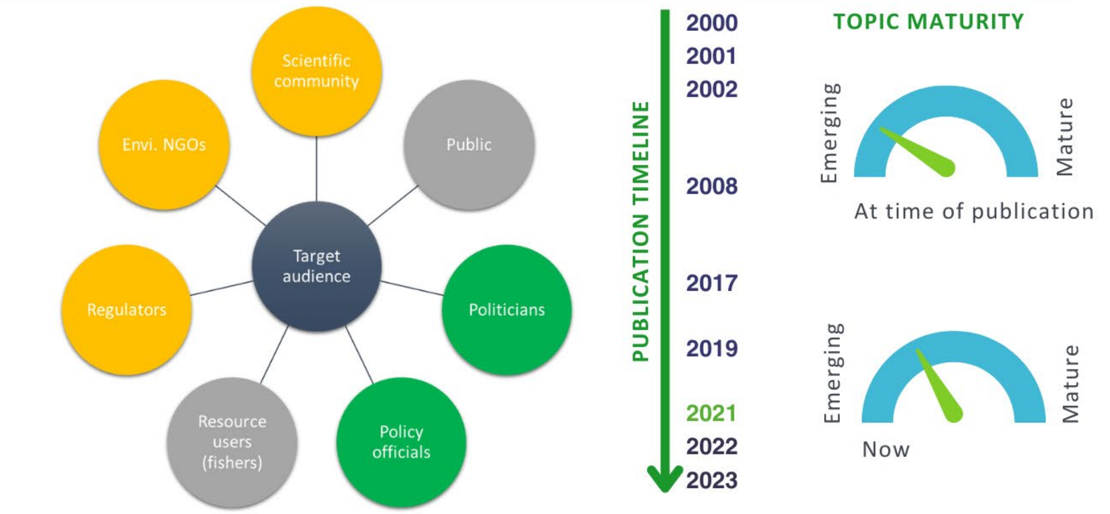
The papers within phase three (publications 7, 8 and 9) focus on influencing policy with the aim of securing significant environmental gains at scale and pace. English waters are the focus of the publications, although they have wider application.

The European Union (EU) referendum in 2016, which had ‘take back control’ as a key message, was tangible to the public, showing the powerful influence of the fishing sector on national and international psyche and politics (Stewart et al., 2022, Phillipson and Symes, 2018). Yet, EU exit provided the opportunity for Government and, by extension, society, to consider what we want from our waters, and how and where is the optimum way to provide this. Government made clear its intention to transition from the Common Fisheries Policy (CFP) (Defra, 2018a) to an alternative ‘world class’ framework, yet it did not have a view of what that framework should look like (Reeves et al., 2018). Thus, the transition from the CFP to the new fisheries management framework provided the opportunity to develop a new regulatory and financial incentive/disincentive regime. This regime could be used to secure improvements in the state of the wider seas by transitioning (aiding or compelling) the industry to less damaging fishing operations in terms of where, when, and how fishing is conducted throughout UK waters. Publications 7 and 9 are disruptive in that they were targeted at specific government departments (e.g. His Majesty’s Treasury (HMT)) with the intention of challenging the status quo of resource management. These papers were possible because of the role of the author working within a Statutory Nature Conservation Body (SNCB) at the SPI during a time of significant policy and legislative flux.

## 4.1 Marinising a terrestrial concept: Public money for public goods [publication 7]

**VAUGHAN, D., SHRIMPTON, E.A., CARPENTER, G., SKERRITT, D.J. AND C. WILLIAMS (2021). MARINISING A TERRESTRIAL CONCEPT: PUBLIC MONEY FOR PUBLIC GOODS.**

- The public money for public goods (PMPG) concept can be considered beyond farming. The marine environment offers one promising application
- PMPG should prompt debate and recognition of the value of marine 'public goods'
- Some forms of aquaculture are well-placed to benefit, while current fishing practices are likely to struggle
- Future research should seek to identify public goods fishers can enhance, distinguish cases where subsidies should be used instead of regulation, and map out a just transition for the application of a new framework



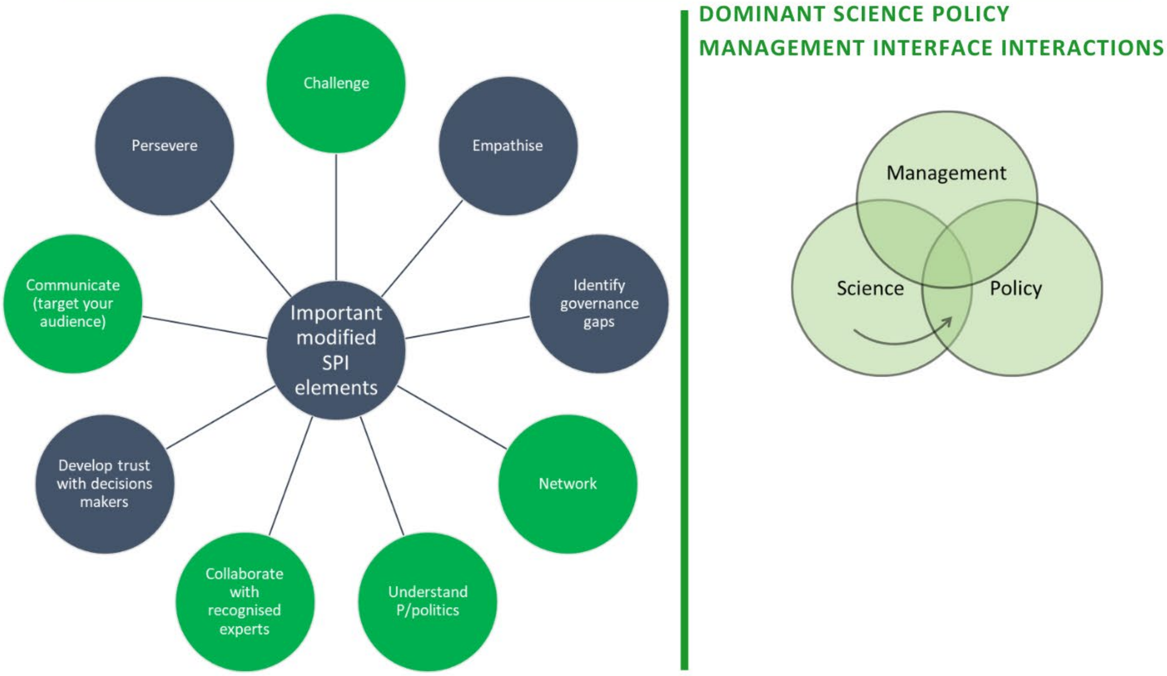




*Figure 10. Vaughan et al. 2021 [publication 7] infographic*

### 4.1.1 How has the work contributed to the scientific evidence base?

Publication 7 sought, for the first time, to consider the application of the concept of using public money for public goods (PMPG) in the marine environment and to provide the peer-reviewed scientific analysis that could be used to influence the embryonic fisheries and marine management policy space, post EU exit.

The analytical foundations for the theory of public goods were laid by Samuelson (1954) in 'The Pure Theory of Public Expenditure'. However, it was not until the early 2000s and critiques of the Common Agricultural Policy (CAP) (Zahrnt, 2009) that the use of PMPG as a concept was developed. The development of this concept into the marine environment is novel.

An early citation of publication 7 considers the PMPG concept and multifunctionality[17] within the Vermont dairy industry and the Maine Lobster fishery through the exploration of farmers' and fishers' perceptions of the value of the tourism industry in New England (Paras et al., 2022). This work identifies that public policy could be used to redistribute the benefits of tourism through (amongst other options) the provision of direct subsidies to farmers and fisheries, recognising the vitally important role they have in attracting visitors to these states.

Although publication 7 was not directed at academics, and had an England focus, the paper is already being used internationally to question the use of public money in the provision of fishing subsidies (Elsler and Oostdijk, 2023).

### 4.1.2 How has the work influenced management and policy?

A transition towards an agriculture subsidy framework deploying the PMPG concept became a possibility following the referendum on the UK leaving the EU and, by extension, the CAP (Bateman and Balmford, 2018). NE was tasked by Defra with delivering the financial aid package (the English Land Management Scheme) for farmers based on the PMPG concept. The principal organisational body within NE engaged on EU Exit policy engagement work was the Green Farming and Fisheries Programme. An internal workshop exploring policy opportunities led to the challenge: could a PMPG concept and framework be applied in the marine environment, and specifically to fisheries and aquaculture? Publication 7 answers that question.

Fishers received significant financial support from government through subsidies (Costello and Mangin, 2015, Reeves et al., 2018). However, fisheries subsidies have long been shown to result in overfishing (Sumaila et al., 2019, Costello et al., 2021) and support for a sector that is recognised as

[17] the characteristic of agriculture to produce not only food and fiber, but also an array of environmental, cultural, and rural development benefits is referred to as multifunctionality Paras et al. (2022).

one of the principal activities exerting pressures leading to the current degraded state of UK waters (United Kingdom Marine Monitoring and Assessment Strategy, 2018). It was for this reason that publication 7 was developed not with an academic audience in mind, but one comprising government ministers and policy officials within Defra (as the implementing government department responsible for fisheries policy), the MMO (as the principal grant management body for fishing in the UK through its role as the Management Body for the UK Seafood Fund), and, critically, HMT (as the government department that can compel other departments to progress policy development that would otherwise be challenging).

It quickly became clear that the concept of the PMPG in the marine environment had not been considered previously, either by government or within the academic literature. It also became clear that because fishers operate within a 'commons' system, i.e. they do not own the sea or the resources they target, the PMPG concept could be applied to fishing and aquaculture, but there would be limited instances that fisheries would be able to receive payment – principally because the very act of fishing is a damaging one (in terms of both removal of stock, a public asset, and damage to the marine environment, also a public resource). As set out within publication 7, fishers currently receive public money to conduct an activity for private gain that can lead to the significant degradation of public assets. This is challenging to reconcile with the application of the polluter pays principle (of which damage is considered pollution). This principle was enshrined in the Environment Act 2021 (UK Government, 2021) and the associated Environmental Principles Policy Statement (Defra, 2023b). These important findings in the paper were shared directly with HMT, Defra, the MMO, and counterparts in the Steering Group of the UK Seafood Innovation Fund (UKSIF). Feedback from policy officials within Defra on the impact of publication 7 was that Defra had been asked by HMT to consider how the UK Seafood Fund (circa £100m) and other future funding schemes could incorporate the PMPG model (Defra, 2022a).

The research outputs from publication 7 were presented at the 2022 Coastal Futures conference to an audience principally constituting academics, conservation advisors, regulators, and environmental policy officials. The application of the PMPG concept to fisheries was also set out within NE's written evidence to the EFRACSC Food Security Inquiry (Natural England, 2022).

### 4.1.3 How could the work be updated?

Understandably, there is a desire within government to continue to support the fishing and aquaculture industry, not least to ensure international competitivity in a business environment that has seen significant trade barriers being introduced following EU exit (Barnes, 2022). A desire to support industry, but also to transition to a financial support framework that increasingly reflects a PMPG concept, is creating challenges. The reality is that fishing as an extractive and damaging activity of a public resource for private profit is difficult to reconcile with a PMPG model. Therefore, consideration of how a model whereby public money is provided to fishers to aid a transition to activities that damage the environment less is recommended.

## 4.2 Same species, same space, different standards: a review of cumulative effects assessment practice for marine mammals [publication 8]

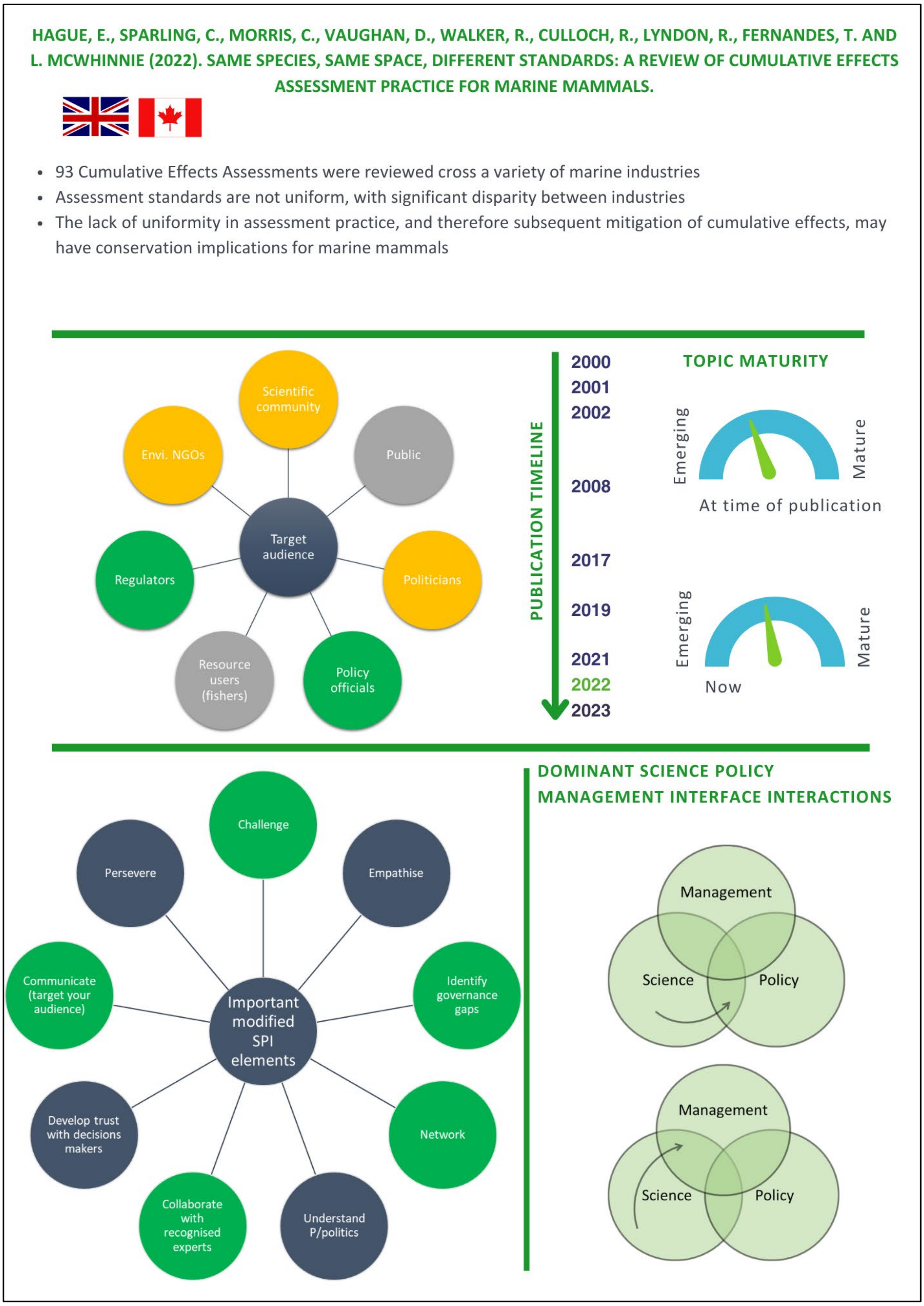



*Figure 11. Hague et al. 2022 [publication 8] infographic*

### 4.2.1 How has the work contributed to the scientific evidence base?

Publication 8 presents a review of Cumulative Effects Assessment (CEA) practice, with a comparison across 11 maritime industries. The work considers assessments for marine mammals; however, the findings and recommendations are relevant to other countries, environments, and species. This is the first study that compares and quantifies differences in the practice of assessing potential cumulative impacts between maritime industries. This is important, as this work identifies non-uniformity between sectors, indicating that some industries may be more rigorous in preventing the cumulative impacts of their respective activities than others. This work provides new insights into the state of practice, whilst providing solutions to address current limitations, with the aim to standardise and ultimately improve the assessment and mitigation of cumulative impacts. This provides a timely and significant contribution towards improving practices that can deliver effective marine conservation. Publication 8 is already being cited, with most citing papers seeking to highlight the disconnect between the need to protect wider seas and the growth of the blue economy (Turschwell et al., 2023, Willsteed et al., 2023).

Publication 8 was submitted as written evidence (UK Parliament, 2022d) to the UK Parliament Marine Mammals inquiry ‘to investigate issues affecting marine mammals and how Britain can play a role in protecting them’ (UK Parliament, 2022b).

### 4.2.2 How has the work influenced management and policy?

The report of the UK Parliament Marine Mammals inquiry (House of Commons, 2023b) highlighted the increasing pressures marine mammals faced which threaten population numbers and welfare. The report also noted that a significant barrier to their protection is the lack of data available. Recommendations for improvements in monitoring and assessing marine mammals were made. Whilst the Government, in its response to the inquiry (House of Commons, 2023a), did not explicitly refer to strengthening marine assessment, it did highlight multiple projects intended to improve the collection of marine mammal monitoring, which, in turn, can inform and indeed improve the environmental assessments of marine developments as sought by publication 8.

The UK government is continuing to consider the role and influence of environmental assessment in the marine environment and the implications this has for: (a) MSPri and the achievement of targets and obligations related to offshore energy generation, and (b) the delivery of Target 3 (Convention of Biological Diversity, 2023), which the UK government interprets as 30% of the UK Economic Exclusive

Zone (EEZ) should be designated as MPAs by 2030[18]. This consideration presents not only opportunities for strengthening environmental assessment (as recommended by publication 8) but also threats of environmental regression.

### 4.2.3 How could the work be updated?

Publication 8 did not consider (a) whether the CEAs were located within an MPA or (b) what the implications of this might be for the quality of the assessments scored by the authors of the paper, i.e. were scores inadvertently higher for those CEAs where the development was located within an MPA? It is recommended that publication 8 is updated to address this gap to ascertain the implications of any differences for marine licensing and consenting.

Interest in wider seas protection and management is increasing (see early discussion in this thesis and publication 6), leading to consideration of 'go fish' and 'go develop' areas within the wider seas that can guide developers to areas that, although not protected, are of lower sensitivity to the pressures from their development activity. Unfortunately, the distribution of habitats at a fine EUNIS[19] classification scale and thus the sensitivity of the UK's wider seas is unclear. To address this, a habitat sensitivity and mapping tool (Natural England Seabed Spatial Sensitivity Tool – NESSST) is being developed to partially fill this evidence gap (Hartley et al., In preparation).

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[18] 30x30 is the UK colloquial term for: Target 3 in the Global Biodiversity Framework *"Ensure and enable that by 2030 at least 30 per cent of terrestrial, inland water, and of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem functions and services, are effectively conserved and managed through ecologically representative, well-connected and equitably governed systems of protected areas and other effective area-based conservation measures, recognizing indigenous and traditional territories, where applicable, and integrated into wider landscapes, seascapes and the ocean, while ensuring that any sustainable use, where appropriate in such areas, is fully consistent with conservation outcomes, recognizing and respecting the rights of indigenous peoples and local communities, including over their traditional territories"*.

[19] For a detailed explanation of EUNIS see: https://www.marlin.ac.uk/habitats/eunis

## 4.3 The use of fuel tax concessions in the UK commercial fishing fleet [publication 9]

**VAUGHAN, D., SKERRITT, D.J., DUCKWORTH, J., SUMALIA, U.R. AND M. DUFFY (2023). THE USE OF FUEL TAX CONCESSIONS IN THE UK COMMERCIAL FISHING FLEET.**

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- Foregone government revenue via fisheries fuel subsidies ranged between £150-180 m. per year
- Segments of the UK commercial fishing fleet would be unprofitable without fuel subsidies assuming no changes in fuel use, engine efficiency or activity
- The most fuel intensive fleet segments are more likely to become unprofitable if fuel tax concessions were removed
- ·The fleet segments that receive the most subsidy burn the most fuel, create the most CO2 emissions and impact the seabed the most





**DOMINANT SCIENCE POLICY MANAGEMENT INTERFACE INTERACTIONS**

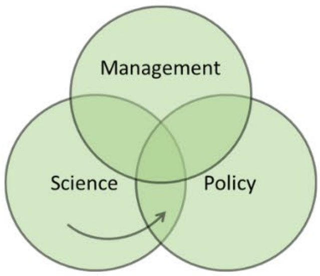


*Figure 12. Vaughan et al. 2023 [publication 9] infographic*

### 4.3.1 How has the work contributed to the scientific evidence base?

Publication 9, for the first time, draws upon public datasets that have been collected and made available with government funding to develop a model (and associated publicly available R script) that can be used to determine the reliance of the UK fishing industry, and different sectors within it, on fuel tax concessions (FTCs). The model can be updated annually as new data are made available. The model can also be modified to replicate this work for other countries.

Harmful fishing subsidies, and in particular fuel subsidies, have long been identified as a barrier to the delivery of sustainable fisheries management, with subsidies resulting in overfishing and environmental impacts (Skerritt et al., 2023). Timperley (2021) sets out that fossil fuel subsidies are one of the biggest financial barriers hampering the world's shift to renewable energy sources; however, it is only recently that scientific attention is being drawn to the relationship between fishing subsidies and $CO_2$ emissions in the fisheries sector (Machado et al., 2021). Publication 9 considers this link in detail for the UK fishing fleet.

It is becoming increasingly clear that fossil fuel subsidies or FTCs (both of which reduce the effective cost of fuel) are increasingly difficult to reconcile scientifically, economically, environmentally, and politically. For policies to be developed that provide for or compel a just transition from fossil fuel reliance by individual sectors (i.e. fishing or agriculture), detailed information is required on the type and magnitude of subsidies that a sector can receive. The importance of this information is reflected in the United Nations Climate Change Conference (COP28) decision text (United Nations Framework Convention on Climate Change, 2023), which requires: 'Phasing out inefficient fossil fuel subsidies that do not address energy poverty or just transitions, as soon as possible'.

### 4.3.2 How has the work influenced management and policy?

Section 1 of the Fisheries Act 2020 (UK Government, 2020) sets out eight Fisheries Objectives. One of these (in what is thought to be a world first; (Stephenson and Johnson, 2021)), is the climate change fisheries objective[20].

In addition, section 2 of the Act requires the government to publish a JFS detailing the policies that will deliver the objectives. As the Senior Advisor for NE leading on the provision of advice to Defra on the development of the draft JFS, I conducted a gap analysis to identify potential new policies that could/would need to be developed to deliver the government's stated ambition of world class fisheries management (Defra, 2018a). One suggested policy was: *fishing subsidies – fuel use: the red*

[20] The "climate change objective" is that—
(a) the adverse effect of fish and aquaculture activities on climate change is minimised, and
(b) fish and aquaculture activities adapt to climate change.

*diesel fuel rebate is phased out for the use within vessels engaged within commercial fishing and aquaculture operations*. This new policy subsequently appeared in a modified form in section 4.2.14.2 of the JFS (UK Government, 2022b).

To potentially inform this policy, publication 9 was developed. The FTCs provision by the UK government (circa £150–180m/yr 2009–2019) established in the draft paper were initially challenged by Defra economists as the understanding at the time (as recorded in the OECD Fisheries Support Estimates) was that the FTCs were significant lower (circa £27m and £32m for 2007 and 2008, respectively). The research highlighted that only one component of the FTCs – Marine Voyages Relief (MVR) – that the fishing industry could receive had been identified/reported, and the total relief provided by the two components of the FTCs (Red Diesel Rebate (RDR) and MVR) had not been estimated. Publication 9 provides a transparent estimate of the combined FTCs accessible by the UK commercial fishing sector (important because this is public money foregone in support of commercial operations), enhancing the evidence base on sector support. Publication 9 provides UK data points that can be considered against FTC figures estimated for the EU fishing fleet, such as those set out in the recent report, *Better Use of Public Money: The End of Fuel Subsidies for the EU Fishing Industry* (Elsler and Oostdijk, 2023).

Shortly after presenting the work to Defra economists, the Defra Minister Victoria Prentis provided evidence at the Environment Food and Rural Affairs Commons Select Committee (EFRACSC) *Inquiry into the UK Seafood Fund* (UK Parliament, 2022c). An improved understanding of the components of the FTCs that the fishing industry could draw up enabled Minister Prentis to set out the following response to a question regarding the government position on supporting fishers during a period of high fuel prices resulting from the war in Ukraine:

*"……. fishing is one of the very few sectors to benefit from duty relief on fuel. There are two types of duty relief. One is the marine voyages relief and the other is the extension of red diesel, which the Chancellor was pleased to give us in this year's Budget, for the fishing sector. Taken in combination, that should add up to 100% of relief on fuel duty costs for fishing. We are not minded to pay fishermen's fuel bills, apart from the relief that they have been given."*

After the EFRACSC UK Seafood Fund Inquiry, NE submitted written evidence to the EFRACSC Food Security Inquiry. This evidence set out the extent of the FTC, links between FTCs and the Climate Change Fisheries Objective, and the state of UK waters resulting from fishing activity (Natural England, 2022).

During the development of publication 9, the research was raised at a UK Government Fisheries and Climate Change Liaison Group (FCCLG). This group was established and hosted by Defra for UK marine policy officials, regulators, and conservation advisors. The potential to use FTCs as a tool to aid the delivery of carbon net zero within the fishing sector was subsequently incorporated in the report, *Carbon emissions in UK fisheries: recent trends, current levels, and pathways to Net Zero* (Engelhard et al., 2022). On 6 September 2023, a follow-up presentation of the FTC paper was provided to the FCCLG to ensure dissemination of the work. Publication 9 was subsequently shared across government, including with HMT.

NE is a statutory consultee under The Environmental Assessment of Plans and Programmes Regulations 2004 (SEA regulations 2004). Under these regulations, NE provided its response to a Defra consultation on six Fisheries Management Plans. The response referenced publication 9 and the Climate Change Fisheries Objective, and noted that some fishing operations utilise disproportionately greater quantities of fuel whilst simultaneously often being the same fisheries that are responsible for the greatest benthic disturbance (Natural England, 2023).

The R model code underpinning publication 9 has been provided to Seafish (a UK public body) to enable the FTC calculations to be replicated and updated annually as new data become available. The work led to modifications being considered of both the Seafish Sea Fishing Fleet Survey and the underlying database for the *UK Ship Register (UKSR) Part 2 – Fishing* to facilitate analysis on the transition to alternative fuel propulsion and the development of policy decisions regarding economic support to the sector in support of carbon net zero delivery.

Publication 9 highlighted the extent to which the sector is reliant on FTCs to operate profitably. If FTCs were removed, this would have serious implications for the fleet. The publication has attracted the attention of the fishing sector (Oliver, 2023) and the media (McVeigh, 2024). The publication is already providing an evidence base for industry to engage in an informed manner with policy officials. Importantly, the work can be used to inform policy and industry discussions regarding what a future fishing fleet composition/structure could/should look like (to deliver the eight fisheries objectives), and how the grant and tax systems can be designed to support this.

### 4.3.3 How could the work be updated?

Although publication 9 sets out the scale of the FTC and the difficulties in reconciling this with the Climate Change Fisheries Objective, the work to date does not set out how the FTCs are distributed, be it by geography (e.g. by administration, i.e. Scotland) or by another metric. Understanding how FTCs are distributed geographically would enable policy officials to better understand how FTC removal/modification would affect different sectors/areas. The available government datasets do not allow this analysis.

The model highlighted inadequacies in both the UK Fishing Vessels Register and the Seafish Fishing Fleet Survey – neither of these record/allow the identification of propulsion type, i.e. electric, fossil fuel, or indeed type of fuel (i.e. diesel/petrol). This information will be important when tracking the transition to renewable/low carbon fuel sources. This publication has led policy officials to seek changes to the Fleet Survey to facilitate this analysis. Discussions are underway with the Maritime and Coast Agency regarding extractions of data from the shipping registry database to aid future analysis.

Publication 7 on the application of the PMPG concept took a terrestrial concept and sought to apply it in the marine realm. It is suggested that in the case of the FTC model, the reverse is explored, i.e. the model is applied to agricultural activities. The challenge of reconciling the provision of harmful fossil fuel subsidies to the agricultural sector with UK Carbon Net Zero 2030 commitments is akin to that faced by the fishing and construction sectors (as set out in publication 9). Like fishing, the agricultural sector avails itself of RDR, which reduces the level of tax applied to fuel. Unlike the maritime sector, the agricultural sector cannot avail itself of the MVR, and therefore it does not benefit from the complete exemption of fuel tax (it merely pays a lower rate of duty).

One of the challenges to the removal of FTCs from the fishing sector set out within publication 9 is that of providing a level playing field in an open access and competitive fishery; if the UK were to remove the FTCs, foreign vessels would still be able to access UK waters to fish their quota, but could bunker within the EU, where FTCs are still available. There are moves in the EU to review FTCs. Were this to occur, and if FTCs were removed ahead of the UK, then countervailing measures (an anti-subsidy measure) could be applied by the EU in the form of tariffs on UK goods such as seafood (European Union, 2016). Conversely, if the UK were to remove FTCs ahead of the EU, then this would put UK vessels (principally small inshore vessels, as larger UK offshore vessels could transit and bunker in the EU, also strengthening the incentive to land outside of the UK) at an economic disadvantage.

The FTC model developed considered the most up-to-date data at the time. The model can be rerun as new data become available, enabling policy officials to consider how exiting the EU has altered fuel consumption and emissions, and how the FTC contributions to fleet segments have changed. Recent work (Scherrer et al., 2023) has highlighted that the fuel intensity of the Norwegian fleet doubled after EU exit.

A similar economic disadvantage may present itself if the UK agricultural sector were to be faced with the removal of the RDR. Both fish and agricultural products are traded globally. Nevertheless, the desire to apply the PMPG concept to agriculture leads one to question the appropriateness of continued tax relief for this sector; - in essence, public money is forgone in support of continued fossil fuel consumption. To inform future anticipated policy discussions on the continued appropriateness of FTCs being applied to the agricultural sector, it is recommended that analysis is conducted by sector, i.e. upland sheep farming, cereal production, etc., to better understand the reliance of each sector on the RDR. This information will be valuable for policy officials as the agricultural sector employs significantly more individuals and contributes more to UK GDP than fishing.

The FTC model within publication 9 could be adapted appropriately, and the analysis completed relatively quickly, as the information required to populate the model appears to be collected and collated in the Defra Farm Business Survey (UK Government, 2023b) with data from 2013 to present covering England and Wales, and with comparable data collected in Scotland and Northern Ireland (SRUC, 2023, DAERA, 2023).

## 4.4 Phase three: Science, Policy, and Management Interface summary

The publications within this phase were written with the express intention of eliciting policy action that would deliver environmental outcomes at scale and pace. Publications 7, 8 and 9 were intended to challenge the status quo of marine management using the opportunity that EU exit afforded through the development of environmental and fisheries management legislation – namely, the Fisheries and Environment Acts – and subordinate legislative/policy framework. The development of these publications and the identification of legislative/policy gaps were made possible through an extensive professional network and previous experience of collaborating with some of the co-authors. These gaps included the application of the PMPG framework to fishing and aquaculture (publication 7), Cumulative Impacts Assessment (publication 8), and the Fisheries Act Climate Change Objective (publication 9).

Publication 9 took almost three years from conception to publication. Much of this time was spent navigating the politics of government bodies to ensure that the rationale for the work, and the policy implications of it, were understood and valued prior to submission of the paper for peer review and publication. Demonstrating empathy with the fishing sector when communicating findings of publications 7 and 9 was crucial to ensure the maintenance of professional relationships. Empathy enabled framing the work as important to fishers so that a just and supported transition away from fossil fuels could be developed rather than the work seeking to undermine public and governmental support for the sector. Networking led to the effective collaboration in publications 7 and 8. The collaboration in publication 7 led to further collaboration in publication 9 and the opportunity to draw in global subject matter experts, adding credibility to the work.

Key elements for working effectively at the SPI identified within this phase were: (a) perseverance to successfully (b) challenge the status quo, which requires (c) empathy and (d) securing the trust of policy officials, which in turn requires (e) political understanding. Fig. 11 highlights one interaction with the missing component (management) from the classic SPI model: science → management.

# 5 Discussion

The development of effective responses to the climate and biodiversity crises that face humanity at the scale and pace required will necessitate scientists, policy officials, politicians, and managers to effectively interact, co-design, communicate, and deliver these responses. To address these crises requires a shift to a systems thinking approach (STA) – a shift recognised by the UK Government (Defra, 2023a)[21]. For this shift to an STA to be effective, those tasked to deliver this will be required to effectively engage at the SPI.

The SPI has evolved and strengthened over the last few decades, which has led to significant improvements in the communication between scientists and policy officials/politicians. Of crucial importance over the next decade will be converting international commitments to address the climate and biodiversity crises into policy, and then into management action that delivers policy effectively, in a just and timely manner.

The publications considered within this thesis were analysed through the lens of a modified SPI incorporating a novel management component (Fig. 2). That this management component is not considered within the SPI literature is therefore an important observation and one recommended for further investigation. Further consideration of this missing component of the SPI may provide opportunities to address the a) need to increase the effectiveness of marine scientists working at the SPI is highlighted by Cvitanovic et al. (2015), b) mismatch between the claims and reality of the effectiveness of knowledge exchange (Karcher et al., 2021).

Lessons learnt whilst working at the SPI throughout the development of the publications considered in each of the three phases of this thesis are important. They have been learnt over two decades working in multiple countries, collaborating with local, regional, and global experts on differing topics and from differing perspectives, i.e. sometimes I have been working at the interface as a scientist and others as a manager or policy official. This has provided me with experience and perspective on effective engagement and, importantly, action that results from that engagement when working at the SPI.

The identification of nine key elements below is intended to aid others working at the SPI to maximise their effectiveness for the benefit of society and the wider environment. These elements are:

1) ***Challenge – be curious and question the status quo***
2) ***Empathise***
3) ***Identify governance and knowledge gaps***
4) ***Network***

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[21] For an overview see: (Aronson, 1996)

5) ***Collaborate with recognised experts***

6) ***Understand politics***

7) ***Develop trust of decision makers***

8) ***Communicate – target your audience***

9) ***Persevere***

These elements are drawn from the work presented in this thesis and are based on the large body of work in the literature as discussed below. The elements identified for each publication are summarised in Table 1.

*Table 1. The elements of key to success when working at the modified Science and Policy Interface identified in each of the publications considered within this thesis.*

| Elements key to success when working at the modified SPI interface | Phase of work | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Phase 1: Baseling and knowledge transfer | | | | Phase 2: Protecting the wider seas | | Phase 3: Influencing policy at scale | | |
| | Publication 1 | Publication 2 | Publication 3 | Publication 4 | Publication 5 | Publication 6 | Publication 7 | Publication 8 | Publication 9 |
| Challenge | | | | | | | | | |
| Empathise | | | | | | | | | |
| Identify governance and knowledge gaps | | | | | | | | | |
| Network | | | | | | | | | |
| Collaborate with recognised experts | | | | | | | | | |
| Understand P/politics | | | | | | | | | |
| Develop trust of decision makers | | | | | | | | | |
| Communicate - target your audience | | | | | | | | | |
| Persevere | | | | | | | | | |

**Curiosity**, time to think and reflect (for yourself and others), and **willingness to challenge the status quo** (ask why or why not?) can provide insights into different perspectives that will help to better understand problems and aid the identification of solutions. Challenging the status quo may mean significant change for stakeholders. This will be the case if we are to address the climate and biodiversity crises. Change may be embraced and/or resisted as there will be winners and losers as society adjusts to the challenges and opportunities presented. It is therefore important to recognise this and demonstrate **empathy** for those affected.

Furthermore, one should seek to **identify governance and knowledge gaps** so that evidence can be collected, and papers developed that can influence how the governance gap is filled (Bergsten et al., 2019, Fried et al., 2022, Karlsson and Gilek, 2020). To do this, **build a network** of people with different perspectives and skill sets that you can draw upon to check thinking, challenge and collaborate with. Collaboration increases the reach of your work and its credibility (Kelemen et al., 2021). **Collaborating with recognised local and global subject matter experts** can maximise the reach and impact of interactions and interventions.

Different government departments and policy officials with the same department may have different and competing agendas. **Try and understand inter- and intra-department politics** in addition to national politics to identify individuals that are blockers and enablers to attempts to influence policy change or development (Watson, 2005, Gluckman et al., 2021). Try to identify the power holders – these may be individuals or government departments which have a different but more powerful agenda that can be used to influence policy, i.e. HMT. In the pursuit of this, building relationships with officials (policy leads and Special Advisers[22]) is critically important.

**Trust, respect, and understanding** are byproducts of working collaboratively through difficult matters and not products in and of themselves. When there is trust, there are more opportunities to collaborate, which moves you from a consultation and transactional relationship to one of co-design or indeed a delegation to you to do the drafting of policy (Lacey et al., 2018). Opportunities may present themselves because policy officials are often extremely busy and may not be subject-matter experts; therefore, if you have their trust, you may be provided with tremendous opportunities to influence. When commenting on draft policies or consultation documents, it can be highly productive to suggest replacement text when seeking amendments. The most effective relationship with policy officials is when you are seen to be a useful collaborator, as this enables you to operate and influence behind the scenes, but also on the record, so that views are known and understood. Recognise that Ministers, regulators, and policy officials, day to day, have contact with stakeholders that have 'skin in the game', and they are also often the ones that shout the loudest. Therefore, it is these stakeholders that are often heard, and as such, they have compelling influence. Wider society or indeed other elements of a stakeholder group can be poorly or imperfectly represented in these day-to-day policy decisions and discussions.

To maximise the impact of any intervention or interaction, it is important to **communicate** well (Spierenburg, 2012). Determining the most appropriate method of communication to reach the most appropriate and influential audience to aid the delivery of the desired outcome is vital; for example, it may be possible to target peer-reviewed papers at government departments to maximise influence on policy.

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[22] Special Advisers are a type of civil servant in the UK. https://www.civilservant.org.uk/spads-homepage.html

Attempting to effect change, let alone change at pace and scale, is incredibly challenging. It requires **perseverance** and a strong support network (De Young, 2011, Rose et al., 2020). I consider myself incredibly lucky to have the second, professionally and personally, throughout the period covered by the publications presented within this thesis.

The critical analysis within this thesis highlights the substantial contribution that the publications considered have made to the scientific literature, policy development and, importantly, marine management interventions. These publications (a) incorporate the development of original ideas such as the application of PMPG to the marine environment, (b) seek to raise the visibility of topics such as fishing effort displacement and spatial squeeze to identify the race for marine space by different marine sectors, (c) seek to offer solutions to addressing marine management challenges such as enforcement techniques, and (d) contribute to understanding implications of the climate crisis and identifying work needed to aid the transition away from fossil fuel activities.

# Glossary

***Arm's Length Bodies:*** a specific category of central government (UK) public bodies that are administratively classified by the Cabinet Office.

***Cumulative Environmental Assessment***: a sub-discipline of environmental impact assessment that is concerned with appraising the collective effects of human activities and natural processes on the environment.

***Environment Food and Rural Affairs Commons Select Committee:*** is appointed by the House of Commons to examine the expenditure, administration and policy of the Department for Environment, Food and Rural Affairs and its associated public bodies.

***Exclusive Economic Zone:*** an area of the sea in which a sovereign state has exclusive rights regarding the exploration and use of marine resources, including energy production from water and wind.

***Fuel Tax Concession***: the combined tax relief provided by Red Diesel Rebate and Marine Voyages Relief in the UK.

***Fisheries Science Partnership:*** a form of fisheries co-management, in that it entails joint research undertaken by fishers and scientists together that is often initiated, funded, and monitored by government, and feeds directly into fisheries management decision-making.

***Highly Protected Marine Area:*** a type of MPA. Areas of the sea (including the shoreline) that allow the protection and full recovery of marine ecosystems.

***Habitats Regulations Assessment***: the process that competent authorities must undertake to consider whether a proposed development plan or programme is likely to have significant effects on a European site designated for its nature conservation interest.

***Joint Fisheries Statement:*** a requirement of the Fisheries Act 2020 and aims to make sure that policies deliver a thriving, sustainable fishing industry and healthy marine environment.

***Marine Conservation Zone:*** a type of MPA. Areas that protect a range of nationally important, rare, or threatened habitats and species.

***Marine Planning:*** a tool for governments to allocate space for different activities and uses of our seas.

***Marine Protected Area:*** defined by the International Union for Conservation of Nature as parts of intertidal or subtidal environments, together with their overlying waters, flora and fauna and other features, that have been reserved and protected by law or other effective means.

***Marine Spatial Planning***: the process by which various stakeholders, adjust their uses in the marine space.

***Marine Spatial Prioritisation***: the process by government prioritises how marine space will be allocated (prioritised) for different marine sectors to deliver governmental commitments.

***Marine Strategy Framework Directive:*** European Union legislation setting out the aim to achieve Good Environmental Status for the EU Member States' marine waters by 2020, applying the Ecosystem Approach.

***Maximum Sustainable Yield***: a theoretical concept used extensively in fisheries science and management. In fisheries, MSY is defined as the maximum catch (in numbers or mass) that can be removed from a population over an indefinite period.

***Net Zero:*** a target of completely negating the amount of greenhouse gases produced by human activity, to be achieved by reducing emissions and implementing methods of absorbing carbon dioxide from the atmosphere.

***Public Money for Public Goods:*** the concept of ensuring public money is spent on securing public goods that are not provided for through markets i.e. clean air or flood protection.

***Small Islands Developing States:*** a grouping of developing countries which are small island countries and tend to share similar sustainable development challenges. These include small but growing populations, limited resources, remoteness, susceptibility to natural disasters, vulnerability to external shocks, excessive dependence on international trade, and fragile environments.

***Special Area of Conservation:*** a type of MPA. Are protected areas in the UK designated under the Conservation of Habitats and Species Regulations 2017 (as amended) in England and Wales.

***Statutory Instrument***: a form of legislation which allow the provisions of an Act of Parliament to be subsequently brought into force or altered without Parliament having to pass a new Act.

***Statutory Nature Conservation Body:*** an organisation charged by government with advising on nature conservation matters.

***Special Protection Area:*** a type of MPA. Areas for birds in the UK classified under the Conservation of Habitats and Species Regulations 2017 (as amended) in England and Wales.

***Science Policy Interface:*** the interaction scientists and other actors in the policy process, and which allow for exchanges, co-evolution, and joint construction of knowledge with the aim of enriching decision-making.

***Systems Thinking Approach***: an approach to problem solving which considers the overall system as well as its individual parts.

***United Kingdom Marine Strategy***: the UK interpretation of the Marine Strategy Framework Directive – a three-stage framework for achieving Good Environmental Status in UK waters.

***Whole Site Approach:*** a developing concept whereby management is directed to protect important and sensitive habitats or species within an MPA that are not designated features.

***Wider Seas:*** the area of a country's Exclusive Economic Area that does not fall within a marine protected area network.

# Acronyms

| | |
|---|---|
| ALB | Arm’s Length Body |
| CEA | Cumulative Environmental Assessment |
| CEFAS | Centre for Environment, Fisheries and Aquaculture Science |
| CAP | Common Agricultural Policy |
| CFP | Common Fisheries Policy |
| Defra | Department of Environment, Food and Rural Affairs |
| EFRACSC | Environment Food and Rural Affairs Commons Select Committee |
| FAC | Fisheries Advisory Council |
| FSP | Fisheries Science Partnership |
| FTC | Fuel Tax Concession |
| GDP | Gross Domestic Product |
| GW | Giga Watt |
| HMT | His Majesty's Treasury |
| IFCA | Inshore Fisheries and Conservation Authority |
| IPCC | Intergovernmental Panel on Climate Change |
| HPMA | Highly Protected Marine Area |
| HRA | Habitats Regulations Assessment |
| JFS | Joint Fisheries Statement |
| JNCC | Joint Nature Conservation Committee |
| MCAA | Marine and Coastal Access Act |
| MCZ | Marine Conservation Zone |
| MMO | Marine Management Organisation |
| MP | Marine Planning |
| MPA | Marine Protected Area |
| MSP | Marine Spatial Planning |
| MSPri | Marine Spatial Prioritisation |
| MSFD | Marine Strategy Framework Directive |
| MSY | Maximum Sustainable Yield |
| MVR | Marine Voyages Relief |
| NE | Natural England |

| | |
|---|---|
| NERC | Natural Environment and Rural Communities Act |
| NFFO | National Federation of Fishermen's Organisations |
| PMPG | Public Money for Public Goods |
| RDR | Red Diesel Rebate |
| SAC | Special Area of Conservation |
| SI | Statutory Instrument |
| SNCB | Statutory Nature Conservation Body |
| SPA | Special Protection Area |
| SPI | Science Policy Interface |
| SSSI | Site of Special Scientific Interest |
| STA | Systems Thinking Approach |
| TCI | Turks and Caicos Islands |
| UKMS | United Kingdom Marine Strategy |
| WSA | Whole Site Approach |

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# Appendix A - confirmation of authorship


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15 June 2023

TO WHOM IT MAY CONCERN

This is to confirm that Duncan Vaughan had the contribution in all the aspects stated below for the following publication:

Cumming RL, Toscano MA, Lovell ER, Carlson BA, Dulvy NK, Hughes A, Koven JF, Quinn NJ, Sykes HR, Taylor OJS, **Vaughan D** (2000) *Mass coral bleaching in the Fiji Islands, 2000*. Proc 9th Int Coral Reef Symp, Bali, Indonesia, Vol. 2, pp 1161-1168.

*Role:* Co- investigator, development of methodology, acquisition of data, analysis and interpretation of data, writing, review, and revision of the manuscript.

Yours faithfully

Dr Robyn Cumming
Collection Manager, Marine

*Ministry of Tourism, Environment, Fisheries and Marine Affairs, Culture and Heritage, Agriculture & Religious Affairs*
*Turks & Caicos Islands Government*
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04 May 2023

**To whom it may concern**

This is to confirm that Duncan Vaughan had the contribution stated below for the following publication:

Clerveaux W, Vaughan D (2001) *An investigation of the effects of increasing fishing efficiency on the productivity of the queen conch (Strombus gigas) and Caribbean spiny lobster (Panulirus argus) fisheries within the Turks and Caicos Islands*. Proc 54th Annu Gulf Carib Fish Inst, 12–17 Nov 2001, Providenciales, Turks and Caicos Islands, p 285–296

Role: Co- investigator, conception and design, development of methodology, acquisition of data, analysis and interpretation of data, writing, review, and revision of the manuscript.

I can confirm that Duncan Vaughan was involved in the all the aspects stated above of this research.

Wesley Clerveaux
Permanent Secretary
Ministry of Tourism, Environment, Fisheries & Marine Affairs
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# INTERNATIONAL PACIFIC HALIBUT COMMISSION

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To whom it may concern

This is to confirm that Duncan Vaughan contributed in all the aspects stated below for the following publications:

Wilson, D.T., **Vaughan, D.**, Wilson, S.K., Simon, C.N. and Lockhart, K., (2008). *Efficacy of a starch-iodide swab technique to detect the illegal use of bleach in a Spiny Lobster (Panulirus argus) fishery*. Fisheries Research, 90(1-3), pp.86-91.

Wilson DT, **Vaughan D**, Wilson SK, Simon CN, Lockhart K (2004) *A preliminary assessment of the efficacy of a chlorine bleach detection method for use in spiny lobster (Panulirus argus) fisheries*. Proc 57th Annu Gulf Carib Fish Inst, 8–12 Nov 2004, St. Petersburg, FL, USA, p 859–868

*Role:* Co- investigator, conception and design, development of methodology, acquisition of data, analysis and interpretation of data, writing, review, and revision of the manuscripts.

Yours sincerely

David T. Wilson, Ph.D.

Executive Director, IPHC

# SOUND SEAS

*Scientific and Social Solutions*

June 14, 2023

To whom it may concern:

This is to confirm that Duncan Vaughan had the contribution in all aspects stated below for the following publication:

**Vaughan D**, Agardy, T. (2019) *Marine protected areas and marine spatial planning - allocation of resource use and environmental protection* In: Marine Protected Areas: Science, Policy and Practice. (Eds. Humphreys and Clark), Elsevier.

*Role:* Principal investigator, conceived and designed the chapter, developed the methodology, acquired data, analysed and interpreted data, as well as writing, review, and revision of the manuscript.

Signed: M. Tun A Date: June 14, 2023

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13th June 2023

**TO WHOM IT MAY CONCERN**

**RE: Confirmation of Contribution**

This is to confirm that Duncan Vaughan contributed to the following publication in the role(s) as set out below:

**Vaughan, D.**, Shrimpton, E.A., Carpenter, G., Skerritt, D.J. and Williams, C., (2021). *Marinising a terrestrial concept: Public money for public goods*. Ocean & Coastal Management, 213, p.105881.

*Role:* Principal investigator, conception and design, development of methodology, acquisition of data, analysis and interpretation of data, writing, review, and revision of the manuscript.

Signed......EAShrimpton...... Date....13/6/23....

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20th June 2023

To whom it may concern,

As first author of the publication:

'Hague, E., Sparling, C., Morris, C., **Vaughan, D**., Walker, R., Culloch, R., Lyndon, R., Fernandes, T. and L. McWhinnie (2022). *Same species, same space, different standards: a review of cumulative effects assessment practice for marine mammals*. Frontiers in Marine Science. 25 (9)',

I am writing to confirm that Duncan Vaughan contributed to the following stated aspects of the publication: *Role:* Co-investigator, collection, analysis and interpretation of data, review and revision of the manuscript.

Duncan also provided a great level of mentorship and support throughout the entire process and was instrumental in securing funding to support the open-access publishing of this work in a high impact factor journal.

Best wishes,

EHague

Date 20th June 2023

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NATURAL ENGLAND

22nd November 2023

To whom it may concern:

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**Vaughan**, **D**. Skerritt, D. J., Duckworth, J., Sumalia, U. R. and M. Duffy (2023) *The use of fuel tax concessions in the UK commercial fishing fleet.* Marine Policy. 155.

*Role:* Principal investigator, conception and design, development of methodology, acquisition of data, analysis and interpretation of data, writing, review, and/or revision of the manuscript.

Mark Duffy

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Principal Adviser – Marine Fisheries

# Appendix B - published papers

Cumming, R.L., Toscano, M.A., Lovell, E.R., Carlson, B.A., Dulvy, N.K., Hughes, A, Koven, J.F., Quinn, N.J., Sykes, H.R., Taylor, O.J.S., and D. Vaughan (2000). Mass coral bleaching in the Fiji Islands, 2000. Proc 9th Int Coral Reef Symp, Bali, Indonesia. Australian Institute of Marine Science, Townsville, Australia [publication 1]

# Mass coral bleaching in the Fiji Islands, 2000

**R.L. Cumming[1], M.A. Toscano[2], E.R. Lovell[3], B.A. Carlson[4], N.K. Dulvy[5], A. Hughes[6], J.F. Koven[7], N.J. Quinn[8], H.R Sykes[9], O.J.S. Taylor[10], D. Vaughan[11]**

**Abstract** The south-western Pacific island countries were largely unaffected by mass coral bleaching during the intense El Nino of 1998, but experienced mass bleaching in 2000 during the subsequent strong La Niña. Nineteen reef locations were surveyed in eight geographic regions within the Fiji archipelago between mid April and early July 2000, to assess the geographic extent and intensity of Fiji's first recorded mass bleaching event. 64% of all scleractinian coral colonies surveyed were bleached (partially or fully, or recently dead from bleaching), and mass bleaching occurred in all regions surveyed except in the far north (north of Vanua Levu).

Bleaching was most intense (>80% of colonies bleached) in southern and eastern sites (south and east from Viti Levu and Vanua Levu, Kadavu and Northern Lau), and lower in some western and one northern site(s). The geographic patterns in bleaching coincide with Fiji's position on the north-western edge of an area of high sea surface temperatures (SSTs), and support the prediction based on SSTs that bleaching should be most severe in the south and east. Seawater temperatures exceeded expected summertime maxima for 5 months and peaked at 30-30.5°C between early March and early April 2000. The bleaching threshold for Fiji appears to be in the range of 29.5-30°C. Our data estimate 10-40% of coral colonies had died from bleaching within four months of the onset of bleaching.

**Keywords** Coral bleaching, Fiji, Seawater temperature, HotSpot, Bleaching intensity, Bleaching mortality, Aerial survey, temperature logger, Degree Heating Weeks, Pathfinder

## Introduction

Coral reefs of the south-western Pacific Islands largely escaped the global-scale El Niño-associated mass bleaching of 1998, but experienced mass bleaching in 2000 during the subsequent strong La Niña. In early 2000, satellite surveillance of sea surface temperatures (SSTs) showed a band of warming water initiating in Papua New Guinea (PNG) and extending down through the Solomon Islands (NOAA/NESDIS HotSpot website: http://psbsgi1.nesdis.noaa.gov:8080/PSB/EPS/SST/climohot.html). By February 1 2000, HotSpot anomalies (HotSpots: SSTs 1°C or more above the climatological maximum monthly mean SSTs) extended across the south Pacific from PNG to Easter Island. Accumulated heat stress was greatest in a pool stretching south-east from Fiji, encompassing Tonga, Niue, southern Cook Islands and Tubuai (Fig. 1). Fiji was on the north-western edge of this pool and experienced within-country gradients in accumulated heat stress, with greatest heat stress in the south and east of the country.

Countries that experienced mass bleaching in 2000 include PNG and Cook Islands (WWF South Pacific Programme 2000a), Solomon Islands (WWF South Pacific Programme 2000b), Tonga (Lovell 2001), Easter Island (Wellington et al. 2001) and Fiji (Cumming et al. 2000; South and Skelton 2000; Lovell 2001). We are aware of only one previous report on mass bleaching in the south-west Pacific region, from Papua New Guinea in 1996 (Davies et al. 1997).

No coordinated country-wide monitoring program existed in Fiji at the time of the 2000 bleaching event. Instead, eight independent groups conducted surveys at 19 locations spread throughout Fiji (Fig. 2). These studies were not standardized and employed a wide variety of survey methods to estimate bleaching severity,

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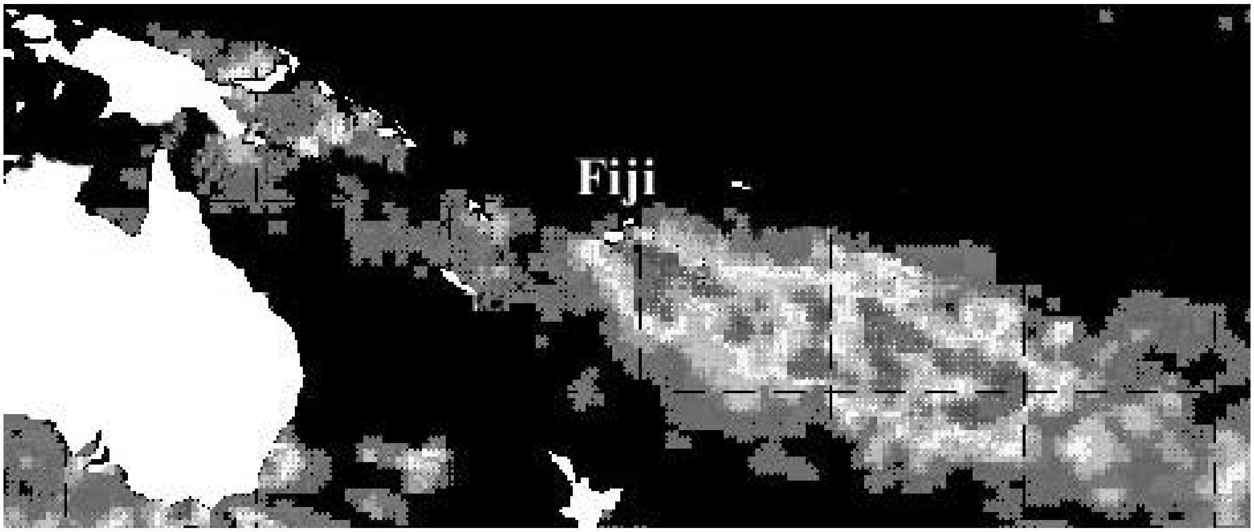



**Fig. 1** NOAA/NESDIS Degree Heating Weeks (DHW) chart for the 90-day period up to 15 April 2000. DHWs accumulate Hot-Spot anomalies over a continuous 12-week period. One DHW is equivalent to one week of satellite-derived SSTs 1°C above the MMM (maximum monthly mean: 28.3°C for Fiji) SST; two DHWs are equivalent to two weeks of SSTs 1°C above the MMM SST or one week of SSTs 2°C above the MMM SST, and so on. Light shading indicates 8-10 DHWs, darker shading inside the light shading indicates 10-14 DHWs. This chart is available in colour on the NOAA DHW website: http://psbsgi1.nesdis.noaa.gov:8080/PSB/EPS/SST/dhw_retro.html

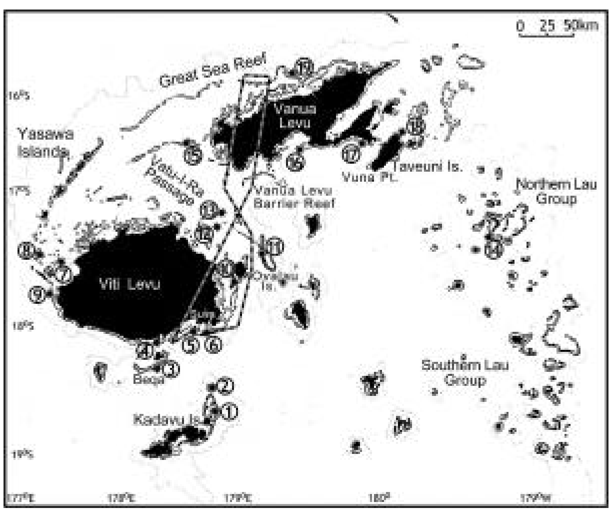



**Fig. 2** Map of Fiji showing surveyed locations and the route of the aerial survey (dotted line) Adapted from UNEP/IUCN (1988). Site numbers, names and descriptions are given in Table 1.

including line intercept transects, belt transects, colony counts, point counts, video transects, video quadrats, coral tagging and aerial surveillance (see Table 1). We standardized these data to provide an overview of the geographic extent and severity of bleaching in Fiji. To investigate seawater temperature as a forcing factor, we accessed both *in situ* and satellite-derived data, and show a geographic pattern of bleaching that corresponds to seawater temperatures.

## Materials and methods

Seawater temperatures

Seawater temperature has been recorded *in situ* since September 1996 at Suva Barrier Reef (site #5, Table 1), and since July 1997 at Vuna Point, Taveuni (see Fig. 2) by the Fiji Seawater Temperature Monitoring Project at the University of the South Pacific, Fiji (http://www.usp.ac.fj/marine/gcrmn/research/seatemp.htm). Hugrun Seamon s/f underwater temperature recorders (with an accuracy of ± 0.05°C) record hourly at approx. 10m depth.

Seawater temperatures for 1985-2000 are from new, high resolution satellite data (Toscano et al. this volume). The NOAA/NASA AVHRR Oceans Pathfinder Program (Vazquez et.al. 1998) introduced an improved archival SST dataset to dataset provide a long (currently 17 years), consistently calibrated time series of global SST fields for climate studies (Kilpatrick et al. 2001). All AVHRR data from 1985 to present have been reprocessed using a Pathfinder version of the current NOAA Non-Linear SST algorithm (Kilpatrick et al. 2001). Pathfinder algorithm coefficients are estimated by regressing the remotely-sensed brightness temperatures to *in situ* best available moored and drifting buoy SSTs (matchups) within 30 min and 0.1° latitude/longitude of each other. Algorithm performance is globally well within a 0.5° C range, with the tropics showing a slight negative bias of 0.1-0.2° C compared to buoys (Kilpatrick et al. 2001). Thus the Pathfinder

**Table 1** Details of locations surveyed. Regions are ordered by latitude (refer to Fig. 2). Superscripts indicate surveyors, numbers match authors as listed on page 1. ‘LIT’ = line intercept transect.

| Region | Date | Location | Site Description | Replicate samples |
|---|---|---|---|---|
| | 7 April | **1A. Great Astrolabe Reef** [5]<br>Sites between:<br>$18^0$ 45.24’ S, $178^0$ 28.00’ E,<br>$18^0$ 43.31’ S, $178^0$ 29.19’ E. | Leeward (north-western) outer barrier. Dominated by *Acropora.* | 3 sets of 0.25 $m^2$ video photos (n=42, 36, 39) at 7 m. Colonies surveyed per set: 129, 111, 121. |
| **Kadavu** | 7-10 June | **1B. Great Astrolabe Reef** [7]<br>(1) $18^o$ 46.50’ S, $178^o$ 31.56’ E.<br>(3) $18^o$ 46.05’ S, $178^o$ 34.06’ E.<br>(4) $18^o$ 43.05’ S, $178^o$ 34.06’ E.<br>(5) $18^o$ 43.68’ S, $178^o$ 27.95’ E. | Five sites: (1),(2) lagoonal sea mounts; (3) windward outer barrier; (4),(5) leeward outer barrier. All dominated by *Acropora* except (4) which was the most diverse site. | 5 1 m belts, 33-69 m long, 1-27.5 m depth. Colonies surveyed per replicate: 150, 232, 249, 153, 113 respectively. |
| | 8 June | **2. North Astrolabe Reef** [7]<br>(1) $18^o$ 37.34’ S, $178^o$ 33.30’ E.<br>(2) $18^o$ 37.20’ S, $178^o$ 33.25’ E. | Barrier reef on north-east corner. Dominated by *Acropora.* | 2 1 m belts, 79 & 27 m long, 2-21.5 m. Colonies surveyed per replicate: 283, 68. |
| | 17 April | **3. Pacific Harbour** [4]<br>$18^o$ 17.51’ S, $178^o$ 4.84’ E. | Two large, lagoonal patch reefs one mile apart. Dominated by *Acropora*. | 2 30x1 m belts at 1.5-3.5 m. Colonies surveyed per replicate: 224, 155 |
| | 14 April | **4A. Beqa** [5] | Beqa Barrier Reef outer sides of channels. High coral cover, dominated by *Acropora* and *Pocillopora*. | 10 60 $m^2$ areas at 4-9 m. Colonies surveyed per replicate: 23, 52, 28, 25, 33, 46, 62,89, 53, 67. |
| **Southern Viti Levu** | 18, 20 April | **4B. Beqa** [4]<br>$18^o$ 27.57’ S, $178^o$ 6.05’ E.<br>$18^o$ 24.12’ S, $178^o$ 11.48’ E.<br>$18^o$ 28.97’ S, $177^o$ 56.02’ E. | Beqa Barrier Reef outer reef slope. High coral cover, dominated by *Acropora* and *Pocillopora.* | 3 30x1 m belts at 3-10 m. Colonies surveyed per replicate: 181, 227, 135. |
| | 19 April | **5. Suva Barrier Reef** [4,9]<br>$18^o$ 09.55’ S, $178^o$ 23.98’ E. | Outer entrance to Suva Harbour. Submerged spur extending seaward from a barrier reef. High wave surge, often turbid. Dominated by *Acropora* and *Pocillopora*. | 1 40 m LIT, 1 30x1 m belt, at 1.5-12 m. Colonies surveyed per replicate: 65, 24 |
| | 20 April | **6. Nukubuco Reef** [1] | Reef crest. High wave energy and strong currents, dominated by flattened *Acropora aspera*. | Two sites of tagged corals, at 0 m. Colonies surveyed per replicate: 21, 14. |
| | 16 April | **7. Vunavadra Island** [4]<br>$17^o$ 41.77’ S, $177^o$ 18.73’ E | Windward fringing reef (south-east facing) with moderate coral cover. | 2 30x1 m belts at 1.5-5 m. Colonies surveyed per replicate: 79, 39. |
| **Western Viti Levu** | 15 April | **8. Tavua Island** [4]<br>(1) $17^o$ 33.90’ S, $177^o$ 17.92’ E.<br>(2) $17^o$ 34.51’ S, $177^o$ 20.76’ E. | Two shallow patch reefs, 2 and 4 miles from Tavua Island: (1) south-east facing, 5 m, (2) top of reef, 2 m, with sparse corals. | (1) 1 10x1 m belt, (2) 1 30x1 m belt. Colonies surveyed per replicate: 21, 62. |
| | 11 June | **9. Navula Reef** [9]<br>(1) $17^0$ 55’ S, $177^0$ 12’ E.<br>(2) $17^0$ 56’ S, $177^0$ 12’ E. | Two sites on the northern (1) and southern (2) sides of a barrier reef passage. Outer barrier reef slope, strong currents, rich in soft corals and *Millepora*, with predominantly small scleractinian colonies. | 6 20 m LITs at 5-12 m (3 at each site). Colonies surveyed per replicate: 20, 25, 31, 18, 24, 35. |

**Table 1** cont’d

| Region | Date | Location | Site Description | Replicate samples |
|---|---|---|---|---|
| **Eastern Viti Levu** | 5-12 June | **10. Caqalai Island** [11]<br>6 sites between:<br>17° 47.25’ S, 178° 43.65’ E,<br>17° 47.60’ S, 178° 44.28’ E. | Shallow reef flats with low coral cover, dominated by *Pocillopora.* | 6 sets of 1x1 m quadrats (n=34, 35, 36, 36, 40, 36) at 3 m, 81 point counts per quadrat, % cover data. |
| | 1-8 June | **11. Wakaya Island** [9]<br>17° 35’ S, 178° 58’ E. | Outer corner of a barrier reef passage. Heavily impacted by crown-of-thorns predation in 1999/2000. Low coral cover, dominated by small colonies of *Pocillopora*. | 2 40 m LITs at 10 m, colonies surveyed per replicate: 24, 22. |
| **Vatu-I-Ra Passage** | 28 May-<br>2 June | **12. Pinnacle** [9]<br>17° 20’ S, 178° 32’ E. | Steep-sided pinnacle in the middle of a deep water passage. Dominated by large *Acropora* and *Porites.* | 2 20 m LITs at 7-18 m, colonies surveyed per replicate: 32, 36. |
| | 29 May-<br>5 June | **13. Vuya Reef** [9]<br>Vanua Levu Barrier Reef<br>17° 15’ S, 178° 34’ E. | Sheltered shallow reef floor, almost flat, behind a barrier reef. High coral cover, dominated by large *Acropora.* | 5 20 m LITs at 12 m, colonies surveyed per replicate: 40, 30, 34, 37, 26. |
| **Northern Lau Group** | 16 April | **14. Vanua Balavu** [9]<br>17° 20’ S, 179° 00’ W. | Southern reef crest inside lagoon. Dominated by *Acropora*. | 1 10 m LIT, 1 20 m LIT, at 1 m, colonies surveyed per replicate: 11, 26. |
| **Southern Vanua Levu** | 27 April-<br>1 May | **15. Yadua Tabu Island** [6,10] | Southern fringing reefs exposed to south-easterly trades. Data collected by volunteers. | 3 sites of 1x1 m quadrats (n=6, 15, 6), at 0-15 m, colonies surveyed per replicate: 61, 186, 92. |
| | 30 June-<br>2 July | **16. Savusavu Bay** [2]<br>Sites between:<br>16° 48.62’ S, $179^0$ 14.35’ E,<br>$16^0$ 50.01’ S, 179° 17.96’ E. | Windward reef slope, approx. 60% coral cover, dominated by *Acropora* and *Pocillopora.* | 6 20-75 m LITs at 3-6 m, colonies surveyed per replicate: 21, 19, 24, 21, 29, 58. |
| | 3-4 June | **17. Rainbow Reef** [9]<br>16° 46’ S, 179° 56’ E.<br>16° 45’ S, 179° 57’ E. | Patch reef in a deep water passage with strong currents. Dominated by soft corals, scleractinian corals dominated by small *Pocillopora.* | 3 20 m LITs at 9-15 m, colonies surveyed per replicate: 17, 15, 15. |
| | 2 June | **18. Honeymoon Island** [9]<br>16° 40’ S, 179° 51’ E. | Shallow fringing reef off a small inshore island. No particular dominants. | 2 20 m LITs at 1 m, colonies surveyed per replicate: 22, 12. |
| **Northern Vanua Levu** | 5-6 July | **19. Great Sea Reef** [3]<br>Sites between:<br>16° 19.75’ S, 179° 18.14’E,<br>16° 14.43’ S, 179° 02.10’E. | Shallow reef slopes on the outer and inner sides of a barrier reef. Approx. 60% coral cover, dominated by *Acropora hyacinthus* | 3 20 m LITs at 2-3 m, colonies surveyed per replicate: 37, 34, 32. |

NLSST algorithm is tuned to bulk SST measurements, closely (or slightly under-) estimating the bulk temperature felt by coral reef organisms.

Reef surveys

We standardized the reef survey data to colony counts for all locations except Caqalai (#10), and to four bleaching categories: (1) Not bleached (normal colouration), (2) Partially bleached (part of the colony white or pastel-coloured, often on the top only), (3) Fully bleached (whole colony white or pastel-coloured), (4) Bleached/dead (bleached colony with new algal turf on the skeleton). Categories (2), (3) and (4) were combined for the proportion of colonies affected by bleaching.

Aerial survey

Reefs between Suva and the Great Sea Reef north of Vanua Levu were surveyed by air on 21 April 2000 (Fig. 2). Reefs surveyed include barrier reefs around Suva and north to Ovalau Island, Vatu-I-Ra passage, Vanua Levu Barrier Reef, southern Vanua Levu, northern Vanua Levu and the Great Sea Reef.

We used photographs taken during the survey to categorize bleaching severity. Categories were: (1) <10%, (2) 10-30%, (3) 30-60%, (4) >60%, of substrate cover bleached. Two of the surveyed reefs (Suva {#5}and Nukubuco {#6}, both barrier reefs adjacent to Suva) were also assessed by SCUBA, allowing us to ground truth the aerial surveys for these sites.

## Results

Seawater temperature and the onset of bleaching

Heat stress was most intense in southern and eastern Fiji. By mid April 2000, south-eastern Fiji (encompassing Beqa, Suva, Kadavu and the Southern Lau Group), had accumulated 12-16 DHWs and south-western Fiji had accumulated 8-12 DHWs. In the north-west, DHWs were ≤4, suggesting a drastic difference in heat stress. The Northern Lau Group had 6-10 DHWs. North of Vanua Levu, 0 DHWs were recorded. These patterns can be seen in detail on the DHW website (Fig. 1).

The onset of mass bleaching was rapid, occurring over only a few days during the first week of March. Minor bleaching occurred up to two weeks earlier (Hendee 2000; Lovell 2001). By 1 February 2000, 2-3 DHWs had accumulated along the southern coast of Viti Levu, increasing to 5 DHWs by 29 February and 6 DHWs (1.5-2.0$^{o}$C HotSpots) in the first week of March when mass bleaching occurred.

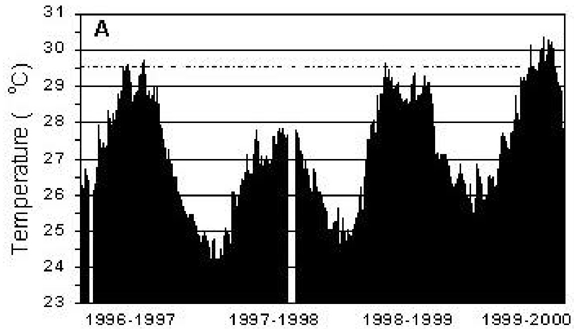
Neither HotSpots nor mass bleaching occurred in the far north (north of Vanua Levu), though this area did exhibit anomalies <1$^{o}$C above climatological maximum monthly mean SSTs (and some minor bleaching).

Seawater temperature recorded *in situ* at Suva Barrier Reef exceeded the maximum monthly mean (MMM: 28.5$^{o}$ C) for five months, remained above 29$^{o}$C for 3.5 months and peaked at 30-30.5$^{o}$C between early March and early April (Fig. 3a). Similar patterns and peak temperatures occurred at Vuna Point, Taveuni, though the MMM is 0.3$^{o}$C lower (Fig. 3b).

Seawater temperatures for 1985-2000 were constructed from both satellite-derived SSTs and *in situ* logger data, for Suva Barrier Reef and Vuna Point (Fig. 4). At Suva Barrier Reef, 2000 was the hottest year during this period, though 1989 was almost as hot. At Vuna Point, seawater temperature in 2000 was hotter than all other years except 1996.

Reef surveys

Eighteen of the 19 locations surveyed (Table 1) had bleached corals (Fig. 5). Bleaching was still widespread as late as July, four months after the onset of bleaching (Fig. 5). Sixty-four percent of all coral colonies surveyed were affected by bleaching, and were either partially or fully bleached, or recently dead from bleaching.



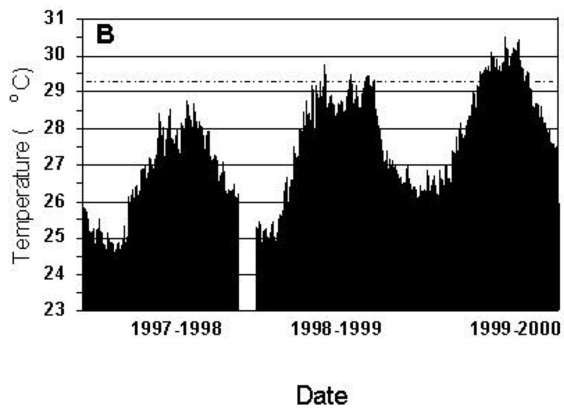




**Fig. 3** Seawater temperature measured hourly by *in situ* temperature loggers at approx 10 m depth. **A** Suva Barrier Reef. **B** Vuna Point, Taveuni.

In the southern survey regions (southern Viti Levu and Kadavu; Table 1) 82% of colonies surveyed were affected by bleaching, with 13% already dead. Site means ranged between 67% and 100% of colonies bleached, and Beqa Barrier Reef outer reef slope (#4B) had the highest proportion of dead colonies (26%). Severe bleaching was not restricted to southern Fiji, however. Vanua Balavu (#14) in the Northern Lau Group had 94% of colonies affected by bleaching, mostly fully bleached, and the highest proportion of bleaching-related mortality of all sites surveyed in April (32%). Bleaching also occurred in the Southern Lau Group (ND pers obs).

The western sites surveyed in April, Vunavadra (#7), Tavua (#8) and Yadua Tabu (#15), had less severe bleaching. Vunavadra had significantly less colonies affected than all other sites surveyed in April (p<0.001; one-way ANOVA, arcsin transform; SNK tests), with only 24% of colonies affected, no bleaching-related mortality and a relatively high proportion of partial bleaching (42% of colonies). Tavua and Yadua Tabu had significantly less bleaching than the most severely bleached sites: Beqa (#4), Vanua Balavu (#14) and Nukubuco (#6). Mass bleaching also occurred in the northern Yasawa Islands (T McLeod, Walt Smith International, pers comm).

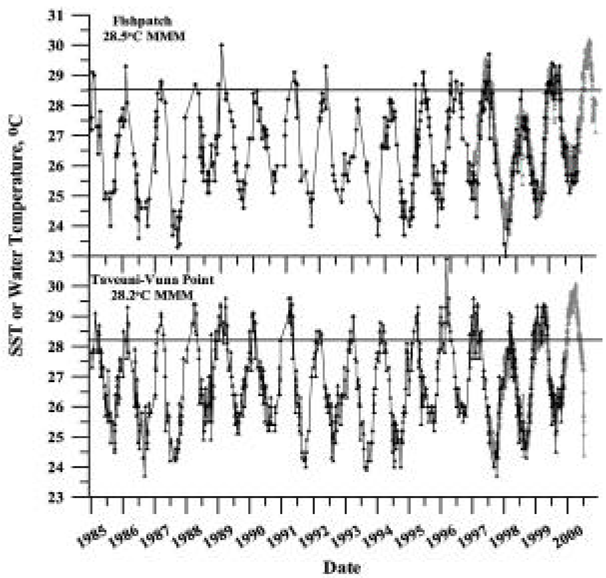



**Fig. 4** Fifteen-year temperature records, combining 9km Pathfinder SST data, for 1985-1999 (black), and average daily *in situ* logger data for 1996/97-2000 (grey).

At sites surveyed in April, most bleached colonies were fully bleached, with relatively few partially bleached and dead colonies. This was reverse in June-July, with most bleached colonies only partially bleached. Less colonies were affected in the June surveys than the April surveys (53% vs. 74%), reflecting probable recovery of some colonies. Nevertheless, >70% of colonies were still bleached in June at the two sites in Vatu-I-Ra Passage. Site means for all other sites surveyed in June ranged between 39% and 57%, except for Honeymoon (#18; 17%) which had significantly less bleaching-affected colonies than all other sites surveyed in June (p<0.001; one-way ANOVA, arcsin transform; SNK tests), and had no bleaching-related mortality. The two most northerly sites, Honeymoon and Great Sea Reef (#19), had the lowest incidence of bleaching of all sites. Great Sea Reef had no bleaching in early July, though minor bleaching was seen during the aerial survey in April.

Mortality from bleaching was low (<15%) at most sites in the April surveys and was higher in the later surveys (Fig. 5). The highest mortality was recorded at Savusavu (#16) and Vuya (#13), where more than 40% of the scleractinian corals had died from bleaching by June/July.

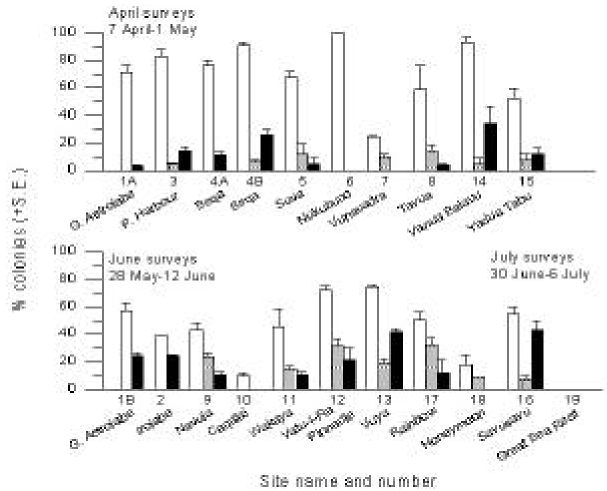



**Fig. 5** Percentage of scleractinian colonies affected by bleaching (partially and fully bleached and recently dead from bleaching; white), partially bleached (grey) and recently dead from bleaching (black). The percentage not bleached is the inverse of the white bar. Site descriptions are given in Table 1. Partial bleaching was not recorded at sites 1A, 1B, 2, 4A and 10. Recent mortality was not recorded at sites 6 and 10. Data from site 10 are % cover.

## Aerial survey

Extensive bleaching (>30% substrate cover bleached) occurred on all reef slopes surveyed south of Vanua Levu. We recorded 30-60% cover bleached through the Vatu-I-Ra Passage and 10-30% on a southern Vanua Levu fringing reef. Reef flats and crests were variable and often lower than 30%, but this may reflect low coral cover rather than low intensity of bleaching. Minor or no bleaching (<10% substrate cover bleached) was observed on all reefs north of Vanua Levu, including the Great Sea Reef.

Suva (#5) and Nukubuco (#6) outer reef slopes were the only reefs for which we recorded >60% cover bleached. Our estimate was lower for the adjacent reef crests (30-60%). The difference reflects the lower coral cover of the reef crests (RC, EL, unpubl data); Nukubuco crest had severe bleaching, with 100% colonies bleached in the two sites surveyed (Fig. 5).

## Discussion

Geographic patterns of bleaching

Our data demonstrate that mass bleaching was widespread in Fiji south of Vanua Levu, and bleaching was only minor north of Vanua Levu. This geographic pattern coincides with Fiji's position on the north-western edge of an area of high heat stress (idenitfied by satellite-derived SST data), with accumulated heat stress greatest in the south and east of the country. More than 40% of scleractinian coral colonies were affected by bleaching at 16 of the 19 sites surveyed, including all seven geographic regions surveyed south of Vanua Levu.

The most severe bleaching (>80% of colonies affected) was recorded in April 2000 in the southern Viti Levu and Kadavu regions, and at a single reef crest site in the Northern Lau Group. The western sites surveyed in April (Vunavadra, Tavua and Yadua Tabu) had significantly less bleaching than the most severely bleached sites (Beqa, Vanua Balavu and Nukubuco), and the second most northerly site (Honeymoon) had significantly less bleaching than all other sites surveyed in June. These trends coincide with the accumulation of heat stress in the south and east. Several sites had low replication and/or small numbers of colonies per replicate, however, and these trends could also be explained by the smaller-scale, within-reef spatial variation in bleaching intensity which is known to occur (Spencer et al. 2000; Marshall and Baird 2000).

The two most severely bleached sites surveyed were both barrier reef crests (Nukubuco and Vanua Balavu). The adjacent reef flat of Nukubuco Reef was much less severely bleached, due at least in part to the predominance of massive and branching *Porites* spp. which were partially bleached or not bleached (RC unpubl data). The third reef crest site (Honeymoon) had much less severe bleaching (than other sites surveyed in June), however, suggesting variable responses on reef crests, though Honeymoon may have been less affected by bleaching because of its northern location.

Links with seawater temperature

Though no previous mass bleaching event is on record for Fiji, minor bleaching (involving a small proportion of colonies and/or a high proportion of partial bleaching) occurred in 1998 and 1999. In 1999, DHWs around south-eastern Viti Levu reached 7-8 (February–April 1999), and prominent bleaching of *Acropora* and *Platygyra* colonies occurred in Suva Harbour. Widespread bleaching occurred again in 2001, and was minor at most sites.

Since minor bleaching occurred in both 1998 and 1999 and major bleaching occurred in 2000, we can use Fig. 4 to estimate that the bleaching threshold lies in the range of 29.5-30$^{o}$C at both Suva Barrier Reef and Vuna Point, Taveuni. This range largely agrees with the bleaching thresholds used for HotSpot mapping of 1$^{o}$C above MMM. For both sites, 2000 is one of the two hottest years since 1985. The other hot year was 1996 at Taveuni and 1989 at Suva, suggesting the north and south of the country are influenced by different hot water masses. In 1996, bleaching occurred in PNG (Davies et al. 1997) and Williams and Bunkley-Williams (1990) provide an anecdotal report of bleaching in southern Fiji in 1989.

Mass bleaching occurred in western and northern areas that experienced thermal anomalies of <1$^{o}$C (Fig. 1). Our data therefore add to accumulating evidence that elevated seawater temperatures <1$^{o}$ C above MMM can also lead to mass bleaching (e.g. Toscano et al. this volume; Goreau et al. 2000).

Mortality from the 2000 bleaching event

Bleaching mortality, estimated as the proportion of bleached colonies that were dead at the time of surveying, was low (<15%) at most sites in the April surveys and reached >40% (Savusavu {#16} and Vuya {#13}) in the later surveys. Mortality recorded in the later surveys may represent a reasonable estimate of ultimate mortality from bleaching because few colonies remained fully bleached (Fig. 5) and recovery of the remaining partially bleached colonies could have been high. On the other hand, the proportion of colonies affected by bleaching was probably underestimated in our later surveys because some bleached colonies recover their colour within four months (Lang et al. 1992; McField 1999; RC unpubl data).

Several of our sites are being monitored regularly as part of the incipient Global Coral Reef Monitoring Network in Fiji (http://www.usp.ac.fj/marine/gcrmn/). Preliminary estimates of impact from post-bleaching surveys and qualitative observations provide the following. By December 2000, <10% of coral cover was lost at both coral-depauperate Wakaya (#11) and coral-rich, *Acropora*-dominated Vatu-I-Ra Pinnacle (#12) (HS). Approximately 40% of coral cover was lost at Vuya (#13), which was also *Acropora*-dominated (HS). Possibly, the steep sides of the pinnacle reduced direct sunlight on the corals, compared with the flat Vuya site.

Beqa Barrier Reef outer reef slope and Pacific Harbour lagoonal patch reefs lost 99% and 80% of *Acropora* colonies respectively between April 2000 and April 2001 (BC). Much of this loss was probably due to the 2000 bleaching event, though this time period also includes a further bleaching event in 2001 and Cyclone

Paula in early March 2001, which damaged southern and south-western reefs.

Suva Barrier Reef outer reef slope (#5), which was *Acropora*-dominated and severely bleached, lost approximately 30% of coral cover and 45% of coral colonies by January 2001 (RC, EL). Nukubuco Reef crest (#6), which was also severely bleached, lost 65% of the dominant *Acropora aspera* colonies by August 2000 (RC).

In the west, no mortality of *Acropora* was detected at Vunavadra (#7), where bleaching was significantly less than at all other sites surveyed in April, and substantial mortality of *Acropora* colonies (47%) occurred at Tavua (#8) (BC).

The most severe bleaching recorded in Fiji (>80% of colonies affected) was of similar intensity to the worst-hit sites in the Indian Ocean in 1998. Mortality of 70-99% of coral cover was recorded at many of these sites (Goreau et al. 2000; Wilkinson 2000).

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# An Investigation of the Effects of Increasing Fishing Efficiency on the Productivity of the Queen Conch (*Strombus gigas*) and Caribbean Spiny Lobster (*Panulirus argus*) Fisheries within the Turks and Caicos Islands

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ABSTRACT

Fishery management decisions throughout the world are often based on incomplete data, fortunately historical landing data exists for the Turks and Caicos Islands (TCI) dating back to 1887 (Sadler 1997). The fishery based upon Queen Conch (*Strombus gigas*) and the Caribbean Spiny Lobster (*Panulirus argus*) is typical of many fisheries because it is characterized by numerous increases, a leveling off and then a decline in catch landings. Lobster catch landings have gradually increased over the past century, from 90,700kg of whole lobster in 1947 to a maximum of 590,758 kg in 1992. Likewise, conch landings have increased from 117,550 kg in 1968 to the present total allowable catch (TAC) value of 736,960 kg.

Fishermen within the TCI have increased their fishing efficiency by acquired knowledge of the best fishing grounds as well as improving their fishing skills and developing new fishing techniques and technologies. The fishery within the TCI has transformed from one dominated by wind powered sailing sloops in the early 1900s to a fishery dominated by fishermen using 85-115 hp (three and four cylinder) engines with their fishing boats. Simultaneously, fishermen have switched from utilizing glass buckets to locate conch and lobster to free diving using mask, fins and snorkel. In the lobster fishery where nooses were used in the past, hooks are the main method now in use, however prohibited substances such as bleach are also used.

Several input and output mechanisms are being considered to manage the fishery. However, many of these mechanisms such as the maximum sustainable yield (MSY) are derived using catch per unit of effort (CPUE) data. Increases in fishing efficiency over time result in increased effective effort, although the nominal/apparent effort may seem to be unchanged. Hence, the CPUE may render misleading information on the economic and biological status of the fishery. To combat such, it is imperative that effort is standardized, and is reflected in local legislation to control technology creep.

KEY WORDS: Effective Fishing Effort, nominal/apparent fishing effort, technology creep

# Investigación sobre los Efectos del Incrementos de la Productividad y la Rentabilidad en las Pesquerías de Caracol y Langosta en las Islas Turcos y Caicos

La administración de pesquerías en el mundo, se encuentra a menudo basada en informaciones incompletas. Afortunadamente, en las Islas Turcos y Caicos, existe información que data de 1904. La pesquería sobre caracol (*Strombus gigas*) y langosta espinosa (*Panulirus argus*) es la típica de muchas pesquerías, pues los desembarques se caracterizan por numerosos incrementos, estabilización y declinación. Para la langosta, las capturas tuvieron un incremento gradual durante el pasado siglo, desde 200 000 lbs. de langosta entera en 1947 a un máximo de 690 846 lbs. en 1998. De igual forma el caracol, aumento de 259 191 lbs. en 1968 a una captura permisible (TAC) actual de 1.6 millones de lbs.

Los pescadores de TCI han incrementado su eficiencia de pesca a través de las prácticas adquiridas en las mejores áreas de pesca así como mejorando sus conocimientos pesqueros y desarrollando nuevas técnicas y tecnologías. La pesca en TCI ha sido transformada desde un predominio de embarcaciones veleras en los inicios de los 1900 a una pesquería dominada por pescadores que utilizan motores de tres y cuatro cilindros (85-115 hp) en sus botes. Simultáneamente, los pescadores han cambiado el mira-fondo de vidrio para localizar la langosta y el caracol por la careta de buceo con snorkel y las patas de rana. En la pesquería de langosta donde el método principal fue el lazo en el pasado, el gancho es ahora el predominante para colectar la especie, sin embargo sustancias prohibidas como el peróxido son también usadas.

Diversos mecanismos de entrada y salida son actualmente utilizados para el manejo de la pesquería. Sin embargo, muchos de estos mecanismos como es el caso de el TAC para caracol, se derivaron de los datos de captura por unidad de esfuerzo (CPUE). El incremento de la eficiencia de pesca a través del tiempo resultó en un incremento efectivo del esfuerzo, no obstante el esfuerzo aparentemente parece no haber cambiado. Para enfrentar esto, es imprescindible que sea estandarizado el esfuerzo, y que esto se vea reflejado en una legislación local para controlar los cambios tecnológicos cambiantes.

PALABRAS CLAVES: Efectos del incrementos de la productividad y la rentabilidad, caracol, langosta

## INTRODUCTION

The Turks and Caicos Islands are an archipelagic overseas territory of the United Kingdom located in the British West Indies (Figure 1). Scale fish are caught for subsistence usage as a by-product of the principle commercial fisheries of lobster and conch.

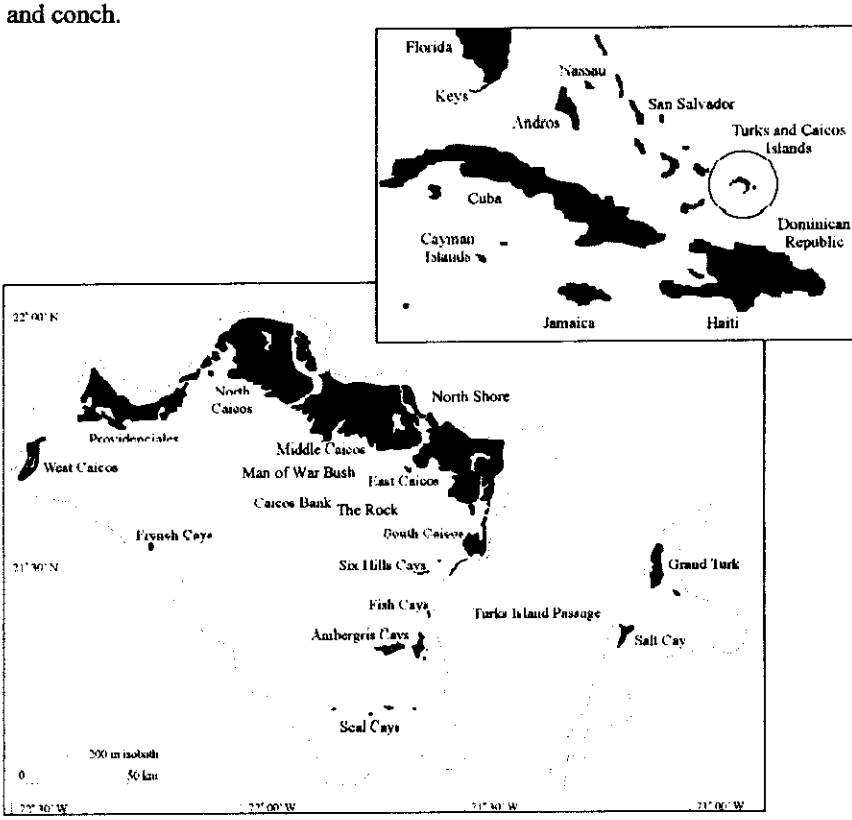


**Figure 1.** The location of the Turks and Caicos Islands in the British West Indies and the main fishing areas referred to in Table 2 later in this paper.

Effective management of any fishery that utilizes the MSY concept to predict fish stocks relies heavily upon often very basic input information to calculate yields through the use of surplus yield models. Such information is often merely catch and effort data (King 1995). It is well established that effort exerted within a fishery increases overtime for a number of reasons and that this 'technological creep' can mask detrimental changes in fish stocks (Pascoe1996, Cunningham 1980). It is this masking of increasing fishing effort within the TCI fisheries that is of interest and concern to those managing this fishery.

Technology creep occurs where fishermen increase their efficiency in catching fish through improvements in fishing gear over time. Unfortunately, technology creep is not reflected in the calculation of effort in most surplus production models. Increased efficiency of fishermen can occur through (Seijo 1998, Pascoe 1996, King 1995, Berg 1989):

i) The utilization and development of different fishing methods or gear,
ii) Changing the method of vessel propulsion thereby enabling fishermen to visit areas with more abundant fish stocks (generally those furthest away from fishing centers),
iii) Increased local knowledge regarding the location of fish stocks, and
iv) Increased skippers' skill.

When referring to fishing effort, it is wise to discern whether or not effective fishing effort or nominal/apparent fishing effort is being referenced. The difference, although appearing slight, is of utmost importance when determining if effort is increasing within a fishery. Eeffective fishing effort refers to the ability of a fisherman to remove a proportion of a mean population size, whereas nominal/apparent fishing effort reflects only the volume and not the effectiveness of resources devoted to removing a proportion of a fish population (Berg 1989, Cunningham 1980). For example, two fishermen in a five meter boat powered by a 150 hp outboard engine and utilizing hooks are far more effective at catching lobster than two fishermen in a four meter sail powered boat utilizing nooses. In this example it is clear that although the nominal fishing effort is the same (i.e. two fishermen and one boat) the effective fishing efforts exerted are disproportionate.

In fisheries where more than one species is fished at a time, effective effort exerted on a particular species can increase even though nominal effort remains unaltered as a result of a decline in a species abundance, forcing fishermen to fish other species, or the price returned on a particular species falls to such a level that it is economically unattractive to fish (Bēnē 2001). Effective effort has increased in the TCI conch fishery even though nominal effort has remained the same, because fishermen are now spending a larger proportion of their time fishing conch over lobster per day due to declining lobster stocks (Berg 1989, Olsen 1985). A TAC, based on MSY calculations and landing data, is currently used to manage the conch stocks within the TCI. It is believed that the TAC derived using a Schaefer biomass dynamic model (Medley 1999) is maintaining stable stocks. However, it is widely accepted that stock estimates derived from catch and effort data are inherently biased, since effort data is not adjusted to take into account changes in efficiency over time (Pascoe 1996). Nevertheless, the slight changes in efficiency over time within the conch industry do not seem to affect the results attained by the fishery dependent model, because the MSY has been corroborated by a recent visual stock assessment (Clerveaux and Danylchuk in press).

However, the lobster fishery is not managed by restrictions or limitations, and as such concerns are being aired by managers of this fishery. Their view is that lobster stocks are prone to overexploitation in the near future, based on anecdotal evidence that the fishery is in decline and effective effort is increasing (e.g. changes in gear), although at present this is not reflected by decreased landings. In effect, fishermen are maintaining landings whilst masking potential overexploitation.

Historical landing data, in conjunction with effort records and interviews with active and retired fishermen, provide an insight to the fishing industry. Reliable data and historical records relating to the fishery will help to determine whether several periods of uncharacteristic landing figures are symptomatic of changes in effective fishing effort within the fishery or other variables such as environmental factors (e.g. water temperature and currents).

**The Development of Gear Type and Efficiency**

Pre-1958 wooden sailing sloops 10 - 12 m in length and drawing 1.5 - 2 m of water constructed of driftwood (Doran 1958) powered by sail were used to fish the conch and lobster grounds for up to a week. Each sloop acted as a mothership for two to four small wooden, oar-powered dinghies. The dinghies were 3 - 4 m dug outs sculled by 3 m oars worked in shallow oarlocks off center on the portside of the transom. Glass buckets (water glasses) were used to first identify conch and lobster by a bowman who carefully worked an 8 m long conch hook underneath the conch and then quickly lifted the animal to the surface (Berg 1989, Doran 1958). Lobsters, on the other hand, were 'bullied' using a small net on a pole, which was placed over the lobster and scooped into the small boat.

The fishery gradually transformed from non-diving to diving with the introduction of mask, fins and snorkel in the mid. 1950s. By the end of the 1950s free diving had increased in prevalence, although conch hooks remained in use until the mid. 1970s. As alternatives to the bully, the 'toss' (a flexible wire noose on a stick) and the 'grabber' (a spring-loaded snake catcher) were introduced to catch lobster. Diving also opened up previously unexploited areas of the fishery as it enabled deeper waters to be fished (Ninnes 1994, Olsen 1985). The grabber was quickly abandoned as it damaged the lobster, thereby resulting in a reduced economic return. These two methods of fishing were quickly replaced by a more efficient method of hooking (a shark hook attached onto a 1.5 m long flexible pole), which allowed fishermen to capture lobsters from within their dens where they aggregate during the day (Bënë 2001).

Free diving from fiberglass boats for conch and lobster aided by the use of the hook increased until it was the norm by the early 1980s (Table 1) (Nardi 1982). So far hooka and scuba which confer high fishing efficiencies, have not been used in the TCI. Their introduction is thought unlikely due to ease of enforcing these currently illegal methods of fishing.

Although changes in nominal fishing are more apparent than changes in effective effort, they have far less impact on the exploitation of a fish stock because of the law of diminishing returns (Pascoe 1996). Changes in gear on the other hand (from the use of a bully to a hook whilst fishing for lobster) and techniques (from non-diving to free diving) are believed to have increased fishermen's effectiveness to exploit the fishery.

Bënë (2001) noted that the shift to hooking is displayed by two distinct bionomic equilibrium's, resulting from two successive switches from conch to lobster and then from lobster to conch, which followed an almost perfect Cobb-Douglas indifference curve (Figure 2).

**Table 1.** The transition of gear in the Caribbean spiny lobster (*Panulirus argus)* and Queen Conch (*Strombus gigas)* fishery of the Turks and Caicos Islands since 1940.

| Year | Prevalent Type of Fishing | Prevalent Gear Type | |
|---|---|---|---|
| | | Lobster Fishery | Conch Fishery |
| 1940s | Non-diving | Bully | Conch Hook |
| 1956 | Diving | Toss & Grabber | Hand Capture |
| Late 1950s | Diving | Toss & Hook | Hand Capture |
| 1960s | Diving | Toss & Hook | Hand Capture |
| 1970s | Diving | Hook | Hand Capture |
| 1980s | Diving | Hook (Bleach) | Hand Capture |
| 1990s - present | Diving | Hook (Bleach) | Hand Capture |

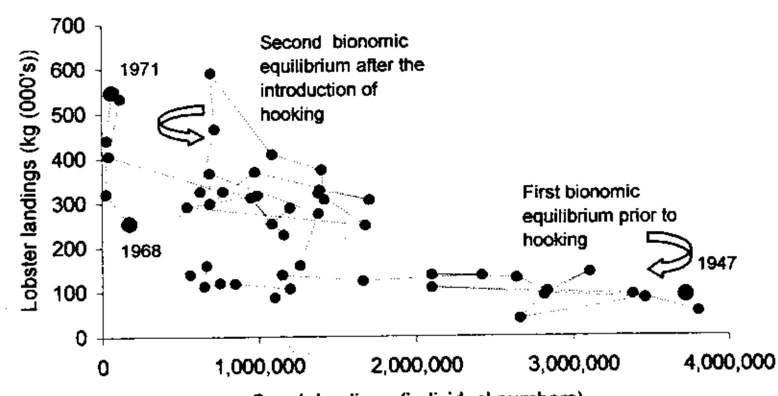


**Figure 2.** Phase diagram of the Caribbean Spiny Lobster (*Panulirus argus*) and Queen Conch (*Strombus gigas*) fisheries of the Turks and Caicos Islands showing two distinct bionomic phases before and after the introduction of the hook as a lobster fishing gear.

**Development of Fishing Locations Over Time**

Stocks are generally not evenly distributed along the seabed, and in many cases accessible areas of higher abundance are fished first to maximize return on investment (e.g. time, money) (Coppola 1996). Coppola also pointed out that fishermen have to extend their search for fish once areas of known abundance have been exploited to their full potential, venturing greater distances from port as a fishery is developed. Over time, fishing areas in the TCI have changed to reflect attempts by fishermen to increase or maintain their catches. Distant sites were explored and developed (Table 2). Until the introduction of high horsepower engines (> 40hp) in the mid. 1970s, fishermen, (in particular lobster fishermen) were restricted to fishing in close proximity to port (8 - 15 km) so as return the same day and land a high quality product. Conch fishermen, on the other hand, at times ventured further (e.g. Ambergris Cay from South Caicos) spending several days fishing and drying conch (Hesse 1977, Doran 1952).

**Table 2.** Fishing site development over time in the Turks and Caicos Islands Caribbean spiny lobster (*Panulirus argus*) and queen conch (*Strombus gigas*) fisheries.

| | Six Hills Cay | Ambergris | Caicos Bank and the Rock | Southern Ambergris (Bush Cay) | Southern Seal Cays | Man Of War Bush | North Shore |
|---|---|---|---|---|---|---|---|
| Pre. 1970 | STB/ D | U | U | STB | STB | U | U |
| Mid. 1970 | D | LTB | U | LTB | STB | D | D |
| 1980 | D | D | D | LTB/ D | D | $D_{DS}$ | LTB |
| Mid. 1980 - present day | D | D | D | LTB/ D | LTB/ D | $D_{DS}$ | D |
| Distance from S. Caicos fishing community | 8km | 23km | 31km | 34km | 37km | 40km | 40km |

Key: D, Divers; U, Undeveloped; LTB, Large Trap Boats; STB, Small Trap Boats; $D_{DS}$, Diving still dominant but stocks are relatively depleted
Small trap boat; < 9m in length, 2 fishermen outboard engine
Large trap boat; > 9m in length, > 2 fishermen inboard engine
Divers; small boat, < 6m in length, 2–3 commercial fishermen

Since the introduction of outboard motor engines for commercial fishing vessels in the early 1950s, engine size and power have increased steadily over the years. By the middle of 1970, the sail powered sloops; seagull and low powered wooden dugout boats were replaced by vessels with fiberglass hulls with two and three cylinder engines, thereby allowing fishermen to venture further, quicker (Table 3). The increase in fishermen's fishing range and ability to carry large amounts of product increased effective fishing effort further allowing effective exploitation of the stocks. Apparent effort changed very little over time period of time.

**Table 3.** Fishermen of the Turks and Caicos Islands have increased their effectiveness over time at exploiting the fish stocks by changing hull composition and increasing engine size, allowing increased load capacity, speed and range.

| Year | Prevalent Propulsion method | Hull material |
|---|---|---|
| 1940 | Sail/oars | Driftwood, dug out canoes |
| 1950 | 2.5-6hp Seagull engine introduced | Locally made wooden boats, first imported fiberglass hull |
| 1960 | 18hp | Wood/Fibreglass |
| 1970 | 40hp | Wood/Fibreglass |
| 1975 | 55hp | Wood/Fibreglass |
| 1985 | 65-70hp | Wood/Fibreglass |
| 1990 | 70-75hp | Wood/Fibreglass |
| 1995 | 85-90hp | Wood/Fibreglass/buoyant foam |
| 2000 | 85-90hp (lobster fishing) | Wood/Fibreglass/buoyant foam |
| 2000 | 105-200hp (conch fishing) | Wood/Fibreglass/buoyant foam |

Source: Department of Environment and Coastal Resources commercial fishing vessel licence application forms, historical records and interviews with commercial fishermen.

**Effect of Increasing Fishing Efficiency**

A simple surplus production model (Fox) was used to model the effects of increasing fishing efficiency using lobster catch and effort data from the mid. 1960s encompassing both pre- and post-hooking years (1966 - 2000) whilst also taking into account the changes in distance traveled from fishing centres over time. The data was then compared to the years when hooking was not as prevalent and there were no significant changes in fishing location (1966 - 1980).

The results indicate that a MSY of 491,650 kg would be obtained at an apparent effort level of 25,000 man-days, had effective fishing effort remained constant, as in pre-prevalent hooking years (1966 - 1980). However, an increase in effective effort causes a shift in the Fox model with the MSY rising more steeply and reaching a lower MSY value of 354,576 kg at 20,000 man-days of apparent effort (Figure 3). The ratio of fishermen to a fishing vessel was also investigated however there was no significant change in this ratio as the fishery developed. The implication here is that this potential area of increasing effective effort is actually static.

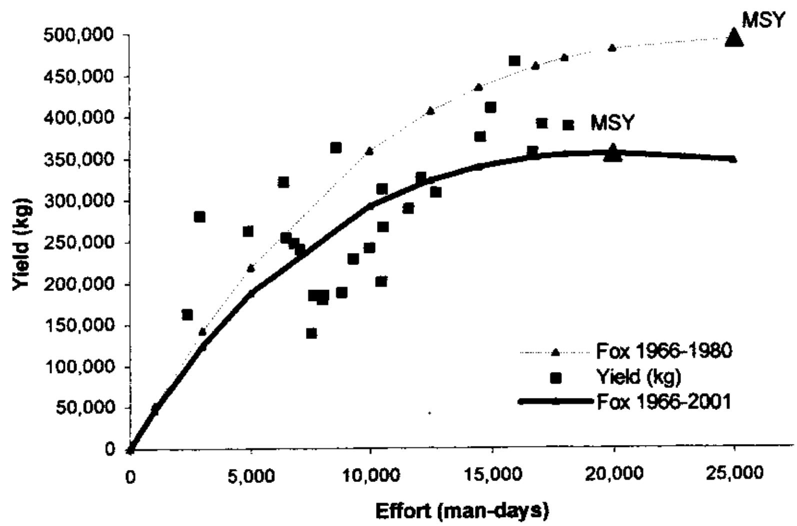


**Figure 3.** Maximum sustainable yield plot from a Fox surplus production model for the Turks and Caicos Islands Caribbean spiny lobster (*Panulirus argus*) fishery illustrating how a change in gear (the introduction of the hook in 1980) results in a decrease in the maximum sustainable yield which could also be attained by a reduced nominal effort (man - days).

## DISCUSSION

Despite concerns which has been raised about surplus production models overestimating stocks and their unpredictability to identify potential changes of the catchability of inputs (Cunningham, 1980), surplus production models are commonly used as a management tool in many fisheries throughout the world (Coppola 1996; Sparre 1992). The reasoning for this is that in contrast to most analytical models, the data requirements are less demanding and can be met with reasonable yield and effort estimates/values being attained over several years (Sparre 1992).

An underlying assumption in the short run catch-effort relationship is that catch in one year is a linear function of effective fishing effort (Coppola 1996, Cunningham 1980) given by:

$$C = qEB$$

where $C$ is the catch, $q$ is the catchability coefficient, $E$ is effort and $B$ is the stock biomass.

A change in gear has the potential to increase both apparent and effective effort exerted upon a stock (Figure 2). The two binomial equilibrium observed in the fishery is a result of an increase in effective fishing effort (e.g. hooking), that led to an increase in apparent effort (e.g. man-days). A shift to free-diving and hooking in the late 1950s and early 1960s increased the effectiveness of fishermen to catch lobsters, hence increasing profit and attracting more fishermen into the lobster industry. As the lobster stocks became depleted, fishermen diverted their effort to fishing conch, thereby producing the second bionomic equilibrium.

Furthermore, the change in fishing gear increased the level of the catchability coefficient ($q$) thereby increasing fisherman's effectiveness at exploiting the lobster stocks. However, lobster landings remained higher in the second bionomic equilibrium than previously, although apparent effort (man-days) declined to previous levels. This phenomenon may provide misleading CPUE information (i.e. abundance of the stock is high). The short-term benefit was the maintenance of high landings at a similar apparent effort level, but larger long-term MSYs were sacrificed.

There has been an increasing trend in the use of reference points for fisheries management, particularly with those that maximize yield (Caddy 1995) (e.g. effort at the MSY ($E_{msy}$)). Increasingly, the use of fishing effort as a reference point would imply that at some point effort would be curbed at the reference level to provide optimum benefit to the resource users. One fishery management tool widely used to achieve this objective (effort reduction) is by exclusion (e.g. limiting the number of individuals allowed to enter the fishery).

The success of limited licence entry programs on a global scale has been quite variable (Austin 1986). Limiting the number of licensed fishermen itself may not prevent biological overfishing, as pointed out by Austin (1986), because only the apparent effort has been managed, and this in fact may have negligible impact on stock protection and management. Nevertheless, if used in conjunction with attempts to limit effective effort such as technology creep, limiting the number of licenses can assist in achieving biological and economic objectives.

It has been suggested that nominal or apparent fishing effort will only be a true reflection of fishing mortality so long as the catchability coefficient remains constant (Cunningham 1980):

$$F = qf$$

where $F$, is fishing mortality, $q$ is catchability coefficient and $f$ is nominal/apparent fishing effort. However, the catchability coefficient does not remain constant due to changes in efficiency, temporal and spatial scales (Tewfik 2000). As such, nominal/apparent effort would need to be periodically standardized and factored to take into account increasing effective effort within the fishery.

## CONCLUSION

Effective and nominal efforts within the TCI fishery are significantly different. Technology creep has over time increased the effectiveness of exploiting the fisheries (e.g. the lobster stocks by shifting the MSY to a lower level of equilibrium). The creep in effective fishing effort caused by changes in gear requires careful management to prevent over exploitation.

It is imperative that fishery managers have a good understanding of the levels of effective and nominal effort being exerted in their fisheries in order to choose the most appropriate manner in which to manage the fishery. If this is not the case, ineffective effort reduction schemes will do little to protect either the economic or biological sustainability of the fishery and may in actual fact be expensive and detrimental.

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# A Description of Fisheries Management in the Turks and Caicos Islands: An Overview of the Problems and Suggestions for Mitigation

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ABSTRACT

Fisheries management is a highly complex and emotive issue in virtually every country in the world. This paper uses the Turks and Caicos Islands and its fisheries as a case study to highlight some of the issues that face fisheries managers though out the Caribbean. The Turks and Caicos Islands fisheries are dominated by free divers collecting Caribbean Spiny Lobster (*Panularis argus*) and Queen Conch (*Strombus gigas*), additionally there are several boats deploying lobster pots and fish traps. There is no significant commercial pressure on the scale-fish stocks, which make the TCI almost unique in the Caribbean, and there are only several commercial game fishing operations at present within the TCI. Overfishing of the lobster and conch populations is a major concern of the Department of Environment and Coastal Resources (DECR) tasked with managing the natural resources of the TCI.

In order for the DECR to meet these challenges, the implementation of several measure are key. These include:

i) The removal of lobster from restaurant and hotel menus during the closed season;
ii) A reduction in the number of foreign fishermen;
iii) Banning of the hook used to catch crawfish;
iv) Development of a method to detect the use of noxious substances used to catch crawfish.

In order for these actions to be implemented, the DECR requires strong political support which needs to be fostered through a comprehensive environmental education program aimed at policy makers, fishermen and the general public. This is especially important because rapid development of the TCI is stretching the resources and capability of the DECR to fulfill its requirements. The rapid growth of the TCI's economy is worrisome for the environment as it is easy to lose sight of the value and importance of the environment both economically in a country where the tag line is "beautiful by nature" and socially where entire communities are dependent upon the viability of the marine resources for there livelihood. Development needs to occur but at a pace where appropriate importance is placed on the natural environment. The TCI is at a crossroads right now because its fisheries are currently in decline but are still among the healthiest in the Caribbean.

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If the TCI does not learn from other Caribbean nations that have already been through a similar rapid development and economic growth phase, then the future of the fisheries resources looks bleak.

KEY WORDS: Conch, fishermen, fisheries management, lobster

## Una Descripción del Manejo de las Industrias Pesqueras en Islas Turcos y las Islas de Caicos: Una Descripción de los Problemas y de las Sugerencias para la Mitigación

La Administración de las industrias pesqueras es un asunto altamente complejo y muy emotivo en virtualmente cualquier país del mundo. Este trabajo utiliza a las islas turcos y las islas de Caicos (TCI)y sus industrias pesqueras como caso de estudio para destacar algunas de los asuntos que enfrentan los administradores de los recursos pesqueros a través del Caribe. Las pesquerías de las islas Turcos y Caicos esta dominada por buceadores a pulmón libre los cuales captura la langosta espinosa del Caribe (*Panularis argus*) y el caracol reina ( *Strombus gigas*), y además hay varios barcos que pescan con nasas de peces y langosta. No existe una presión comercial significativa en la pesquería de peces lo que hacen al TCI un caso único en el Caribe. Sin embargo la sobre pesca de las poblaciones de la langosta y del caracol es una preocupación muy grande para el Departamento del Ambiente y de los Recursos Costeros (DECR) encargados del manejo de los recursos naturales del TCI.

En la orden para que el DECR resuelva estos desafíos, la puesta en práctica de varias medidas es fundamental. Éstos incluyen:

i) El retiro de la langosta de menús de restaurantes y del hoteles durante la temporada de veda;

ii) Una reducción en el número de pescadores extranjeros;

iii) La prohibición del anzuelo de capturar langosta;

iv) El desarrollo de un método para detectar el uso de sustancias nocivas para capturar langosta.

Para que estas practicas puedan ser implementadas el DECR requiere de una fuerte ayuda política que debe ser fomentado a través de un programa de educación ambiental comprensivo dirigido a políticos, pescadores y al publico en general. Esto es especialmente importante porque el desarrollo rápido del TCI está estirando los recursos y la capacidad del DECR para cumplir con sus obligaciones. El crecimiento rápido de la economía de TCI es preocupante para el ambiente pues es fácil perder de vista el valor y la importancia del ambiente desde el punto de vista económico y social, especialmente en un país donde su emblema es "bello por naturaleza" y que depende de los recursos marinos para su sustento. El desarrollo necesita ocurrir pero a un paso donde se le de la importancia apropiada al medio ambiente natural. El TCI está en una encrucijada ahora porque sus industrias pesqueras están actualmente en declinación pero todavía está entre los más sano del Caribe. Si el TCI no aprende de

otras naciones del Caribe que han pasado por una fase similar de rápido desarrollo y crecimiento económico, entonces el futuro de los recursos pesqueros de nuestro país parece muy oscuro.

PALABRAS CLAVES: Langosta, caracol, pesquerias, manejo de pesquerias

## INTRODUCTION

The Turks and Caicos Islands are an archipelagic overseas territory of the United Kingdom located in the British West Indies (Figure 1) (Sadler 1997). Scale-fish are caught for subsistence usage as a by-product of the principle commercial fisheries of lobster and conch (Clerveaux 2001c).

The effective management of any fishery relies upon the application of restrictions based on the life histories, location and abundance of target species and considerations regarding the number and efficiency of fishermen operating in the fishery. An understanding of the political, social and economic environment that the fishery operates in is also important (Jennings 2001, King 1995). This paper uses the fishery of the Turks and Caicos Islands to highlight some of the problems facing fisheries managers and suggests some solutions that may be acted upon to ensure the continued existence of a viable fishery in the TCI.

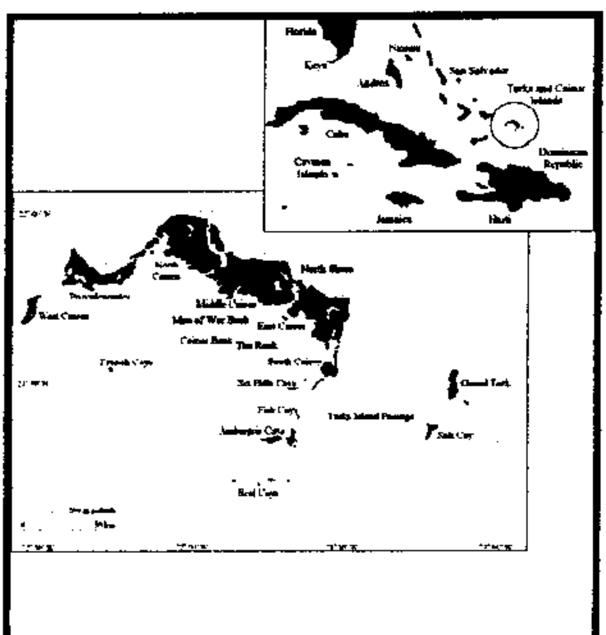

**Figure 1.** The location of the Turks and Caicos Islands in the British West Indies

RESPONSIBILITY OF THE FISHERY

The Ministry for Natural Resources controls the management of the fisheries by delegating day to day responsibility to the Department of Environment and Coastal Resources, encompassing the Fisheries Department (FD) and the Protected Areas Department (PAD). There are three DECR offices throughout the islands (Grand Turk, South Caicos, and Providenciales) at which Fisheries Officers operate and a National Environmental Center on Providenciales at which the PAD is based. The Director of the DECR is based on Grand Turk (the governmental administrative center) whereas the Chief Fisheries Officer is based on South Caicos (the fishing capital) (Bennett 2000). Both Fisheries Officers and Protected Area Officers have extensive law enforcement powers. Police Officers and members of Her Majesties Navy are also conferred the powers of Fisheries Officers. There is a Marine Branch of the Royal Turks and Caicos Islands Police Force (RTCIPF) that assists in Fisheries Enforcement with personnel (armed support), a spotter aircraft and offshore capable vessels (Table 1). The Environmental Health Department is involved in the management of the fishery by determining and enforcing the sanitary standards that the processing plants and seafood markets have to operate under.

**Table 1.** Law enforcement capabilities of the Department of Environment and Coastal Resources and the location of these assets and personnel throughout the Turks and Caicos Islands.

| Assets/Personnel | South Caicos | Providenciales | Grand Turk |
|---|---|---|---|
| Fisheries Officers | 4 | 5 | 2 |
| Environment Officers | 1 | 0 | 1 |
| Scientific Officers | 1 | 1 | 1 |
| Protected Areas Officers | 4 | 0 | 0 |
| Research vessels | 2 | 1 | 0 |
| Protected areas patrol vessels | 2 | 0 | 0 |
| Fisheries patrol vessels | 3 | 2 | 1 |
| Patrol vehicles | 1 | 2 | 1 |
| Administrative personnel | 1 | 4 | 3 |

Stakeholder involvement in the management of the fishery is encouraged through regular public fishermen meetings hosted by the DECR. In addition to this, bimonthly meetings are held by the Fisheries Advisory Committee (FAC) which are used to advise the Minister for Natural Resources on activities within the fishery (Bennett 2000). The FAC consists of retired and active commercial fishermen, processing plant owners, a representative from the national parks committee and representatives from the Ministry for Natural Resources and the DECR. The FAC is facilitated by the DECR, but is deliberately not lead by it since the aim of this committee is to provide stakeholders with an opportunity to assist in the management of the fishery and to create a forum for stakeholders to raise concerns within the fishery. The FAC also uses this opportunity to direct research conducted by the DECR.

An additional committee involved in the management of the conch fishery is the Convention in Trade in Endangered Species of Wild Flora and Fauna (CITES) Management Authority (Bennett 2000). The TCI CITES Management Authority is used to review scientific research and recommendations by the DECR pertaining to the annual conch quota. The committee sets the: amount of conch that can be landed; total amount of conch and conch trimmings that can be exported; and reviews size limits and the closed season for conch, in addition to ensuring that monitoring of exports and record keeping are conducted to comply with CITES. The TCI is not a full party to CITES, but adheres to the convention since it exports virtually its entire conch product to countries that are full parties to CITES (e.g. 99 % to the USA).

## THE FISHERY

The TCI fishery can be characterized as a predominately commercial fishery utilizing free divers to collect Caribbean spiny lobster and Queen conch (Table 2) with a small amount of scale fish caught as a by-product of the free diving operations for subsistence usage (Béné 2001). Use of pots and traps by fishermen to catch lobster and scale-fish respectively is limited. There are limited game fishing activities throughout the islands, primarily on Providenciales whereas commercial fishing for lobster and conch is based on South Caicos and Providenciales with a small number of fishermen and vessels operating from Grand Turk (Table 3). There are no operational commercial, scale-fish/ pelagic/ aquarium/ live-rock/ sponge/ bêche-de-mer fisheries within the islands. Virtually all of the fish sold in restaurants and hotels throughout the islands is imported from the United States. There is one commercial conch mariculture operation located on Providenciales which collects conch egg masses from the wild and then raises the conch for export.

**Table 2.** Catch and export figures for the lobster, conch and scale-fish fisheries of the Turks and Caicos Islands.

| Year | Lobster landings (kg) | Conch landings (kg) | Scale-fish landings (kg) |
|---|---|---|---|
| 2000-2001 | 288,852 | 810,502 | 625 |
| 1999-2000 | 252,889 | 730,770 | 1,084 |
| 1998-1999 | 310,878 | 640,310 | 0 |
| 1997-1998 | 228,285 | 781,425 | 0 |
| 1996-1997 | 179,505 | 730,935 | 2,065 |
| 1995-1996 | 307,665 | 956,925 | 1,058 |
| 1994-1995 | 373,860 | 946,305 | 13,356 |

Source: Department of Environment and Coastal Resources landing sheets and export data sheets.

Table 3. Commercial fishing effort within the Turks and Caicos Islands.

| License period 2001-2002 | South Caicos | Providenciales | Grand Turk |
|---|---|---|---|
| Commercial Fishermen | 163 | 175 | 38 |
| Commercial fishing vessels | 78 | 92 | 18 |
| Game fishing vessels | 3 | 28 | 6 |
| Processing plants | 3 | 2 | 0 |

Source: Department of Environment and Coastal Resources commercial fishing vessel license application forms.

**Caribbean Spiny Lobster (*Panularis argus*)**

Caribbean spiny lobster is the species that fishermen concentrate upon during its open season (1st August – 31st March) due to the high economic value of this species (approx. $7.22/kg landed live weight at the processing plants). Lobster (legal size 8.3cm carapace length) are caught by fishermen free diving using fins and a snorkel enabling them to reach depths sometimes in excess of 18m (Clerveaux 2001c, Bennett 2000, Anon. 1998). When diving, the fishermen utilize a fishing hook attached to a fiberglass rod to impale the lobster and bring it to the surface. About 90 % of crawfish are caught during the 2001-2002 season using the hook. This practice - although illegal according to the Fisheries Protection Ordinance - has been ignored by the DECR since the expiration of a temporary waver in 1990. Fishermen utilize the hook because this is more efficient than either the bully and net or the toss. Unfortunately once a lobster is "hooked" in the abdomen it often dies. Due to almost total mortality of lobster following hooking, little can be done if the lobster is subsequently found to be either egg bearing, undersize or tar spotted (possession of lobster exhibiting any of these traits is prohibited) (Bennett 2000). Undersize lobster caught intentionally/unintentionally are not usually landed at the processing plants, as this is where Fisheries Officers conduct most of their inspections. Illegal lobsters are either consumed by the fishermen and their families or sold on the "black market" to hotels and restaurants which serve the lobster as "lobster hash". To ensure a closed season, the DECR requires establishments selling lobster to file an annual return on the last day of the season stating how much lobster they have in their possession (the aim of this is to remove the "black market" demand for lobster during the closed season). However, this does not work. Commonly the lobster holdings of the establishment selling lobster do not diminish. To negate this problem and to reduce fishing effort during the closed season on lobster, lobster should be removed from the menu on restaurants and hotels, as has been done in other Caribbean countries. Unfortunately this is proving difficult to garner political support for as it is thought that this may effect tourist revenue – despite the fact that this scenario not been experienced by other Caribbean nations that do not allow the sale of crawfish by restaurants during their closed seasons.

Another deleterious practice occurring commonly in conjunction with the use of the hook is the use of gasoline, bleach or washing up liquid. The liquid is squirted into cracks and crevices to irritate the lobster into coming out into the open so they can be easily caught. The introduction of hydrocarbons, chlorine or phosphates (from the washing up liquid) may have deleterious effects upon the flora and fauna comprising the substrate in the immediate vicinity of the use of this liquid (Bennett 2001, Bennett 2000). The fishing community is highly vocal about this method of catching lobster, with many of the older fishermen being opposed to its use and stating that reefs are dying due to this reason. The DECR is researching a technique to identify lobsters caught with bleach to provide a method of enforcing legislation that already bans the use of noxious substances to catch marine products. Bleach or gasoline is also used to wash off the eggs on berried lobster. Evidence of fishermen using bleach to catch lobster can often be observed as the clothes of the fishermen are often covered in bleach stains (Vaughan 2001 pers. obs).

**Conch (*Strombus gigas*)**

Conch is the staple protein for the population of the TCI (Sadler 1997). Conch is collected by free divers and is "knocked" (shelled) onboard the fishing vessel (vessels used by fishermen throughout the TCI for the harvest of lobster and conch are predominately 5m Boston Whaler type hulls with a center console and a 75hp outboard engine with a crew of three) (Clerveaux 2001c). The meat is then landed at processing plants where fishermen are paid approximately $1.80/kg. Conch is managed by a closed season for export (15th July - 14th October), size restrictions and a quota for 2002-2003 currently set at 747,776kg of landed product (meat plus trimmings), which is divided between the five processing plants. The processing plants then clean the meat obtaining 37 % meat 10 % trimmings and 43 % waste (Bennett 2001). The meat and the trimmings are then either sold locally or exported. Fisheries Officers regularly conduct inspections of the processing plants and the exported marine product. A backwards step in the management of the fishery was made in 2002 with a decision to revoke a requirement that all exported marine product had to be inspected and taped by a Fisheries Officer prior to export. This requirement had been in place for one year and allowed accurate tracking of the conch export, as well as ensuring that no illegal export of conch could occur.

**Illegal Fishing**

Poaching (the collection of marine product) by unlicensed fishermen is a considerable problem in the TCI. The problem originates from two sources – fishermen from within the islands and fishermen based outside of the TCI. In the first instance, fishermen (predominantly Haitian illegal immigrants) go out to sea and fish either together or with a licensed foreign commercial fisherman (the licensed foreign commercial fishermen are also illegally fishing as they must fish with a "Belonger" on board the fishing vessel). These illegal fishermen exert considerable fishing effort since they also fish even in the worst weather when the DECR and other fishing vessels do not venture out to sea to either apprehend or report them (Bennett 2001, Bennett 2000).

A further problem for the TCI fishery and other Caribbean nations is the high incidence of poaching by fishermen from the Dominican Republic (DR). During the

2000-2002, nine vessels and approximately forty dinghies with hooka compressors with a total of one hundred fishermen were apprehended (the captain and engineer from each vessel received custodial sentences with the remainder of the crew being repatriated). The DECR subsequently confiscated and either sold or sunk the vessels. The development of a policy under which each poacher vessel is sunk is desperately needed since many of these vessels sold to Belongers find there way back to the DR only to reappear within TCI waters poaching again. The Marine Branch of the RTCPF conduct joint operations with the DECR utilizing offshore capable patrol vessels. Location of poaching vessels is achieved using a Fisheries Officer as a spotter in the Police aeroplane that conducts regular patrols of the TCI economic exclusive zone. A further concern is that foreign fishermen employed legally to assist TCI fishermen gain a great deal of knowledge regarding the fishing grounds of the TCI and can pass on this information to the poachers. The Dominican poachers conduct their fishing activities using hooka and spear guns (both of which are illegal in the TCI). A major problem with illegal fishermen based both in/outside the TCI is that they plan only to be fishing in the TCI over a limited time frame (months or hours). Therefore they do not have a long-term interest in the health of the fishery, demonstrated by the fact that the majority of their catches consist of undersize, out of season, berried crawfish taken by illegal methods.

## THE STATE OF THE FISHERY

The lobster fishery is exhibiting signs of overfishing, Clerveaux (2001c) demonstrated that the ratchet effect of increasing effort to maintain fishing yields is occurring with fishermen deploying faster larger vessels and travelling further afield to land comparable amounts of lobster. Bioeconomic analysis of the lobster fishery advocates a reduction in the number of fishing vessels gradually from 198 in 2002 to 73 by 2010 with no concomitant increase in vessel size or numbers of fishermen (Puga 2002). It is clear from mathematical modeling that the MSY for lobster in the TCI has been surpassed and that increased effort maintaining catches is actually masking a decline in the lobster stocks. It is therefore of grave concern that the lobster fishery will subsequently crash, placing additional pressures on the conch and scale-fish populations if fishing effort is not reduced. At present the conch stocks are healthy and well managed by a quota: determined by a recent nationwide visual census and mathematical modeling (Clerveaux 2001a, Clerveaux 2001b, Medley 1999, Ninnes 1994, Berg JR 1989). However, there is no room for increasing the quota, which both processing plant operators and the fishermen will advocate to maintain marine product landings and revenue. Furthermore, the high percentage of legal foreign fishermen and the trend for Belonger fishermen upon retiring applying for and obtaining permits for foreign fishermen to work for them is a worrying trend as fishing effort continues to increase (Table 4). Scale-fish populations are not fished commercially to any extent, offering an opportunity for fishermen to diversify into. Although for a scale-fish operation to be financially viable, a significant investment by fishermen in time and labor is required. There is currently little expertise in the islands amongst the local population to operate a commercial scale-fish, fishing vessel. For a scale-fish fishing vessel to be economically viable, it must stay at sea fishing for several days. This is a big hindrance to the local fishing populace who are used to spending no more than eight hours at sea each day and have no desire to change this.

**Table 4.** Number and percentage of licensed commercial foreign fishermen exerting effort within the TCI fishery.

| Year | Commercial fishermen | Foreign fishermen | Percentage of foreign fishermen |
|---|---|---|---|
| 2001-2002 | 376 | 52 | 13 |
| 2000-2001 | 350 | 59 | 17 |

Source: Department of Environment and Coastal Resources license application forms.

## APPROACH TAKEN TO MANAGE THE FISHERY

During 2001 a more rigorous approach to enforcement of fisheries legislation was taken. The decision to take a top down approach was taken as fishermen were blatantly flouting the fisheries legislation, especially with regard to the landing of undersize lobster (up to 70 % were undersize on the 1st of August 2001). A successful prosecution of a processing plant and a Belonger fisherman lead to a dramatic reduction in the undersize lobster being landed (Vaughan 2001 pers. obs.). Fisheries Officers are often constrained in their jobs by strong family and fishing ties inherent within the islands, indeed many of the Fisheries Officers are ex-commercial fishermen. Therefore, warnings are commonly given rather than legal action, acting as little future deterrent to the fisherman. By stressing that each Fisheries Officer has a duty to enforce the legislation, and establishing an environmental education program which started with the establishment of two Environment Officers, the DECR started to take a holistic approach to fisheries management in 2001. The Environment Officers were tasked with raising environmental awareness amongst the general public and running a turtle tagging/monitoring program. Awareness of the importance of leaving undersize lobsters was stressed by issuing each commercial fisherman with an information sheet and a plastic caliper to measure the lobsters carapace length.

To improve the effectiveness of the extensive MPA system throughout the country (established to protect lobster, scale-fish and conch stocks and to provide managers with a tool to regulate development), the MPA boundaries were depicted for the first time in 2003 on commercial readily available/affordable navigation charts. Substantial effort and expense has been channeled into demarcating the MPA boundaries - this process is ongoing. A review of the Fisheries Protection Ordinance was conducted in 2002. Alterations to tighten up the legislation are under consideration with the Attorney General. Once reviewed the changes will go to the FAC for discussion and for the members to gather comments from other stakeholders.

The DECR is fairly well funded and equipped compared with other Caribbean nations (Gravestock 2002), nevertheless the extent and number of MPAs precludes thorough and effective management. Indeed, the very fact that so much of the territorial waters are protected causes a great deal of conflict between the DECR and fishermen.

Rapid development throughout the islands, and the governmental view that cruise ship tourism is the way for the islands to press ahead economically, will further stretch the resources of the DECR; this in turn compromises its' ability to secure the sustainable and wise use of the natural resources of the TCI for future generations unless the DECR is further strengthened with personnel, equipment and expertise.

## CONCLUSION

In order to maintain an economically and ecologically viable fishery, several steps need to be taken to reduce effort within the fishery and to ensure that the local people, who have a long-term stake in the environmental integrity of the fishery, are those that benefit directly from the fishery. Rapid development of the TCI, in conjunction with rapidly increasing fishing efficiency and effort through the use of foreign fishermen, larger vessels, larger engines etc. are the main threats to the marine resources of the islands.

In order to constrain fishing effort emphasis should be on:

i) The implementation of a cap on the number of commercial foreign fishermen that are issued licenses - these fishermen should only be issued licenses in accordance with the existing legislation: i.e. they are to assist an elderly or infirm "Belonger" fisherman. The assistance should be limited to one commercial foreign fisherman to each such person and no foreign fisherman should be issued if the "Belonger" ceases to fish.

ii) Removal of the power to issue foreign fishermen licenses from the Minister for Natural Resources. This capability should be transferred to the FAC to negate political pressure being exerted on the Minister each year by "Belongers" wanting foreign fishermen to work for them.

iii) Banning hooking to catch lobster, implemented within a year following an extensive retraining program whereby the older fishermen (proficient in the use of the toss and the bully to catch lobster) are used to train fishermen that have no experience with this gear type.

iv) Continue the process of demarcating the boundaries of the marine protected areas and follow through with the campaign to raise awareness of their use and purpose amongst marine resource users.

v) Ban the sale of lobster at markets, restaurants, shops and hotels during the closed season, thereby reducing the incentive for fishermen to poach off season since the demand for this product is immediately removed.

Development is already eroding the integrity of the MPA system within the TCI; this is due to the fact that the islands have become very accessible due to the high number of scheduled flights from the USA and Europe. Tourism brings with it benefits for a country's economy, however a country must decide at what part of the spectrum is it most suited for economically, environmentally and socially. With a country aiming for exclusive tourism, the economic benefits can be high and although the environmental resources per tourist are also high, the overall impact upon the environment is much less than in a situation where a country aims for mass marketing and medium/low end tourism, i.e. package holidays and cruise ship developments. With cruise ships now docking on a regular basis on Grand Turk, the pressure upon MPAs around Grand Turk is mounting. As tourism increases throughout the islands and economics favor the development of scale-fish resources, then this resource will come under pressure. Unfortunately this resource is likely to be exploited by a select few people utilizing foreign labor to catch the fish. This, in turn, allows the common marine resources of the islands to be eroded by those that have little or no long-term stake in the environmental integrity of the Turks and Caicos Islands.

It is the continued commitment and effort of those people in the TCI acting as the stewards of the marine resources that hold the key to the long-term protection and utilization of those resources for future generations. A holistic approach to management that encompasses the involvement of stakeholders through the utilization of an effective FAC and regular fishermen's meetings, and the continued active enforcement of fisheries/national parks legislation through a "top down" approach to management should be continued. This, in-conjunction with the environmental education and outreach approach "bottom up" offers the most hope for the sustainable and wise use of the marine resources of the TCI.

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# Efficacy of a starch-iodide swab technique to detect the illegal use of bleach in a Spiny Lobster (*Panulirus argus*) fishery

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**Abstract**

The destructive and illegal practice of using chemicals (bleach, dishwashing liquid, gasoline) to catch spiny lobster (*Panulirus argus*) is thought to be common throughout much of the Bahamian Archipelago. Injection of a chemical irritant into a lobster den will result in either a rapid escape response or a subduing effect, both of which make it easier to capture spiny lobster. We used both laboratory and field trials to determine the efficacy of a starch-iodide swab technique to identify lobsters that had been exposed to bleach solution (NaOCl). All lobsters exposed to bleach tested positive immediately following exposure and for varying periods thereafter. No false positives were detected on control lobsters. The average length of time that bleach remained detectable on the exoskeleton was 6.2 and 9.6 h in the laboratory and field, respectively, with some individuals testing positive 12 h after exposure. The swab technique will provide fishery officers with a powerful tool to reduce or eliminate the illegal use of bleach for harvesting spiny lobster.


*Keywords:* Chlorine bleach; Illegal fishing techniques; *Panulirus argus*; Caribbean spiny lobster

## 1. Introduction

The use of noxious substances to capture marine and freshwater fishery products is prevalent in many fisheries around the world (Hensley and Sherwood, 1993; Jones and Steven, 1997; Bennett and Clerveaux, 2005; Inogwabini, 2005; Mak et al., 2005). Cyanide is the most commonly used with fisherman in the aquarium trade using cyanide to stun or asphyxiate ornamental fish for collection (Jones and Steven, 1997; Barber and Pratt, 1998; Halim, 2002; Mak et al., 2005). Chlorine bleach has also been employed widely, with uses ranging from the stripping of eggs from ovigerous (egg-bearing) females in the Maine rock lobster (*Homarus americanus*) fishery (Austin, 1995; Cogger and Bayer, 1996; Lobster Institute, 1997; Heckman et al., 2000), to its use as a subduing agent in both reef fish and crustacean (spiny lobster) fisheries in the Pacific (Hensley and Sherwood, 1993), Bahamas (Campbell, 1977) and the Turks and Caicos Islands (Lang et al., 1998; Clerveaux and Vaughan, 2001; Rudd, 2003; Tewfik and Bene, 2004). The deleterious effects of cyanide and chlorine bleach on marine organisms are well documented (Carpenter et al., 1972; Campbell, 1977; Lehtinen et al., 1988; Rosemarin et al., 1994; Jones and Steven, 1997), with effects ranging from decreased productivity to partial or total mortality of benthic communities (Campbell, 1977). Specifically, corals and other organisms are killed by the chemicals, with fleshy and turf algae rapidly recolonizing the area, thereby making the recolonization by corals and other organisms difficult and slow (Carpenter et al., 1972; Lehtinen et al., 1988; Rosemarin et al., 1994).

In the Maine lobster fishery, ovigerous lobsters are taken by trap fisherman and the eggs are subsequently removed by dipping the tails into a seawater and bleach solution (Heckman et al., 2000). Bleach solutions used by Maine fishers frequently exceed 20% concentrations and result in the complete removal of eggs in less than 2 min, quicker at higher concentrations

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(Cogger and Bayer, 1996). To combat this illegal egg removal method, researchers developed three techniques to detect the use of bleach. Firstly, Cogger and Bayer (1996) developed a microscopic examination technique, where chemical scrubbing of ovigerous lobster using bleach was detectable from the damage caused to the plumose setal hairs on the lobster tail. However, this technique required the removal of the swimmerets for microscopic examination, thereby causing physical damage to the lobster (Lobster Institute, 1997), making them less suitable for sale. Smith (1999) then developed a test whereby a 0.1% hematoxylin solution was applied to a lobster tail and resulted in bleach exposed tissue turning purple. Finally, a less invasive technique developed by Heckman et al. (2000), involved the use of a simple swab test, that combined two chemicals (potassium iodide and a starch indicator solution) to detect the use of concentrated (20% or greater) solutions of chlorine bleach (5.25% NaOCl) on lobster.

In the Turks and Caicos Islands (TCI), Bahamian Archipelago, fishers illegally inject household bleach, or liquid detergent sometimes mixed with gasoline into spiny lobster dens using a small plastic bottle (Clerveaux and Vaughan, 2001; Rudd, 2003). These methods have either a ‘subduing’ effect (bleach) or result in a rapid escape response (detergent and gasoline) and aid in the harvesting of lobster. Chlorine bleach is the preferred chemical by fishers in the TCI as it causes the least irritation to themselves, is cheap, and results in the ‘subduing’ of lobsters inside their dens (Clerveaux and Vaughan, 2001). Specifically, bleach causes a chemical reaction in the gill tissue which blocks the transfer of oxygen and causes the lobster to slowly asphyxiate.

The present paper aims to develop and validate a portable detection method that would assist fishery officers to reduce or eliminate the use of bleach in the TCI’s spiny lobster (*P. argus*) fishery. The methods of Cogger and Bayer (1996) and Smith (1999) were considered unsuitable for field use in the TCI fishery, given the requirements for laboratory examination of plumose setal hairs, and a 0.1% hematoxylin solution applied to lobster tails, respectively. Thus, the swab test developed by Heckman et al. (2000) was considered the most adaptable to the TCI fishery. However, given that in the TCI bleach is injected by fishers into the water column rather than used as a concentrated dipping solution, the dilution of the bleach solution may make the swab test developed by Heckman et al. (2000) ineffective for detecting lobster caught in this manner. Thus, we aimed to test the efficacy of the potassium iodide and starch indicator swab technique over time on lobsters exposed to bleach injected into the water column, both in the laboratory and in the field. In addition, we provide information on the efficacy of the test with respect to lobster size and sex.

## 2. Methods

### 2.1. Study site

The spiny lobster fishery is the most valuable marine export product to the TCI economy each year. At South Caicos (Fig. 1), marine resource harvesting (e.g. conch, lobster) is the principle component of the economy and employment, and in combination with fisheries from the other islands, contributes approximately 2.4% of the TCI GDP (2000 estimate). TCI regulations stipulate that lobster fishing is only permitted from 1st August to 31st March and minimum carapace length (CL) for harvested lobster is 3 1/4” (83 mm) (Fishery Protection Ordinance, 1994). During the closed season, no lobster can be exported or served in local restaurants. However, there is no harvest quota, thus local stocks are subject to intensive fishing pressure during the open season (Rudd, 2003).

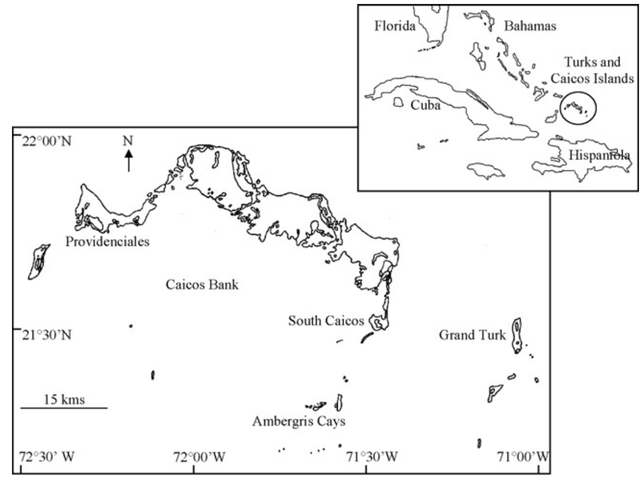


Fig. 1. Location of the Turks and Caicos Islands at the southern end of the Bahamian Archipelago.

Spiny lobsters are harvested predominantly by “free diving” (i.e. without underwater breathing support—the use of SCUBA and Hookah for fishing purposes is illegal) with the remainder being caught with traps. Free diving is in itself a form of effort limitation on the rate and quantity of harvesting, due to the depths that the divers can achieve. Free divers fishing for lobsters currently utilize several illegal fishing methods, most notably “the hook” (a large fishing hook fixed to the end of a 3–5 ft fiberglass rod) and noxious chemicals to force lobsters from their dens (Clerveaux and Vaughan, 2001; Rudd, 2003).

The efficacy of a starch-iodide swab technique was tested both in a laboratory environment and in the field at South Caicos, TCI (Fig. 1), from August through November 2004, and from January through February 2005, respectively.

### 2.2. Starch-iodide swab technique

The swab test consisted of two primary chemicals (potassium iodide and a starch-indicator). The potassium iodide crystal (Spectrum Chemical® code #P1335-10) was prepared as a 0.1% solution in distilled water. The starch-indicator solution (Fisher Scientific® code #SS408) was used in its manufactured concentration (0.1% Starch). Swabs used in the trials were Dynarex® brand (#4304) cotton swabs that were sterile and unbleached. The swab test involved removing a cotton swab from its individual protective packaging and applying five drops of each chemical from a dropper bottle, to ensure that neither the swab nor test chemicals were contaminated during repeated trials. The cotton swab was then rubbed on the smooth areas of the carapace, between the rostrum (horns), around the base of the antennae, cephalothorax and the front legs. During a series of initial trials, these anterior areas of the lobster were found to retain bleach for longer periods than posterior areas (pers obs). A positive test result was recorded when white swabs turned purple after being rubbed on the bleach exposed exoskeleton, a result of the chemical reaction between the swab chemicals and bleach.

### 2.3. The efficacy of the starch-iodide swab technique in a controlled laboratory environment

Using the aquarium facilities located at the Department of Environment and Coastal Resources (DECR) South Caicos base, five raceways measuring $3 \times 0.5 \times 0.5$ m (length × width × depth) were used in the experimental manipulations. Lobsters for experimental trials were collected using industry standard lobster traps to minimize potential injury/mortality attributable to free diving fishing methods, and to ensure no previous exposure to bleach. Prior to each trial the water in each raceway was tested for the presence of bleach using a standard chlorine test kit. In all cases the test produced a negative result. Five lobsters were then taken from a main holding tank and were swab tested for the presence of bleach. If negative (all were negative), each lobster was measured to the nearest mm carapace length, sexed, then placed into a separate raceway. This gave four treatment raceways and one control for each trial ($n = 5$). Water flow was then turned off and 40 ml of household bleach (Clorox$^{TM}$) was added to each experimental tank using two 20 ml syringes (40 ml) (not the control) and left in situ for a period of approximately 60 s. Forty milliliters of bleach was used for the trials following discussions with both DECR fisheries officers and former lobster fisherman in the TCI. These individuals provided anecdotal evidence that free-diving lobster fishers would inject at least 40 ml of bleach into a lobster den containing 2–5 lobsters. Thus, we chose to use 40 ml, the lower limit of estimates given, for all trials.

Following treatment all lobsters were removed from raceways and placed in separate holding baskets. The lobsters were swab tested using the starch-iodide solution (including the control) to test for the presence of bleach and the result recorded as either a "+" or "−". A positive result occurred when the white swabs turned purple after being rubbed on the bleach exposed exoskeleton of a lobster. Lobsters were placed back into their separate holding baskets for further swab tests to be carried out at 2, 4, 6, 8, 10, 12 and 14 h after initial testing. At the end of each trial, the raceways were drained, flushed with fresh seawater and left to dry until the next morning. Prior to the next trial, the raceway was flushed again and filled with seawater. The experiment and protocol was repeated ten times using a total of 25 male and 25 female lobsters ranging in size from 50.5 to 114.0 mm CL.

To compare the length of time (2 h intervals) that bleach was detectable on lobster carapaces after exposure between male and female, legal sized ($\geq$83 mm CL) and undersized (<83 mm CL) spiny lobsters, a two-way ANOVA, with sex and size as fixed factors was used. A logistic curve (non-linear regression) was then used to determine the predictability of lobster testing positive to bleach at intervals post-exposure using the two hourly test data combined for sex and size.

### 2.4. Independent validation

To further validate the swab technique using an independent validation, three fishery officers from the Department of Environment and Coastal Resources (DECR) were given a blind test using the swab technique described above. For the validation, 10 lobsters were put through the same exposure ($n = 6$) and non-exposure ($n = 4$) regime as described previously and the results recorded. Four hours after the treatment regime was completed, the fishery officers each swab tested the 10 experimental lobsters for the presence or absence of bleach and the result compared to the treatments.

### 2.5. The efficacy of the starch-iodide swab technique in the field

Lobsters for the field trials were collected using industry standard traps. The field site consisted of an open sandy area in approximately 3 m of water, in close proximity to natural lobster dens containing spiny lobsters. Field trials were carried out using artificial lobster dens rather than actual lobster dens in the reef structure to standardize the methodology and to prevent damage to the natural benthic habitat. The artificial dens consisted of a cage measuring $0.85 \times 0.65 \times 0.41$ m with a mesh size of $5 \times 3$ cm. Three sides and the top were covered with plastic to simulate a semi-enclosed lobster den that still allowed some water movement. Only lobsters with a carapace length (CL) greater than the minimum legal size of 83 mm (84–123 mm CL) were used in the field trials to more closely represent the local fishery.

Five lobsters were removed from the aquarium facility's main holding tank and transported to the field site. Prior to placement in each of five test cages, lobsters were swab tested for the presence of bleach. As no bleach was detected on any of the lobsters they were measured to the nearest mm (CL), sexed and a single lobster placed in each cage. The five experimental cages were then placed approximately 10 m apart at the field site (four treatment cages and one control). Household bleach (Clorox$^{TM}$) was then squirted into the open entrance of the artificial den towards the lobster using two 20 ml syringes (40 ml), in each of the four treatment traps. The lobsters in the cages were left for approximately 60 s before they were hauled from the water back into the research vessel. The lobsters were removed from the cages and placed in separate holding baskets. All lobsters, including the control, were then swab tested using the starch-iodide solutions and the result recorded. Lobsters were then placed back into separate holding baskets for further swab tests at 2, 4, 6, 8, 10, 12 and 14 h after initial testing. The trials were repeated five times on a combination of male and female lobsters as the laboratory trials found no difference in bleach detectability between sexes. A logistic curve (non-linear regression) was used to determine the predictability of lobster testing positive to bleach at intervals post-exposure. The analysis was carried out on the two hourly swab test data expressed as a percentage of lobster testing positive to bleach at two hourly intervals after exposure.

## 3. Results

### 3.1. The efficacy of the starch-iodide swab technique in a controlled laboratory environment

Trials produced clear results with all lobster that were exposed to bleach testing positive using the swab technique immediately after exposure with no false positives. The percentage of male and female lobsters that remained positive (two hourly swabs) slowly declined with time (Table 1). The comparison of test efficacy between male and female lobsters showed no significant difference between sexes (F = 0.385, d.f. 1, 36, $P = 0.538$). There was a significant difference between legal and undersized lobsters (F = 10.549, d.f. 1, 36, $P = 0.003$) (Fig. 2) with bleach being detectable for longer periods on larger lobster carapaces. There was no significant interaction between sex and size. Results for male and female lobsters were then combined as a total percentage testing positive post-exposure to bleach (Table 1). The logistic curve analysis of the combined data shows that approximately 90% of lobster returned positive results 4 h after exposure to bleach. Detectability of bleach on lobster declined after this initial 4 h period, however bleach was detectable on one lobster 12 h after initial exposure to bleach (Fig. 3).

Table 1
Laboratory swab test results expressed as a percentage of treated lobsters that tested positive to bleach following initial exposure (0–14 h)

| | *n* | Time (h) | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | 0 | 2 | 4 | 6 | 8 | 10 | 12 | 14 |
| Male % | 20 | 100 | 100 | 95 | 70 | 40 | 15 | 5 | 0 |
| Female % | 20 | 100 | 95 | 80 | 65 | 45 | 15 | 0 | 0 |
| <83 mm CL | 28 | 100 | 96 | 89 | 57 | 32 | 7 | 4 | 0 |
| >83 mm CL | 12 | 100 | 100 | 100 | 100 | 83 | 33 | 0 | 0 |
| Total % | 40 | 100 | 98 | 88 | 68 | 43 | 15 | 2 | 0 |

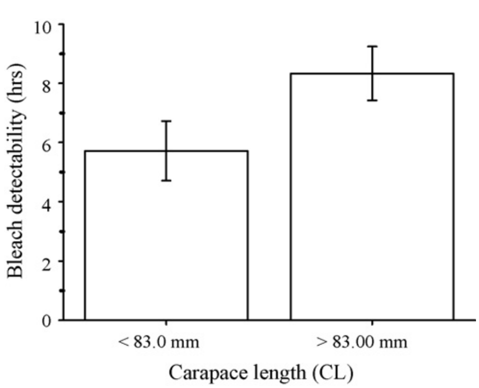


Fig. 2. Laboratory trials. Comparison of bleach detectability between lobsters under and over the legal minimum size of 83 mm carapace length (CL). Error bars = 95% CI.

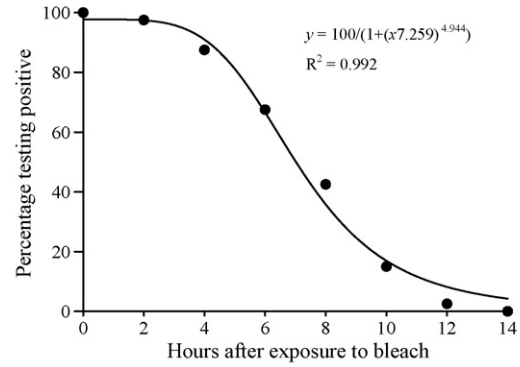


Fig. 3. Laboratory trials. Predictive logistic curve applied to the total percentage data (dots) of the maximum length of time that spiny lobsters tested positive to chlorine bleach.

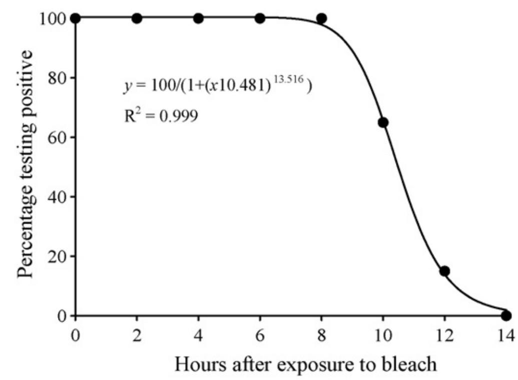


Fig. 4. Field trials. Predictive logistic curve applied to the total percentage data (dots) of the maximum length of time that spiny lobsters tested positive to chlorine bleach.

### 3.2. Independent validation

Blind swab tests carried out on 10 individual lobsters by DECR Fishery Officers correctly matched the initial treatment regime. The six lobsters exposed to bleach tested positive. The four lobsters that were not exposed to bleach tested negative 4 h after the initial treatment.

### 3.3. The efficacy of the starch-iodide swab technique in the field

Field trials produced clear results with all lobster that were exposed to bleach testing positive using the swab technique immediately after exposure (Fig. 4) with no false positives. The percentage of lobsters that remained positive stayed at 100% until at least 8 h after exposure, after which bleach detectability decreased rapidly until 14 h after exposure when no bleach could be detected. The logistic curve analysis shows that the detectability of bleach on spiny lobster is greater than 60% up until 10 h after exposure (Fig. 4).

## 4. Discussion

The most important aspect of the swab test, is that it is consistent, easily interpreted (white swabs turn purple when rubbed on bleach exposed exoskeleton), and is 100% accurate with no false positive results. In addition, the test is fast and does not cause any physical damage to the lobster. The chemicals that form the test are also non-toxic (potassium iodide and starch indicator) and have been approved for food and drug use by the U.S. Food and Drug Administration (Food and Drug Administration, 2005). As such, the swab test poses no threat to the integrity of the fishery product for human consumption (Heckman et al., 2000).

A typical day's fishing for spiny lobster in the TCI is less than eight hours. This involves departing the docks around 07:00 h followed by a 30–60 min boat ride to the desired fishing ground, with fisherman returning to the landing docks between 15:00 and 17:00 h. Thus, from the time fisherman arrive at the fishing ground and begin fishing (around 08:00 h) until they land their catch back in port, is approximately eight hours or less. As shown in our field trials, 100% of bleach exposed lobster tested positive eight hours after exposure. After this time, bleach detectability declined rapidly until 14 h after exposure when no treated lobsters tested positive. The likelihood of detecting the use of bleach, by swabbing lobster at the landing docks is therefore extremely high.

The swab test developed in the present study will be most effective when used in the field by fishery officers, as the detectability of chlorine bleach was shown to decrease over time. Ideally, enforcement and conservation officers would board fishing vessels (as allowed by the law) and swab test the most active lobsters in the vessel, as well as several which appear to have been caught early in the day (dead or in poor condition). Field swabbing of lobsters will circumvent problems associated with fishers only using bleach early in the day to catch spiny lobsters. Employing the swab test in the field has the added advantage that other fishery and boating regulations could be enforced simultaneously, such as the taking of undersized lobster, egg-bearing, or molested lobster (pleopods removed to disguise egg bearing females).

One of the unusual findings of our study was that bleach was detected for longer periods in the field than the laboratory trials. All lobsters exposed to bleach in the field tested positive up to eight hours after exposure, while in the laboratory, only 43% of the exposed lobsters tested positive after the same time period. Reduced detectability in the laboratory is likely to have been a function of the size of lobsters used in the trials and hence, the carapace surface area available for repeated, two hourly swab tests. On average lobsters in the laboratory trials were smaller (77.7 mm ± 2.53 s.e. CL) than those used in the field trials (103.9 mm ± 2.21 s.e. CL). This was supported by our analysis of bleach detectability for laboratory tested lobsters larger and smaller than the legal minimum size of 83 mm CL. The analysis indicated that larger lobsters (≥83 mm CL) retained bleach on their carapace for a longer time than undersized individuals (<83 mm CL) in the laboratory trials. Of the treated lobsters greater than the minimum legal size, all tested positive to bleach at six hours after exposure, 83% at eight hours, with a rapid decline in bleach detectability soon thereafter (Table 1). These percentages are more aligned with the field trials where 100% tested positive at eight hours after exposure, followed by a rapid decline soon thereafter (all lobsters in the field trials were larger than 83 mm CL). Given that lobsters under the minimum legal size will be seized and retained by fishery officers with a possible prosecution being brought against the fisher, the difference between our laboratory and field trials, in terms of bleach detectability over time is not considered problematic.

Chlorine bleach detection kits, containing re-sealable containers of potassium iodide and starch-indicator solutions, with individually packaged swabs are ready for immediate use in spiny lobster fisheries. The swab technique will be an effective fisheries management tool if used correctly and frequently. The swab test will provide fishery officers with a powerful tool to reduce or eliminate the illegal use of chlorine bleach for harvesting lobsters throughout the Bahamas and wider Caribbean. In addition, the test has the potential to be highly effective in detecting bleach usage for a wide range of marine taxa and as such, the success of this technique may have much wider ramifications for illegal bleach fishing around the world.

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# Fishing effort displacement and the consequences of implementing Marine Protected Area management – An English perspective

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ABSTRACT

The creation of Marine Protected Areas (MPAs) and MPA networks is increasing globally. This trend is reflected in England's waters, where 34.7% of waters are protected. MPA network creation can displace activities (primarily fisheries) that are thought to be incompatible with the habitats and species of conservation importance that the network has been established to protect. There is also an obligation on the UK Government to ensure that all of its waters achieve Good Environmental Status (GES) by 2020 under the Marine Strategy Framework Directive. The designation of MPAs and the subsequent introduction of management measures that displace activities may result in unintended impacts/consequences on protected benthic habitats or species within (a) the MPA where management measures have been introduced, (b) other MPAs or (c) wider UK or international waters. An incomplete understanding of the extent and type of fishing that is occurring within the MPA network (and throughout English waters in general), coupled with a paucity of information regarding how fishing effort is displaced as a result of MPA designation, may hinder the achievement of both GES by 2020 and MPA management goals. Better understanding of fishing effort displacement can inform the siting of future MPAs, aid marine spatial planning and improve existing MPA management. To aid the better description and understanding of the various facets of fisheries effort displacement, this paper proposes for the first time a structure to differentiate the types of fisheries displacement. Measures to mitigate the consequences of displaced fishing effort are also identified.

## 1. Introduction

### 1.1. The MPA network in England – moving from designation to management

The concept of Marine Protected Areas (MPAs) has gained prominence in the dialogue on marine conservation and fishery management since the early 1990s. Agenda 21, which urged coastal states to maintain biological diversity and productivity of marine species and habitats under national jurisdiction, was adopted at the 1992 UN Conference on Environment and Development (UNCED). This international instrument and others, including the Convention on Biological Diversity (CBD) [1] and the World Summit on Sustainable Development (WSSD) [2] in Johannesburg, 2002, encouraged the designation of protected areas. As a signatory to the CBD and the Oslo and Paris Convention (OSPAR) [3], which requires contracting parties to establish an ecologically coherent and well-managed network of MPAs across the North-east Atlantic by 2016, the UK is obligated to achieving this.

The establishment of a comprehensive, effective and coherent MPA network within England inshore and offshore waters[1] [4,5] is well underway with 132 sites (Table 1) [6] being designated representing 34.7% and 79,682.6 km$^2$ of these waters. In England, the MPA network comprises Natura 2000 sites (consisting of Special Areas of Protection (SPAs) and Special Areas of Conservation (SACs)) as well as Marine Conservation Zones (MCZs) and Sites of Special Scientific Interest (SSSIs) – although this designation type makes a limited contribution to protecting intertidal habitats. Additional MCZs (called Tranche 3 sites) are currently being considered as are boundary extensions to existing SAC and SPAs.

One of the activities with the greatest potential to damage features designated for protection is fishing. As such, management of fishing activities may be required. The development of management measures for MPAs is now underway.

### 1.2. Fisheries management in England and the provision of conservation advice

The regulation of marine fisheries in England is ultimately the

[1] Inshore waters are 0–12 nm from baselines as set out in The Territorial Sea (Baselines) Order 2014. Offshore waters are 12–200 nm and extend out to the limits set out in The Exclusive Economic Zone Order 2013.

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**Table 1**
The extent of Marine Protected Area coverage in England inshore and offshore waters as of May 2017 (JNCC 2017).[2]

| | Total area ($km^2$) | Total Marine Protected Areas* | | | Special Areas of Conservation with marine components | | | Special Protected Areas with marine components | | | Marine Conservation Zones Tranche 1 & 2 ** | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | No. | Area $km^2$ | % | No. | Area $km^2$ | % | No. | Area $km^2$ | % | No. | Area $km^2$ | % |
| English inshore + offshore waters | 229,779.2 | 132 | 79,682.6 | 34.7 | 39 | 57,853.2 | 25.2 | 43 | 8233.1 | 3.6 | 50 | 20,424.2 | 8.9 |
| English inshore waters | 51,716.0 | 117 | 20,727.2 | 40.1 | 34 | 14,863.1 | 28.7 | 43 | 7864.4 | 15.2 | 40 | 3982.9 | 7.7 |
| English offshore waters | 178,063.2 | 24 | 58,955.4 | 33.1 | 9 | 42,990.1 | 24.1 | 1 | 368.7 | 0.2 | 14 | 16,441.3 | 9.2 |

responsibility of the Government's Department for Environment, Food & Rural Affairs (Defra), which superseded the Ministry of Agriculture Fisheries & Food (MAFF) in 2002. Defra delegates regulatory responsibilities to the Marine Management Organisation (MMO), which licences commercial fishing boats, and ten Inshore Fishery and Conservation Authorities (IFCAs) who regulate the waters within their districts (0–6 nm) through local byelaws and other management measures [7]. The Marine and Coastal Access Act (MACAA) not only established the IFCAs and the MMO but provided the mechanism with which to designate MCZs and to develop marine plans throughout English waters [8].

Conservation advice is provided to the fishery regulators in England by two Statutory Nature Conservation Bodies (SNCBs). The first, Natural England, acts as the Government's advisor for inshore waters (and in English waters out to 200 nm for offshore renewable energy). The second, the Joint Nature Conservation Committee (the umbrella body through which the four national SNCBs deliver their statutory responsibilities for the UK as a whole), provides advice from 12 to 200 nm.

### 1.3. Fisheries structure and distribution of fishing effort in England

In England in 2015 the fishing industry had 3139 registered fishing vessels, of which 2598 were less than 10 m in length. Although not all active, the number of smaller vessels in the English fishing fleet is indicative of the scale and relative importance as a component of commercial fishing in England. The landings of all species of fish and shellfish into England by UK registered fishing vessels in 2015 were 101,000 t with a value of £161.3 million [9].

Information on the location of inshore fishing activity in England is limited (as there is no statutory satellite monitoring of smaller vessels (limited to vessels > 15 m length before 2012, > 12 m thereafter) although significant efforts have been made to fill this gap in knowledge [10–12]. The activities of fishing vessels > 12 m that have Vessel Monitoring Systems (VMS) have to be inferred from positional and course and speed data. There is no requirement to have fishing gear deployment sensors integrated to the VMS.

### 1.4. The blue belt v blue growth

There is a commitment at both a European and a national level to drive economic growth in the marine environment; this is termed "blue growth"[13]. The UK government is also committed to developing a "blue belt"[14] in England, which in essence equates to development of an ecologically coherent MPA network. Therefore the challenge is to balance economic growth against a backdrop of increasing environmental protection. While the two aims are not mutually exclusive, achieving sustainable development of England's coastal waters will be challenging due to the many competing demands for marine space.

The current impact assessments conducted during MCZ designation (required under the MACAA) do consider some socio-economic aspects of displacement but they do not provide for a fuller ecological assessment of the impact that introducing an MCZ will have if fishing effort is merely displaced, and issues arising from that displacement remain unaddressed. Clearly there is a need to take a more holistic approach to assessing and mitigating fishing effort displacement.

### 1.5. Why fishers fish in the way they do

Most economic models of fisher behaviour – both theoretical and empirical – are based on the general premise that the key objective of the individual fisher is to maximise their individual profits from fishing. Profit-maximising behaviour does not necessarily mean that fishers obtain the highest level of profits possible. Instead, they respond in a way that would potentially increase their individual profitability. For example, fishers will switch gear if the benefits from the use of an alternative gear exceed the benefits of the current gear and the costs of switching gear (by way of example, such switches in gear could grant them access to areas from which they are currently excluded or they could be allocated additional quota). Similarly, fishers will not go to sea if the expected revenue from the catch does not cover the fuel and other running costs associated with the trip (as doing so would reduce their profit [15]). Fishers may, however, engage in marginal or unprofitable activities for the purpose of developing or maintaining a track record of fishing a particular species or area.

A number of alternative hypotheses have been proposed to explain fisher behaviour. In particular, personal habit has been thought to be characteristic of fisher behaviour [16,17]. That is, fishers are assumed to prefer to fish in the same areas with the same gear year after year. Similarly, Shepherd and Garrod [18] and Placenti et al. [19] assumed "inertia" existed in the fishery, with major improvements being necessary to encourage fishers to change their behaviour. In some studies, this "habit" or "inertia" has been linked to risk aversion [20]. That is, fishers are assumed to prefer to go where they know the likely outcome rather than try somewhere new, where the outcome is generally unknown. However, while habit, inertia and risk aversion may influence fisher behaviour in the absence of any changes in their regulatory, economic or natural environment, any disruption to this environment is likely to result in a response based on the economic incentives that

[2] The total MPA values* do not include the contribution of Sites of Special Scientific Interest. These statistics should not be used as a direct indication of seabed protection as they include mobile species MPAs (such as harbour porpoise SACs) which direct protection at species in the water column and not at the seabed. Note however that the Conservation Objectives for the harbour porpoise SACs include reference to the protection of habitats on which the animals are dependent. All of these statistics are based on the site boundaries of MPAs and therefore assume that all of the area within an MPA is protected. In practice, protection may only be given to individual features or management zones within the site and not the entire extent of the site. These statistics therefore overestimate the true areal extent of MPA protection. MPAs can overlap each other, especially between designation types but also within designation types in exceptional cases. The 'Total MPAs' columns account for all of these overlaps. ** The eleven Isles of Sicily MCZs are treated as one site.

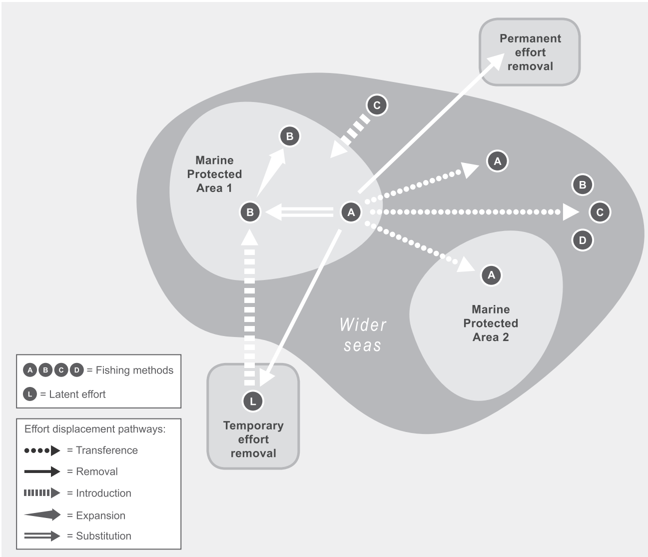


**Fig. 1.** Potential effort displacement pathways resulting from management measures that dictate that fishing type A is incompatible with the species and habitats protected within Marine Protected Area 1.

exist.

Fishers may be assumed to be making choices that mean they are already fishing in the most efficient manner and location for their individual circumstance. Fishers may be pushed into improving gear selectivity or directed towards alternative fishing grounds (to avoid choke species[3]) to ensure that they are able to comply with the Landings Obligation (LO) (more commonly referred to as a discards ban) introduced on 1 January 2015 and 2016 (for pelagic and demersal species respectively) under the reformed Common Fisheries Policy (CFP)[21]. Altering the fishing opportunities available to the fishers will inevitably cause conflict and introduce economic inefficiencies. Whilst this may be considered a bleak view, this is predicated on the fact that (a) fishing opportunities are a limited commodity, and (b) there is already significant latent capacity across many sectors in the fleet [22]. Development of alternative options for fishers to progress, such as aquaculture operations as advocated through blue growth [13], may lead to further displacement by excluding fishers from areas identified for aquaculture infrastructure/operations, leading to the subsequent concentration of existing fishing operations and impacts.

This paper aims to provide the reader with an understanding of the various facets of fishing effort displacement in the context of a rapidly expanding MPA network in England. It sets out the regulatory bodies that are responsible for assessing fisheries displacement in England and the legislative requirements for displacement to be assessed. This paper uses an existing definition of fishing displacement and expands on this to provide a framework that can be used to better describe fishing effort displacement. The paper then sets out the potential environmental implications of this displacement before suggesting management solutions.

## 2. Fisheries displacement

### 2.1. What is fishing displacement?

McLeod [23] defined displacement as: *"the changes in fishing behaviour and patterns that could occur in response to new management measures"*. Changes in fishing behaviour could be *"the adoption of a new fishing method, or target species, or stopping fishing ",* whereas changes in fishing pattern could be *"moving to other fishing grounds near or far"*.

### 2.2. Fisheries management in England and the consideration of fisheries effort displacement

In England, all competent authorities must undertake a formal assessment (Habitat Regulations Assessment - HRA) of the implications of any new plans or projects which are capable of affecting the designated interest features of Natura 2000 sites before deciding whether to undertake, permit or authorise such a plan or project. The requirement for HRAs to be conducted is set out in The Conservation of Habitats and Species Regulations 2010 [24], which transposes into English Law the EU Habitats and Birds Directives [25,26]. Historically, the position in England has been that many commercial fisheries occurring within SACs/SPAs have not been routinely subject to HRA. Recently, Defra announced a change in its approach to the management of fisheries activities within Natura 2000 sites [27], and fisheries are now generally interpreted as a plan or project. Under this approach, action was taken (and is ongoing) to manage fishing activities within Natura 2000 sites where there was evidence (that may be inferred from other studies from which there may be good evidence of a likely effect) that these activities were incompatible with achieving the Conservation Objectives for the protected area (for a detailed description of the process, see: Clark 2017 [7]). The outcome of this project will be a step change in the management of fisheries and their impacts on protected species and habitats. For many sites, fisheries management measures have already been, or are expected to be, introduced that will restrict the temporal and spatial exploitation of fisheries.

### 2.3. Fisheries displacement and the European dimension

At present, Defra, on behalf of the UK Government, in its submission of *Joint Recommendations,*[4] recognises that effort may be displaced and there may be associated impacts. This submission is based on guidance developed by the European Commission [28]. Displacement is one of eleven topics *Joint Recommendations* are expected to address, although limited evidence has enabled only very rudimentary consideration of displacement effects.

The definition developed by McLeod [23] provides a good starting point for displacement discussions and a model to build upon; however, the interpretation of fisheries displacement deserves further consideration to start to fill the existing guidance void for managers.

### 2.4. Elements of fishing effort displacement

Fig. 1 illustrates various potential pathways for fishing effort displacement from a spatial perspective; the coexistence of fishing and protected species in the same area at the same time should also be considered. Fishing effort can be altered in several ways in response to MPA management.

1) Fishing pressure is ***removed*** from the system entirely either permanently (e.g. removal of licence/permit/authorisation completely from the vessel) or temporarily (with the vessel being laid up and the associated licence becoming dormant, i.e. becoming latent effort).
2) There is the ***substitution*** of effort by the existing fishing fleet. In other words, a change in fishing practice within the site, e.g. fishers change their gear type – i.e. from mobile to static gear – but still

[3] Fish species for which quotas are so limited relative to local or general abundance that the imposition of a landing obligation in a mixed fishery is liable to result in fishing vessels having to cease operations well before they have caught their main quota allocations.

[4] A Joint Recommendation is a scientific and technical information package that is required by the European Commission when Member States request fisheries management measures to be introduced under the CFP (Regulation No 1380/2013 - Article 11).

fishing broadly the same grounds.

3) ***Expansion*** of other fishing effort already occurring within a site that fills the area vacated by the fishing activity that was incompatible with the protected features. There is ***transference*** and potential ***intensification*** of fishing effort where the original fishing pressure exerted within a site is relocated outside of the site (due to an unwillingness or inability of fishers to change gear types or species targeted to enable fisheries to continue within the site).
4) There may also be the ***introduction*** of new fishing activities that do not already occur within the MPA (*cf*: ***expansion***) that may backfill the opportunities vacated by the original fishing pressure e.g. static gears replacing mobile gears.

### 2.5. The potential implications of fishing effort displacement

Where there is substitution of fishing effort (a change in the gear type used or the species targeted) within an MPA, the new fishing activities and the pressures they exert may: (1) interact with the same habitats and species of conservation importance in a different manner; (2) exert pressure at a different stage of the species' life history; (3) interact with other species or habitats that the protected species relies upon or interacts with (predator–prey relationships or ecosystem services such as shelter); or (4) interact with other protected species or habitats within that site.

Transference and potential intensification of fishing effort into other MPAs may potentially expose protected species and habitats to pressure. Similarly, the fishing effort may now be directed in such a way whereby species or habitats sensitive to the pressure but outside of the MPA network become exposed; these areas could currently be lightly or unfished. Certain areas, thus far less fished, may attract effort [23]in response to the constraints on other areas with the potential risk of increasing fishing impacts on more pristine seabed habitats that have similar sensitivities to the fishing pressures [29]. This may result in unintended consequences for the marine ecosystems by increasing the mortality of other species or other fish life stages [30]. In the worst cases, this can lead to depletion of certain stocks and biological resources [31–33].

It is worth noting that fishers largely operate in such a way to maximise their returns. Therefore, if they are targeting species within the MPA network and now have to target them elsewhere, this may in turn lead to increased conflict with other marine users and fishers (for example, demersal trawling competing with potting grounds).

Indeed, if fishers are displaced from the areas that are the most efficient to fish, they may have to fish more intensively to maintain catch rates or profitability against increased costs such as fuel [23] from greater steaming time. Consider the following example. A beam trawler with a quota for plaice (*Pleuronectes platessa*) exhausts its quota in five days sweeping an area of 5 $km^2$. If displaced, the fisher may have to sweep the new ground more intensively to utilise their quota. The result of this is that habitats/species outside of the MPA network that were not previously exposed may now be exposed to greater pressure. This intensification of fishing effort within the wider seas may potentially undermine progression toward Good Environmental Status as required under the Marine Strategy Framework Directive [34]. Ideally, to avoid this, effort should be taken out of the system to retain the sustainable balance or at the very least to promote diversification of other less impacting gears so as to relieve the pressure on the habitats/species. A push towards the intensification of effort exerted by fishing activities will likely drive technological change, and future efficiencies in operation may lead to future increased pressures on both stocks (principally those that are not managed by quota or to Maximum Sustainable Yield (MSY)) and the wider marine environment. As with all management actions, there may be unintended consequences.

A possible result of designating MPAs inshore is that effort will be displaced offshore. This may result in the development of larger, faster, more capable vessels that can exert increased fishing pressure on the available resources. This could lead to existing quotas being taken up more quickly, with fishers then focusing their effort on other stocks and areas. In essence, a negative feedback loop could be introduced, i.e. stocks in general are exposed to increased effort and associated habitats are subjected to increased exposure to fishing impacts. Therefore, there may be a requirement to alter future quotas to reflect the implications of spatial closures to mitigate against adverse effects.

Pushing smaller and often single-handed inshore fishing vessels further offshore can also increase the risk of accidents (increased exposure to poor weather, greater transit times and therefore longer working days). Fishers will also have to contend with an increased emergency response time in the event of an accident.

## 3. Management solutions

### 3.1. Understand the magnitude of the potential problem

There is a requirement for marine planning authorities/UK administrations to further the understanding of displacement issues within the marine management community as set out within the UK Marine Policy Statement [35]. There is also an imperative to understand current and potential fishing displacement. This understanding is required so that fisheries managers can work collaboratively to prevent an influx of fishing gear/methods as fleets are displaced from historically favoured grounds. (Notwithstanding Natura 2000 sites, in most instances MCZs were selected largely to avoid the most fished/valuable areas. In addition, management measures typically take the form of effort/zonal management which is designed to minimise the impact on fishers.)

It is unlikely to be appropriate to use emergency byelaws to manage displaced/replacement fishing effort (particularly within the MPA the original effort has been displaced from), as in some circumstances these may be regarded as foreseeable.[5] Were this to be the case, the use of emergency byelaws could be deemed *ultra vires*. The implication of this is that byelaws will have to be developed through the regular process, which can be time-consuming, during which impacts on protected features may continue, or the use of Statutory Instruments to prevent damage will have to be explored. It is therefore advisable to consider the implications of effort displacement at the same time as developing management measures. However, it is imperative to document the consequences of management once in place, to enable regulators/advisors to refine measures to meet Conservation Objectives.

To this end, it is recommended that (1) an overarching regional-seas-scale assessment is conducted of the degree to which fishing effort has been displaced to date as a result of the designation of an MPA network within English waters and (2) a site-specific assessment forms an integral component at the point when management measures are being identified. Where effort displacement/substitution impacts are envisaged, then appropriate mitigation strategies should be explored within the assessment.

It is suggested that the assessments are compiled or commissioned by Defra (governing body) with input from the IFCAs, MMO (management bodies), and the Centre for Environment, Fisheries and Aquatic Science (scientific advisor), whereby the expected displacement is identified and the impacts are ascertained in conjunction with appropriate SNCB.

### 3.2. Removal of effort from the system so that it does not continue to exert pressure

The arbitrary and/or compulsory revocation of fishing licences is legally and politically unattractive in England. The removal of active

[5] Section 157(2)(b) prevents an IFCA introducing an emergency byelaw without the consent of the Secretary of State if the circumstances could have reasonably been foreseen.

fishing licences would also be considered the antithesis of blue growth. However, significant vessel decommissioning schemes were used throughout the 1990s and early 2000s to reduce the capacity within the English commercial fishing fleet to bring capacity more in line with the available quota. This decommissioning was conducted for fisheries management rather than environmental management purposes. Whilst there is little desire to revisit decommissioning schemes for solely fishery management purposes (the schemes were only partially successful in reducing capacity – much of the money received from decommissioning vessels was spent on upgrading existing vessels or on commissioning new, more capable vessels [36]) – there is no impediment, other than cost, to introducing a decommissioning scheme targeted at displaced vessels on environmental grounds. The removal of effort is further complicated because one should not consider a vessel to be the fishing effort in itself, but the fishing licence and any quota or access rights associated with it, as these provide the ability of a vessel to access fishing opportunities and these can be moved from one vessel to another in many cases.

Because the MCZ designation process requires the socio-economic impacts of designation and not subsequent management to be taken into account (although some basic assumptions were made in the MCZ Impact Assessments), it may be that through analysing attributable displacement/substitution it becomes clear that there will be significant impacts both economically and environmentally as a result of the displacement. It may therefore be advantageous to remove effort from the fishery voluntarily through the recovery of fishing licences/permits using financial incentive schemes, whilst at the same time introducing legislation that prevents future incursion into particular areas by new vessels. It is recognised that government may be wary of this in case this implies that compensation should be paid for the removal of the right to fish; however, the right to fish in general is not being extinguished, merely the location that a fisher can legitimately fish.

The proposal above is potentially possible because the UK has recently agreed its operational framework for the provision of funding under the European Maritime and Fisheries Fund (EMFF)[37]. Further exploration of the ability to use the fund as above is warranted.

### 3.3. Support appropriate substitution

Substitution activities may be appropriate and compatible with the Conservation Objectives of the MPA if they are considered in a proactive manner. The introduction of the LO and the requirement for fisheries to reduce the levels of discarding is leading to significant developments in fishing gear technology and techniques. The EMFF operational programme for the UK has been developed to support appropriate changes to fishing practices. A key tenet of this funding stream is the support for fishers to adopt more environmentally benign activities; therefore, there may be scope to support appropriate substitution of fishing activities – this could be explored at both an individual and a fishery level.

Substitution of fishing effort within Natura 2000 sites would be subject to HRA and potentially appropriate management measures, e.g. effort caps as well as spatial and temporal measures, to ensure that the new fishing effort would be compatible with the interest features of the site.

However, there has been a recent trend of reducing the ability of fishers to change target fisheries, i.e. from finfish to shellfish (driven by concerns regarding the status of these stocks and the pressure being exerted upon them – these stocks are already exposed to significant levels of latent effort)[22,38,39]. The drive to ensure sustainable fishing of finfish and basing quotas on pressure stocks around MSY may have pushed fishing effort into activities that exert greater impact on the seabed, i.e. scallop dredging. This curtailing of fishers' ability to change between fisheries is due to the categorisation of commercial fishing licences, whereby fishers may or may not have shellfish entitlements attached to their white fish licences. However, securing some degree of fishing effort substitution may be the best approach if compatible with the MPA objectives. This issue is further compounded as fishers exploring substitution as an option at present are likely to encounter problems, especially if they lack are required to provide evidence of a historical track record of fishing either within a particular area or for a particular species, but they cannot do so.

Relative stability[6] and national allocations of quota species may also hinder the ability of fishers to diversify. By enabling and/or promoting appropriate substitution, fishers are able to maintain their connection with the area, thus providing continued employment and social stability. Where displacement forces fishers to exploit new areas with which they lack a historic connection, then fishers may be less inclined to take a stewardship view of the resources that they are targeting, as they will be concerned with maximising their income because of additional or new costs to their operation as they adapt techniques, relocate their shore-based activities or increase expenditure on fuel [40–43]. Where fisheries are forced into more nomadic fishery practices rather than new, but fixed, areas, the stewardship connection may become further weakened.

To promote effective substitution of activities (changes in gear type), lessons should be learnt from recent offshore industry developments, in which significant attempts have been made to deliver the co-location or co-existence of activities, i.e. offshore windfarm installations and commercial fisheries. It was initially thought that the co-location of these activities was not feasible; however, over time and through the use of improved technology and liaison with developers, fishers have gained the confidence to fish between structures.

Over time, the profitability of vessels might be reduced (from higher costs or altered catch rates, and constant fish prices) and the adaptive capacity of the fishing sector impaired, forcing fishers to leave the sector. Knowledge is still limited on how fishers react to the various constraints such as area restrictions, increased fuel expenses, and changed stock distribution and dynamics. Enhancing our knowledge of these dynamics is an important step in predicting the economic and ecological impacts of fishing displacement [43–48]. One way of achieving this would be to promote monitoring efforts that enable the displacement from existing (and soon to be implemented) management measures to be discerned.

### 3.4. Proactively prevent substitution through the release of latent effort

For substitution of fishing effort to occur in a controlled manner, the issue of latent effort must be tackled. Defra [22,38] consulted on the subject of reducing unused licences (latent capacity) in the English 10 m and under fishing sector. The response [39] highlighted that government were of the opinion that latent effort required addressing, and in the short term a decision has been made to proceed with temporary restrictions on licences for quota species and a temporary suspension of shellfish entitlements. If the issue of latent effort is not addressed, it will be more difficult to assist active fishers to change their fishing operations if they then have to compete with latent effort potentially being released and competing for access to areas where fishing is allowed. The response highlighted a previously made commitment to work with industry on any suggestions or proposals for an industry-funded decommissioning scheme. This would suggest that there is scope to further explore the use of the EMFF to address displacement impacts.

### 3.5. Invest in co-management institutions

As the introduction of management measures for sites gains

[6] Total Allowable Catches (the amount of a particular commercial fish or shellfish species can be caught which have been derived from scientific analysis and/or political agreement) are shared between EU countries in the form of national quotas. For each stock, a different allocation percentage per EU country is applied for the sharing out of the quotas (*national allocation*). This fixed percentage is known as the *relative stability* key.

momentum, it is imperative that appropriate co-management opportunities/institutions are supported and provided with adequate resources to allow meaningful engagement. Successful co-management and, by extension, effective application of the outcomes approach[7] [44] is very resource intensive and often iterative, with feedback loops allowing learning and adaptation to take place. These institutions need to be active listeners and effective communicators, with managers taking an enabling role in encouraging collaboration and conflict resolution between different stakeholder groups. The most appropriate organisations to do this within England are the IFCAs and the MMO. It is clear that managing displacement is not only about managing the environmental implications of displacement but also about conflict management. The marine environment is a very busy place, with many different sectors vying for the same space. In some instances there are opportunities for co-location, but in many cases the activities taking place are mutually exclusive – this applies not only to fishing type A and fishing type B, but also fishing and other sectors. In the best case scenario there may be opportunities for activities that were previously excluded from an area used for fishing to move (back) into that area.

### 3.6. UK Marine Policy Statement, Marine Spatial Planning and marine licensing

Expanding the use of marine plans may help address some elements of fishing effort displacement. The UK Marine Policy Statement [35] specifically mentions displacement and need for authorities to consider it:

> 3.8.10 Marine plan authorities should consider the potential social and economic impacts of other developments on fishing activity, as well as potential environmental impacts. They should, for example, have regard to the impacts of displacement and whether it is possible for vessels to relocate to other fishing grounds. They should also consider the potential impacts of this displacement on the viability of fish stocks and on the marine landscape in the alternative fishing grounds. They will also wish to consider and measure the impacts on local communities of any reduction in fishing activity, redistribution of fishing effort or associated impact on related businesses as the result of a marine development. Marine plan authorities should engage with other regions to where activity is displaced to ensure that a comprehensive picture of impacts is developed and unintended consequences are avoided. Wherever possible, decision makers should seek to encourage opportunities for co-existence between fishing and other activities. Inshore Fisheries Groups in Scotland and Inshore Fisheries and Conservation Authorities (IFCAs) in England will be expected to participate fully in wider marine planning. …………..

There appears to be a presumption within the Marine Spatial Planning (MSP) regime that fisheries can occur throughout UK waters unless direct management interventions restrict their access. Neither MSP nor associated marine licensing (in particular of large infrastructure projects) considers in depth the implications of displacing fishing effort on the marine environment. Because fisheries are dependent on fish habitat and fisheries resources are variable in space [45], spatial restrictions due to MSP may lead to closure of highly productive marine areas or preferred fishing grounds, exacerbating the magnitude of effort displacement [46].

Given that the management structures to implement fishery management plans are already in place, they may provide a more cost-effective way of minimising the overall environmental impact of fishing than MPAs in many circumstances. The identification and management of core fishing grounds in a spatial planning framework is also a 'bigger picture' approach, which considers the impacts of fisheries on the overall state of the environment rather than local environmental gains that might be achieved in MPAs despite costs elsewhere. The challenge will be to combine fisheries management and MPA management through the use of MSP to identify and secure access for fishers to key fishing grounds, as well as identifying those areas to avoid, and then manage both in a more co-ordinated manner [47–49]. Whilst this is likely to be initially very challenging, this may provide a long-term mechanism to reduce conflict and subsequent fisheries displacement.

## 4. Conclusions

In England, MPAs of all designations are recognised as multi-use sites. There is no imperative for MPAs to be exclusive of fishing activities, merely that the fishing activities that occur within them are compatible with the conservation goals of the habitats and species for which the sites are designated. Where this is secured as an outcome, this will help achieve blue growth but in a sustainable manner. However, it is clear that as there is a shift in focus from designation to the management of MPAs (as the network of MPAs is nearing completion), the opportunity to develop a strategic assessment framework that predicts the degree of potential fisheries displacement from the MPA network has been missed. The focus of managers should therefore be on assessing the spatial and temporal changes to fishing practices that have resulted from the introduction of management measures to date to better understand the implications of introducing MPAs and their management on both the environment and the fishing fleet. This understanding should aid in future management of MPAs as this will lead to reduced conflict between stakeholder groups and increase buy-in to the objectives of the MPA. This information should also help regulators consider and manage fishing activities that impact the marine environment outside of the MPA network. This understanding, when coupled with the development of a fishing effort displacement assessment template that managers can use, will assist in securing the long-term sustainable management of the marine environment. Increased understanding of how fishing effort displacement can occur and the likely outcomes from introducing MPAs and subsequent management can be used in the location analysis for future MPAs – although this information may have greater utility to countries that do not yet have extensive MPA networks in place.

The need for management tools that can identify fisheries displacement and the potential problems that arise from it, along with a clear framework of how these problems can be addressed, is required not only in England but also in any nation that is designating MPAs.

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CHAPTER

# 2

# Marine protected areas and marine spatial planning – allocation of resource use and environmental protection

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## Abstract

Rarely is the strong link that exists between Marine Spatial Planning (MSP) and Marine Protected Areas (MPAs) explicitly recognised. MSP is the process by which the use of marine space is identified and used to inform development decisions made by regulators. Marine areas that are important for marine conservation/ecology form one of the most common data layers within marine plans. Some of these marine areas will be formally adopted/designated and have legal protection as MPAs; other marine areas may be protected culturally or through informal agreements. Where MPAs do not exist, marine plans can aid in the identification of areas where they could be sited optimally (taking into account environmental, social, economic and political considerations). MPAs and marine plans are generally based on current information, be it habitat/species distribution or marine uses; however, both marine plans and MPAs may be used as tools to drive future sustainable use of the marine environment. This requires recognising existing uses and identifying how these uses may be affected by climate change, economic development, marine users' social licence to operate and also how the government of the day sees the future use of its seas.

**Keywords:**

Marine protected areas; MPA; Marine spatial planning; MSP; Dynamic ocean management; Real time closures; Real time incentives; Co-location; Fishing effort displacement; Ecologically coherent network.

## Introduction

Whether you consider yourself a practitioner (conservation ecologist, philanthropist, funder, manager, government advisor or regulator) in Marine Protected Area (MPA) management or a marine spatial planner, at the heart of your work will be the identification of particularly important or valuable marine space and the management of activities within


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it. Such management may include the proactive allocation of activities within that space so that marine space is used in the best way to deliver all of the activities that occur within the marine environment along with environmental protection to enable the continued and sustainable use of that marine space.

The terms Marine Spatial Planning (MSP), maritime spatial planning and marine planning are used interchangeably by many practitioners. Marine planning can be thought of as the general process of understanding marine spatial use whereas marine spatial planning can be thought of as the actual development of plans and policies that set out how marine space is to be used. Although there are subtle differences in the definitions, for the purposes of this chapter, the authors use the term marine spatial planning with the following definition - the identification of marine natural resources and the current and potential use of those resources, and the allocation of marine space through a formal framework.

Although MSP underlies the identification of sites important for conservation and influences the design of future MPAs, professionals from the MPA and MSP domain rarely work together. This chapter is aimed at practitioners and attempts to highlight the myriad ways that the two subject specialisms can work hand-in-hand to deliver more for the marine environment. At the most basic level, effective MSP can ensure that new MPAs are sited appropriately and that MPA network outcomes are better delivered.

## The drivers behind MPA designation and MSP adoption

Whereas the impetus for the designation of MPAs can generally be easily defined (in many cases this is derived from international commitments), the drivers for MSP will vary from country to country. MSP may be used as a mechanism to deliver blue growth or an ecosystem based approach to management (Santos et al., 2014). MSP may be used to proactively identify suitable areas for MPA designation (Smith et al., 2008) or for the allocation of space for marine industries such as offshore wind energy (Azzellino et al., 2013). In some cases, the drivers for MSP may not be well defined (Collie et al., 2013; Foley et al., 2010). The United Kingdom (UK), for example, has an explicit Marine Policy Statement (MPS) (2011) that sets out the government's vision and expectations on how the marine environment should be used and protected. A comprehensive marine planning system in England is being developed to implement the MPS; however, the MPS provides little in the way of direction, or ranking or hierarchical guidance in how marine space should be used. This lack of clarity makes it difficult to determine the effectiveness of marine plans once they are developed and implemented. The Netherlands takes a very different marine planning approach compared with England, in that five strategic and hierarchical elements are set out which implicitly establish winners and losers in the race for marine space (Vaughan, 2018).

MSP is also used within the context of large, multiple-use MPAs to accomplish zoning, by which particularly important and sensitive marine areas are strictly protected. A good example of this is the Great Barrier Reef Marine Park in Australia, which, through its representative areas program, accomplished a zoning plan across its 345,000 $km^2$ expanse (an area roughly the size of Japan) (Fernandes et al., 2005).

In the developing world, MSP is commonly used to identify areas for new MPA designation, or for protected zones within a marine plan that become somewhat analogous to

MPAs. In Belize, for instance, scenario planning and trade-off analysis has led to the identification of coastal and marine areas of particular importance (Verutes et al., 2017). These areas are then afforded extra protections in coastal management. In Rodrigues Island in the Indian Ocean, MSP specifically for the aim of maximising conservation benefits has been employed to identify various types of MPAs (Pasnin et al., 2016). In South Africa, a participatory planning process has led to the recent establishment of an extensive network of MPAs (Mann, 2018).

Differing approaches to marine planning can be traced back to the social, political, cultural and legal constructs of a country, the strategic importance of a country's maritime space, the economic value of the resources within that marine space, the current and historical use of that space, and the extent of a country's marine space, as well as the financial resources and technical ability that a country has to map and manage its marine space (Cormier et al., 2018). Whether marine plans provide benefits to the marine environment depends on what the plan was intended to achieve and how the plan is implemented.

## The importance of scale and management response

The requirement to manage our marine environment in a holistic manner, focusing not only on those areas of sea that fall within MPAs but also on the wider seas, is now widely understood and essential with widespread impacts of human pressures (Ehler and Douvere, 2009; Agardy, 2010; Ardron et al., 2008; Halpern et al., 2008, 2015). Our seas are interconnected, with fish, marine mammals and seabirds moving vast distances between breeding, nursery and feeding areas. We understand the importance of ocean gyres in entraining and also dispersing marine larvae. We also understand that the interconnectedness of our seas and oceans means that no one country is able to manifestly improve the health of the seas, yet each country has the ability to damage this global resource.

Collective action can be taken by governments to minimise the dangers of unregulated use or poor management. In many regions of the world, collaborative frameworks are adopted, such as that taken by those Member States of the European Union (EU) through the Marine Strategy Framework Directive (MSFD) (2008).

The EU has mandated that EU maritime member states develop marine plans through the implementation of the EU Maritime Spatial Planning Directive (2014a). Indeed, marine plans form one of the key measures that countries will rely upon to achieve Good Environmental Status as required under MSFD.

At the regional seas level, countries that share a body of water have, in many cases, developed international agreements to set standards on pollution control and other marine management, share information on marine systems and their use, develop specific protocols on marine biodiversity and habitat protection, and provide the framework for trans-boundary cooperation. Regional Seas Conventions exist in the North Atlantic (Convention for the Protection of the Marine Environment of the North-East Atlantic or OSPAR Convention; this came into force in 1998), the Baltic Sea (the Convention on Protection of the Marine Environment of the Baltic Sea; this came into force in 1992), the Mediterranean (the Convention for the Protection of the Marine Environment and Coastal Region of the Mediterranean, or Barcelona Convention; adopted in 1995), the Caribbean (Convention for the Protection and

Development of the Marine Environment of the Wider Caribbean Region; established in 1983), the West African region (The Convention on Cooperation in the Protection, Management and Development of the Marine and Coastal Environment of the Atlantic Coast of the West, Central and Southern Africa Region, ratified in 1984) and East Africa (the Nairobi Convention for the Management, Protection, and Development of the Marine and Coastal Environment of the Western Indian Ocean, entered into force in 1996), among others. Regional Seas programs are administered through the UN Environment Programme (UNEP) and help countries share information and build capacity to undertake marine planning and to manage their seas sustainably. Many of these regional seas have initiatives to undertake MSP, and in the process perform gap analyses to detect what important marine areas are missing from the suite of MPAs in that region.

At an even larger scale, collective action is needed to respond to the challenges posed by global warming and associated sea level rise, ocean warming and acidification. Just as there is a growing global realisation that we need to protect our seas at a macro level, there is a global thrust for blue growth whereby counties look to develop and monetise their marine resources. Countries seek to do this using the United Nations Convention of the Law of the Sea (UNCLOS) (1982) to either establish or extend existing Exclusive Economic Zones (EEZs). Commercial operations also look to exploit seabed minerals or fishery resources in the high seas and the Area Beyond National Jurisdiction (ABNJ).

The 'high seas' comprises all parts of the sea that are not included in the EEZ, in the territorial sea or in the internal waters of a State, or in the archipelagic waters of an archipelagic State (UNCLOS article 86). The 'Area' is the seabed, ocean floor and subsoil thereof, beyond the limits of national jurisdiction (UNCLOS Article 1).

The increasing demand for marine space within the ABNJ and the pressure that this demand places on species, habitats and the ecosystem services that the marine environment provides, require appropriate and effective institutions and the legal framework to ensure the sustainable use of the marine environment. Although under both UNCLOS and the Convention on Biological Diversity (CBD) (1992), nations are committed to preventing harm to the environment and biodiversity beyond national jurisdiction, few countries have developed assessment procedures or other oversight mechanisms to identify potentially harmful activities under their jurisdiction or control (Ardron et al., 2008; Maes, 2008).

To put the current extent of marine protection in the ABNJ into context, 61% of the planet's ocean lies within this area (2018). Within this area there is limited effective protection of species and habitats. Where there is management in place, this is often sectoral and directed at activities such as shipping (e.g. the International Maritime Organisation) or fishing (e.g. Regional Fisheries Management Organisations (RMFOs)) (Ardron et al., 2008; Blanchard, 2017). In the case of the RMFOs, their remit can vary widely. For some RFMOs, their stated mandate is the management of a particular fish species such as tuna (Indian Ocean Tuna Commission (IOTC)), while the mandate of others can extend to wider marine resources within a particular region (e.g. Commission for the Conservation of Antarctic Marine Living Resources (CCAMLRs)). This sectoral approach to managing activities and the pressures from these activities within the ABNJ mirror the approach most commonly taken to managing these sectors within EEZs. And just as MPA designation and management seek to provide protection for species and habitats inshore, this approach is also being advanced within ABNJ. To date, 12 MPAs within ABNJ have been established – two in the Southern Ocean and 10 in the North-East Atlantic region – and more are proposed (Smith et al., 2017). The Southern Ocean MPAs were adopted by members of CCAMLR whereas those in the North-East Atlantic were established under the OSPAR Convention.

Recognising that MPAs alone within will not provide sufficient protection of the marine environment, the UN adopted resolution 69/292 in June 2015 to develop an international legally binding instrument under the UNCLOS on the conservation and sustainable use of marine biological diversity of ABNJ (2015a). The UN subsequently, in December 2017 through resolution 72/249, decided to convene an intergovernmental conference in September 2018 to consider an international legally binding instrument under the UNCLOS on the conservation and sustainable use of marine biological diversity of areas beyond national jurisdiction, with a view to developing the instrument with negotiations addressing measures such as area-based management tools, which include MPAs (2017). And on 24 December 2017, nations agreed to convene an intergovernmental conference, leading to a legally binding treaty under UNCLOS for ABNJ protections that would be negotiated over two years. Time will tell whether an adequately ambitious and integrated approach to management of the ABNJ is agreed, subsequently developed, and adopted.

## A brief history of MPA development

MPAs can trace their roots back to tenurial arrangements and taboos, through which community authorities decreed areas prohibited to fishing or other uses. Many of these marine tenure patterns reflect an innate understanding that some areas of the sea are critical for maintaining ocean health and productivity; for example, fish spawning sites form the base for many taboo areas in Oceania. Fisheries managers started planning fishing closed areas in the late 1950s with the specific aim of protecting stocks and enhancing fisheries production. As protected areas on land began springing up all over the world, marine authorities began to designate multiple-use MPAs, including Marine National Parks, and later, Marine World Heritage sites. Modern MPAs were designated in large numbers beginning in the 1970s, when Pacific nations began pioneering spatial restrictions to protect 'the commons' (this was spurred by the first World Park Congress, held in Japan in 1975).

There are currently 15,334 protected areas covering 26,945,395 km$^2$, which represents 7.44% of our seas and oceans. In terms of marine space under national jurisdiction, the current coverage of marine protected areas is 17.23% (UNEP-WCMC and IUCN, 2018).

MPAs, whether free-standing, in multiple MPA networks or as protected zones within larger multiple-use marine managed areas, are designated based on both need and opportunity. Ecologically important, vulnerable or particularly valuable areas do get highlighted through systematic and strategic planning efforts on the part of coastal and marine management regulators and international agencies that backstop regional seas agreements (e.g. the Regional Activity Centre for Specially Protected Areas, supporting the Mediterranean regional seas member states), and then these key areas sometimes become designated as MPAs. More often than not, however, MPAs are designated because a threat to a particular place arises and needs to be countered with spatial protection, or an opportunity to holistically manage an area arises. In general, these opportunistic MPAs can be considered low-hanging fruit, and they have value beyond the protection of biodiversity or habitat they encompass, in that they can serve as demonstrations of the benefits that MPAs can provide.

## A brief history of MSP development

While it is not within the scope of this chapter to synthesise the history of marine planning in its entirety, it is worth noting that many countries embark on MSP at the national or sub-national scale only after designating MPAs. This was certainly case in the UK. In 1986, the Lundy Marine Nature Reserve was designated as the first MPA in the UK. In 2014, 28 years later, the East Inshore and East Offshore Marine Plans were adopted in England (2014b). These were the first large-scale marine plans to be developed in the UK. The pattern of a country first designating MPAs and then developing marine plans at a later date is mirrored around the world. Virtually all coastal states have implemented at least one MPA ranging on the IUCN scale of protected areas categories from Ia (strict nature reserve) to VI (protected area with sustainable use of natural resources). Very rarely have marine plans been established in places without designated MPAs.

In terms of managing marine space, MSP is already catching up with MPAs as a management discipline with a major influence over the future use of our seas and oceans. Whilst countries have been designating MPAs for several decades, the first International Marine Protected Areas Congress (IMPAC I) was held in Geelong, Australia, in 2005. This was only a short time before UNESCO held the first International Workshop in 2006 on MSP, which is thought of as the birthplace of MSP internationally. The seminal text (*Marine Spatial Planning: a step-by-step approach towards ecosystem-based management*) on the subject, authored by Ehler and Douvere, was published in 2009 (Ehler and Douvere, 2009). And while there is no MSP dataset comparable to the World Database on Protected Areas that tallies the number of marine plans globally, in August 2018, approximately 70 countries were preparing or had prepared approximately 140 MSP plans at the national, regional or local levels (UNESCO, 2018). Approved MSP now cover almost 10% of the world's EEZs (UNESCO et al., 2017).

## How the aims of MPAs can be assisted by MSP

There are at least two ways that MSP can help create MPAs that deliver benefits to humans and nature simultaneously: first, by utilising MSP to locate MPAs and delimit their boundaries in such a way that they capture the most important ecological processes and productivity; and second, by utilising MSP to design zonation of MPAs to ensure maximisation of benefits. Examples of the former abound in the academic literature, but few organisations have implemented plans of strategically designed MPA networks. Nascent examples include North Ari planning in the Maldives (Agardy et al., 2017) and UK MPA planning. Another noteworthy example of a case in which MSP was used to create zonation within a large, multi-use MPA is the Great Barrier Reef Marine Park (Day et al., in prep). Recently, an MPA network was designed in South Africa based on systematic conservation planning and MSP (Haupt et al., 2017). And in the Northern Adriatic region of the Mediterranean, ecosystem services assessment and analysis of trade-offs has allowed MSP to identify priority areas for management and MPA designation (Gissi et al., 2018).

The involvement of relevant stakeholders, who ultimately need to be on board in order to ensure the efficacy of MPAs, can also be enhanced by MSP. The reason that this is the case relates to how people perceive MPAs versus MSPs. The general perception is that MPAs serve conservation purposes; they are often cast as tools for safeguarding nature, not nurturing people. In contrast, MSP is often perceived as a process that can lead to economic development and a blossoming Blue Economy (commonly understood to be the sustainable use of ocean resources, for economic growth, improved livelihoods and jobs, and ocean ecosystem growth (World Bank, 2018b)). In fact, many governments commit to funding MSP processes because they anticipate that the resulting plans will unlock the 'blue growth potential' of their territorial seas and EEZs (Howard, 2018).

MSP can therefore encompass a significant stakeholder-driven process. This often requires that those involved not only set out their ambitions for marine space as well as their concerns, but also engage with other stakeholders. This creates a shared understanding, if not agreement, of different stakeholder requirements on how marine space should be used. Ultimately, a better and shared understanding of competing needs should aid marine management, including that applicable to MPAs. Stakeholder involvement is key for societal acceptance of management measures. Effective stakeholder engagement can smooth MPA designation and the designation of boundaries, along with regulations and management regimes within them. The engagement of people is a good starting point and should not be undervalued.

Many MPAs have been designated opportunistically, and though they may have conservation value, they are sometimes not in the optimal place to confer either significant conservation benefits or benefits to humans. Many large MPAs are in areas of historically low pressures; therefore, interventions often do not result in tangible environmental improvements. These large and relatively pristine MPAs in remote areas do have value in future-proofing (Leenhardt et al., 2013), but it must be recognised that this potential future-proofing may be undermined by climate change. The marine environment is under pressure, and for environmental gains to be made, interventions are generally needed to bring about change.

Similarly, many MPA networks established within a country or across a trans-boundary area are developed for very specific purposes of protection of species or habitats. These MPA networks thus have conservation value but limited wider value – i.e. limited recovery of seas/ocean areas that were degraded and limited enhancement of benefits to human users.

The most egregious examples of opportunistic MPAs, established with little ecological understanding, have been of limited utility and rarely demonstrate the effectiveness potential of well-designed MPAs (Agardy et al., 2016). The drivers behind such rushed declarations include a country's need to keep commitments made under international treaty negotiations (e.g. Aichi Targets of the CBD), commitments made under soft law or declarations made in public fora (e.g. Sustainable Development Goal declarations), or even domestic politics and legacy concerns (Leenhardt et al., 2013; Rife et al., 2013).

Enduring effectiveness of MPAs is challenged by a number of factors. In some cases the MPAs can actually enhance some threats while abating others – an example is the case of displacement of native species in the Mediterranean by invasive species, which is sometimes thought to have been enhanced by the establishment of no-take MPAs that act as stepping stones for alien species (Giakoumi et al., 2016). In other cases MPAs can act as islands of protection, but without enough impact to prevent the seas in which they sit from spiralling downhill. Some MPAs can also be impacted from land-based activities, and interventions outside of the MPA may be needed to enable the MPA to deliver its intended outcomes. This calls for more integrated marine management and is a further argument for integrated coastal zone management and MSP to go hand-in-hand with MPA establishment (Agardy et al., 2011).

MSP can aid in greater integration of management activities within an area, and a more holistic approach: first, because MSP is meant to include a number of different sectors in spatial management, and second, because MSP is meant to have at its foundation the ecosystem-based approach to management that recognises ecological and human connections across vast landscapes/seascapes (Douvere, 2008; Ehler and Douvere, 2009). Using MSP to identify where new MPAs should be designated requires an understanding of the broader ecosystem ecology, and is usually based on some sort of gap analysis that looks at not only the existing array of protections, but also their effectiveness.

## How MPAs can assist in the delivery of MSP

The general goal of MSP is to steer marine use in a direction that is sustainable. This may be accomplished by limiting uses that are degrading, enhancing or expanding uses that are not, and treating interconnected ecosystems in such a way that linkages are maintained and ecological processes continue to provide benefit flows to communities and countries. MSP and the ecosystem-based management approach it encompasses are the foundation for emerging Blue Economies around the world.

By protecting key pockets of biological diversity and key ecosystem processes, MPAs can serve as a foundation for continued delivery of the things that people value in coastal and marine areas: resources such as fisheries and minerals, recreational values, sites for tourism activities, cultural and spiritual values, etc. (Jones et al., 2017; Lillebø et al., 2017). In essence, MPAs can act as the blueprint for continued ecosystem functioning and delivery of ecosystem services — but only when MPAs are designed carefully, systematically and with the big picture in mind (Agardy et al., 2011). MPAs can act as refugia (Green et al., 2014), insurance policies, and the bank of natural capital through which people can live off the interest. MPAs can also, importantly, enhance production through spillover, thus increasing value over a wider area (2009; Harrison et al., 2012; Roberts, 2012).

The EU Blue Growth strategy supports the goals of the Europe 2020 strategy for smart, sustainable and inclusive growth. Blue Growth seeks to support sustainable growth in the marine and maritime sectors as a whole. Maritime economic activities are supported by marine ecosystem services in combination, or not, with abiotic outputs from the marine natural capital (Lillebø et al., 2017). In order to balance concurrent sectoral interests and achieve sustainable use of marine resources, there is the need to consider the ecosystem's capacity to provide the required marine ecosystem services. Blue Growth options require navigating trade-offs between economic, social and environmental aspects. Fundamentally, MPAs may ensure the continued maintenance of ecosystem services that then enables marine development to be conducted in a measured manner (Agardy, 2019).

In line with global targets agreed under the CBD (1992), the number of MPAs is increasing rapidly, yet socio-economic benefits generated by MPAs remain difficult to predict and under debate (Mizrahi et al., 2018). MPAs often fail to reach their full potential as a consequence of factors such as illegal harvesting, regulations that legally allow detrimental harvesting, or emigration of animals outside boundaries because of continuous habitat or inadequate size of the reserve (Agardy et al., 2011). Edgar and colleagues have summarised the elements that contribute to MPAs that successfully meet conservation goals, based on analysis of 87 MPAs investigated worldwide: MPAs designed as no take, or restricting all extractive activity, MPAs that are well enforced, MPAs that have been established for >10 years, MPAs that are large (>100 $km^2$), and MPAs isolated by deep water or sand (Edgar et al., 2014). These results, and the findings of other assessments of MPA effectiveness in delivering broad positive outcomes, suggest that MPAs should be carefully planned, sufficient in their coverage, regulations and enforcement, and taken as a part of broader planning and management that can be achieved with MSP.

## From data to information, to insights, to decision-making

Critical to ensuring sustainable use of marine resources is a basic understanding of five parameters: (1) where resources occur throughout marine space, (2) how, if at all, the location and quantity of those resources changes over time and (3) how the resources are exploited, (4) the impacts on the wider environment that arise from that exploitation and (5) who is doing the exploitation. An additional important element to consider is why a resource is being exploited in any particular way. This information provides the context for the resource use and provides resource managers with important insights that enable them to engage with the resource users, understand the concerns of stakeholders, empathise with them and ultimately develop management measures, if needed, that are appropriate and will be accepted, or at least understood, by the resource users.

Our collective understanding of the state of the marine environment and the pressures to which it is exposed is furthered by the work conducted to underpin MSP development and MPA designation as well as the subsequent assessment of: (a) the effectiveness of marine plans in delivering their objectives and (b) the condition of protected species and habitats within MPAs.

Accountability and transparency throughout the designation process for MPAs and marine plan adoption can be secured through legislation. Legislation can require those responsible for designating MPAs or making plans to: consult in a meaningful way with stakeholders; set out options for achieving the objectives for the MPA or the marine plan and; and set out potential management measures. Legislation can require management bodies to report on the condition of MPAs as well as the efficacy of marine plans and the policies within them. The very fact that there is a requirement to report on the effectiveness of MPAs or marine plans should drive improvements in management and, therefore, protection levels. For reporting on MPA designation or marine plan effectiveness to be meaningful and enlightening, condition monitoring of the marine environment will often be required, which further increases the knowledge base upon which to make management decisions (Pomeroy et al., 2005; Day et al., 2002; Bennett and Dearden, 2014).

A meaningful stakeholder engagement process can increase awareness of the MPA or marine plans and how they may impact the stakeholder. Increased understanding of the process by which end products are delivered can help ensure that difficult discussions between managers and stakeholders are then focused on management measures rather than the processes (i.e. disagreement on the evidence – such as data layers – that is used to base management decisions on) that have led to management measures being developed and introduced. In essence, individuals or stakeholder groups may not agree with or like the outcomes, but can understand how they have been arrived at. If those outcomes are borne out of a transparent, accountable and evidence-based process, then there is an increased likelihood of acceptance for the outcomes and therefore voluntary compliance with management measures and, ultimately, success of the MPA or marine plan.

The MSP process whereby the spatial and temporal distribution of marines resources and marine activities are captured and presented in a way that can be understood by the various stakeholder groups (e.g. from fishers to policy makers) is critical. The ability to effectively present the evidence base and the confidence in that information upon which decision-making will rely are extremely important.

As in many other situations, there are in essence three ways to proactively bring about change in the marine environment. These are to start, modify or stop an activity. This can be achieved through various mechanisms ranging from voluntary agreements to incentives, compensation schemes, formal regulation, enforcement and sanctions. Bringing about change invariably incurs costs, to those impacted by the change and also those bringing about change. Therefore, this provides a further imperative to understanding not only how the marine environment is being used, but also how it is changing.

## More than the sum of its parts

Whilst the origins of MPA designation within any one country will have been the establishment of a single or small number of MPAs to protect a specific species or habitat, there has been a gradual shift to the development of MPA networks within a single country and at a regional seas scale. One example of this is the Regional Seas Convention for the North-East Atlantic (OSPAR), which facilitates the development of an ecologically coherent MPA network in the North Atlantic. Creating representative networks of MPAs as part of an ecosystem-based management approach is generally advocated to protect the full spectrum of marine ecosystems and vulnerable species (Johnson et al., 2014). This shift towards the development of MPA networks is directly linked to the CBD Decision X/2 of COP10 (2010), which requires that by 2020, 10% of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscape and seascape. As Johnson et al. (2014) point out, ecological coherence in itself is not the end-point or 'Holy Grail'; rather, the ambition is ultimately effective management of such networks to secure conservation objectives.

Just like a single MPA that protects a single species or habitat directly, but fails to provide protection for important supporting habitats or species within that MPA, a network of MPAs also has the potential to overstate the level of protection that it can provide. Whereas a network of MPAs may provide protection for habitats, sessile species or for mobile species at particular stages in their life cycle where their movements are spatially constrained, networks provide limited protection for more mobile species (see Evans (2018) for a detailed analysis of the strengths and weaknesses of area-based management versus issue-based conservation measures). The conservation benefits of an ecological network of MPA are also unlikely to be achieved unless the ecological connections between MPAs are maintained.

Recognising this, the use of MSP and the regulatory/licencing mechanisms that are either embedded or ancillary to it can provide the joined-up management that provides the levels of environmental protection for vulnerable mobile species and the water-borne life-history stages of species that form the habitats of MPAs. Even in the situation in which an ecologically coherent and extensive MPA network is in place and is effectively managed, there is a growing realisation that this network provides a starting point for protection and the raising of environmental protection. It is now recognised that the remainder of the marine environment is an important source of natural capital. It is worth noting that the spatial coverage of MPAs is not a true reflection of the actual spatial footprint of protection; in many cases, an MPA only confers protection to a proportion of the benthic habitat within that MPA. Within the unprotected area of a country's marine space there will be areas of high ecological value that could and should be protected. MSP provides a process and a mechanism to achieve this. MSP can aid in the identification of activities that interact spatially and temporally with important ecological processes and migratory routes. MSP and its associated management mechanisms can ensure that the ecological connectivity of MPAs is protected by minimising harmful interactions. As such, MSP is an essential tool for delivering an ecosystem approach and should add value to existing management measures for the marine environment (Crowder and Norse, 2008; Gilliland and Laffoley, 2008).

## Moving the goal posts or upping the game?

The marine environment and its use are highly dynamic; therefore, static management measures may need to be augmented in order to optimise both marine use and environmental protection. Recognising this, there is increased focus on securing the health of the wider seas by governments, their advisors and regulators. This pivot (from designating MPAs and introducing management measures within them, to wider seas management measures) may be viewed by some sectors as moving the goal posts in terms of environmental protection. It is likely that further constraints on their activities will be deemed unpalatable yet inevitable. Attention needs to be paid to these sectors and their concerns to ensure that wider seas management measures are workable and effective. Many of the types of measures that can be introduced rely upon marine users working collaboratively in order for the measures to be successful.

### Dynamic ocean management

The ability to protect Migratory Marine Species (MMS), which constitute a large portion of marine taxa, is becoming more sophisticated. Management measures have historically looked to (a) protect specific habitats that species rely upon during specific periods of their life cycle through conventional MPA designation, and (b) licence activities that can have an adverse effect on the species, requiring marine activities to design their operations to avoid, minimise, mitigate or compensate for their impacts.

Dynamic Ocean Management (DOM) is now gaining attention as a useful management tool (Lascelles et al., 2014) to augment existing management responses. DOM is defined as management that rapidly changes in space and time in response to changes in the ocean and its users through the integration of near-real-time biological, oceanographic, social and/or economic data (Maxwell et al., 2015). These changes in environmental parameters (such as sea surface temperature) could be instigated as a result of predicted or measurable changes in those parameters. DOM can refine the temporal and spatial scale of managed areas, thereby better balancing ecological and economic objectives. DOM was developed out of the need to identify a more responsive management tool to protect MMS. Marine users can now modify their activities to minimise the impacts of their activities on the marine environment when they are presented with the information on which to act. Maxwell et al. (2015) sets out how passive acoustic buoys and aerial surveys are used to detect the real-time presence of North Atlantic right whales (*Eubalaena glacialis*) along the US East Coast to reduce lethal ship strikes of this critically endangered species. DOM area locations are distributed to ship captains via mobile applications to alert them to the whales' presence and to recommend avoiding areas or reducing speeds when whales are present (Wiley et al., 2013; Conn and Silber, 2013; Silber et al., 2012). Dynamic management areas are also paired with traditional seasonal closures of the whales' breeding grounds (Van Parijs et al., 2009).

DOM also has a significant and growing role to play in fisheries management. Bycatch of threatened species in capture fisheries remains a major impediment to fisheries sustainability. Management measures designed to reduce bycatch can result in significant economic losses and even fisheries closures. Static spatial management approaches can also be rendered ineffective by environmental variability and climate change, as productive habitats shift and following the introduction of new interactions between human activities and protected species (Hazen et al., 2018; Little et al., 2015). Increased accessibility to environmental data, computing power and the need to reduce the impact of fishing activities and to minimise the catch of choke species (a term used to describe a species with a low volume quota, which, if reached, would lead to vessels having to stop fishing even if they still had quota for other species) has also led to the development of both Real Time Closures (RTCs) and Real Time Incentive (RTI) fisheries management.

## Real time closures

Real-time spatial management in fisheries, a type of DOM, uses near real-time data collection and dissemination of information to reduce susceptibility of certain species (or age classes of that species) to being caught in mixed fisheries (Woods et al., 2017). RTCs for managing fisheries bycatch and discards can be implemented under either a co-management or a self-governance approach (Little et al., 2015). Real-time catch and discard information is shared among fishers to incentivise and encourage vessels to leave areas of high bycatch which may include protected species. RTCs can therefore augment the protection that static MPAs provide MMS.

## Real time incentives

Under an RTI fisheries management approach, fishers are allocated fishing-impact credits to spend according to spatiotemporally varying tariffs. Fishers choose how to spend their credits, e.g. by limited fishing in sensitive areas and fishing longer in less sensitive areas (Kraak et al., 2012, 2014, 2015). One can argue that DOM, RTC and RTI are examples of recent MSP advances as they all seek to manage the use of marine space. The use of these dynamic management tools can deliver significant environmental benefits to the wider environment;

however, their use has the potential to increase conflict between different marine stakeholder groups as activities are moved or displaced as a result of the dynamic management.

## Future-proofing marine protection

### Scenario analysis

MSP could be used as a tool in several ways to aid the delivery of environmental protection and sustainable use of the marine environment. MSP provides a snapshot of what activities are occurring where and when in the marine environment. It also reflects how marine space is allocated at that particular time. This information provides a baseline for discussions on how the marine environment should, or could, be both developed and protected in the future. An important task of natural resource management is deciding between alternative policy options, including how interventions will affect the dynamics of resource exploitation. Yet predicting the behaviour of natural resource users in complex, changeable systems presents a significant challenge for managers (Davies et al., 2015). To develop a better understanding of how the existing situation may change, which in turn may impact the effectiveness of policy options, varying scenarios can be developed (although to date, most spatial planning processes have not selected specific outcomes, such as preferred use scenarios (Collie et al., 2013)). Scenarios can be simplistic in that they may only consider how a particular marine sector is likely to change over a period of time (i.e. offshore wind), or they can be far more complex. In the Celtic Sea, three scenarios (ABPmer and International, 2016) were considered as part of a project to bring together key stakeholders to support the implementation of environmental and maritime policy. The scenarios were:

**1)** *Business as usual:* The marine economy develops as expected. There are no major changes in attitudes, priorities, technology or economics. Economic growth remains the priority, with society and industry reluctant to adopt environmental policies that radically change the status quo.

**2)** *Nature at work:* The environment takes centre stage. Population growth, new technology and making the most of a healthy environment are the driving forces. Environmental protection is strong, with an extensive network of strongly managed protected areas.

**3)** *Local stewardship:* Society seeks greater local self-sufficiency. More decisions are taken locally and there is increased pride in local produce. Environmental policy varies across the region as decisions reflect local issues and concerns. Tourism and recreation grow strongly as people choose to holiday at home.

Each scenario considered variables such as population, economic equality, technical innovation, globalisation and environmental policy. Furthermore, each scenario highlights that, as is the case now, there will be trade-offs, winners and losers in terms of economy, society and the environment.

Scenario analysis is helpful as it highlights difficult decisions that will need to be addressed by policy makers in the future, and in some situations can provide a time frame for those decisions to be made within. Scenario analysis can therefore guide stakeholder discussions and prioritise research to inform decision-making. In the context of marine protection it can help managers anticipate the future use of existing MPAs. This is especially useful when the current effectiveness of the MPA in delivering its objectives is known because this will enable managers to consider how management may need to change to continue delivering effective management. Anticipating not only the future needs of existing marine space users but also those needs of potential new users enables policy makers to proactively allocate and prioritise in a hierarchy how marine space should be used in the future. Scenario analysis may also aid the resolution of potential problems relating to cumulative environmental impacts proactively, i.e. recognising that multiple users of marine space will have a future impact on protected species and therefore measures can be put in place to mitigate or offset those impacts before they are realised.

## Improving feature-based protection

Many, if not most, MPAs have been established to protect specific species or habitats (features) rather than all species or habitats within that particular site (Solandt, 2018). However, there is increasingly a realisation that this approach can have significant limitations, especially where the MPA is of limited size. In many cases, the boundaries of MPAs are drawn close to the protected features of the sites, recognising that management measures to protect the features will impact marine users; therefore, to ensure that the designation of the site is secured, efforts are taken to provide the required level of protection using the smallest spatial footprint possible. For this reason, MPAs may not deliver significant 'additional' protection other than that directed at the formally designated features. Ironically, as some MPA networks are deemed to be nearing completion – in that they provide an ecologically coherent network of MPAs – there is concern that climate change is altering the distribution and abundance of the very features that the network of MPAs has been designated to protect. This may not be a concern if the network has adequate network connectivity and functioning as it was intended to.

In some instances, protected features may migrate out of the MPA, requiring: (1) the MPA boundaries to be altered, (2) the MPA to be de-designated and (3) a replacement MPA to be identified and designated, or (4) those features that were to be protected to be added as protected features within other MPAs where those features are not already protected. In the latter case, suitable management measures to protect those previously unprotected features would need to be introduced. Where this happens, stakeholders may consider that the goals of the original MPA have altered significantly and thus they may be less inclined to support the MPA and the management regulations applicable to it.

This feature-based approach makes it hard for the MPA and the protection it provides to adapt to changing environmental conditions (e.g. temperature, salinity, acidification, sea level, wave exposure, ocean currents). Conservation management approaches are often primarily focused towards the designated features, with no specific conservation measures applying to other (non-feature) areas within the site boundary. Areas of 'non-designated habitat' may still harbour significant biodiversity interest and support the designated features by providing additional resilience to human impacts. Recognising this, the UK Government has set out its intention to move to a whole-site approach to protect sites of the greatest biodiversity interest (UK, 2018a).

Through the incorporation of data layers that indicate how climate change and human activities are likely to alter, MSP can show how pressures on protected features may change over time. These data layers may also highlight where protected features are likely to migrate or be displaced to over time. Once this information has been discerned, then a determination can be made as to how effective the existing protection will be in the future and how the MPA network will need to evolve in order to continue providing adequate, or indeed improved, levels of protection.

There is a growing recognition that there is a need to increase the protection of features outside of MPAs. One method of achieving this was recently employed by the Scottish Government. It adopted a list of 81 Priority Marine Features (Wilding et al., 2016.). To produce the list, species and habitats on existing conservation schedules were assessed against criteria that considered: the abundance/extent of the feature, its conservation status (threatened, in decline, etc.) and the functional role that the feature plays. The list will be used to: focus future conservation action and marine planning, direct research and education, and promote a consistent approach to marine nature conservation advice. The Scottish National Marine Plan states that ‘Development and use of the marine environment must not result in significant impact on the national status of Priority Marine Features’, thus conferring these features with additional protection (Scottish Government 2015b).

A focus on a feature-based approach is not without its problems, especially when this is extended across an MPA network. The requirement to provide timely and accurate conservation advice on features, their location and condition is resource intensive (Rush and Solandt, 2017). In England, for example, conservation advice packages for MPAs set out the protected features, the objectives for the site, the conditions of the features and the sensitivity of features to pressures exerted by different activities.

## The need for integration

### Co-location and space partitioning: considering displacement of activities

It is clear that competition for marine space is increasing. With this, the interactions and potential conflict between different activities and the marine environment is also increasing and is set to continue to do so. The drive to designate MPAs (especially those with higher protection classifications) results in the removal of marine space available for certain activities. If the area to be designated was identified separately from a wider MSP process, it is possible that the use of marine space may be suboptimal. It may have been the case that the desired protection could have been achieved through the designation of other areas that would have had a reduced impact on other activities. The suboptimal use of marine space may also work the other way, in that the planning of the MPA may not maximise conservation goals when concessions are being made to users to maintain their access to space and resources. Under this scenario, a decision to introduce either an MPA or an MPA network may require that the protection is introduced in such a way that existing activities are not, or are minimally, impacted. This approach may be taken for a variety of reasons — stakeholder support (or lack of it) for the process, strength of different stakeholder groups or political will. A lack of support for the MPAs may subsequently result in a lack of compliance with management measures.

It must be noted that the use of marine space need not be mutually exclusive (all MPAs within England, for example, are considered multiple-use sites) (Solandt, 2018). There are many instances in which activities can either be co-located (both activities occur in the same location and at the same time) or managed to ensure that a different users of the same piece of marine space can be accommodated at different times of the year (Stelzenmüller et al., 2016; Yates et al., 2015; Hooper and Austen, 2014). The ability to co-locate activities and manage this process is potentially more difficult where there is a lack of a clear policy steer on which activity is to take precedence. In this situation, there is the danger that the engagement between differing stakeholders becomes adversarial rather than cooperative. Where activities cannot co-locate, two issues arise: (1) increased environmental impacts may result from a displaced activity (Vaughan, 2017) and (2) potential financial compensation for displacement of an activity may be raised.

The impacts resulting from fishing effort displacement have been considered within fisheries management, but the displacement of fishing and other activities is rarely considered in MPA planning in depth. McLeod (2014) defined displacement as: 'the changes in fishing behaviour and patterns that could occur in response to new management measures'. Changes in fishing behaviour could be 'the adoption of a new fishing method, or target species, or stopping fishing', whereas changes in fishing pattern could be 'moving to other fishing grounds near or far'. It is likely that activities (that exert pressures on the marine environment) other than fishing that are displaced from an area through competition for marine space, or in response to explicit MSP measures, potentially expose protected species and habitats to new, or additional, pressures within MPAs and also the wider seas. This mirrors the fishing effort displacement threat (Vaughan, 2017; ABPmer, 2017). Recognising that MSP may lead to the displacement of activities enables a strategic view to be taken on how best to avoid, minimise, mitigate or compensate for the impacts of displaced activities on protected species, habitats and the wider environment. However, assessing and managing the cumulative impacts of human activities on the environment remains a major challenge to sustainable development (Willsteed et al., 2017).

To ensure that co-location is effective, significant efforts must be expended ensuring that different marine space users are engaged in meaningful and timely dialogue so that their needs are understood and are accommodated as far as possible (Hooper and Austen, 2014; Vaughan, 2017; Gray et al., 2005). In the case of offshore renewable energy, this may involve the use of fisheries liaison officers employed by developers.

Where multiple activities seek to operate in the same marine space, this may result in the displacement of fishing activities either voluntarily or involuntarily. In the Netherlands, for example, fishers are excluded from fishing within offshore windfarms, yet in the UK, restrictions on fishers may be limited to construction phases of the development. While not prohibited from fishing within windfarms once they are operational, some fishers may not wish to continue fishing within windfarms due to safety concerns or a requirement to incorporate differing fishing patterns or gear types, and therefore they look to operate elsewhere (Christie et al., 2014; Hooper et al., 2015; Mackinson et al., 2006).

The development of comprehensive data layers to inform MSP can aid discussions on co-location and also compensation payments (e.g. between fishers and offshore wind developers), as these can provide a robust and third party evidence base, i.e. information on how and when fishing vessels use marine space, derived through satellite vessel monitoring systems (Campbell et al., 2014), as well as verified catch and landing data that can be linked back to sea areas, e.g. International Council for the Exploration of the Seas statistical rectangles.

Marine plans that have high levels of spatial specificity within them bring to the fore the issues of equity, justice and power. Consideration of compensation for those individuals whose activities may be altered as a result of policy decisions may also be required. These issues have been considered extensively in the context of MPAs (Jones, 2009), yet they are likely to increasingly occur outside of MPAs as demand for marine space increases.

MSP is developing as a management tool, and there is increasing interest in exploring how proactive co-location of offshore developments and protected areas can develop (Yates et al., 2015; Christie et al., 2014; Ashley et al., 2014). However, developers have significant concerns, as to date they have sought to actively avoid, where possible, interactions with the MPA network. This interaction can increase development costs, time scales (project inception to operation) and the general regulatory burden, whereby marine users need to demonstrate that they are not having an adverse impact on the protected species or habitats of the MPA. In many cases, this requirement is ongoing. There is also the concern that declines in the conservation status of protected species or habitats within MPAs that are co-located with developments may therefore require action to be taken on behalf of the developer to curtail their activity, therefore placing investment returns at risk (Christie et al., 2014).

## Political imperative

Santos states that 'large investments in MSP around the world have resulted in many planning processes that have not been implemented, or will likely not be implemented, because of resource constraints or sociopolitical and "realpolitik" factors' (Santos et al., 2019).

Managing the need for marine space and the conflict that is sometimes generated by competing or incompatible interests requires decisions to be made on how that marine space should be used. It is usually the case that different marine sectors are managed by differing regulators with departments or ministries for shipping, defence, environment, fisheries, coastal resources, protected areas and energy (Lloyd et al., 2011). In these situations, there is the danger of considering the activities under the remit of these regulators in isolation. Regulators are assessed on their performance in managing activities under their responsibilities by government and also stakeholders. As such, these regulators have little to gain by stepping back and taking a view on the wider use of marine space as this may result in difficult decisions to be taken and implemented. It may also result in regulators ceding their power to other departments or ministries. MSP provides both a mechanism and outputs that can encourage and compel a more holistic approach to the use and management of marine space than could or would otherwise be taken by individual regulators acting alone. Where plans are not sufficiently spatially prescriptive and rely on overarching policy statements that do not explicitly set out winners and losers, regulators can make decisions in accordance with the plan, but this may not represent the best use (depending on your perspective) of that marine space.

Unless there is a significant push for an integrated approach to MSP from the outset, without potential changes to a nation's marine governance arrangements to facilitate this (such as the development of an overarching maritime regulator), it is likely that marine plans will be initially developed with non-spatial, sectoral polices prevalent. Where policies are not spatially explicit enough and apply to significant proportions of the area covered by the marine plan, the policies may lack value in that they do not aid decision-makers. Furthermore, unless there is integration of these policies and their coverage, these policies may either compete with each other or be so broad in scope and interpretation that prioritisation of marine space for certain activities does not occur. Ultimately, it is likely that MSP will require a political process that leads to the allocation of sea space to meet social, ecological and economic objectives (Qiu and Jones, 2013) to resolve conflicts through prioritisation and prescriptive policies (Sander, 2018).

## Conclusions

Although the two disciplines of marine management embodied by MPA planning and MSP are distinct, each is focused on how we allocate and use our marine space. Practitioners in these disciplines will naturally have different marine management goals, and will seek to achieve these goals using their experience, skills and knowledge. Yet, these professionals will invariably liaise with the same stakeholder groups. This can be confusing for stakeholders and therefore lead to suboptimal outcomes because of stakeholder fatigue and lack of engagement, especially where there is a lack of clarity on the problems practitioners are trying to be solve, over what time frames and what the outputs will be from either process.

For MSP, marine protection is but one of the key drivers, and may not be the main driver, whereas with MPA management and designation, marine protection is the key driver. Because of this, understanding the aims and objectives of those working on MPAs and MSP is important when considering how best environmental benefits can be secured through collaborative working. This is important because working in an interdisciplinary manner is required if publicly acceptable and effective marine management is to be introduced that will provide the long-term protection and enhancement of the marine environment. We do not have a blank piece of paper when drawing up MPAs and/or undertaking MSP within marine space. Developers such as those engaged in aggregate dredging may view MPAs as a restriction on their existing or future activities, whereas conservationists may view current aggregate dredging sites or leases as a constraint when seeking to locate an MPA.

The act of designating MPAs has the effect of displacing and condensing marine developments and activities in the remaining marine space, which has the potential to increase stakeholder conflict. Designating MPAs may have the unintended effect of weakening management in areas outside of MPAs, by lending credence to the view that the remainder of marine space can now be developed with limited oversight. Yet it is vitally important to ensure that the wider marine space is managed so that it is not degraded and thus fails to provide important ecological services and MPA connectivity. MPA designation continues across the globe; continued designation of these sites is important as they play a key role in protecting core areas of conservation benefit. However, there is the danger that human impacts in general and in particular marine use and the impacts from this use, continue to increase, and therefore lead to the degradation of these MPAs. The risk therefore is that once designated, we expend a disproportional amount of management resources trying to manage change within MPAs and MPA networks at a local scale when we need to be taking a broader view and ensuring that impacts in the wider environment are understood and addressed – in essence, where should we deploy our limited management resources for best effect? Recognising that MPAs might be best treated as one part of a unified conservation strategy means that MSP needs to be harnessed to create better, more durable MPAs. Conversely, incorporating MPA planning into MSP can reduce conflict for marine space through optimisation of that space.

There is real potential for MSP to provide the mechanism for driving change in the protection and use of the marine environment as our global understanding of the marine environment, its importance and its interconnectedness develops. This is because marine plans generally have: (a) spatial elements that identify how marine space is currently used and could be used in the future; and (b) a requirement to update the plans in terms of not only how marine space is being used and is anticipated to be used, but also how the marine environment has changed or is likely to change. These requirements enable plans to be updated to reflect the economic, environmental, social and political requirements at any particular time. When MPAs that are located within a marine plan undergo assessment as to their effectiveness, a decision can be taken when updating the plan to reflect recommendations from this assessment where appropriate. Similarly, MSP may highlight that marine use is evolving in a manner such that additional measures may be required within MPAs, or indeed that marine use is changing such that existing measures are no longer appropriate. Marine plans are likely to evolve slowly because the process to develop, agree, adopt, implement and review them is generally lengthy, resource intensive and iterative. Thus, plans provide the users of marine space with some degree of certainty regarding their use of that space.

The demand for marine space and the impacts on the marine environment continue to increase. This requires the increased proactive consideration of these demands. MSP provides a mechanism to take forward stakeholder/societal discussions on what appropriate use and protection of the marine environment should be, at both a micro and a macro level. Regardless of the level considered, managing marine space is complicated, and often the responsibility for doing this is shared across different governmental regulators and advisers at a local or regional level, and across different governments and global institutions at a macro level. MSP, by its very nature, requires, encourages and/or compels these different bodies to share information and their vision on how marine space should be used, which should enable a view to be developed (locally, regionally, nationally and internationally) that also sets out how the marine environment should be protected and enhanced over the short, medium and long term. It is clear that MPAs and MSP are both key elements required to achieve this vision.

## Disclaimer

The views and opinions expressed in this report and its content are those of the authors not of Natural England which has no responsibility or liability for any part of the chapter.

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# Marinising a terrestrial concept: Public money for public goods

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ABSTRACT

Exiting the EU allows the UK to unilaterally change the frameworks that govern its environment and natural resources. This opportunity is timely given the urgent need to address the biodiversity and climate emergencies, and deliver the necessary policy changes to meet associated international agreements. The UK's divergence from EU environmental policy has already begun. The new Agriculture Act uses the concept of "public money for public goods" (PMPG) to seemingly revolutionise direct agricultural subsidies, replacing the much-maligned funding mechanisms under the Common Agricultural Policy and making the provision of their replacement dependent upon actions delivering societal gain. However, the potential benefits of transposing this concept to marine fisheries and aquaculture are yet to be recognised despite similar criticisms of funding mechanisms under the Common Fisheries Policy. This paper therefore considers the key distinctions between our use of marine and terrestrial environments and how PMPG could be applied to fisheries and aquaculture. The findings suggest that some forms of aquaculture are well-placed to benefit from a 'marinising' of the PMPG concept. Currently, capture fisheries, because they do not have ownership over marine space and interact with the marine environment in an extractive manner, have a greater challenge to adapt their business models to receive public money under this framework.

## 1. Introduction

On 1st January 2021, the United Kingdom (UK) became an independent coastal state following the end of the transition period of exiting the European Union (EU). In doing so, it regained the ability to make unilateral decisions regarding many of the policies that regulate how its environment and natural resources are managed. While EU-exit will impact the governance and management of many sectors of the UK economy, the agricultural and fisheries sectors will perhaps be most profoundly affected. Political saliency has been heightened as fisheries were at the forefront of the negotiations in the run up to the UK leaving the EU (Popescu and Scholaert, 2021). Importantly, EU-exit includes the withdrawal from the Common Agricultural Policy (CAP) and the Common Fisheries Policy (CFP), and the associated funding mechanisms that determine how the corresponding sectors are financially supported. EU State aid rules no longer apply in the UK[1] and the UK Government has made clear that it intends to establish a new UK subsidy regime (Dept of BEIS, 2020; Dept of BEIS, 2021). The UK-EU Trade and Cooperation Agreement (TCA) ensures that the EU and UK will each have in place its own independent system of subsidy control (with neither being bound to follow the rules of the other) (EU and UK, 2020). It also offers the UK the opportunity to determine the principles that support how its natural resources are managed and how direct public payments, or subsidies, are provided to farmers, fishers and aquaculturalists alike - and to align with a more environmentally focused vision for the UK's future.

Opportunity for policy reform and simultaneous commitments by the UK government to seek a sustainable and environmentally focused future are timely. The climate and biodiversity emergencies are well

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[1] It is acknowledged that EU state aid rules still have a bearing to some extent in Northern Ireland due to the operation of the Northern Ireland Protocol to the Withdrawal Agreement - Cabinet Office (2020) *The Northern Ireland Protocol*. London, UK: Cabinet Office.

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documented, prompting an urgent response by the government. Its obligations on the international stage under the Paris Climate Change Agreement and the Sustainable Development Goals of the United Nations (SDGs) further mandate this urgent need for reform (UN, 2015; UNFCCC, 2015). Adapting to and mitigating the impacts facing the UK requires an unprecedented response in pace and scale, in both terrestrial and marine environments. In addressing these obligations, consideration is required of how the UK's assets can support adaptation and mitigation; how this response is best implemented; how the balancing of the three pillars of sustainability (economic, social and environmental) will be achieved; and how to reform policy equitably and justly.

Such considerations of how terrestrial natural assets are best used to sustainably benefit wider society have already begun. The Department for Environment, Food and Rural Affairs (Defra), Command Paper on the future of food, farming and the environment included an entire chapter on PMPG, with environmental enhancement a priority (DEFRA, 2018c). The requirement for a domestic replacement for the CAP led to the Secretary of State for the Environment securing the incorporation of the PMPG concept into the proposed Environmental Land Management (ELM) scheme (DEFRA, 2020a), through which farmers will be paid for producing public goods (Lewis, 2021; DEFRA, 2020a). The Sustainable Farming Incentive is the first in a package of ELM schemes, which will provide a mechanism for farmers to be paid for producing public goods (identified as including cleaner water, cleaner air and carbon reduction) (Lewis, 2021). This is in stark contrast to the CAP 'basic farm payment' where landowners were paid by area, with few other conditions other than simply owning land. Payments therefore benefitted the largest landowners most, rather than supporting principles of equitable distribution (Bateman and Balmford, 2018; DEFRA and Government Statistical Service, 2018). To date, over £3.34 billion of public money has been spent securing environmental improvements in land management in 2019 (DEFRA et al., 2019). Following the reforms, direct agricultural subsidies will no longer incentivise land cultivation but will instead be repurposed to deliver public goods or public benefits; payments to landowners and farmers are transitioning over time to prioritise environmental considerations.

For decades, these direct public income support payments to the private agriculture sector were the topic of much debate. On the one hand, the European Commission argued their existence in the CAP provided vital financial safety nets that allowed the agricultural sector to continue to produce food and products, particularly for certain forms of farming (Rizov et al., 2013). On the other, the lack of meaningful environmental conditionality meant they were seen to be contributing to the ongoing environmental degradation of land and waterways (Kirsch et al., 2017; Pe'er et al., 2020). With a dichotomy of views, the debate for the removal, redirection or reform of support had been lengthy. Furthermore, these payments masked other financial problems and resulted in high subsidy dependence in industrial agriculture across Europe. Against this background PMPG has been introduced into UK agricultural policy. However, its introduction as a concept appears at least to have enabled the continued overall level of financial support to the sector (Conservative and Unionist Party, 2019), while seemingly supporting the government's commitment to leave the environment in a better state than which it inherited (DEFRA, 2018b).

As in agriculture, such subsidies have been at the centre of a long-standing sustainability debate in fisheries (Sakai et al., 2019; Sumaila et al., 2019; Tipping, 2016). Indeed, even Adam Smith raised concerns regarding the sustainability of public money being transferred to the private fisheries sector (Smith, 1999). Given the transboundary nature of fish stocks and the global nature of fishing fleets, these concerns are not only domestic; the international struggle to address harmful fisheries subsidies has been ongoing for more than two decades. During this time there have been several attempts to form multilateral agreements on their reform, most notably via the ongoing World Trade Organization (WTO) negotiations, the Aichi Biodiversity Targets, and the SDGs, which commenced in 2001, 2010 and 2015, respectively. Yet, reformed rules for the provision of fisheries subsidies remain elusive, even though many of them have been shown to exacerbate environmental degradation and undermine biological sustainability within the EU (Skerritt et al., 2020).

In this light, this study suggests that the PMPG concept, as recently applied to direct public income support for UK farmers, should be explored and considered as a framework for reforming direct fisheries and aquaculture subsidies in the UK. Each of the fisheries administrations within the UK is responsible for developing their own domestic replacement to the European Maritime and Fisheries Fund (EMFF)[2] the current CFP funding mechanism for providing direct subsidies to the fisheries and aquaculture sectors. These new schemes and their future iterations provide opportunities to consider and incorporate PMPG. The ambition shown in the UK's course change from the CAP towards the ELM scheme could be mirrored in the UK's inevitable divergence away from the EMFF scheme of the CFP.

This article places the debate within the policy context of challenges and opportunities for fisheries, aquaculture and the marine environment in the UK. It then sets out what constitutes a *public good* in the context of the new UK Agriculture Act, before exploring what might constitute a public good in the marine environment. It then outlines the current forms of public money transfers under the outgoing EMFF and whether they could be considered to provide public goods, before outlining what the principles of fisheries and aquaculture subsidisation might look like under the concept of PMPG. Lastly, the paper discusses future policy, how the PMPG concept may be expanded, and how it can be implemented through a just transition. We discuss how this concept may be applied more broadly to address long standing concerns of how public money and natural assets are currently managed and whether they are done so for the benefit of all of society. Specifically, we suggest that the requirements for continued financial support under the replacements to the EMFF are reshaped and linked to the requirement to provide positive environmental outcomes and other public goods. Having commenced the exploration of PMPG in the marine environment we conclude by suggesting further areas of research.

## 2. Political context

EU-exit provides a unique opportunity to revisit environmental legislation and policy. In terms of the UK Government's current stance towards the environment, there is recognition that transformational change is required. The desire for the UK to be a world leader in the global response to the environmental crisis led to the Government introducing the legal requirement to achieve Carbon Net Zero (CNZ) by 2050, and subsequently a commitment to reduce carbon emissions by 78% by 2035 (compared to 1990 levels) (UK Gov, 2021a), the publication of the 25 Year Environment Plan (DEFRA, 2018b), and the successful bid to host the 26th UN Climate Change Conference (UK Gov and UN, 2021)). Tackling climate change and preserving the planet's biodiversity forms one of the four priorities for the UK's G7 Presidency in 2021 (UK Gov, 2021b). From an international marine perspective, through its leadership of the Global Ocean Alliance, and as ocean co-chair of the High Ambition Coalition for Nature and People, the Government is championing the 30 by 30 target, which advocates for the protection of at least 30% of the global ocean within Marine Protected Areas (MPAs) by 2030 (which aligns with global protection of at least 30% of land by the same year) (UK Parliament, 2021).

This desire to lead is exemplified in the concerted effort the UK has taken to address its carbon emissions and take advantage of an extensive Exclusive Economic Zone (EEZ), with favourable bathymetry and meteorological conditions to become the global leader in the development of offshore windfarms (OWFs) (Prime Minister's Office, 2020). The

[2] The overall EU EMFF budget for 2016–2020 was € 6.2 billion, with the UK allocated €243 million (of which €92.1 million was allocated to England for that period) (MMO, 2016).

rapid expansion of OWFs and their demand on marine space is currently the most obvious facet of the UK's Blue Growth agenda and are an example of the changing appreciation and recognition of the high value of the UK's marine assets. With increasing pressures from greater use leading to growing competition for space, comes increased scrutiny of environmental impacts occurring in an already degraded ecosystem but also increased recognition around the important ecosystem services that can be derived from its inherent assets (Ruckelshaus et al., 2013). Recently there has been a shift in focus from the terrestrial to the marine environment regarding the extent that carbon sequestration occurs (Green et al., 2021). It is already being advocated that restrictions on bottom trawling are required to maximise the extent that this ecosystem service can be provided (Sala et al., 2021). As this will require restrictions on fishing activities, this further raises the question as to how the UK's assets (natural and financial) should, or could, be used to benefit wider society. The expansion of OWFs, has coincided with attempts to increase protection of the marine environment with the designation and management of MPAs being the most obvious manifestation of this (the UK MPA network now consists of 371 sites covering 38% of the UK EEZ (Pow, 2021)) the other being the development of the UK Marine Strategy.

The ambition and the legal requirement to improve the state of the UK's wider seas and to deliver Good Environmental Status (GES) of its waters (by 2020) is established within the 25 Year Environment Plan, the Marine and Coastal Access Act (2009) and associated UK Marine Strategy Regulations 2010 in addition to the Fisheries Act (2020) (*Marine and Coastal Access Act, c.23*, 2009; DEFRA, 2018b; DEFRA, 2019; Fisheries Act, 2020). The most recent status update regarding GES delivery was conducted in 2019 with eleven of the fifteen elements reported on were either Red (GES not being achieved) or Amber (GES only partially achieved) (DEFRA, 2019). This update stated that the predominant human pressures preventing GES being achieved include commercial fishing. The marine environment has been degraded through the act of fishing and climate change is exacerbating the pressures on both fish stocks and the wider environment. This status update is clearly at odds with the ambition set out within the Fisheries White Paper - Sustainable fisheries for future generations (the precursor to the Fisheries Act 2020) (DEFRA, 2018a), to develop 'world class' sustainable fisheries.

As a result, access to these stocks and fishing grounds was a key area of tension during the TCA negotiation. It is noteworthy that the TCA has 'carve outs' for subsidies applicable to both agriculture and farming and that the UK is consulting on a domestic subsidy control regime. The WTO agreement on subsidies and countervailing measures (ASCM) applies to fisheries subsidies, as there is currently no sector specific WTO agreement in place akin to the WTO Agreement on Agriculture. Negotiations for a fisheries agreement under the WTO are ongoing (WTO, n.d). The UK therefore has an opportunity to consider how to address long-standing concerns regarding subsidy application in both sectors. This opportunity also comes at a time when scrutiny around the use of public finances will increase as the as the UK plans its economic recovery post-COVID-19 ('build back better' (HM Treasury, 2021) and establishes new policies and regulations following EU-exit.

Alongside traditional fisheries, the UK has natural conditions to support aquaculture (including the farming of oysters, mussels, scallops, and clams, using various techniques) while at the same time providing ecosystem services such as water filtration that can aid the delivery of GES. While the provision of ecosystem service benefits, from water purification to habitat creation and carbon sequestration have been widely documented (Brumbaugh 2008, Northern Economics Inc, 2009; National Research Council 2010), there are also instances where ecosystem services may be lost by an expansion of bivalve shellfish aquaculture, e.g. through loss of soft sediment habitats and food resources for wading birds (Herbert 2016). Some of the ecosystem services provided are dependent on-site specifics, bivalve species selected and life stages, making trade-offs (Sequeria, 2008) a key consideration. The shellfish aquaculture sector already contributes around £35.6 million annually to the UK economy (Hambrey et al., 2016). However, this sector is facing considerable spatial constraints including from the MPA network, issues with water quality in some sites, threats from disease and/or invasive non-native marine species (INNS) in some locations (e.g. Wales (Jenkins 2021) and the Solway Firth (Solway Firth Partnership 2017)), and opposition from commercial fishers where the public right to fish is perceived to be impinged by the consenting of aquaculture sites (Hambrey et al., 2016; Black and Hughes, 2017; MMO, 2020).

Increasing demand for marine space and the opportunity to establish a new fisheries/aquaculture legislative framework and management regime coupled with a desire to improve not only the state of the UK's marine environment but to provide global leadership provides an opportunity for new concepts and principles to be debated and developed. In looking for solutions to support fisheries and provide the economic and social goods, one may look to how complex ecosystems and environmental impacts have been addressed on land. Acknowledging the apparent success in farming (in that the concept seems broadly accepted and supported by a diverse range of stakeholders (NFU, 2021; Sustain, 2021; Woodland Trust, 2018)), it is here proposed that PMPG should be debated as a central tenet of marine governance, and specifically in the reform of fisheries and aquaculture.

## 3. Public goods

### 3.1. What are the public goods in UK agricultural policy?

Despite its widespread usage in agricultural policy discussions, there is considerable ambiguity regarding the definition of public goods with at least three distinct concepts evoked in different instances: how goods can be accessed and depleted, whether goods have beneficial outcomes that are widely enjoyed, and the ownership of goods.

The definition of a public good used in economics, first proposed by Paul Samuelson (1954), is a good (i.e. something that provides utility) that cannot be easily depleted (i.e. it is non-rivalrous) and is difficult to prevent others from accessing (i.e. it is non-excludable). This is the definition used in key government technical documents, for example in HM Treasury's Green Book for Central Government Guidance on Appraisal and Evaluation (HM Treasury, 2020). Private goods have the opposite characteristics (i.e. they are rivalrous and excludable) while common goods (non-excludable but rivalrous) and club goods (non--rivalrous but excludable) share one but not both features.

Outside of economics the term public goods is often used in a less technical and more generalised sense to refer to goods that are enjoyed by a broad population (sometimes referred to as *the public good* or the common good) or in other instances to goods with non-market benefits (i.e. a *public benefit* or externality). These two generalised uses may overlap in some cases but not all.[3] A third use of the term public goods relates to ownership, particularly in the case of marine resources where some goods are referred to as *public assets*. However, a public good (under either the technical or the general definition) need not imply public ownership, nor does a private good imply private ownership.[4]

[3] For example, a gift of flowers to a friend is a positive externality with only a single beneficiary (Holtermann, S. E. (1972) 'Externalities and Public Goods', *Economica,* 39(153), pp. 78–87.) and a beautiful sunset has many beneficiaries but is not an externality as no economic agent was involved in its production (Reddy, S. (2015) 'Externalities and Public Goods: Theory OR Society', Available: Institute for New Economic Thinking. Available at: https://www.ineteconomics.org/perspectives/blog/externalities-and-public-goods-theory-or-society (Accessed 26th March 2021).

[4] These three definitions do not constitute an exhaustive list, merely the most common uses. For example, Timmermann (2018) describes three additional variations of the term: a *normative* public good, a *visible* public good, and *joint action* public good.

While the multiple uses of the term public good refer to distinct concepts, it is common for articles on agricultural public goods to define public goods according to the technical economic definition but then apply the generalised definition that covers a much wider range of potential public benefits, only some of which are technically public goods. In some cases, this shift in definition is subtle (Global Justice Now, 2017; Hird, 2021; Sustain, 2021), while in other cases it is more explicit, for example, by arguing that all goods are situated on a spectrum of "publicness" (Cooper et al., 2009), or acknowledging the technical definition but questioning its relevance (Kipling, 2019).

One potential reason for retaining the link to the technical definition of public goods is that in welfare economics, public goods are viewed as a valid reason for government intervention and could thus justify public money being spent. However, this link between public goods and government intervention is disputed, with critics noting that there can be non-governmental solutions to public good problems (Cowen, 1992), or alternatively, that public spending need not require the identification of a public good (Devlin and Wheatley, 2017, p.19). Many economic theorists have questioned the use of public goods theory in welfare economics altogether as the public goods concept combines multiple dimensions (Woolley, 2006), neither of which are 'natural' properties but rather determined by a mix of evolving factors such as institutions, ideology, technology and costs (Cowen, 1985; Goldin, 1977; Vivero-Pol, 2017; Sheng, 2020). Despite these critiques, public goods theory continues to be promoted and its flexible use in practice has placed it at the heart of UK agricultural policy reforms with the support of diverse stakeholders.

In the 2020 Agriculture Act, the twelve identified areas for financial assistance appear to use the generalised definition of public goods as *public benefits* (Agriculture Act 2020 c.21). No explanatory note is provided to justify the areas of financial assistance and despite the term 'public goods' appearing frequently in consultations in the various stages of the Bill and in the 25 Year Environment Plan (DEFRA, 2018b), the term does not appear in the Agriculture Act itself.

The twelve areas for financial assistance range from environmental issues, to animal welfare, to health and well-being. Although not included in the Act, several other areas of potential financial support have been identified as public goods such as beautiful landscapes (Bateman and Balmford, 2018; Cooper et al., 2009; DEFRA, 2018c; OECD, 2015; Vojtech, 2010), to rural vitality (Cooper et al., 2009; DEFRA, 2018c; Global Justice Now, 2017), to employment (DEFRA, 2014; Gerrard et al., 2011; Global Justice Now, 2017), to public health (Hird, 2021; Sustain, 2021), and to democratic accountability (Global Justice Now, 2017). Some of these areas were mentioned in Defra consultations (DEFRA, 2018c) and in the initial development of the ELM (DEFRA, 2020a) but do not appear in the Act. While the production of food (sometimes with qualifiers like 'healthy' or 'secure') was frequently proposed as a public good (DEFRA, 2014; Gerrard et al., 2011; Global Justice Now, 2017; Hird, 2021; NFU, 2018a; NFU, 2018b; Timmermann, 2018; Vivero Pol, 2013; Lochhead, 2009 cited in Almas, Campbell and Marsden, 2012) other authors have specifically noted that food production is not a public good as it closely fits the definition of a private good (Bateman and Balmford, 2018) and has a weak link to consumption outcomes due to many subsequent stages of the supply chain and international trade (Bateman and Balmford, 2018; Helm, 2016). Furthermore, while food is certainly an important good, if producing something desirable is a public good deserving of public money than other sectors from residential construction to energy generation would be equally deserving (Carpenter, 2018), as would all other actors in food supply chains from farm equipment manufacturing to supermarkets (Helm, 2016).Another contrast between the Act and earlier consultations is that the areas for financial assistance identified in the Act refer to processes rather than outcomes, for example *improving* the quality of soil, rather than the *outcome* of a specified soil quality metric.

### *3.2. What are the public goods in the marine environment?*

The areas of financial assistance in the Agriculture Act can serve as a model for what constitutes public goods. Many of these identified public goods in agriculture are directly applicable to fisheries and aquaculture, stemming from the fact that both these sectors impact on, and are impacted by, the natural environment and have significant government financial inputs and oversight (Agriculture Act 2020 c.21). The areas of financial assistance of an environmental nature can be reinterpreted in the marine context while the areas of animal welfare, health and well-being are directly applicable (Table 1). In some cases, there is alignment between the areas of financial assistance in the Agriculture Act and areas of financial assistance in the Fisheries Act, although the latter were developed based on a different, unstated concept.

Climate change mitigation and animal welfare are included as areas of financial assistance in the Agriculture Act but neither appears in the areas of financial assistance in the Fisheries Act despite its direct relevance. These public benefits are less developed in fisheries and aquaculture than in terrestrial agriculture, although the Fisheries Act does contain a climate change Fisheries Objective. Conversely, three areas of financial assistance in the Fisheries Act that are not included in the Agriculture Act are: personal expenses of workers, the health and safety of workers, and the training of workers. This difference may indicate that consideration of workers is further developed in fisheries and aquaculture than in terrestrial agriculture.

While this comparison indicates a substantial overlap in the areas of potential financial assistance between agriculture, aquaculture and fisheries, how that financial assistance is justified and implemented diverges significantly. These differences also explain some of the different sector experiences with public funding to date with direct public income support forming a significant portion of farm income but with direct support rarely used and in smaller amounts in the fisheries sector.

Unlike the relationship between farmers and agricultural land, fishers do not actively manage the marine environment.[5] Much of this difference is explained by feasibility. While farmers can take actions to improve agricultural land, fishers cannot actively restore the marine environment and certainly not in the magnitude to prevent natural disasters. This is not to say that fishers cannot have a large impact on the environment – they do – only that the impact is to the detriment to the marine environment compared to taking no action (i.e. not fishing). By its very nature, fishing is an extractive activity that kills fish, can result in bycatch, damage habitats and affect food webs. While minimising impact (e.g. switching to lower-impact fishing gear) can still improve the state of the marine environment compared to current fishing practices, there is not the same support for the principle of "public money to erode public goods to a lesser extent". Under some interpretations of public goods, the financial flows should be in the opposite direction with fishers paying penalties for the creation of negative externalities (Ryan et al., 2014). There are some limited exceptions where the act of fishing is beneficial to the environment (removal of invasive species, litter, as in Table 1). This challenge to apply PMPG to capture fisheries is not nearly as acute in bivalve or algae aquaculture where careful management can lead to additional public benefits in the form of water quality improvements, habitat creation and climate change mitigation.

Another key difference between payments for public goods in terrestrial and marine contexts is the ownership structure of the resource. While the degree to which fishers fully own fishing opportunities (e.g. licences, quota) is an area of active debate (Appleby et al., 2018), they do not generally own the marine resources themselves (e.g.

[5] Aquaculturists who lease areas of the seabed from the Crown Estate under Regulating and/or Several Orders share features with both farmers and fishers (e.g. Historically, native oyster fishers have sometimes actively managed the environment in question by harrowing the beds and/or laying cultch. This harrowing is conducted to keep the cultch clean for spat settlement).

**Table 1**
Areas for financial assistance specified in the Agriculture Act and their possible marine equivalent (Fisheries Act clause identified where relevant).

| Agriculture Act Clause | Examples given in the Explanatory Notes | Marine equivalent[c] | Capture fisheries examples | Aquaculture (AQ) Fisheries (F) | Aquaculture |
|---|---|---|---|---|---|
| 1. a) protects or improves the environment | tree planting | conservation, enhancement or restoration of the marine and aquatic environment (Fisheries Act 33. 1. a) [a] | shift to lower-impact fishing gear Williams (2019), removal of marine litter and fishing gear | Applies to both AQ + F | benevolent habitat forming shellfish aquaculture (Madricardo et al., 2020; Fodrie et al., 2017) |
| 1. b) supporting public access | facilities for educational visits … share information about agroecology | promotion or development of recreational fishing (Fisheries Act 33. 1. i) | recreational fishing policy, facilities for educational visits, data collection | Applies to both AQ + F | |
| 1. c) restores or enhances cultural heritage or natural heritage | maintenance of historic farm buildings, dry stone walls and conservation of limestone pavement … contributing to research, education, recreation and tourism | cultural heritage in coastal communities | maintenance of historic fishing vessels and portside infrastructure (van der Schatte Olivier et al., 2020), natural heritage, traditional fish related festivals, education around local fish and how to cook them (Everett and Aitchison, 2008; Michael Hall and Sharples, 2008) | Applies to both AQ + F | |
| 1. d) mitigate or adapt to climate change | peatland restoration | blue carbon sequestration[a] | habitat forming shellfish aquaculture (Fodrie et al., 2017), fishing less to restore natural carbon sinks (Sala et al., 2021) and to let biomass sink (Mariani et al., 2020) | Applies to both AQ + F | |
| 1. e) reduce or protect from environmental hazards | improving soil porosity | restoration of natural features for storm protection | shift to lower-impact fishing gear (Williams, 2019) and areas away from inshore features that provide flood protection, nutrient cycling, erosion protection, sediment stabilisation in aquaculture (van der Schatte Olivier et al., 2020) | Applies to both AQ + F | |
| 1. f) protecting or improving the health or welfare of livestock | participation on health or disease control schemes animals have access to materials that allow them to express their natural behaviours | health and welfare of fish through reduced bycatch and increasing survivability of bycatch which is discarded[a] | shift in species (e.g. bivalves), shift to lower-welfare impact fishing gear (Waley et al., 2021), lower stocking densities in aquaculture | Applies to F | |
| 1. g) conserving native livestock, native equines or genetic resources relating to any such animal | rearing rare and native breeds or species | Conserving native species, removal of Non-Native Invasive Species (NNIS) | encourage the targeted removal and commercial and/or recreational utilization of dead NNIS, removal of NNIS (Giakoumi et al., 2019; MacLeod et al., 2016), restocking schemes: oysters (Native Oyster Network, 2021), sturgeon (Blue Marine Foundation 2021). | Applies to both AQ + F | |
| 1. h) protecting or improving the health of plants | reduce the risk of introduction and spread of harmful plant pests and disease | conservation, enhancement or restoration of the marine and aquatic environment (Fisheries Act 33. 1. a)[a] | shift to lower-impact fishing gear Williams (2019). | Applies to both AQ + F | |
| 1. i) conserving plants grown or used in carrying on agricultural, horticultural or forestry activity, their wild relatives or genetic resources | conserve and utilise crop wild relatives | no equivalent, plant nurseries are a different sector[b] | not applicable | Could potentially apply to AQ | |
| 1. j) protecting or improving the quality of soil | Assistance for soil monitoring and research … practices which protect and enhance soil health | conservation, enhancement or restoration of the marine and aquatic environment (Fisheries Act 33. 1. a)[a] | shift to lower-impact fishing gear (Williams, 2019), water quality improvements in shellfish aquaculture (van der Schatte Olivier et al., 2020) | Applies to both AQ + F | |
| 2. a) starting, or improving the productivity of, an agricultural, horticultural or forestry activity | precision application equipment for slurry | the promotion or development of commercial fish or aquaculture activities (Fisheries Act 33. 1. b) the reorganisation of businesses involved in commercial fish or aquaculture activities (Fisheries Act 33. 1. c) improving the arrangements for the use of catch quotas or effort quotas (Fisheries Act 33. 1. h) | participation in scientific research (e.g. gear trials to reduce bycatch) | Applies to both AQ + F | |

(continued on next page)

Table 1 (*continued*)

| Agriculture Act Clause | Examples given in the Explanatory Notes | Marine equivalent[c] | Capture fisheries examples | Aquaculture (AQ) Fisheries (F) | Aquaculture |
|---|---|---|---|---|---|
| 2. b) supporting ancillary activities carried on, or to be carried on, by or for a producer | activities carried on by a producer … or someone acting for them | the economic development or social improvement of areas in which commercial fish or aquaculture activities are carried out (Fisheries Act 33. 1. g) | improvement of launching facilities for recreational vessels | Applies to both AQ + F | |

[a] Fishers can only minimise or reduce their own harm rather than taking a beneficial action.
[b] Fishers cannot have a significant impact.
[c] Includes the catching sector, the aquaculture sector, and recreational fishing.

fish stocks, marine habitat). Aquaculture is situated in between the two where ownership can be in the form of a lease of areas of seabed which can provide a financial return to the state to support management and data collection.

Following from this difference in ownership, the justification for direct public income support in the agricultural sector to influence how owners use their private property is absent in the fisheries sector. As the marine environment is owned and managed on behalf of the public, the government can more easily legislate a change to how marine space is used compared to an area of terrestrial land under private ownership. Thus, a change in policy regarding resource use can be achieved in the marine environment both at scale and at a faster pace than would be the case in the terrestrial environment. Still, public payment is a powerful tool and this key difference in ownership may lead to different forms of public payment rather than a forgoing of public payments entirely.

## 4. Do public payments to fisheries and aquaculture deliver public goods?

There are three broad arguments for transferring public money to the private fisheries and aquaculture sector that may deliver public goods in their broadest sense. The first is to incentivise sectoral development in a manner that may not otherwise occur. Such support was integral to the post-war expansion of fisheries (Schrank, 2003; Tickler et al., 2018) and more recently for developing nations' fisheries (Cisneros-Montemayor et al., 2013; Espinoza-Tenorio et al., 2011). The second is to address distributional and social equity issues, such as to improve the conditions of marginalised groups (Harper and Sumaila, 2019; Schuhbauer et al., 2020). The third is to address conservation concerns such as limiting carbon emissions or undoing harm previously caused (Balmford and Whitten, 2003; Cullis-Suzuki and Pauly, 2010).

Over the past two decades, much of the subsidisation of EU fisheries and aquaculture was provided with the intention to incentivise development (Skerritt et al., 2020). While such development was once an important policy for increasing food production, fears of food shortages in the EU have largely receded and many exploited fish stocks have reached or exceeded their ecological limits (a recent audit of 104 UK fish and shellfish stocks found that only 36% were healthy in terms of stock size (Guillen, 2021)). As such, subsidy provision to increase capacity is no longer necessary, particularly as fishing capacity is estimated to increase by 2–4% annually through technological advancements alone (Palomares and Pauly, 2019; Munro and Sumaila, 2002; Eigaard et al., 2014).

However, many fisheries subsidies continue to enhance fishing capacity (Skerritt et al., 2020; Sumaila et al., 2019). Such subsidies have been shown to cause harm by distorting markets, contributing to unfair trade practices and unequal competition within countries (Schuhbauer et al., 2020), and, importantly, they undermine the natural resources that the sector relies upon by encouraging overcapacity, overproduction and subsequently overfishing (Sakai et al., 2019; Schuhbauer and Sumaila, 2018). This has led to the pervasive view that the continuation of subsidies to the fisheries sector are now intended to lower fishing costs to offset declining catches (Sumaila et al., 2019), rather than to deliver public goods *per se*.

However, not all fisheries subsidies are damaging. Sufficient evidence exists to classify certain forms of subsidies based on their likely impact on fish stock and environmental sustainability, noting that the status of the resource (Arthur et al., 2019), characteristics of recipient fisheries (Quinn and Ruseski, 2001), and cultural and institutional differences (Sakai et al., 2019), have all been shown to alter the observed impact a particular subsidy may have. Certain subsidies can have positive, or at least neutral, impacts upon environmental sustainability and may even provide direct public goods. Indeed, the EU has made steady progress towards redirecting many damaging forms of subsidies towards less damaging, potentially beneficial, forms of support (Skerritt et al., 2020). These beneficial subsidies are thought to act as an investment in natural resources while also conferring sectoral benefits. For example, the UK is currently investing in the establishment and maintenance of MPAs that intend to directly conserve portions of the marine environment (UK Parliament, 2021), which may result in benefits to the sector through spill-over effects (Halpern et al., 2010; Lenihan et al., 2021).

While the nature of fisheries subsidies is the subject of much research, few studies have quantified aquaculture subsidies (but see Guillen et al., 2019; Love et al., 2017) - despite significant public money being transferred to the sector annually, especially in the EU where EUR 1.17 billion was transferred between 2000 and 2014 with the key aim of developing the aquaculture sector (Guillen et al., 2019). However, unlike wild capture fisheries, increasing aquaculture production does not necessarily have direct negative environmental impacts, and may even provide public benefits in the case of bivalve or algae culture.

While reducing the impact of fishing on the marine environment has been a clear policy goal in the UK, the opportunity for habitat restoration and creation through bivalve shellfish aquaculture, for example, has not been the focus of specific policies or subsidies. The PMPG approach to subsidies however opens this realm of possibility. Specifically, subsidising excess production or 'set aside' areas on leased shellfish beds (through Several or Regulating Orders – where public authorities can lease areas of seabed for cultivation of bivalve shellfish) can generate public goods (e.g. climate change mitigation, water purification, enhanced biodiversity, food for wading birds and other wildlife (National Research Council, 2010; Grabowski et al., 2012; Herbert et al., 2016; Herbert et al., 2012; Northern Economics Inc, 2009; Rodriguez-Perez et al., 2019; Watson et al., 2020; Williams and Davies, 2018)) without the corresponding risk of distorting markets or supporting specific companies or sectors unfairly. Care must be taken to ensure that overall environmental degradation does not happen by trading one set of ecosystem benefits for another. The approach could consider an ELMs-like approach to delivering multiple co-benefits in the public interest through supporting bivalve shellfish aquaculture if well situated and regulated to consider possible trade-offs (Rodriguez-Perez et al., 2019).

Clearly, the environmental outcomes of fisheries and aquaculture subsidies are complex, and policy interventions in the marine environment can lead to unexpected, or unintentional (and potentially perverse), outcomes, not least because increasing fishery production directly puts pressure on fish stock sustainability and leads to increased environmental degradation, simply through the action of increased fishing activity. With this nuance in mind, we outline the types of direct public payments that the UK fisheries and aquaculture sectors were known to receive via the EMFF and, to the extent possible, describe whether they are likely to provide or undermine public goods. Each relevant form of public payment via the EMFF was categorised as either having positive (+ve), negative (-ve) or neutral (0) impacts on public goods. The definitions in the regulations of what the public payments set out to achieve, or to the extent possible the types of projects that were funded under each payment type, were used to inform this categorisation. Our definition of a public good is taken from the proceeding section of this paper, and as such does not consider the production of food or jobs in its definition. The likely outcome in terms of providing public goods and supporting arguments are provided (Table 2).

The UK's specific objectives for transfers of public money via EMFF were defined by four main policy goals: 1. To transition the fleet to sustainably managed and discard-free fisheries; 2. To foster growth potential across the fisheries, aquaculture and processing supply chains; 3. To support the efficient use of natural resources; and 4. To fulfil the UK's enforcement and data collection obligations. Although some public goods are reflected in these broad goals, they tend to focus on growth, efficiency and management of the current sector, rather than on providing direct public benefits.

These overarching goals are reflected in the likely environmental outcomes of the specific EMFF public payments. We determined that five EMFF payments provided clear public goods, including two directed specifically towards fisheries; one focuses on the removal of litter and ghost fishing gear, the other provides life jackets to fishers and crew. However, the majority were considered to have neutral effects. This is probably because fishers, unlike farmers and landowners, cannot easily restore the marine environment, but can only erode the natural environment to a lesser degree. As such, any subsidy that enhances fishing capacity, by its nature will be increasing environmental degradation and pressure on fish stocks, and therefore some of the EMFF payments clearly deliver negative impacts.

Some of the subsidies identified that do intend to provide benefits only go as far as aiming to reduce the impact that fishing or aquaculture have on the environment, rather than remove it or undo its impact altogether (restoration). This includes public payments that aim to reduce incidental mortality of commercial and non-commercial fish, broaden participation in environmental decision-making, and improve energy efficiency. The reality of these subsidies is that they reduce the impact of fishing on the marine environment, rather than remove the impact or begin to undo the harm previously caused. This contrasts with the Agriculture Act Clause 1.a, which aims to protect or improve the environment by actively planting trees, rather than to reduce the process of cutting down trees.

Furthermore, public money for new equipment including vehicles, ice machines, power generators and more efficient engines on fishing vessels, have been shown to potentially lead to increases in fishing capacity and therefore can lead to further fishing and further environmental degradation (Sumaila and Pauly, 2006). This is particularly true for the replacement of vessel engines, or any modernisation that increases a vessels ability to find, catch or store fish, which has been shown to increase fishing effort (Palomares and Pauly, 2019), even in non-open-access systems (Munro and Sumaila, 2002). The intention is to increase efficiency but not capacity, however, the contradiction of providing funding for vessel modernisation while simultaneously requiring these investments not to increase the vessel's ability to catch fish was highlighted by the European Court of Auditors (European Court of Auditors, 2011) and the European Commission now recognises that vessel modernisation without increasing fishing capacity is not always achievable (European Commission, 2019).

Findings that public money spent via the EMFF largely have a neutral or negative impact on delivering public goods is reflected, to some degree, by a recent evaluation of the environmental benefits flowing from

**Table 2**
Types of support provided to UK fisheries and aquaculture allocated from the European Maritime and Fisheries Fund (EMFF), and whether they likely provide public goods (+ve), undermine them (-ve), or are neutral (0) in their outcome.

| Stated subsidy intention | Specific public payments | Example(s) from allocated EMFF funds in the UK | Likely outcome on providing public goods |
|---|---|---|---|
| **Fisheries** | | | |
| Reduces impact of fisheries on the environment, including avoidance and reduction of unwanted catch. | Design and implementation of conservation measures. | Develop knowledge of live wrasse fishery to inform management and development. | -ve Fishery development adds additional pressure to environment. |
| | Limit impact of fishing on environment and adapt fishing to protect species. | Replacement fishing gear to reduce by-catch or gear loss. Replacement nets with larger mesh sizes to improve selectivity. | 0 May reduce impact on fish stock, but not wider environmental impacts. Replacing gear can potentially increase fishing capacity. |
| | Innovation linked to conservation of marine resources. | Determine effects of offshore aquaculture installations on fisheries. | 0 May limit impact but does not offer benefit. |
| | Protection and restoration of marine biodiversity – collection of lost gear and litter. | Fishing 4 Litter aimed to remove 25 tonnes of litter from Cornwall and reduce wildlife fatalities. | +ve Removal of litter. |
| Protection and restoration of aquatic biodiversity and ecosystems. | Protect and restore marine biodiversity. | Establish sustainable seaweed farming. Collecting seabed data so impacts of fishing can be monitored. | 0/-ve Fishery development adds additional pressure to environment. |
| Enhancing competitiveness and viability of fisheries and improving safety or working conditions. | Advisory services. | Scallop stock assessments in English Waters. | 0 May lead to reduced impacts but does not offer benefit. |
| | Health and safety. | New machinery, safety equipment, crew comfort. | 0 Could deliver benefit or increase fishing capacity. |
| | Added value, product quality and use of unwanted catch. | Updating refrigeration systems. | 0 If food is not considered a public good, there is no clear benefit. |
| | Improving fishing ports, landing sites, auction halls and shelters. | Upgrading fuel systems and LED light retrofit to reduce electricity consumption. | 0 Could deliver benefit or increase fishing capacity. |
| Support to strengthen technological development and innovation. | On board investments. | Replacement of anti-fouling paint with copper coat. | 0 Could deliver benefit or increase fishing capacity. |
| | Replacement or modernisation of engines. | New engines to reduce fuel consumption. | -ve (net) Potentially less $CO_2$ but likely to increase fishing capacity. |
| Professional training, new professional skills and lifelong learning. | Training, networking, and support to spouses. | Supply of lifejackets and training. | +ve Provision of lifejackets 0/+ve Training (for alternative professions as this removes fishing effort). |

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**Table 2** (*continued*)

| Stated subsidy intention | Specific public payments | Example(s) from allocated EMFF funds in the UK | Likely outcome on providing public goods |
|---|---|---|---|
| Development and implementation of the Integrated Maritime Policy. | Protection of environment and sustainable use of resources. | Development of North Devon Marine Natural Capital Plan. | 0 May limit impact but does not offer benefit. |
| Improved knowledge and data collection. | Data collection. | Data collection framework national correspondent. | 0 No clear benefit. |
| Promoting economic growth, social inclusion and job creation. | Local development strategies. | New equipment including vehicles, ice machines, quays, power generators and chillers. | -ve Likely to increase fishing capacity. |
| **Aquaculture** | | | |
| Support to strengthen technological development, innovation and knowledge. | Innovation. | Feasibility of UK scallop hatchery and Black Soldier Fly meal as a replacement to fish meal. | +ve |
| | Management and advice for aquaculture. | Benthic survey equipment. | 0 May limit impact but does not offer benefit. |
| Enhancing competitiveness and viability of aquaculture. | Investments in aquaculture. | Increasing mussel farm capacity. | +ve May reduce pressure on fish stocks and the wider environment from fishing. |
| Promotion of aquaculture having high level of environmental protection, and animal health and welfare. | Promoting human capital and networking. | Fish health training course, upgrading management skills and knowledge, employee training and upskilling. | +ve |
| **Fisheries and Aquaculture** | | | |
| Investment in processing and marketing | Processing of fisheries and aquaculture products. | New fuel-efficient vans, ice machines and solar panels. | -ve Likely to increase fishing capacity. +ve Reduction in $CO_2$ emissions |

the EMFF to fishing and aquaculture in England (Arthur et al., 2019). This report demonstrated that of the 1172 projects funded at the time of its publication, 396 projects, or a third, were classified as having overt intentions of providing environmental benefit. While the report identified some direct positive outcomes from the EMFF, including reductions in $CO_2$ emissions and unwanted catches from fisheries, projects that aimed to address environmental concerns in less direct ways, e.g. through research or participatory planning processes, were more difficult to identify.

Regarding aquaculture subsidies, the evaluation found that bivalve shellfish aquaculture systems likely provided some environmental benefits (acting as artificial reefs, or to protect/create habitats). The evaluation also sought to capture additional, and sometimes unintended impacts which could be termed public goods as they go beyond food production, income and employment (which are private benefits). The environmental benefits of EMFF funded habitat forming aquaculture projects in England were focussed on shellfish, with 13 'habitat forming' projects on mussel, native oyster, or seaweed aquaculture reported (Arthur et al., 2019). While the habitat forming aquaculture projects were successful in increasing Natural Capital and generating employment and income, the potential to scale up is often limited by the extent and availability of suitable habitats and existing poor water quality. As bivalve stocks increase, initial benefits e.g. improved water quality, could later result in dis-benefits resulting from density-dependent factors.

Overall, public money currently used to support the UK fisheries sector at best have a neutral impact on delivering public goods, and at worst appear to work to undermine public goods. Those that support the aquaculture sector are less clear cut, and, particularly for bivalve shellfish aquaculture production may indeed provide clear public benefits, such as cleaner water and carbon sequestration (however, the research into the impacts of these subsidies lags that of the agricultural and fisheries sector considerably).

This exploratory analysis does not mean that the concept of PMPG cannot work in the marine environment, but the question persists of how we build upon those subsidies that provide public goods, or are at least are neutral in their outcome, and how do we redirect those that are likely undermining this concept. The concept of PMPG questions what payments are made to whom and for what outcome, and provides a framework for that debate. It provides a philosophy that underpins the design of those subsidies and potentially helps answer that question.

## 5. Discussion

### 5.1. The concept

We have highlighted that payments via the EMFF often had neutral and at worst negative impacts on providing public goods. As a result, it was arguably negative in the mid to long term for the future of the marine environment, socially and economically. We have shown that the PMPG concept applied to land, will apply differently in the marine environment, mostly because of the public assets being utilised (i.e. space, fish) and the ownership of those assets, but there are parallels with the possibilities for ecosystem improvement as well as addressing how space is most advantageously used.

PMPG does not provide a prescriptive answer of what public payments should look like, but it does provide a philosophy for how those payment policies could be designed. In this section we explore this further by considering the impact of the concept on future policy, how the concept can be expanded, the key gains that can be achieved and how it can support a just transition.

### 5.2. Future policy

With the UK revisiting its subsidy control scheme and the development of domestic replacements to the EMFF (noting that the PMPGs concept was not included in the first iteration of the Fisheries and Seafood Scheme for England when launched in April 2021 (MMO, 2021)), it is incumbent on the government to re-evaluate what the public pays for and what private industry should deliver in return for the funding received. The concept of PMPG could be incorporated into the policy statement on the application of five environmental principles emanating from the Environment Bill.[6] Further still, it has been argued it could be incorporated in itself as an additional subsidy control principle (Natural England, 2021). Application of the environmental principles is open for debate, as these principles will post date the enactment of the Fisheries Act. Express consideration of how these principles may be applied to a future fisheries management framework remains live. For example,

[6] The five principles of the Bill being - integration, the adoption of the 'prevention principle', which means policy should prevent, reduce or mitigate harm, the 'polluter pays principle', the rectification at source principle and the 'precautionary principle', which states that a lack of scientific certainty on the potential environmental damage of an activity should not postpone measures to prevent it DEFRA (2021) 'Consultation Launched on Environmental Principles', [press release], Available: UK Gov. Available at: https://www.gov.uk/government/news/consultation-launched-on-environmental-principles (Accessed 22nd March 2021).

commercial fishing is an anomalous marine industrial sector as the application of the polluter pays principle is not currently applied to this in any meaningful way – fisheries that damage the marine environment are not required to restore or provide compensation. Fishing is a specific case in that we already subsidise the damage of a public good or at least fail to manage fisheries strictly enough to prevent damage (e.g. through the funding of the supporting regulatory/management framework and reduced red fuel duty). Consideration of how to move to a point whereby the industry pay for damage caused to the marine environment and the management costs incurred by wider society to enable it, is warranted. It is notable that section 38 of the Fisheries Act does enable secondary legislation to be introduced that allows the imposition of charges for fisheries management upon fishers (Fisheries Act, 2020 c.22 s.38). PMPG could be used in designing a charging regime.

The application of the PMPG approach is not just relevant to the UK but has wider resonance. Were the UK Government to introduce the concept fully, it would be well placed to further this concept internationally because of its increased engagement in international fora and the wider geographical footprint of the UK provided through the Overseas Territories. Whilst the UK is uniquely placed to explore and operationalise this concept because of the requirement to revisit its legislative frameworks in full, this may be significantly more challenging for other countries to achieve, or for Areas Beyond National Jurisdiction where international collaboration and qualified majority decision making adds complexity.

### 5.3. Expanding the concept

The application of the PMPG in English agriculture has some particularities that have shaped the present analysis. First, the public goods concept was applied to agriculture in a very general sense (Table 1), referring to what may loosely be termed 'public benefits', although four of the twelve areas have a much stronger private element than public benefit (animal welfare, productivity, ancillary services and perhaps soil quality too). This form of application has thus shaped the equivalent application of the PMPG concept to the marine environment (Section 3).

Second, because the PMPG concept is currently only being applied to direct financial support for agriculture, the scope of the analysis for fisheries and aquaculture was likewise limited to direct support (Section 4). A wider application of PMPG could be considered, however, and there is even the potential for the discussion of PMPG in the marine environment to leapfrog its application in the terrestrial environment by considering indirect support too.

One prominent example of indirect support is the red diesel tax rebate which is received by both the agricultural and fishing/aquaculture sectors. In sharp contrast to the UK Government's climate ambitions, this subsidy reduces the incentive to reduce $CO_2$ emissions and distorts relative prices in favour of the most carbon-intensive fishing methods such as scallop dredging. This indirect subsidy comes at a significant public expense. Based on data provided by Seafish for 2019 (Motova, 2021), there was nearly 107 million litres of fuel consumed by the English fleet over the year. Whilst this figure does not distinguish between types of fuel consumed, the potential benefit to the sector is considerable. The current full tax rate for diesel being nearly 58p per litre as opposed to the effective rate after rebate on red diesel of around 11p per litre (HMRC, 2021).

A second example, unique to the fisheries sector, is the allocation of access rights such as fishing quotas. While not traditionally viewed as a subsidy, the free allocation of access rights to a publicly owned resource functions as a benefit in kind and contrasts with systems of auctions or royalty payments that are used for other resources in the UK (e.g. forestry, aggregate extraction, water abstraction) as well as the allocation of access rights in some international fisheries (e.g. Iceland, the Faroe Islands, Australia, New Zealand and regions of the US). This consideration is particularly relevant as the TCA resulted in EU quota shares (considered to represent 25% of the value of the EU landings from UK waters) to be gradually transferred to the UK over a 5.5-year period (ABPmer, 2021; Popescu and Scholaert, 2021). A PMPG approach could consider tying the allocation of fishing quotas to the delivery of public goods i.e. lowering the impact of fishing could be the basis for preferential allocation of fishing opportunities and public money in support of this. Indeed, a Government consultation on quota allocation during the development of the Fisheries Act found widespread support for "criteria-based allocation" (DEFRA, 2020b).

Just as the application of PMPG could be expanded to include indirect support, there is also the potential to expand its use beyond food-based applications in the terrestrial and marine environments to a more universal application to any sector receiving public money. Such an expansion of the concept would align with the Government's ambition to establish the five principles of environmental governance. An expansion would also align with the recognition during COVID-19 support programmes that public money should be conditional to leverage resources to tackle the biggest challenges of our time and 'build back better' (HM Treasury, 2021b).

### 5.4. Key gains

Under the EMFF, Table 2 suggests public good can be achieved through payments supporting activities such as litter removal, aquaculture research and development and training. Cross referring with Table 1, there is further potential for PMPG to support lower impact fishing gear (Williams, 2019) (whilst addressing issues of the potential to increase capacity), encouraging a shift in species selection, education in marine systems (education and public engagement being a key requirement under the Dasgupta report (2021)), engagement with scientific research, exploration of blue carbon sink potential and maintenance of portside infrastructure, amongst others. With a PMPG lens, funding towards algae or bivalve shellfish aquaculture also has significant public good potential in terms of nutrient cycling, water quality, habitat creation, biodiversity and sediment stabilisation. This alongside switching gear came up repeatedly in Table 1 suggesting a potential focus for subsidy policy.

Environmental and social gains are not only made by decisions on where money is paid, but also where it is not. Using a PMPG lens several existing payments or concessions are difficult to justify particularly around fuel and funding more efficient fishing equipment (linked to increased capacity and subsequently overfishing). Unless these payments can be supported through other government agendas, they are difficult to defend.

### 5.5. Just transition

Just as with the terrestrial environment it is hoped that reformed financial support will bring about the transformational changes needed. However, transformational change that impacts the fishing sector must not threaten already disadvantaged coastal communities; areas where the government is already seeking to 'level up the economy' (HM Treasury, Ministry of Housing Communities and Local Government and Dept of Transport, 2021).[7]

The need for a 'just transition' is enshrined in the Paris Agreement (UNFCCC, 2015) and reflects that in addressing the needs of the environment, the transition towards a more sustainable world needs to be equitable and just. In other words, in bringing about the change needed to deal with the marine environmental crisis, the social and economic impacts on fishers and communities affected by the change need to be considered, with the aim to 'leave no-one behind' (UNFCCC, 2015). PMPG provides an opportunity to not only expand the concept to include

[7] Coastal communities are expressly referred to in the UK Government's levelling up campaign as regions requiring support in the strive to address economic differences and inequalities across the country.

indirect support and quota allocation reform but also to address distributional inequalities. It is important that the 'PMPG' concept can be used to support these wider goals and help drive a just and equitable transition (rather than working against it).

To support that drive, what is considered a public good worthy of payment and how benefits and burdens should be distributed, should be rooted in participatory processes through procedural justice principles. As such, it is not for the authors to dictate what PMPG should look like for a specific region in detail; what PMPG should do is kick-start that debate and ensure all interested parties are at the table. Respect and recognition of all interested parties should underpin environmental policymaking (Schlosberg, 2007a, 2012) and the application of PMPG is no different. In addressing distributional inequalities, that fishers should be engaged and respected in that process is a given. One of two key concerns raised by fishers and other interested groups in a recent study in Newfoundland was the issue of engagement and participation in the changes being suggested (Kahmann et al., 2015), the lack of buy-in being the consequence of a failure to address those concerns (alongside wider issues over the efficacy of policies that do not take into account those viewpoints (Hart, 2021)).

But if fisheries are a public asset, then those participatory processes should also actively engage with the public and wider society (including future generations) who may be the silent majority of beneficiaries, and perhaps removed from and unaware of the nuances within marine governance. Taking this one step further, it is arguable that as a public asset, the wider societal needs should take precedence (Bean, 2020) including decisions over how that marine space is used. This is undoubtedly difficult but not impossible and the balancing of competing interests is not a tension solely experienced by this concept as attested to by the general principles and applicability of Aarhus (UNECE, 2018). There are provisions both in the EU-exit agreement and the Fisheries Act for wider participation, although of course it remains to be seen how that is implemented and whether this is successful.

Who is represented at the table and how is a contentious and complex issue. How future generations should be represented is a matter of current debate. In England, in the House of Lords, the Well-Being of Future Generations Bill stalled (House of Lords, 2020) and at the time of writing the Parliamentary website notes no date is set for a second reading of Bill No2 in the House of Commons (House of Commons, 2020). The Bills follow the introduction of the Well-Being of Future Generations (Wales) Act (2015) which, amongst other matters, requires public bodies to consider the long-term impact of their decisions (Well-being of Future Generation (Wales) Act, 2015 anaw 2) – in effect considering the effects on future generations and providing them with a proxy seat at the table. The effectiveness of the Welsh Act in practice remains a matter of debate (Dickins, 2018), although is commendable for its attempts to grapple with this complex issue. Again, this is not simply an issue for PMPG. However, if PMPG is at the foreground of thinking, it is possible some of these concerns can be addressed, for example, through the application of the sustainability and climate change objectives in Fisheries Act, thus integrating PMPG with those objectives.

How non-human life can 'participate' or be represented at the table is also complex and contentious. There is increasing acceptance that human and non-human life are interconnected and human life, including economic prosperity, is dependent on ecosystem health (Dasgupta, 2021) (although, the rights of non-human life that conflicts or are not overtly connected to human prosperity is more difficult (Holland, 2014)). Either way, how non-human life and interests are given a seat at the table (albeit and inevitably by 'proxy') remains an unresolved issue (Schlosberg, 2007b). At a bare minimum, the authors suggest that PMPG policies are drafted that include and build on the collection of scientific data on the state of the environment and ecosystems and that learn, reflect and adapt to that data. In other words, a policy that is not static but allows for learning or is reviewed when certain metrics are achieved (or not). This is an example of good, adaptive governance in any event (and could and should be extended to social as well as environmental goals) (Akamani, 2016) as well as arguably a form of 'communication' or at least feedback, from non-human life.

A 'just' transition requires consideration of how assets and burdens are distributed between societal groups, regions, non-humans and intergenerationally. It requires that consideration be undertaken with principles of participatory justice, respect and recognition in mind. As with governance of any complex system it requires feedback mechanisms through data gathering for responsive adaptive governance. It also requires those concepts to be applied with a view to enhancing the well-being of living things. PMPG is a tool that can help focus attention and address those issues. It forces open dialogue on distribution (of public money) with a view to enhancing the ecosystem's ability (which by definition includes human and non-human) to flourish as a public good.

### 5.6. *Conclusion and areas for further research and debate*

Applying PMPG to the marine environment is a departure in current thinking that could aid transformational change in the appreciation and use of the marine environment. It is a framework that can guide that change. It is accepted that the marine environment is a complex system and as such impacts are not always predictable or linear and can be contested. The authors hope to start a healthy debate and garner perspectives on the use of our marine environment and payments made to support it.

In considering the three broad reasons for public money to be paid 1) development, 2) distributional and social equity issues, 3) conservation issues, the following research and debate areas are suggested:

- How can we incentivise sustainable sectoral development using PMPG as a concept, supporting diversification and innovation that would not otherwise take place, and how do we do so without increasing (or even by reducing) capacity?
- How can PMPG support the future, sustainable prosperity of our coastal communities, and to what extent would such objectives be an appropriate and efficient use of either fisheries or agricultural policy budgets, as opposed to targeted, means-tested social security support?
- How can fishers enhance ecosystem services and are there other public goods they provide that are currently unrecognised? Research and data collection linking enhancement or reduction of ecosystem services to specific marine habitats and the impacts of fishing on these are a key area for the future.
- How can changes in the marine environment be attributed to the actions of individual fishing operations seeking subsidies given the shared nature of the marine space?
- If the current marine regulatory regime (licencing and marine planning) precludes fishers from leasing marine space (unlike those engaged in aquaculture), could this be considered to foster stewardship through the application of the PMPGs concept?
- How are the benefits of aquaculture subsidies captured (including habitat restoration and creation), how are unintended outcomes monitored, and how can PMPG support positive overall outcomes?

And more generally:

- How can we build upon the public subsidies linked to the provision of public goods (and how can we replace those that do not)?
- How can we integrate science and data at the start of PMPG policy making, what indicators for success should there be, what measurable outcomes and outputs should flow from the application of this concept?
- How can the concept of PMPG be used to continue supporting the fishing industry (and the UK's coastal communities that rely upon

this sector) without exacerbating negative environmental impacts resulting from fishing?

- How can we engage, and where necessary educate, the wider public to form a vision for the future use of direct and indirect public subsidies impacting our oceans, and how can PMPG be used to support that vision?
- How can PMPG be utilised and expanded into other spheres, beyond food production?
- Could PMPG be applied on the international stage to support the recovery of marine ecosystems as global public goods?

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# Same Space, Different Standards: A Review of Cumulative Effects Assessment Practice for Marine Mammals

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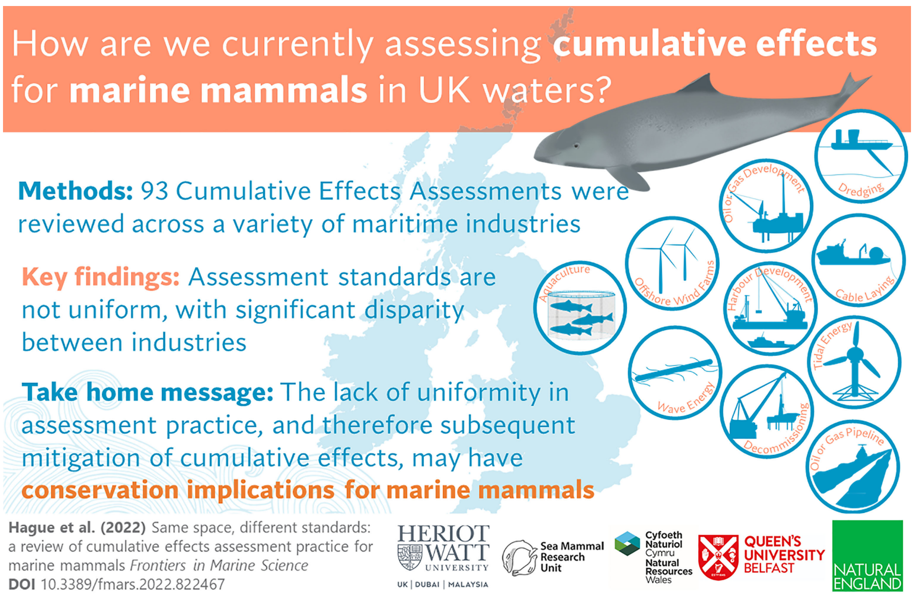
Marine mammals are vulnerable to a variety of acute and chronic anthropogenic stressors, potentially experiencing these in isolation, successively and/or simultaneously. Formal assessment of the likely impact(s) of the cumulative effects of multiple stressors on a defined population is carried out through a Cumulative Effects Assessment (CEA), which is a mandatory component of the Environmental Impact Assessment (EIA) process in many countries. However, for marine mammals, the information required to feed into CEA, such as thresholds for disturbance, frequency of multiple (and simultaneous) exposures, interactions between stressors, and individual variation in response, is extremely limited, though our understanding is slowly improving. The gaps in knowledge make it challenging to effectively quantify and subsequently assess the risk of individual and population consequences of multiple disturbances in the form of a CEA. To assess the current state of practice for assessing cumulative effects on marine mammals within UK waters, 93 CEAs were reviewed across eleven maritime industries. An objective framework of thirteen evaluative criteria was used to score each assessment on a scale of 13-52 (weak - strong). Scores varied significantly by industry. On average, the aquaculture industry produced the lowest scoring CEAs, whilst the large offshore windfarm industry (≥ 20 turbines) scored highest, according to the scoring criteria used. There was a significant increase in scores over the sample period (2009-2019), though this was mostly attributed to five industries (cable, large and small offshore wind farms, tidal and wave energy). There was inconsistency in the language used to define and describe cumulative effects and a lack of routinely applied methodology. We use the findings presented here, along with a wider review of the literature, to provide recommendations and discussion points aimed at supporting the standardisation and improvement of CEA practice. Although this research focused on how marine mammals were considered within UK CEAs, recommendations made are broadly applicable to assessments conducted for other receptors, countries and/or environments. Adoption of these proposals would help to ensure a more consistent approach, and would aid decision-makers and practitioners in mitigating any potential impacts, to ensure conservation objectives of marine mammal populations are not compromised.

**Keywords: cumulative effects assessment (CEA), cumulative impact assessment (CIA), anthropogenic activities, management policy and practice, marine mammals, maritime industry**




**GRAPHICAL ABSTRACT** | How are we currently assessing cumulative effects for marine mammals in UK waters? A graphical abstract for Hague et al. (2022).

## 1 INTRODUCTION

In a global survey of more than 2000 scientists, understanding the individual and interactive effects of cumulative stressors was the top ranked research priority out of a possible sixty-seven distinctive research questions, whilst developing approaches for monitoring cumulative effects was ranked the fourth highest priority (Rudd, 2014). The ongoing question of how to address the complexity of multiple stressors has even been described as one of the 'holy grails' of modern conservation (Simmonds, 2018).

Cumulative Effects Assessment (CEA) and Cumulative Impact Assessment (CIA) (hereafter collectively referred to as CEA) are examples of methodical procedures that attempt to address identify, predict and evaluate the significance of multiple effects, or impacts, from one or multiple activities on a specified receptor (Judd et al., 2015). The receptor considered could be a species (e.g. harbour porpoise), group of species (e.g. marine mammals) or habitat (e.g. benthic environment). These assessments are usually completed as part of an Environmental Impact Assessment (EIA), which is one of the main tools utilised by regulatory agencies to ensure that the environment, and the receptors it supports, are adequately protected (Hawkins et al., 2020). The assessments should identify the potential for stressors to individually and cumulatively have significant effects on a receptor, and if so, should make suggestions for appropriate mitigation in order to ultimately reduce or prevent impacts (Judd et al., 2015). In doing so, these assessments should ultimately make human activities more sustainable (Duinker et al., 2013). In this way, a CEA is regarded as having the potential to deliver the most meaningful component of the EIA, in terms of providing a more complete understanding of the overall consequences of the development or activity (Cooper and Sheate, 2002).

In the UK, completing an EIA and an associated CEA is mandatory for a number of specific project types and activities. Whilst the regulations pertaining to EIA vary by the industry that the project falls under (**Figure 1**), and in

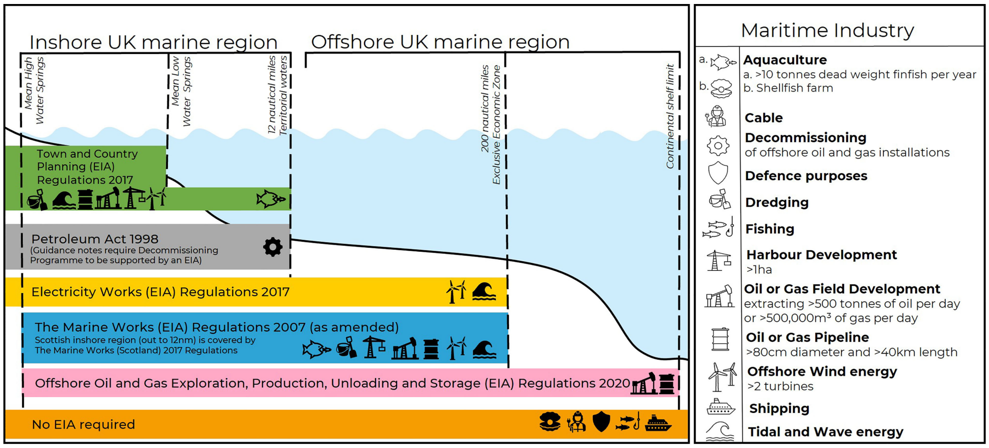



**FIGURE 1** | Legislative regimes pertaining to the requirement for an EIA, and associated CEA, for various maritime industries within UK waters. Note, there are nuances in regulatory regimes and EIA requirements across devolved UK nations which are not captured within this figure. Thresholds for requirement of an EIA are also nuanced, and may vary subject to screening, for example if the proposed project is in a 'sensitive' area. This figure is presented for high level illustrative purposes only. Adapted from MMO Marine Licencing (2020)[7].

some instances by devolved administration (Scotland, Wales and Northern Island), the policy wording that outlines the information required to be included in an EIA, including for the consideration of cumulative effects, is identical across the regulations (for example, [123456]). This states the requirement for; '*a description of the likely significant effects of the project and the regulated activity on the environment resulting from … the cumulation of effects with other existing or approved projects, taking into account any existing environmental problems relating to areas of particular environmental importance likely to be affected; … and the descriptions of the likely significant effects on the factors … must cover the direct effects and any … cumulative … effects of the project and the regulated activity. This description must take into account the environmental protection objectives established at EU or at national level which are relevant to the project and the regulated activity*'.

These requirements cover a number of different types of activity and projects. Some categories, known as Schedule A1 (or '1') projects, always require completion of an EIA and associated CEA. Examples include projects that plan to extract over 500 tonnes per day of petroleum, or to construct an oil or gas pipeline that is over 40 km long and over 800 mm in diameter. A second category of project, Schedule A2 (or '2'), may first submit optional screening documents to the appropriate authority, giving brief details of plans and potential impacts. The authority will consider whether it seems likely that the size, nature or location of the project will mean it is likely to have significant environmental effects. If so, then an EIA and associated CEA will be required. Example of Schedule A2 projects include installation of a wind farm consisting of more than two turbines, or an intensive fish farm that intends to produce more than 10 tonnes of fish per year.

Whilst the policy requirements may be similar or identical across industries and their respective sectors, each sector is managed by a sector-specific regulator, which may also differ by administration, and in turn may also have a different body from which they seek advice, for example, a specific Statutory Nature Conservation Body (SNCB). Regulators can provide support, publish associated guidance to aid completion of EIAs and CEAs, and ultimately may steer the requirements and standards of such assessments. However, despite the regulations and regulatory support, CEAs are a long-recognised area of weakness within the EIA process (Burris and Canter, 1997; Cooper and Sheate, 2002; Gunn and Noble, 2011; Duinker et al., 2013; Judd et al., 2015; Willsteed et al., 2018; Durning and Broderick, 2019), with CEA practice recently described as 'woefully deficient or simply absent' (Sinclair et al., 2017). Whilst standards and practice of CEA may vary by practitioner experience, specific expertise, geography and role (Kågström, 2016; Foley et al., 2017), assessing potential impacts of multiple stressors is, in itself, an inherently difficult task. This is exacerbated, in most cases, by a limited understanding of how receptors respond to various stressors,

[1] The Marine Works (Environmental Impact Assessment) Regulations 2007, The Marine Works (Environmental Impact Assessment) (Amendment) Regulations 2017 and The Marine Works (Environmental Impact Assessment) (Scotland) Regulations 2017.
[2] The Offshore Petroleum Production and Pipelines (Assessment of Environmental Effects) Regulations 1999 (as amended) – replaced by the 2020 EIA Regulations: The Offshore Oil and Gas Exploration, Production, Unloading and Storage (Environmental Impact Assessment) Regulations 2020.
[3] The Offshore Petroleum Production and Pipelines (EIA and other Miscellaneous Provisions) 2017.
[4] Town and Country Planning (EIA) Regulations 2017, and The Town and Country Planning (Environmental Impact Assessment) (Wales) Regulations 2017 and The Town and Country Planning (Environmental Impact Assessment) (Scotland) Regulations 2017.
[5] The Electricity Works (EIA) (England and Wales) Regulations 2017 and The Electricity Works (EIA) (Scotland) Regulations 2017.
[6] The Environmental Impact Assessment (Miscellaneous Amendments Relating to Harbours, Highways and Transport) Regulations 2017.
[7] https://www.gov.uk/guidance/do-i-need-a-marine-licence

and the associated thresholds for response. Whether or not some, or all, of this information is available dictates where on the spectrum of quantitative to qualitative a CEA can be, and how much expert judgement has been relied upon in the absence of empirical evidence. This ultimately may have implications on the associated confidence, uncertainty and reproducibility of the assessment. Furthermore, it is also largely unknown whether stressors interact in an additive manner, to be greater than the sum of their parts (synergistic) or in a diminishing manner to be less than the sum of their parts (antagonistic) (Cocklin et al., 1992; Crain et al., 2008). The approach has therefore often been to assess the impact of each stressor separately assuming either no interaction and/or that pressure-effect relationships are linear, which likely over- or under-estimates impacts (Judd et al., 2015), rather than considering multiple stressors together. However, in the context of the definition of CEA, it is evident that evaluating stressors separately does not constitute a true assessment of cumulative effects.

## 1.1 Marine Mammals and Multiple Stressors

Marine mammals are widespread in UK waters (Reid et al., 2003; Hammond et al., 2013; Hammond et al., 2017), with all species being vulnerable to acute and chronic anthropogenic disturbances (Avila et al., 2018). Stressors are often not experienced in isolation (either spatially or temporally), and instead an individual or group may be simultaneously exposed to a number of stressors, from one multi-stressor activity and/or from multiple activities occurring simultaneously or consecutively. Responses to disturbance can be reflected as temporary (Kyhn et al., 2015) or permanent avoidance (Morton and Symonds, 2002), changes in behaviour [e.g. reduction in foraging vocalisations (Pirotta et al., 2015a)], and in the longer term may result in a change in population size, fecundity or health (Pirotta et al., 2018). In more serious cases, the consequences of anthropogenic disturbance may be fatal (Fernández et al., 2005). The effects of experiencing multiple stressors at the same time, either acute or chronic, are largely unknown for marine mammals, mainly due to a paucity of data as marine mammals are inherently difficult to study (Simmonds, 2018).

Currently, few quantitative tools exist to assist practitioners in the assessment of potential impacts to marine mammals. Sparling et al. (2017) detail and evaluate the sensitivities and utility of common approaches to population level assessment. This review identified two principal approaches: rule-based methods and predictive population modelling. An example of a rule-based methods, is Potential Biological Removal (PBR), which calculates the number of deaths a population could sustain while ensuring the population size remains at or above the 'optimal sustainable population size' (e.g. Williams et al., 2016). Predictive models, attempt to quantify the magnitude of impacts on a population trajectory. Examples of predictive models specifically designed for marine mammals include the interim Population Consequences of Disturbance model (iPCoD) (King et al., 2015) and the harbour porpoise (*Phocoena phocoena*) specific Disturbance Effects on the Harbour Porpoise Population in the North Sea[8] (DEPONS) model. iPCOD is based on a Leslie-matrix population model with the impacts explicitly modelled in terms of their effect on the population's vital rates (fecundity and stage-specific survival). DEPONS is an individual- or agent-based model where the population consequences emerge from the results of simulations of the effects of the impact on the behaviour and energy balance of many individuals. Predictive models require a considerable amount of information, including population demographics, energetics parameters and estimates of disturbance response (e.g. the iPCOD model requires an estimate of the relationship between days of disturbance and individual survival and reproduction rates). This requirement has limited satisfactory parameterisation of such models for many marine mammal species (Harwood et al., 2016). However, predictive models have been parametrised with combinations of empirical information and knowledge acquired through expert elicitation processes and implemented for some of the species recorded within UK waters, including harbour porpoise (Nabe-Nielsen et al., 2018; Cervin et al., 2020), grey seal (*Halichoerus grypus*) (Silva et al., 2020), harbour seal (*Phoca vitulina*) (Thompson et al., 2013), bottlenose dolphin (*Tursiops truncatus*) (Pirotta et al., 2015b; Schwacke et al., 2017), minke whale (*Balaenoptera acutorostrata*) (Christiansen and Lusseau, 2015), long-finned pilot whale (*Globicephala melas*) (Hin et al., 2019), sperm whale (*Physeter macrocephalus*) (Farmer et al., 2018) and humpback whale (*Megaptera novaeangliae*) (Costa et al., 2016). For many other species data limitations preclude such assessments.

Marine mammals are highly protected under a range of international and national legislation, enabling protection at the individual, population, species and site level. Furthermore, the status of cetaceans and seals are key indicators used to assess progress towards achieving Good Environmental Status, as part of the UK Marine Strategy[9]. This strategy also outlines both the need to evaluate potential cumulative impacts of anthropogenic pressures on marine mammals, and to improve the overall evaluation process of cumulative impacts (HM Government, 2012; Government, 2019).

The aim of this study was to review the current state of practice for assessing cumulative effects on marine mammals within UK waters, comparing practice between 2009 – 2019 across eleven maritime industries.

Following a review of CEA practice, we highlight examples of best practice, as well as ongoing challenges and limitations of conducting a CEA. From this, we provide recommendations to improve future assessment practice broadly relevant to receptors beyond marine mammals, and to other geographic regions. We use marine mammals as the receptor of interest due to their high conservation value and degree of legal protection, their vulnerability to a variety of anthropogenic disturbances which occur in UK waters (Avila et al., 2018), the identified knowledge gap of the consequences of cumulative effects to marine mammals (Nelms et al., 2021) and the strong correlation between their distribution and areas of human impact (Pompa et al., 2011). The UK borders one of the busiest maritime areas worldwide, the North Sea, an area that is recognised as having many countries carrying out marine activities in the region, and therefore an area where cumulative effects are already a significant concern (Gușatu et al., 2021). It is

[8] https://depons.eu/

[9] https://moat.cefas.co.uk/introduction-to-uk-marine-strategy/

therefore imperative to ensure such effects are being considered, managed and mitigated in a broader, holistic manner. CEA is mandatory for many, but not all, industries within UK waters (**Figure 1**), and therefore this work utilises current CEA practice to act as a knowledge provider for other marine industries and countries that are yet to adopt CEA as part of their sustainable management strategies.

# 2 METHODS

## 2.1 Scope of the Review

We collated Environmental Impact Assessments (EIAs), Environmental Statements (ESs), Environmental Appraisals (EAs) (hereafter, collectively referred to as EIAs) and any associated documentation covering marine developments in UK waters, published between 2009 and 2019. We did not exclude any industries from the collation or review process, and instead attempted to find EIAs and associated CEAs for as many maritime industries as possible.

The devolved administrations hold and make available records of such documents in different ways. For England and Wales, the majority of these documents were available on the National Infrastructure Planning (NIP) website[10], an online repository that holds applications for Nationally Significant Infrastructure Projects (NSIPs). However, five years after a project receives a decision from the Secretary of State, the associated documents are removed from the NIP website, and so the process to obtain these documents is more challenging. For EIAs within Scottish and Welsh waters, we used the Marine Scotland Information[11] search engine and the Natural Resources Wales public register[12], respectively. There appears to be no publicly available equivalent database for Northern Ireland, consequently obtaining the appropriate documents was more challenging, which is reflected in the number of CEAs reviewed from Northern Ireland. EIAs were also collated from developer websites, Tethys[13], the Marine Management Organisation Marine Case Management System[14], The Crown Estate's Marine Data Exchange[15] and the 4C Offshore Winds Database[16].

Within the Department for Business, Energy & Industrial Strategy (BEIS)[17], the Offshore Petroleum Regulator for Environment & Decommissioning (OPRED)[18] make available recently submitted EIAs related to oil and gas field development on a dedicated webpage[19], though after a short period of time the documents are removed and replaced by a 1-4 page document that summarises the reviewed EIA[20]. The latter summary documents were not sufficient for the purpose of this review, and so an online search was required to attempt to find the full EIA. Where an online search was not successful, the relevant authority (BEIS) was contacted to request missing documentation, who directed the author to contact individual oil and gas operators in the first instance, though individual requests were not feasible within the timeframe of the review.

Oil and gas field decommissioning EIAs were downloaded from OPRED[21], which makes such EIAs accessible whilst the associated 'Decommissioning Programme' is under consideration. Once a decommissioning programme has been approved by OPRED, the EIA document is removed from the website, again making the process to obtain these documents more challenging.

Gaps in the repository were filled, where possible, by the authors' own resources, by further online searches (using terms such as 'Project Title + EIA'), or through direct requests to colleagues. Whilst every effort was made to build a comprehensive repository, we acknowledge that there are some EIAs that may be missing from this analysis. The search resulted in a total of 93 project EIAs and their associated CEAs, from eleven marine industries.

## 2.2 Critical Evaluation Criteria

To ensure each CEA could be compared quantitatively, we scored each CEA based on evaluation criteria developed and outlined in Table 2 of Willsteed et al. (2018). The criteria within the framework were developed following a review of legislative documents where the assessment of cumulative effects is explicitly or implicitly required, and a review of key cumulative effects and marine ecosystem management literature and theory [for further detail, refer to Willsteed et al. (2018)]. This framework separates four categories (Procedure, Space and Time, Pathways and Receptors and Cumulative Effects) into twenty-one attributes used for evaluation. We discounted the attributes that were not relevant for the purposes of our specific aim (reviewing the current state of practice for assessing cumulative effects on marine mammals), as these attributes related to the scoping process rather than the CEA itself (Attributes 6, 8, 12-15), or to effects in isolation (Attributes 6, 8), to the future condition of receptors (Attribute 16), or to the assessment of cumulative effects on ecological connectivity overall (Attribute 19). This resulted in final grading of the 13 remaining attributes (**Supplementary Materials; Table 1**). Attributes were scored on a linear scoring system, where the minimum score awarded was 1 (very weak), and thus the lowest a CEA could score was 13. The highest an attribute could be graded was 4 (very strong), and thus the highest a CEA could score was 52. The primary reviewer (lead author) assessed all CEAs (n = 93). To ensure scoring bias was negligible, 36 CEAs were reviewed by at least one additional reviewer (a co-author).

[10] https://infrastructure.planninginspectorate.gov.uk/
[11] marine.gov.scot
[12] publicregister.naturalresources.wales
[13] https://tethys.pnnl.gov/
[14] https://marinelicensing.marinemanagement.org.uk/
[15] https://www.marinedataexchange.co.uk/
[16] http://www.4coffshore.com/offshorewind/
[17] https://www.gov.uk/government/organisations/department-for-business-energy-and-industrial-strategy
[18] https://www.gov.uk/government/organisations/offshore-petroleum-regulator-for-environment-and-decommissioning
[19] https://www.gov.uk/government/collections/eia-submissions-and-decisions-2021
[20] https://www.gov.uk/guidance/oil-and-gas-environmental-statements-reviewed
[21] https://www.gov.uk/guidance/oil-and-gas-decommissioning-of-offshore-installations-and-pipelines#approved-decommissioning-programmes

All reviewers are experienced in writing and/or reviewing EIAs and associated documents, and are professionals associated with policy and/or marine mammal science. No reviewers scored CEAs where there was any potential conflict of interest (e.g. reviewer had been involved in writing or providing advice to the regulator). To ensure a baseline understanding of the Willsteed et al. (2018) framework, all reviewers discussed how each attribute should be interpreted and this helped ensure continuity when implementing scoring criteria. The average difference between the primary and secondary reviewers total score was 3.8 (*SD* = 3.7, *IQR* = 4.4), with no bias towards higher or lower scoring by the primary reviewer identified (the secondary reviewers total score was higher for 53.3% of reviewed CEAs, whereas the primary reviewer scored higher or the same as the secondary reviewers on 44.4% and 2.2% of occasions, respectively). Therefore, we are satisfied that the CEAs that were not double reviewed will also be representative and any scoring bias will be negligible. Where a CEA had been scored by more than one reviewer, the average of the scores was taken forward for the final analysis.

Alongside scoring each CEA according to the Willsteed et al. (2018) framework, reviewers also collected information from each CEA which would further the understanding of the state of assessment practice, including the age of the CEA, data sources used, guidelines followed and assessment methodology.

## 2.3 Statistical Analysis

A Kruskal-Wallis Test was used to check for differences in quality scores between industries. This non-parametric test was selected as the total scores were not normally distributed (Shapiro-Wilk test; W = 0.92, $p$ <0.005), with variance not equal across industry groups (Levene's test; F (10, 82) = 2.73, $p$ = 0.005).

To check for changes in score over the ten-year study period, we used a multiple linear regression model, where the 'total CEA score' was the response variable and 'Year' and 'Industry' were the predictor variables. The total scores were log-transformed, as the scores were non-parametric with a moderately right skew (skewness score: 0.53). The transformation improved the skewness of the total score data (skewness score: 0.08), and the linear model then fitted the assumption of homoscedasticity, and the residuals were normally distributed. As n=1 for two of the industries (dredging and oil or gas pipeline), their respective data were excluded from the multiple linear regression analysis, as it would not be possible to document trends based on only one data point. For all other industries, n ≥5.

Analysis was undertaken in R (R Core Team, 2020), using packages 'moments' (Komsta and Novomestky, 2015), 'rstatix' (Kassambara, 2021), and figures were created using packages 'ggplot2' (Wickham, 2016) and 'tidyverse' (Wickham et al., 2019).

# 3 RESULTS

We reviewed 93 marine mammal CEAs, completed between 2009 and 2019, covering 11 maritime industries [aquaculture (n = 7), cable (n = 9), oil and gas decommissioning (n = 10), dredging (n = 1), harbour development (n = 8), oil or gas field development (n = 10), oil or gas pipeline (n = 1), offshore wind farm (OWF) large (≥ 20 turbines, n = 22), OWF - small (≤ 19 turbines, n = 9), tidal (n = 11), wave (n = 5)]. For clarity, the total score each CEA received from the grading of the thirteen Willsteed et al. (2018) attributes is simply referred to as the 'score'. CEAs scoring between 13 and 22.5 were categorised as '*very weak*', 23-32.5 '*weak*', 33-42.5, '*strong*' and 43-52 '*very strong*' (**Supplementary Materials; Table 1**).

The results of the review are separated into four parts: Overall Score, Assessment Procedure, Spatial and Temporal Scale, and Cumulative Effects. The latter three correspond with the three attribute categories of the Willsteed et al. (2018) evaluative framework.

## 3.1 Overall Score

The average CEA score across the 93 CEAs was '*weak*' (average = 24.9; range = 13 - 48) (**Table 1**). Overall, 75% (n = 70) of the individual CEAs were '*weak*' or '*very weak*', whilst only 4% (n = 4) were '*very strong*', all of which were large OWF CEAs.

### 3.1.1 Variation by Maritime Industry

Score varied significantly by industry (Kruskal-Wallis test: $\chi 2$ (10) = 54.4, $p$ = <0.005, n = 93) (**Figure 2**; **Table 1**). The effect size was very large (as indexed by eta2[H] = 0.54), indicating 54% of the variance in score was explained by which industry the CEA was attributed to. On average, the aquaculture industry produced the lowest scoring CEAs (mean = 14.6; range = 13 - 18.75; n = 7), whilst large OWF produced on average the highest scoring CEAs (mean = 35.2; range = 17 - 48; n = 22) (**Figure 2** and **Table 1**).

The higher the ranking of the CEA, the higher the average number of pages the assessment (**Table 1**) (*F-value* (1,92) = 89.66, $r^2$ = 0.48, $p$=<0.005). Furthermore, the average number of pages of each CEA varied by industry (**Table 1**); large OWFs had on average the lengthiest CEAs (mean = 24.2 pages; range = 0.5 - 53), whilst aquaculture CEAs were on average the shortest (mean = 0.17 pages; range = 0 - 0.5).

### 3.1.2 CEA Scores Over Time

Overall, CEA scores improved significantly over time (F (9, 81) = 26.12, residual s.e. = 0.09, adjusted $r^2$ = 0.71, df = 81, $p$ = <0.005). The best model to explain this change included industry as an explanatory variable, which found that for five industries, scores significantly improved over the study period. For cable, large and small OWF, tidal and wave industries, CEA total scores increased per year by 2.46, 1.80, 1.57, 2.09 and 1.56 points, respectively, over the study period (all $p$ = <0.005) (**Table 1** and **Figure 3**). The scores of CEAs from four industries did not significantly change over time (aquaculture, decommissioning, harbour development, oil or gas field development: all $p$ = >0.05). Two industries (dredging and oil or gas pipeline) could not be assessed for change in score over time, as n=1.

## 3.2 Assessment Procedure

### 3.2.1 Definition, Methodology and Guidelines

*Definition* 26% (n = 24) of the CEAs provided no definition for cumulative impacts or effects, a further 9% (n = 8) only provided vague definitions. The most frequently used or para-phrased

**TABLE 1 |** Average score for the thirteen attributes used to grade each Cumulative Effects Assessment (CEA), by industry (from Willsteed et al., 2018).

| Willsteed et al. (2018) *Attribute* | *Attribute description* | Aquaculture | Cable | Decom-missioning | Dredging | Harbour Development | Oil or Gas Development | Oil or Gas Pipeline | OWF - Large | OWF - Small | Tidal | Wave | Average score overall |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Total CEAs reviewed (n = 93)** | 7 | 9 | 10 | 1 | 8 | 10 | 1 | 22 | 9 | 11 | 5 | |
| *1* | The CEA explicitly defines cumulative in context of the CEA, reflecting the three components of cumulative environmental change | 1.14 | *2.86* | 1.25 | *2.75* | *2.19* | *2.03* | *2.00* | *2.72* | *2.03* | *2.59* | *2.00* | *2.20* |
| *2* | The purpose and scope of the CEA specifically are clearly set out in the supporting documentation | 1.11 | *2.58* | 1.13 | **3.50** | 1.84 | 1.78 | **3.00** | **3.19** | *2.32* | *2.43* | *2.60* | *2.28* |
| *3* | The CEA documents and applies a clear, systematic CEA methodology, from scoping through to mitigation | 1.04 | *2.75* | 1.20 | **3.50** | 1.81 | 1.48 | *2.50* | **3.07** | 1.89 | *2.02* | *2.10* | *2.11* |
| *4* | The assessment makes use of appropriate data, tools and analytical methods, makes use of quantitative and qualitative methods where data allows. Assumptions and uncertainties are clearly stated and incorporated into the assessment | 1.14 | *2.19* | 1.28 | *2.00* | 1.38 | 1.60 | *2.00* | **3.26** | 1.89 | *2.07* | *1.60* | *2.05* |
| *5* | The conclusions of the CEA are accessible and are compiled in a document that clearly states predicted impacts before and after proposed mitigation measures, assumptions and uncertainties | 1.32 | *2.31* | 1.20 | *2.75* | 1.41 | 1.78 | *2.50* | *2.92* | *2.25* | *2.45* | *2.20* | *2.13* |
| *7* | The **temporal** extent of pressures associated with other activities included in the CEA are identified by a scoping process and documented | 1.07 | *2.53* | 1.08 | **3.50** | 1.28 | 1.20 | 1.50 | *2.66* | 1.88 | 1.95 | *2.50* | 1.91 |
| *9* | The **spatial** extent of pressures associated with other activities included in the CEA are identified by a scoping process and documented | 1.25 | *2.47* | 1.08 | *2.00* | 1.50 | 1.43 | 1.50 | *2.67* | 1.99 | *2.18* | *2.60* | 1.99 |
| *10* | The CEA applies appropriate **temporal** boundaries relative to the receptors selected for assessment in the CEA | 1.11 | 1.75 | 1.00 | *2.75* | 1.25 | 1.10 | 1.00 | 1.99 | 1.35 | 1.55 | 1.40 | 1.48 |
| *11* | The CEA applies appropriate **spatial** boundaries relative to the receptors selected for assessment in the CEA | 1.14 | *2.08* | 1.03 | *2.00* | 1.50 | 1.18 | 1.00 | *2.72* | *2.07* | *2.18* | 1.80 | 1.88 |
| *17* | The effects of multiple stressors **from the proposed activity** on receptors are assessed | 1.14 | 1.78 | 1.23 | 1.00 | 1.16 | 1.48 | 1.00 | *2.33* | 1.59 | 1.59 | 1.30 | 1.63 |
| *18* | The effects of multiple stressors from the **proposed activity and other activities** on receptors are assessed | 1.11 | 1.94 | 1.15 | *2.50* | 1.44 | 1.43 | 1.50 | *2.68* | 1.90 | *2.11* | *2.10* | 1.89 |
| *20* | A clear rationale for determining impact significance is presented and conclusions clearly relate to predicted change against an appropriate measure of population change | 1.04 | 1.72 | 1.20 | *2.00* | 1.47 | 1.40 | 1.50 | *2.93* | 1.70 | *2.07* | 1.50 | 1.87 |
| *21* | Uncertainty is explicitly considered and clearly identified | 1.04 | 1.67 | 1.10 | 1.75 | 1.22 | 1.20 | *2.00* | *2.10* | 1.39 | 1.50 | 1.50 | 1.52 |
| ***Average total score (**Willsteed et al., 2018**) (min, max score = 13, 52)*** | | 14.6 | *28.6* | 14.9 | *32* | 19.4 | 19 | *23* | **35.2** | *24.3* | *26.7* | *25.2* | *24.9* |
| ***Predicted increase in total score, per year, during the study (only reported if, p = <0.05)*** | | | 2.46 | | | | | | 1.80 | 1.57 | 2.09 | 1.56 | |
| ***Average number of CEA pages*** | | 0.17 | 5.54 | 0.69 | 1 | 0.94 | 0.53 | 1.3 | 24.2 | 3.72 | 5.11 | 4.3 | **7.70** |
| ***Average number of sources used to define baseline per CEA*** | | 1.71 | 5.78 | 5.9 | 11 | 7.12 | 4.8 | 7 | 10.8 | 8.22 | 9.73 | 9.4 | **7.64** |
| ***% of CEAs that provided a definition of cumulative effects/impacts*** | | 0 | 89 | 30 | 100 | 75 | 50 | 100 | 95 | 78 | 100 | 80 | **73** |
| ***% of CEAs with significance based on professional judgement*** | | 100 | 67 | 100 | 100 | 100 | 90 | 0 | 18 | 100 | 64 | 100 | **71** |
| ***% of CEAs that used maps to illustrate spatial scale of cumulative effects*** | | 0 | 11 | 0 | 0 | 0 | 0 | 0 | 45 | 22 | 0 | 20 | **15** |

*Boxes are shaded by score grade; white = very weak (average attribute score of 1 to <2), light grey/italic font = weak (average attribute score of 2 to <3), dark grey/bold font = strong (average attribute score of ≥3). The base of the table shows six additional measures that contribute to CEA standard.*

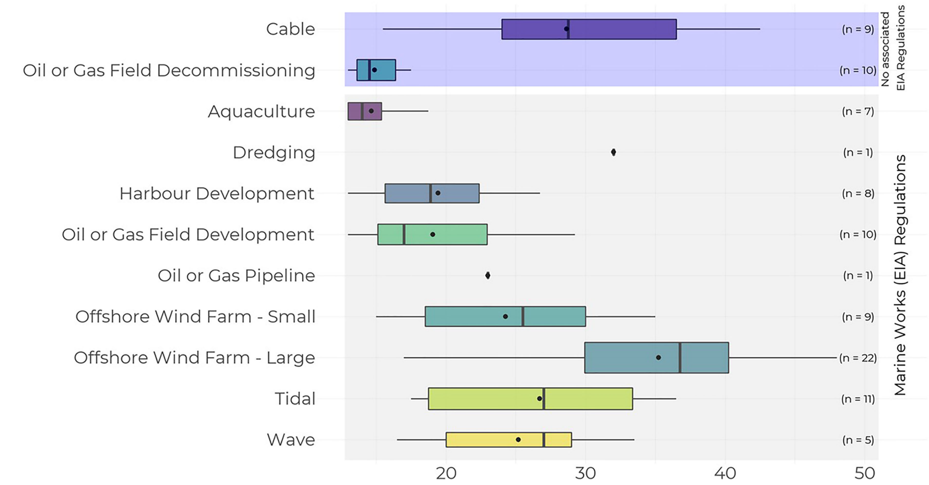



**FIGURE 2** | Mean and median scores of marine mammal Cumulative Effects Assessments (CEAs) across maritime industries. Industries are grouped based on the regulations associated with the required EIA and CEA (for more detail, see **Figure 1**). The industries that fall under the Marine Works (EIA) Regulations 2017 commonly also fall under the remit of other EIA Regulations, though the policy wording *via* other regulations is identical, hence only presenting the Marine Works (EIA) Regulations here for clarity. The boxplot shows the median (line through box), mean (dot), and lower and upper hinges (first and third quartile). Widths of boxes are proportional to the square-roots of the number of observations (CEAs reviewed) in the groups.

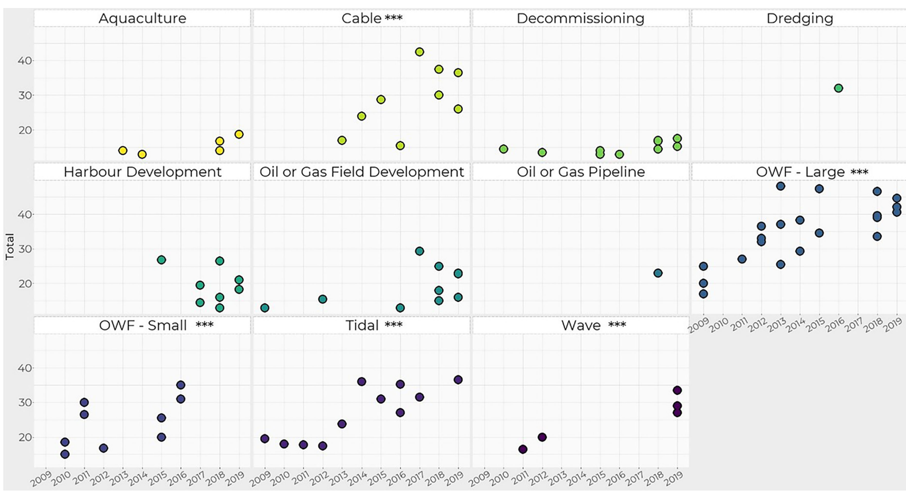



**FIGURE 3** | Total score of each Cumulative Effects Assessment (CEA) over the 10-year study period, by maritime industry. Asterisks*** next to industry name are displayed where change over time was significant, indicated by a multiple linear regression model, where p = <0.05.

definition was provided *via* the European Commission guidelines (Hyder, 1999); *"Cumulative impacts are impacts that result from incremental changes caused by other past, present or reasonably foreseeable actions together with the project"*, which was presented in 41% (n = 38) of CEAs that did provide a definition.

*Methodology* Setting out the purpose and scope of the CEA was on average the highest scoring attribute out of all the thirteen Willsteed et al. (2018) attributes that were assessed, with three industries (dredging, oil or gas pipeline and large OWF) all scoring 'strongly' (indicated by the scoring of Att. 2).

*Guidelines* 70% (n = 65) of the CEAs referenced some type of guidelines or documentation to guide the CEA process, with the Chartered Institute of Ecology and Environmental Management (CIEEM, previously IEEM) and the Institute of Environmental Management and Assessment (IEMA) guidelines referenced 47 and 29 times, respectively.

### 3.2.2 Sources for Baseline Characterisation

*Marine mammal baseline data* The average number of sources provided to characterise the baseline environment (i.e. marine mammal presence, abundance and distribution) for the CEA was 7.64, with dredging, large OWF and tidal CEAs on average citing the highest number of sources (11.0, 10.8, 9.73 sources, respectively). In contrast, CEAs from the aquaculture industry provided on average only 1.71 sources. '*Very strong*' and '*strong*' CEAs provided on average ≥11.7 sources of baseline information, whilst '*very weak*' CEAs provided on average 5 sources, and so the overall CEA score did appear to be reflective of the number of sources used to characterise the baseline environment. 3% (n= 3) of CEAs cited zero sources and did not provide any information on what their assessment considered as a baseline.

### 3.2.3 Effects Considered

Construction activity noise was the most frequently included type of stressor within the assessed CEAs (45%; n = 42), followed by vessel noise (29%; n=27). Often, there was an initial qualitative discussion with regards to the likelihood of occurrence of a number of potential sources of disturbance, and subsequent justification as to which impacts were taken forward into a more formal cumulative assessment. In this way, of the 426 times that potential sources of disturbance were considered in the 93 CEAs, they were scoped out of further assessment on 213 occasions (50%), whilst 213 were formally assessed in some way. Further assessment then consisted of either qualitative discussion, a quantitative analysis, or a mixture of both approaches (see *3.4.6 Quantitative vs. Qualitative*). Potential sources of disturbance may have been scoped out of inclusion in the CEA due to, for example, potential effects being deemed insignificant when experienced in isolation, or, lack of data or knowledge precluding assessment.

## 3.3 Spatial and Temporal Scale

*Spatial scale* The two Willsteed et al. (2018) attributes that directly considered the spatial scale of the CEA both scored '*very weak*' (Att. 9: mean = 1.99; Att. 11: mean = 1.88) (**Table 1**). 43% (n = 40) of CEAs did not explicitly state the spatial scale of the assessment. 14% (n = 13) of CEAs based their spatial boundary of pressures on a threshold relevant to the marine mammal(s) included in the assessment, using a defined boundary such as the reference population or management unit (MU) extent (e.g. 13 CEAs used the IAMMWG (2015) MU boundaries). Alternatively, 13% (n = 12) of CEAs defined the spatial scale of the assessment based on a set distance from the project itself, ranging from 0.5 km to 100 km.

15% (n=14) of CEAs presented a map to demonstrate the spatial scale of the project and to put into context the location of the proposed activity in relation to other projects or developments, and/or marine mammal distribution.

*Temporal scale* The two Willsteed et al. (2018) attributes that directly considered the temporal scale of the CEA on average both scored '*very weak*' (Att. 7: mean = 1.91; Att. 10: mean = 1.48) (**Table 1**). 63% (n = 59) of CEAs did not explicitly state the temporal scale of the CEA. 20% (n = 19) used the duration of the project as the temporal scale for the CEA, some also with the addition of a number of years post-project. Three CEAs used a defined scale of 5, 6 or 7 years in advance, whilst 5% (n = 5) of CEAs used the 'reasonably foreseeable future' as a temporal scale.

## 3.4 Assessment of Cumulative Effects

The conclusions of 25% (n = 23) of CEAs were accessible and compiled clearly, with supporting assumptions partially or fully addressed (indicated by scoring ≥3 for Att. 5). In contrast, 75% (n = 70) of CEAs had conclusions which were scattered, difficult to access, with the conclusions unclear (indicated by a score of ≤2.9 for Att. 5).

*Uncertainty* Att. 21 considered whether 'uncertainty is explicitly considered and clearly identified', and on average scored '*very weak*' (mean = 1.5). For 68% (n = 63) of CEAs, uncertainty was not explicitly considered (indicated by a score of ≤1.9 for Att. 21), and for a further 27% (n = 25) uncertainty was referenced in the methodology without being defined or formally considered within the assessment itself (indicated by a score of >2 but ≤2.9 for Att. 21). Only five CEAs clearly considered uncertainty within the assessment (scored ≥3 for Att. 21), all were from large OWFs.

### 3.4.1 Considered Effects of Multiple Stressors

85% (n = 79) of CEAs considered only like-for-like stressors together (i.e. construction noise together with vessel noise, rather than construction noise together with vessel collision risk). 49% (n = 46) of CEAs did not consider the effects of their activity combined with other activities (indicated by score of <2 for Att. 18) (**Table 1**). No CEAs for aquaculture, oil or gas pipeline and decommissioning industries considered multiple stressors of their activity combined with other activities. Of the 51% (n = 47) of CEAs that did consider the effects of multiple stressors from the project in question and other activities, 14 of those provided a clear rationale for selection of stressors with reference to marine mammals.

*Interactions* 16% (n = 15) of CEAs discussed the potential for multiple impacts to interact (i.e. additive, synergistic, multiplicative, compensatory or antagonistic), however this was not formally assessed within any CEA.

### 3.4.2 Quantitative vs. Qualitative

The method to determine impact significance was qualitative, based solely on professional judgement, for 71% of CEAs (n = 66). Only 6% (n = 6) of CEAs determined impact significance based on wholly quantitative methods, all these were from large OWFs. 23% (n = 21) used a combination, using quantitative analysis where feasible, supported by professional judgement where quantification was not possible. 100% of CEAs from aquaculture, decommissioning, dredging, harbour development, small OWF and wave were solely qualitative (**Table 1**). All CEAs that were categorised as '*very weak*' were qualitative assessments. Only four of the 93 CEAs (4%) scored a '*very strong*' overall score, and of these, one used solely quantitative methods, whilst three used a combination of qualitative and quantitative methods.

77% (n = 72) of CEAs used no quantitative model to predict the magnitude of effects. 21% (n = 20) of CEAs modelled cumulative exposure to underwater noise, most frequently using either the INSPIRE 'fleeing animal'[22], SPEAR or SAFESIMM (Donovan et al., 2012) models, the latter of which is designed specifically for marine mammals. 10% (n = 10) of CEAs modelled population consequences of disturbance, using either iPCoD (n = 5),

[22] https://www.subacoustech.com/services

Population Viability Analysis (n = 4) or another stage-based modelling approach (n = 1). Only large OWF used iPCoD within their CEAs. A further eight CEAs described an awareness of and/or an enthusiasm to utilise the iPCoD or DEPONS model to aid the CEA process, however such models were described as not being used due the complexity of the model, data required to parameterise the model not being available (e.g. piling schedule), or, the model not being available at the time of writing.

## 4 DISCUSSION

In the UK, CEA has been a required part of EIA practice for a number of decades, with current understanding and ability to assess cumulative impacts evolving substantially since assessment practice began to gain traction in the 1970s (Hodgson and Halpern, 2018). However, this review found differences in approach and varying standards of CEAs between industries and over time (2009-2019), according to the adopted scoring metrics. Before providing a thorough discussion of these findings, it is important to highlight the inherently difficult nature of completing a CEA. In some cases, the challenges are industry specific (e.g. more/less data for impacts relevant only to certain industries), whilst other challenges are reflected across industries, receptors and CEA practice (e.g. lack of knowledge on how stressors interact). It is therefore important to bear such challenges in mind when considering the findings of this review.

A caveat that we were not able to properly account for, but should be considered when interpreting the results, is that the level of detail required by the regulator for a CEA is usually proportionate to the size of the project (Lonsdale et al., 2017). For example, smaller scale projects may require less thorough CEAs. In some cases this could be due to smaller predicted impact footprints, or more localised activities (e.g. harbour maintenance and development), meaning, proportionate to the predicted risk, the regulator may only require a qualitative CEA, for example. This proportionality may be reflected in a lower score, according to our scoring criteria, though the CEA may in fact have fulfilled the regulatory requirements. We were only able to account for this, in part, for the OWF industry through categorisation of 'small' versus 'large' offshore wind farms. Comparatively, CEAs from smaller OWFs did score less than larger OWFs, on average, and were also shorter in terms of number of pages (**Table 1**). The review also found higher ranking CEAs had on average a higher number of pages. Previous work has struggled to confirm a clear relationship between project spatial scale and EIA length (Fernández et al., 2018), though Bond et al. (2014) found good practice had been eroded through recent streamlining of impact assessments themselves.

Further limitations to the review process itself are discussed in Section 4.2.

### 4.1 Variation in Cumulative Effects Assessment

#### 4.1.1 Between Industries

To our knowledge, this review provides the first comparison of CEAs across industries, and has highlighted a lack of standardisation within UK practice when assessing potential cumulative effects to marine mammals. The legislative wording with regards to the requirements for an EIA and associated CEA is identical or extremely similar for all but two (cable and decommissioning) of the eleven industries considered in this review (refer to Section 1 and **Figure 1** for an overview of the associated legislative regimes), yet there is a considerable variation in average CEA score between industries. The renewables industry (OWF, tidal and wave energy) scored the highest, whilst CEAs from the aquaculture, decommissioning, oil or gas field development, and harbour development industries scored lowest. The disparity in scoring across industries may to some degree be explained by differing interpretations of this legislation by regulators and practitioners, with similar disparities in interpretation and understanding reported within Swedish cumulative effects practice (Wärnbäck and Hilding-Rydevik, 2009). In the UK, the regulatory bodies who are responsible for the implementation of the associated legal requirements differ across industries and administrations. As such, the application of the legislation, including the production of associated guidance and support to improve practice, may also vary between regulators. To ensure the interpretation and application of the legislation is uniform across industries, we recommend that regulators should publish guidance, in conjunction with SNCBs, to support the production of CEAs in the marine environment, which should be applicable and adaptable to support the assessment process across relevant industries. The guidance should be designed to achieve the legislative requirements, whilst being adaptable enough to allow practice to evolve with new knowledge and best available science, and should aim to achieve effective cross-industry cumulative assessment.

The siloed approach between industries to the regulation and completion of these assessments has no doubt meant that industries have missed opportunities to share best practice and lessons learned. It is important to note that additional to the lack of synergy between industries, there are long standing disparities in the interpretation and approach to cumulative effects between scientists and practitioners (Hodgson et al., 2019). For example, scientific evaluations more commonly focus on the effects of environmental pressures on species or ecosystems and track this back to human activities, whilst policy-based CEA interpretation considers the effects of activities, projects or plans (Judd et al., 2015; Hodgson and Halpern, 2018). This disparity in interpretation may have led to incompatibility in emerging methodologies (Judd et al., 2015), with latest scientific evidence or tools not necessarily suitable to support practical CEA requirements or aid improvement of practice. The formation of a collective cumulative effects community across science, policy and practice could work to address the disjuncture identified across these fields, fostering an improved and more holistic understanding for all. Hodgson et al. (2019) suggest cumulative effects research be formalised as a subdiscipline, with the development of a community of practice, active conversations and support opportunities such as workshops and conferences. Such synergies could provide an ideal platform for innovative and applied research questions to be addressed, with research led by practitioner requirements. Examples of the success of this approach are evidenced by the development of tools such as iPCoD (King et al., 2015).

To accompany and support a more standardised approach across industries, the development of a systematic database used as a source and archive for CEA associated guidance, data, evidence and good practice would be highly beneficial (Clarke Murray et al., 2014; Foley et al., 2017; Dibo et al., 2018; Durning and Broderick, 2019; Hodgson et al., 2019; Caine, 2020). This would provide a standard platform of activities and their associated effects to be recorded, and when required, incorporated within baseline characterisations and CEAs, encouraging a more collaborative overall 'CEA mindset' (Sinclair et al., 2017; Gușatu et al., 2021). This would likely be more cost effective than the current disjointed approach, where data are not readily shared due to being commercially sensitive and expensive to collect (Connelly, 2011). Further, sharing of data with subsequent dissemination of long-term pre-, during- and post-construction monitoring data would allow industries to learn from past developments and improve the collective knowledge of the true cumulative effects and consequent impacts of consented projects (Masden et al., 2010; Hawkins et al., 2017; Dibo et al., 2018; Caine, 2020).

Parallel to industry-level support through a cumulative effects community, it is important that cumulative effects are also addressed over the broader scale. The adoption of a more 'integrated ocean management' approach could provide a wider holistic solution to the currently fragmented approaches between industries and administrations (for a further discussion, see Winther et al., 2020). If effective measures and an integrated approach spanning all marine industries is in place at a wider strategic scale then adverse cumulative environmental change at an individual project level may be avoided (Gunn and Noble, 2011). Such approaches would capture the incremental effects from smaller projects and activities that are not subject to EIA (Cooper and Sheate, 2004), which are unlikely to be captured by project-level CEAs. Further, a strategic ocean management approach may somewhat alleviate the, in some cases, disproportionate expectations for individual projects to complete comprehensive CEAs. This is especially important, as all industries considered in this review exploit similar or directly adjacent areas of the marine environment (Van den Burg et al., 2019), yet we document a great disparity in their assessment of the potential impacts of multiple stressors to marine mammals. A lack of standardisation is a notable limitation of current management and conservation efforts. Activities which may impact the same receptors and marine space, either individually or cumulatively, should be subject to the same rigour and standards when it comes to proportionately assessing their potential impacts; long-term sustainability can only be achieved if best practice is applied across all industries (Winther et al., 2020).

#### 4.1.2 Over Time

Notably, there was an increase in CEA scores over the sample period, though this improvement was attributable to only five industries [cable and renewables (large and small OWFs, tidal and wave energy)], with four industries exhibiting no significant improvement (aquaculture, decommissioning, harbour and oil or gas field development). This may be a reflection on differing levels of pressure imposed on each industry to improve and evolve practice. For example, 'newer' industries (such as marine renewables) have worked together to develop specific renewable-focused guidelines to support the assessment of environmental effects (e.g. RenewableUK, 2013), with parallel academic research feeding into the evolving understanding of industry impacts (e.g. Sparling et al., 2018; Whyte et al., 2020). In contrast, this pressure and collective effort to improve practice may not be imposed on 'older' industries, hence not showing an improvement over time, according to our scoring metrics.

#### 4.1.3 Terminology

Previous research has found how (and if) practitioners initially define 'cumulative effects' and 'impacts' to be one of the main pre-cursors to subsequent inconsistency in practice (Cooper and Sheate, 2002; Duinker et al., 2013; Foley et al., 2017). 26% and 9% of the CEAs reviewed here provided either no definition or a vague definition, respectively. This may be due to an uncertainty by practitioners as to what cumulative effects are, and/or due to a lack of universally utilised precise working definition (Baxter et al., 2001; Cooper and Sheate, 2004; Duinker and Greig, 2006; Gunn and Noble, 2011; Judd et al., 2015). Furthermore, historic earlier definitions are now being recognised as insufficient (Duinker et al., 2013), and so we support the persistent calls for the establishment and widespread adoption of a consistent, comprehensive, definition of impact and of cumulative effects, based on best available science (Cooper and Sheate, 2002; Canter and Ross, 2010; Judd et al., 2015; Foley et al., 2017), which is complex enough to guide practice and scientific work (Duinker et al., 2013), for example adoption of the definitions and concepts outlined by Judd et al. (2015). It should be clear which effects could contribute to cumulative impacts, and should be standard practice to state which effects, impacts, pressures, sources and receptors are investigated, in order to reduce the present ambiguity (Cooper and Sheate, 2002; Foley et al., 2017; Hodgson et al., 2019; Elliott et al., 2020).

#### 4.1.4 Baseline Characterisation

Characterising the baseline using appropriate data and background information is an essential part of the CEA process, as it not only describes the level at which potential negative effects are being assessed against, it also has implications for how significance is evaluated (Clarke Murray et al., 2014). Ideally, the baseline should describe species presence, including information on abundance, distribution and seasonality, and should acknowledge existing and potential stressors in the surrounding area.

Choosing a point in time at which to define the baseline is challenging. Whilst using a historic non-affected environment would be ideal (Masden et al., 2010; Clarke Murray et al., 2014; Foley et al., 2017; Dibo et al., 2018), in practice this is extremely challenging, with a survey of CEA practitioners across the Pacific Rim finding that past conditions only tended to be included in the baseline definition by more experienced CEA consultants (Foley et al., 2017). Instead, it is common practice to use current conditions as the baseline (Bérubé, 2007; Clarke Murray et al., 2014; Foley et al., 2017), despite this approach not taking into account 'shifting baseline syndrome'. This is the phenomenon where each new generation of scientists or practitioners uses the current environment during their career as a baseline against which to evaluate changes, which results in a gradual shift in baseline over time, as each successive generation may use a more disturbed benchmark than the last (Pauly, 1995). In terms of UK practice, the regulations relevant to CEA and EIA in the UK (outlined in **Figure 1**) suggest using the present environment as the baseline, stating the explicit requirement for a description of the current environmental state, along with an outline of how this may evolve in future without the project. The regulations also state that this should be based on relevant information and scientific knowledge, though anecdotally the present review found the most up-to-date and/or appropriate data and sources were not always utilised (taking into account when the CEA was undertaken), with 3 of the reviewed CEAs not providing any information on what their assessment considered as baseline. For marine mammals around the UK, there are a number of data sources that can aid in characterising the baseline environment, these range from wider scale survey data [e.g. SCANS surveys (Hammond et al., 2013; Hammond et al., 2017)], to finer scale local data. Future practice should aim to ensure the most appropriate sources are utilised.

#### 4.1.5 Effects Considered

Impacts included in CEAs tended to be limited, focusing on similar projects (e.g. an OWF only including other co-occurring OWF in their CEA), or including only projects or stressors where there was high confidence in the data available. Furthermore, many of the CEAs reviewed only carried forward impacts that were deemed significant in isolation into the CEA itself, similar to the findings of Baxter et al. (2001) and Olagunju and Gunn (2013). Only including impacts that the assessment deemed significant in isolation does not take into account that two or more potential impacts concluded as non-significant in isolation could result in significant cumulative impacts (Clarke Murray et al., 2014). There is a growing body of literature documenting how cumulative exposure to individual stressors that were deemed non-significant, when considered cumulatively result in biologically significant consequences for marine mammals [e.g. bioaccumulation of organochlorines in prey impacting reproductive health of NE Atlantic bottlenose dolphins and killer whales (*Orcinus orca*) (Jepson et al., 2016)]. Furthermore, this review found potential sources of disturbance may be scoped out from inclusion within the a CEA due to lack of data or knowledge, limiting further assessment of potential cumulative effects. However, only including stressors with sufficient information, or indeed only similar projects, does not truly reflect the real-world scenarios that marine mammals are experiencing (e.g. chemical pollution), and so has implications for the robustness of the assessment. Despite this being common practice, 90% of surveyed practitioners agreed that *all* disturbances should be considered when evaluating baseline conditions (Dibo et al., 2018).

#### 4.1.6 Spatial and Temporal Scale

It is essential to define the extent of the focal area to determine the scope of the assessment, and which activities and pressures to include within the assessment (Judd et al., 2015). However choosing the appropriate scale, spatially, temporally and in terms of level of detail, is one of the key challenges to effective assessment of cumulative effects (Therivel and Ross, 2007; Clarke Murray et al., 2014), particularly for highly mobile species such as marine mammals (Gușatu et al., 2021). Scale can be difficult to define, for example, stressors at a global scale (e.g. climate change) will impact on receptors at a local scale (Clarke Murray et al., 2014). Similarly, the frequency and duration of pressure(s) will influence the nature and scale of response by the receptor (Judd et al., 2015). Within this review 63% and 43% of CEAs did not explicitly state the temporal or spatial scale of their assessment, respectively, meaning it was unclear how other projects, stressors or activities were selected for inclusion within the CEA in terms of their spatial and temporal proximity.

The appropriate spatial scale of a CEA is context specific and so should be tailored to each project and to each source of disturbance, as some impacts may be better suited to being considered at a defined radius whereas some impacts may require consideration at an ecological scale. For the latter, this could correspond with the receptor's range (Clarke Murray et al., 2014), or be defined by receptor-specific ecological boundaries (Dibo et al., 2018), though only 14% of CEAs reviewed defined spatial boundaries with relevance to marine mammals [e.g. marine mammal management units (IAMMWG, 2015) or seal management units]. In contrast, 13% of CEAs assessed defined a spatial parameter based on a set figure, e.g. a 5 km radius from the project. This approach is more appropriate for certain activities where a radius of disturbance can be defined (e.g. piling noise), though may still not necessarily be appropriate for mobile species. It is most common for spatial scale to be defined using the footprint of expected impact or activity across all CEA receptors (Foley et al., 2017), although in some cases this may be more suitable for sedentary receptors or benthic habitats rather than highly mobile or migratory species like marine mammals or seabirds, therefore consideration to the ecology and behaviour of the receptor is required.

In practical terms, the temporal scale of a CEA should ideally accommodate the complete life cycle of all elements of the project: exploration, construction, operation and decommissioning (Lonsdale et al., 2017). In fact, surveyed CEA professionals suggested a CEA should consider impacts beyond the life cycle of a project itself (Dibo et al., 2018), as project duration may not adequately consider that behaviour may change over time in response to the action, and that this change may be lagged (Masden et al., 2010). The consideration of only high impact, short-term pressures (e.g. impact pile driving) with the preclusion of the effects from persistent long-term pressures does not represent the reality of the cumulative pressures imposed on receptors. Nevertheless, the majority of professional practice tends to scale their analysis based on the duration of the proposed activity or impact only (Foley et al., 2017). In the present review, it was found that when a temporal scale was defined, project duration was the most used temporal limit (20%). This approach may be appropriate at a single project level when there is evidence that the effect of the stressor associated with the activity is alleviated when the activity ceases [e.g. harbour seals re-distributed as per non-piling within two hours of cessation of pile-driving activity (Russell et al., 2016)]. However, where effects of stressors persist long after an activity (e.g. permanent decline in gray whale habitat use following an increase in human-induced disturbances from fishing and other shipping activity; Findley and Vidal, 2002) then this approach is not suitable. It is recommended, where possible, that the best approach in defining a temporal scale for a CEA should incorporate the length of time required for the ecological components to recover (Masden et al., 2010; Foley et al., 2015; Foley et al., 2017). The challenge here is that receptors whose response is lagged to certain stressors may mean the temporal boundary of the assessment extends beyond the operational timeline of a project.

### 4.1.7 True Assessment of Multiple Stressors?

CEAs tend to consider cumulative effects in one of two ways; either as multiple stressors from a single activity, or as single stressors from multiple activities (like for like) (Clarke Murray et al., 2014). Consideration of multiple stressors from multiple activities is rare (Clarke Murray et al., 2014); indeed of the CEAs reviewed 49% never considered multiple stressors from multiple activities. Notably, this was completely absent in CEAs from the aquaculture, decommissioning, and oil or gas pipeline industries. 85% of all CEAs reviewed considered only like for like stressors together (e.g. construction noise together with vessel noise, rather than construction noise together with vessel collision risk). However, for marine mammals and other receptors, it is unlikely that stressors are only experienced in this way, and so this approach may not be representative of the impacts on ecological components (Clarke Murray et al., 2014).

Only 16% of the CEAs discussed the potential for impacts to interact, with the surrounding uncertainty associated with these interactions never expressly considered. In practice it is often necessary to assume effects interact only additively (or do not interact at all), however it is important to acknowledge this assumption, and the consequent potential for under- or over-estimation of impacts which further affects the effectiveness of any management or mitigation measures (Judd et al., 2015; Singh et al., 2020). To address this knowledge gap, further empirical research is required to identify and distinguish effects of single vs multiple disturbances, and to build an understanding of how stressors interact (whether additive, synergistic, multiplicative, compensatory or antagonistic) in order for this aspect of the CEA process to improve (Clarke Murray et al., 2014).

### 4.1.8 Methodology: Qualitative Versus Quantitative

There are some quantitative methods and tools being developed, or already available, to assist the practitioner when assessing cumulative effects on marine mammals, for example, the iPCoD model (King et al., 2015; Booth et al., 2020), and other stepwise quantitative approaches [e.g. Piet et al., 2021) (refer to Section 1.1 for further examples]. However, few CEAs reviewed used wholly quantitative methods to determine significance (6%, all from large OWFs), and instead 71% relied on professional judgement or did not describe how significance was determined. All CEAs from aquaculture, decommissioning, dredging, harbour development, small OWF and wave energy industries were solely based on professional judgement. This is a higher proportion than other reviews of CEA, one of which documented 26% of CEAs described cumulative effects qualitatively, whilst another documented 66% of CEAs relied solely on expert judgement (Cooper and Sheate, 2002; Korpinen and Andersen, 2016). Professional judgement has historically been relied upon by necessity, due to the absence of established thresholds of response or tools to aid the assessment process (Cooper and Sheate, 2002; Connelly, 2011; Clarke Murray et al., 2014; Lonsdale et al., 2017; Singh et al., 2020). For CEAs to be robust, reliable and transparent, the onus is on practitioners to support their judgements with the best available science.

Predictive models are a potential additional or alternative method to professional judgement, and can be used as a tool to simulate and predict the potential for significant impacts (Duinker et al., 2013). Geographic Information Systems (GIS) are another tool that can be extremely useful in facilitating the CEA process (e.g. Lonsdale et al., 2020; Gușatu et al., 2021), but again, only 15% of the reviewed CEAs presented a map as part of the marine mammal CEA. Maps can be used to superimpose the spatial extent of projects, activities and stressors with receptor presence, to aid interpretation of potential impacts, in addition to the quantitative analysis or qualitative assessment (e.g. Halpern et al., 2008; Batista et al., 2014), and have been used to map potential cumulative impacts on four marine mammal species in the California Current (Maxwell et al., 2013) and cumulative impacts on Hong Kong's pink dolphins (Marcotte et al., 2015).

*Uncertainty* It is important that CEAs acknowledge the limitations of their chosen approach, and are transparent in how this contributes to the uncertainty and consequent interpretation of the CEA outputs (Judd et al., 2015). In the UK, including *'details of difficulties (for example, technical deficiencies or lack of knowledge) encountered compiling the required information and the main uncertainties involved'* is specifically outlined in the Marine Works (EIA) Regulations 2007[1] as a requirement to be included within the EIA and associated CEA. Yet, only 5% of the CEAs reviewed explicitly considered uncertainty (all from large OWFs). Considering the high level of uncertainty surrounding marine mammal individual and population responses to disturbance, and often the uncertainty surrounding the stressor itself (e.g. definite construction period), identifying and making decision-makers aware of the degree of uncertainty associated with these assessments is essential.

## 4.2 Limitations

While every effort was made to ensure this review was conducted in a robust manner, there are some important caveats that should be understood when considering the findings of this study. First and foremost, we were not able to ascertain whether the scores given as part of this review correlated with effectiveness of the assessment themselves, though it is plausible that this is the case. CEAs were scored using a framework developed by Willsteed et al. (2018), to allow comparison of scores across industries and through time. The attributes of the framework were designed based around UK legal obligations for CEA, principles of marine ecosystem approaches to management and key principles of assessing cumulative effects, and were trialled by Willsteed et al. (2018) to score benthic and fish and shellfish ecology CEA chapters from the marine renewable energy industry. As those same legislative requirements span nine of the eleven industries reviewed, there may be a degree of bias in the scoring criteria towards those, with bias against the decommissioning and cable industries. However, it was felt that the attributes were broad enough to represent a number of features that all CEAs should include to be satisfactory (e.g. definition, quantitative methods, clear methodology, uncertainty acknowledged) and as such we do not expect this has significant implications on our findings.

This review assessed documents from all marine industries that legally require CEA completion, and so we therefore expect these findings to accurately reflect the state of CEA practice within UK seas. However, there were differing levels of accessibility associated with the documents reviewed across sectors, as has been noted by other studies (Ball et al., 2013; Lees et al., 2016). Only one CEA was found for the dredging and the oil or gas pipeline industries, despite a thorough search. This may be a reflection of the rarity within that industry to complete a CEA, or it may be a problem of access, but without review of further CEAs from those industries it is not possible to confirm whether the scores here are representative of their respective industries practice. EIAs and their associated CEAs are, in principle, public documents, and are inherently useful when looking at how to improve future practice and address knowledge gaps, and so we recommend accessibility to such documentation is improved (Lees et al., 2016). Furthermore, where practical, supporting documentation and data should be made available, as simply reviewing completed CEAs is not always enough to determine the methods and tools practitioners have undertaken (Foley et al., 2017). This may have led to an underscoring in the present review, as the final document is often a summarised version of a long, nuanced and complex process.

Finally, the scores of the reviewed CEAs do not account for the scoping advice received from the regulator or statutory advisors, or for the influence of practitioner expertise or experience, which has previously been identified as a factor that may influence CEA standards and practice within other countries (Wärnbäck and Hilding-Rydevik, 2009; Kågström and Richardson, 2015; Kågström, 2016; Foley et al., 2017). In some cases, practitioners may be employed by project proponents, which may lead to potential bias and/or conflicts of interest (Duinker and Greig, 2006). Whilst we again could not consider this in the present review, standardising practice through sharing of guidance and minimum standards of approaches would potentially somewhat alleviate this influence on CEA practice.

## 5 CONCLUSION

Cumulative Effects Assessment serves as a tool to estimate the overall expected impact on a receptor of interest (Judd et al., 2015). The aim of a CEA should be to provide decision-makers with sufficient information to support consenting decisions which ensure the sustainable development of marine spaces in parallel with conservation of the receptors considered. Inadequate or inclusive assessment of potential impacts can lead to the consenting of developments and projects that put significantly underestimated levels of stress on local marine environments (e.g. King and Pushchak, 2008). The robustness

**TABLE 2 |** Recommendations to improve and standardise Cumulative Effects Assessment practice.

**Approach**
1. CEAs to consider *multiple* stressors from *multiple* activities or sources
2. All impacts initially considered through a risk screening and prioritisation process uniform across industries, documented by a clear audit trail
3. Temporal scale of each impact considered within the CEA based on the length of time required for the receptor to recover from that specific impact, with lags in response time incorporated where required, as per the best available science/current knowledge
4. Spatial scale of the CEA tailored to the appropriate context per impact, taking into consideration the spatial range of the receptors and the scale of impacts, as per the best available science/current knowledge
5. Where possible, use predictive models to assess cumulative effects, acknowledging caveats and surrounding uncertainties to the chosen approach. If this is not possible and/or proportionate, ensure professional judgement is based upon best available science
6. Use of Geographical Information Systems as a tool to aid the CEA process, to demonstrate spatial and temporal overlap of multiple stressors

**Transparency**
7. Standard practice to state which impacts, effects, pressures, sources and receptors are considered and which are scoped out, with reasoning
8. Thorough and transparent description of CEA methodology
9. Transparency in describing the knowledge gaps, and the implications this has on uncertainty in the CEA process

**Management**
10. Consistency and standardisation regarding the assessment of cumulative effects across industries
11. Adaptive CEA management informed by regular reviews of CEA practice

**Further work required**
12. Development of a standard comprehensive definition of impact and effects
13. Systematic database used as a source and archive for CEA associated data, evidence, guidance and good practice
14. Synergies developed across science, policy and practice through the formation of a cumulative effects community. Support opportunities such as workshops and conferences to provide platforms for active conversation which may aid development of innovative approaches and provide more holistic understanding
15. Development of thresholds for disturbance
16. Field research to identify and distinguish effects of single vs multiple disturbances, building further understanding of how stressors interact, coupled with the development of tools and frameworks that allow findings to be integrated into assessments

and validity of these assessments therefore plays a pivotal role in the protection of marine mammals, and any other receptors, from anthropogenic impacts.

We document a disparity in how cumulative effects are being considered across the same marine space, with considerable discrepancies in the efficacy of CEAs across maritime industries, with some (aquaculture, harbour development, decommissioning and oil or gas field development) not showing any signs of improvement over the study period. Considering the findings of this review and a wider consideration of the scholarly literature, we offer recommendations (summarised in **Table 2**) which may go some way to ensuring cumulative effects are considered in a consistent manner, and appropriately mitigated for, across the marine environment. In providing these recommendations, we acknowledge the very significant challenges to doing CEA well, and that a 'one size fits all' approach is not always appropriate. Despite this, it is expected that these findings, and recommendations, are broadly applicable to global CEA practice, including industries and receptors within both the terrestrial and marine environment. This is timely considering reviews of CEA practice elsewhere have also found CEA implementation to be less than satisfactory, with similar challenges to those identified in the present review also reported elsewhere [e.g. in Canada (Baxter et al., 2001; Duinker and Greig, 2006; Sinclair et al., 2017), the United States (Ma et al., 2012; Schultz, 2012), Sweden (Wärnbäck and Hilding-Rydevik, 2009), in the Brazilian Amazon (Athayde et al., 2019) and in the Arctic (Kirkfeldt et al., 2017)].

This work highlights a siloed, sector by sector, non-uniform approach to assessing cumulative impacts on marine mammals. Future work could explore whether the variation in practice highlighted here has in fact resulted in sustainable development being non-uniform across UK industries and waters. Long-term sustainability of the marine environment can only be achieved if all industries work to the same standards in terms of protecting the environment from significant harm (Winther et al., 2020), and so we suggest the development of a cumulative effects community approach in order to facilitate standardisation in approaches and sharing of best practice. Adoption of a more holistic, rather than fragmented, approach would help to ensure the continued development of the marine environment does not compromise the conservation of marine mammals, and indeed other species and habitats.

## DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

## AUTHOR CONTRIBUTIONS

EH and LM contributed to the conception and design of the study. EH, AL, CS, CM, DV, RW, RC and LM reviewed and scored the CEAs. EH organised the resulting database, performed the statistical analysis and wrote the first draft of the manuscript. All authors contributed to further manuscript drafting and revision, and read and approved the submitted version.

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## SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2022.822467/full#supplementary-material

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# Revisiting fuel tax concessions (FTCs): The economic implications of fuel subsidies for the commercial fishing fleet of the United Kingdom

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ABSTRACT

Fuel forms a significant portion of the total expenditure for many commercial fishing vessels and in some cases, profitability can be dictated by fuel costs. In many nations, including the UK, these fuel costs are reduced by cost-reducing subsidies. There is evidence of growing support from various channels that public opinion is moving towards a reassessment of fuel subsidies. Analysis of the economics of the UK fishing fleet, using publicly available industry-supplied data, implies that the nominal annual value of fuel tax concessions for diesel is between £ 150–180 million per year (2009–2019). That support is largely provided to the most fuel-intensive fishing methods, such as mobile demersal trawls and dredges. Results show that, without the current fuel tax concession, several fleet segments would be deemed unprofitable. This paper outlines the current value of fuel tax concessions for fishing vessels and potential policy considerations for reform.

## 1. Introduction

One of the most substantial and variable costs of fishing is fuel expenditure [1]. However, the proportion of total costs that it constitutes varies between different fleet segments. For example, large vessels likely have higher fuel costs in proportion to revenue, than smaller vessels that might stay closer to port. Indeed, in 2018, small-scale fishing fleets in the European Union (EU) collectively used about 147,000 litres of fuel for every €1 million of revenue generated, while their large-scale counterparts used more than twice the amount for the same revenue [2]. Furthermore, the species being targeted and the type of gear being used will influence fuel consumption [3,4]. Targeting small pelagic species in large volumes is often more efficient than targeting more evasive species i.e. demersal beam-trawls have much higher fuel costs than pelagic trawls or drifting gillnets [5].

Fuel also represents one of the most heavily subsidised costs of fishing. Globally, fuel subsidies are estimated to have totalled $8 billion in 2018 and represent more than a third of all harmful fisheries subsidies [6]. According to the World Trade Organization [7], fuel subsidies can be government payments that directly subsidise fuel costs such as grants and loans, or indirect contributions in the form of foregone government revenue, such as exemptions from normal rates of fuel tax.

The UK provides commercial fishers support through fuel tax concessions; the '*Red diesel rebate*' (RDR) [8] and the '*Marine voyages – relief from fuel duty*' [9]. RDR is a fuel tax concession available to several sectors of the UK economy. While the fishing sector uses this same red diesel,[1] the '*marine voyages – relief from fuel duty*' is specific to commercial marine vessels and entitles them to a further discount compared to what other users pay for the same fuel type. Certain commercial vessels, including UK fishing vessels and foreign-owned fishing vessels refuelling in the UK [9], are able to avail themselves of this relief and thus pay zero fuel duty. However, because different fishing methods and vessel types use different amounts of fuel for the same volume of output, the benefits enjoyed from this relief (i.e. as a proportion of their variable costs) is likely to favour more fuel-intensive methods [10,11].

From April 2022, access to the RDR was removed for some non-marine sectors [8,12], but no change is currently planned for commercial fishing. The policy objective driving this announcement was to "*help meet its [the UK Government's] climate change and air quality targets*" and "*ensure that the tax system incentivises users of polluting fuels like diesel to improve the energy efficiency. invest in cleaner alternatives, or just use less fuel*" [13]. These justifications for changes to the RDR could be applied

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[1] Some fishing vessels use petrol, which does not qualify for the RDR, but it does qualify for *marine voyages – relief from fuel duty*.

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to the '*marine voyages – relief from fuel duty*'. In fact, '*marine voyages – relief from fuel duty*' entitles the fishing sector to pay no excise taxes, so would need to be reformed alongside RDR. We herein refer to the combined tax relief provided by RDR and the '*marine voyages – relief from fuel duty*', as Fuel Tax Concessions (FTCs), with the understanding that the role of FTCs (specifically RDR) is already being revisited in other sectors.

For any future transition, changes to FTCs must be fully informed, just and deployed in a phased manner. To enable this, we must have a fundamental understanding of the overall value of FTCs and an awareness of the differential importance across the UK fishing fleet, to fully understand the current reliance on the system and implications of potential policy changes. This study, therefore, aims to explore for the first time the effects of FTCs on the costs and profits of fishing for the UK commercial fishing fleet. With this in mind, we will estimate the overall value to the FTC and look at the sector-level impacts of FTC removal on profitability. Finally, we will disaggregate these estimates to understand the economic importance of FTCs for each segment of the UK commercial fishing fleet.

## 2. Material and methods

This analysis draws upon several publicly available data sources to provide an exploratory re-analysis of the economic performance of the UK's commercial fishing fleet, by fleet segment, and the effect of FTCs on profitability. The methodological approach consisted of collecting official available information via the following broad steps:

- Annual financial fleet performance data from 2009 to 2019 (both overall and across fleet segments) were taken from the Seafish[2] website [14];
- Information regarding the rate of FTC for UK fishers, in terms of pence per liter (ppl) savings, were taken from government disclosed values on fuel subsidy use in the UK [8,9];
- Information on engine manufacturer for each commercial fishing vessel on the UK Shipping Registry (Part 2) enabled estimation of the proportion of diesel use for each fleet segment;
- Annual cost of subsidized diesel purchased at the pump by the fishing industry were provided by Seafish;
- The above data were used to calculate total annual cost savings through the purchase of fully tax-free red diesel for each fleet segment from 2017 to 2019;
- Economic performance indicators and cost saving estimates for each fleet segment were then standardized by total landed tonnage of catch per fleet segment; and
- Net profit margins for each fleet segment with and without FTCs were generated by subtracting total annual costs from total annual revenue and dividing this by total annual income.

### 2.1. Sources of data

Annual financial performance indicators regarding the UK commercial fishing fleet were taken from the Seafish website [14]. Seafish conduct annual surveys of UK fishing vessels to gather data on: their activity, catch, income streams, expenditures and vessel specifications (e.g. length, power, Gross Tonnage (GT) and age). Data were downloaded separately for each of the 30 fleet segments that are tracked by Seafish (See; Table S.1 and Section 2.2) and the overall fleet data. Two fleet segments are excluded by Seafish from the fleet segment specific data; these are the UK pelagic fleet and miscellaneous segments. The specific metrics used in this study are: total income, fuel expenditure, total expenditure (including all costs), number of active vessels and total landings (tonnes). Data from 2009 to 2019, inclusive, were downloaded, providing us with an eleven-year period of the most recent and complete data at the time of the study. Financial data are the nominal values and do not account for inflation, to give a comparable account of foregone revenue for a given year.

The average annual cost of diesel (ppl) for UK fishers from 2009 to 2019, inclusive, was obtained from Seafish [15]. This data records the monthly price paid at the pump for diesel. We derived an annual average price of marine diesel for our calculations. The fishing industry pays no excise tax on diesel due to the '*marine voyages – relief from fuel duty*' [9], therefore the excise tax on the diesel price provided by *Seafish* is 0.00ppl. The full rate of excise tax for diesel, with no FTCs, is 57.95ppl [12]. Our analysis assumes all diesel-powered vessels uptake these available FTCs.

Information on 4231 of the 5422 registered UK fishing vessels was provided by the UK Shipping Registry from 'Part 2 – Fishing' of the registry (11/11/2021 extraction). Our preliminary analysis of these data showed them to be representative of all size categories and thus representative of the UK commercial fishing fleet. This data contained information on vessel length, engine make, model and power (kW). A proportion of vessels of lengths greater than and less than 10 m using diesel was determined by checking engine make and model specification (Table 1).

### 2.2. Defining fleet segments

As described by Ulrich et al. (2012), a fleet (or fleet segment) is a group of vessels with the same length class and predominant fishing gear during the year, and a métier is a group of fishing operations targeting a similar (assemblage of) species, using similar gear, during the same period of the year and/or within the same area and which are characterized by a similar exploitation pattern [16]. Here, we refer to each fleet segment, to reflect the terminology used by Seafish [15]. This is vital information for the analysis of the implications of FTCs as the rate of fuel use and the proportion of costs that fuel constitutes is known to vary by vessel type, gear used and species targeted.

The present study analyses 30 unique fleet segments, separated by vessel size and power, fishing location and fishing method (Supplementary Table S.1). Analyzing the segments in isolation allows us to look at how each would be affected by the removal of FTCs. Financial data for fleet segment 8, representing vessels greater than 10 m with an annual fishing income less than £ 10,000, were highly variable and not representative of the fleet, therefore, results for this fleet segment are presented in Tables S.1 & S.2, but are excluded from the main body of results.

### 2.3. Data analysis

Some of the metrics (relating to finances) were available as averages per vessel within a fleet segment, while others (total landings) were totals. All metrics which were available as averages were multiplied by the number of vessels within the fleet or fleet segment such that all metrics used in analyses represented fleet segment totals. Seafish data make several assumptions, including assuming uniform engine types within a fleet segment, which will introduce some degree of deviation

**Table 1**
The proportion of fuel use assumed to be diesel in a UK commercial fishing métier[31] where the vessels are either, over 10 m in length or predominantly under 10 m in length.

| Vessel length | Diesel use (%) |
|---|---|
| > 10 m | 99.7 |
| < 10 m | 77.5 |

[2] Seafish is a UK non-departmental public body funded through a levy on the first sale of seafood products in the UK. Its remit is to support the seafood sector.

from true values [17].

Calculation of the increased cost of fuel was based on increasing the annual price of duty-free diesel by the sum of the FTCs, 57.95ppl (the RDR provides a rebate of 46.81ppl and the marine voyages provides a rebate for the final 11.14ppl) across the study period. The annual cost of fuel consumption was multiplied by the proportional increase of diesel with the rebates from FTCs added, to give the annual cost of fuel use without FTCs (Eq. 1).

$$FE \quad without \quad FTC = FE * \frac{AD + 57.95}{AD} \tag{1}$$

Where, *FE* is the fuel expenditure, either for an individual fleet segment or the entire fleet, and *AD* is the annual average price of marine diesel. The overall value of FTC was calculated by subtracting the original fuel expenditure from the cost of fuel with FTC removed.

To account for some fuel use being petrol rather than diesel within a fleet segment, we reduced the cost of the fuel rebate by a fixed percentage, based on the size of the vessels within each fleet segment. Each segment was classified as either having most vessels greater or less than 10 m in length (Table S.1) and assigned a percentage for diesel use based on the fuel use from the UK Shipping Registry (Table 1). As the data for the whole fleet contained some records outside of the 30 fleet segments, we estimated the overall expenditure on diesel attributed to fleet segments greater than 10 m in length. This calculation was based on 88.9% of the total fuel expenditure being accounted for by fleet segments predominantly over 10 m and of these vessels 99.7% used diesel. Likewise, for vessels under 10 m, the proportion of the whole fleet's fuel expenditure was 11.1% and diesel use across vessels was 77.5%. Therefore, the overall predicted percentage of fuel expenditure on diesel was 97.2% across the whole fleet. To account for this in our valuation of the FTC, we reduced the value of the FTC by the fixed percent (97.2%) when looking across the fleet as a whole, but applied the fleet segment specific percent (Table 1) when conducting fleet segment specific analysis.

This method means that our valuation of FTCs excludes the contribution of petrol. However, because the underlying data calculates all fuel expenditure based on the price of diesel, calculation of FTCs for petrol vessels is not possible. Our method also assumes that all diesel and petrol vessels within a segment have the same financial indicators. We also make a significant simplifying assumption that FTCs will not lead to changes in fuel use, and that all vessels will uptake available FTCs. We also make the simplifying assumption of no changes to variable costs, such as crew share, after the introduction of FTCs. It is also assumed that fishers cannot transmit cost increases to price increases. In other words, the price of fish remains constant.

Analysis for fleet segment specific costs and profit margins uses only the most recent data from 2017 to 2019, to ensure the most recent estimate for the effects of FTC removal while retaining enough data to show annual variation. Fleet segment specific fuel expenditure with the added cost of FTCs was calculated (Eq. 1), using the same process as for the whole fleet. New fleet segment specific total expenditures were also generated by adding the additional cost of FTCs to the total fleet segment expenditure. Profit margins were calculated using Eq. 2, in scenarios with and without FTCs added onto the total expenditure. When calculating the total expenses of the fleet, we included all annual outgoings presented by Seafish including: total costs (insurance, repairs, gear, hire and maintenance, other vessel expenses, commission, harbour dues, fuel costs, subscriptions and levies, shore labour, boxes, ice, bait, crew travel, food stores, quota leasing, days purchased, other fishing expenses, crew share), depreciation, interest on loan repayments and other financial costs.

[3] A metier is a group of fishing operations targeting a similar assemblage of species, using similar gear, during the same period of the year, and/or within the same area and which are characterized by a similar exploitation pattern.

$$Profit \quad margin = \left(\frac{Total \quad income - Total \quad E}{Total \quad income}\right) * 100 \tag{2}$$

Where, *E* is the expenditure, either for an individual fleet segment or for the fleet as a whole. The annual value of FTCs was calculated by subtracting the estimated fuel expenditure from the fuel expenditure without FTCs. Analyses on fleet segments are summarized across 2017–2019, inclusive, and show mean and standard deviation. All analysis was done in R version 4.0.3 [18].

## 3. Results

The incurred cost of fuel for the UK commercial fishing fleet has fluctuated over the past decade, ranging from £ 159 million in 2012 to £ 95 million in 2016 (Table 2). The total amount of FTCs that the UK fleet has benefitted from, however, has varied only slightly over the same time period, from £ 176 million in 2009 to £ 151 million in 2019 (Fig. 1. A). As such, the proportion of the total realised fuel costs (fuel cost incurred by fisher plus FTCs) that the FTC represents changed through time, from a maximum of 64% to a minimum of 50% (Table 2). In 2015 and 2016, the estimate of FTC represented 63% and 62% of the total realised cost of fuel for the fishing fleet, respectively, effectively representing a ca. 62% discount in the real cost of fuel for those years where the cost of fuel was low.

In terms of economic performance, the UK fishing fleet has remained profitable over the last decade in terms of the realised costs and revenues, as the total annual costs have remained less than total annual revenue (Fig. 1. B). However, if we consider the reduction in costs due to FTCs, we see that the profitability of the fleet is less clear cut. Between 2009 and 2015, the true total costs of fishing, i.e., annual fuel costs with FTC removed, was higher than annual revenue (Fig. 1. B). However, in the latter half of the period, between 2016 and 2019, revenue was generally above the estimated total cost of fishing without FTCs. This indicates that the fishing fleet in aggregate would not always have been profitable in the absence of FTCs. The impact on individual fleet segments will vary.

The removal of FTCs proportionately affects those fleet segments with the largest fuel expenditure per tonne of landed catch (Fig. 2; Table S.2 provides a complete breakdown of estimates). Many of those fleet segments with the highest fuel use were estimated to be unprofitable without FTCs (Fig. 3). Specifically, the segments most affected, going from marginally profitable (where the standard deviation overlaps with zero) to unprofitable, were 'North Sea Nephrops' and 'beam trawlers'). Additionally, the 'South West beam trawlers' also switched from being profitable to either marginally profitable or unprofitable

**Table 2**
Maximum and minimum fuel cost and realised fuel cost (fuel cost plus fuel tax concession, FTC) incurred by the UK commercial fishing fleet across the study period (2009–2019).

| | Maximum amount (£ m.) | Minimum amount (£ m.) | Maximum proportion FTC constitutes (%) | Minimum proportion FTC constitutes (%) |
|---|---|---|---|---|
| Fuel cost (Fuel cost incurred by the fishing sector) | 159.0(2012) | 94.6(2016) | - | - |
| Realised fuel cost (Fuel cost incurred by the fishing sector + FTC) | 317.0 (2012) | 235.2(2016) | 63.7 (2009) | 49.9 (2012) |

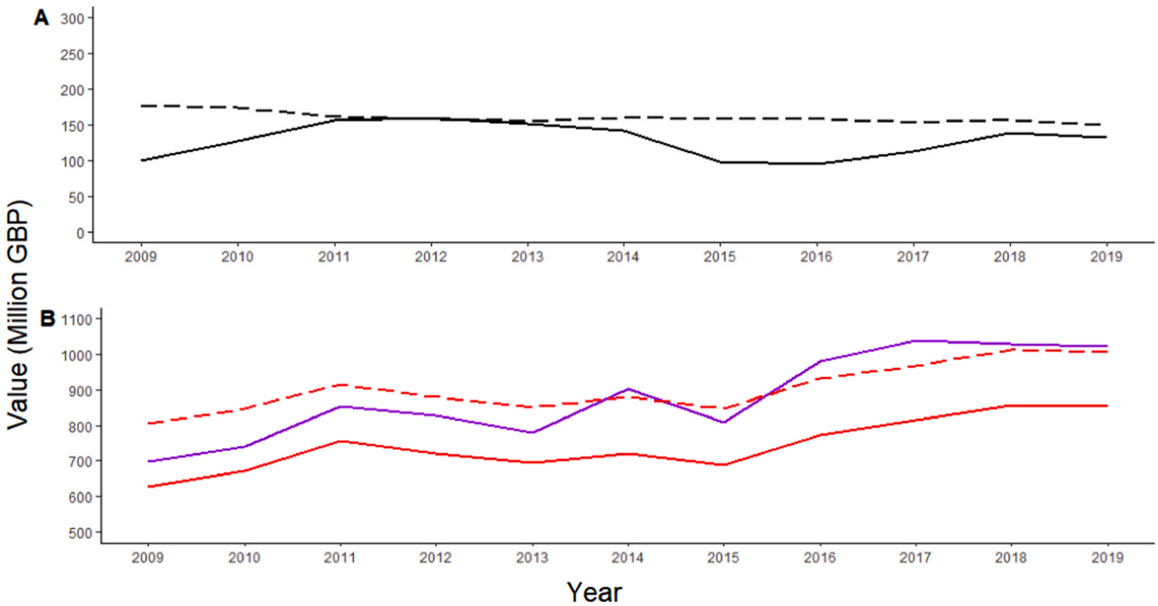


**Fig. 1.** The financial performance of the UK commercial fishing fleet between 2009 and 2019. A; The total annual fuel costs (black line) and the annual estimated FTCs (dashed black line). B; Total annual costs (red line) and revenue (purple line) of the UK fishing fleet, and the estimated total annual costs with FTCs removed (dashed red line).

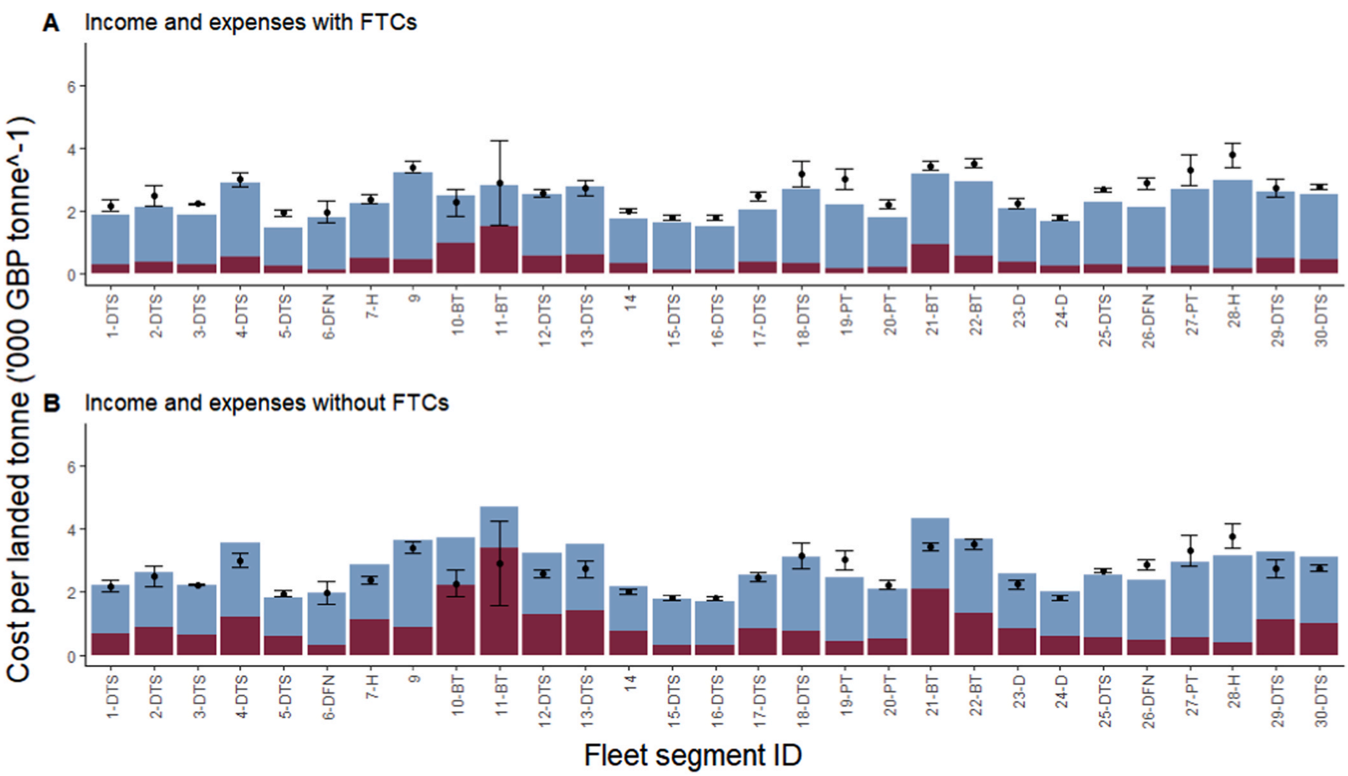


**Fig. 2.** The fuel costs (red bar), total expenditure (blue bar) and income (black point) of the fleet. A; with the current taxation rate of 0ppl. B; with the full tax rate of 57.95ppl added onto fuel and total expenditure. Amounts are shown as thousands of GBP (£), standardised by annual total tonnes of catch per fleet segment. Where income exceeds expenditure, fleet segment is operating profitably. All amounts represent the mean, while error bars show the standard deviation over 2017, 2018 & 2019. Letters following the fleet segment ID refer to primary fishing gear. For fleet segment ID and primary fishing activity information, refer to S. Table 1.

after considering the additional costs of removing FTCs (Fig. 2 & 3). Overall, most segments that were unprofitable after the removal of FTCs were vessels associated with: beam trawling, Nephrops and scallop dredging. Most of the fleet segments remaining profitable without FTCs are those associated with pots, traps and hooks (Fig. 3). All of which generally have a lower fuel use per landed tonne (Fig. 2). The other important aspect for a fleet segment to remain profitable seemed to be the size of the current profit margin, with segments which had a high fuel use per unit of catch remaining profitable due to large initial profit margins, e.g., 'Area VII BCDEFGHK trawlers 10–24 m′ (segment 5).

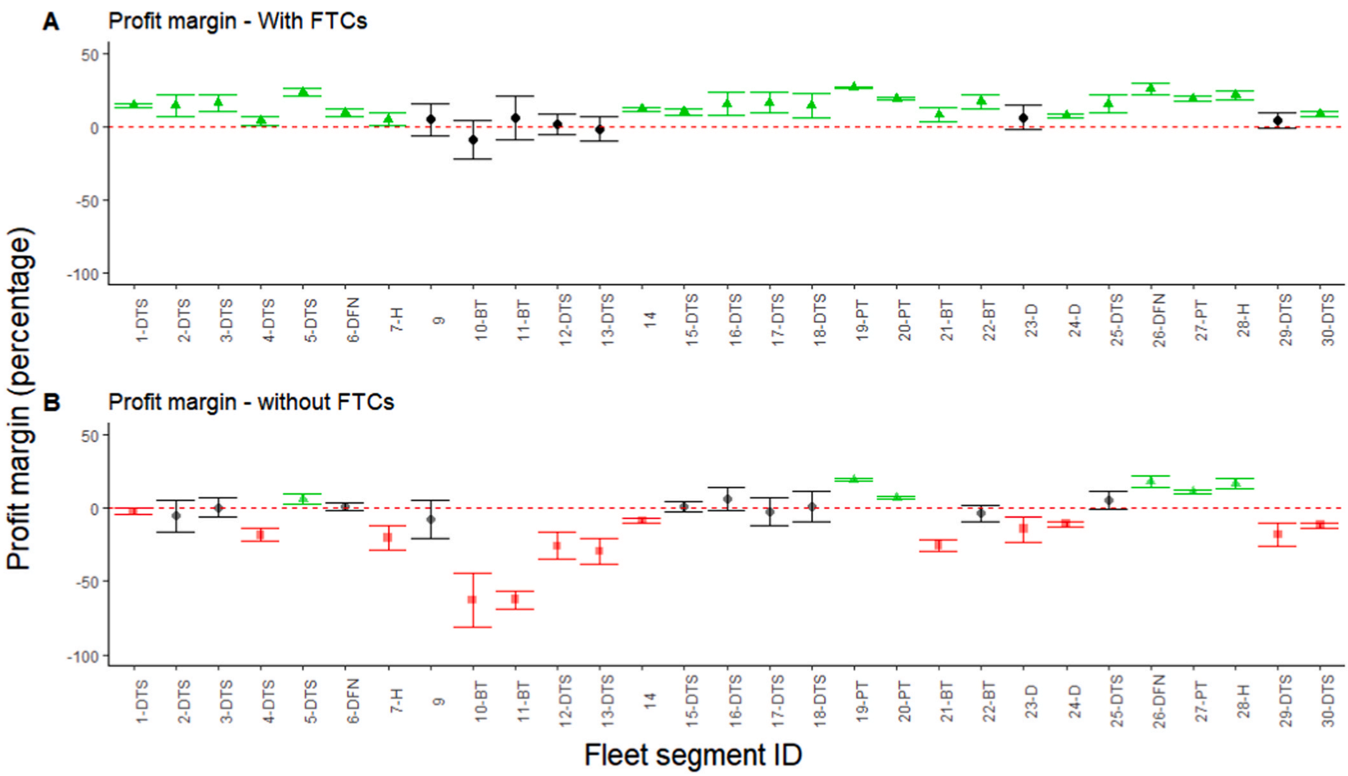


**Fig. 3.** The mean and standard deviation (error bars) of the net profit margins of each of the fleet segments between 2017 and 2019. A green triangle indicates the average and standard deviation are wholly above 0. A black circle indicates the standard deviation crosses 0. A red square indicates the mean and standard deviation lie wholly below 0. Panels show (A) profit margins with FTCs and (B) after the removal of FTCs. For fleet segment ID descriptions, refer to S. Table 1.

## 4. Discussion

Our primary aim was to understand the role of FTCs in the economic performance of the UK fishing fleet. We found that the total amount of FTCs, i.e., the amount of foregone revenue due to RDR and the '*marine voyages – relief from fuel duty*', ranged from £ 176.1 m in 2009 to £ 150.6 m in 2019. Our results suggest that if FTCs were immediately removed without adaptation to reduce fuel consumption on average, the UK fleet would likely have remained profitable only in recent years. Prior to 2016, with the exception of 2014, the fleet's annual income was less than or equal to the estimated range of real expenditure. This is in contrast to the findings of Isaksen et al. [19] who demonstrate that FTCs were not a prerequisite for a profitable Norwegian fishing fleet, suggesting that economically sustainable fisheries are possible without fuel subsidies. They went on to show that the impact on small coastal vessels was minor compared to larger vessels [19], as did Carvalho & Guillen [2] due to them being more fuel efficient (i.e. they use less fuel for the same output).

At the level of individual fleet segments, we found that all are estimated to be profitable or marginally profitable between 2017 and 2019, whilst receiving FTCs. However, when the cost-reducing effect of FTCs was removed, 14 of the 30 segments analysed became unprofitable, some significantly so, and nine moved from being profitable to marginal (Fig. 3 & S. Fig. 2). Those segments which have the highest fuel expenditures naturally benefit most from FTCs and therefore are most negatively impacted by its removal. Conversely, segments with lower fuel costs, generally the smaller vessels, were less affected and remained profitable or marginally profitable in the absence of FTCs. The fleet segments most impacted are generally larger vessels using active or mobile gears, such as dredges or demersal beam-trawls (Tables S.1 and S.2). Providing support to these segments via FTCs enables them to remain economically viable, despite significant fuel requirements. These results, which corroborate previous findings [2,19], provide an initial projection of how the potential removal of FTCs might impact the UK fishing fleet and where support to meet climate change commitments could be targeted.

Arguments could be made that keeping these otherwise unprofitable segments viable supports food and livelihood security, and generates income for potentially vulnerable communities. However, given that much of the catch is currently exported and is not consumed by the domestic market [20], the food security argument is weaker than first seems. Furthermore, the intention of this study is not to argue for the removal of economic support for these segments, but that the strategies and rationales for their subsidisation are justifiable in the context of ongoing commitments to ensure sustainable and equitable use of our marine resources and in reaching carbon net zero [21]. In the absence of clear and explicit policy objectives for the continuation of FTCs for fishing, it is challenging to construct arguments in support of, or against, their utility. Furthermore, certain fishing methods have environmental consequences beyond the direct removal of fish and production of Green House Gases (GHG) - the release of 'Blue carbon', for example, especially via fishing methods that significantly disturb the seabed [22].

### *4.1. Consequences of FTC removal*

Many industries are preparing for significant shifts in how they operate to reduce GHG and wider ecological footprints. While the fishing industry continues to benefit from FTCs, it would be prudent to consider how its modification could be achieved in an equitable and just manner.

Removing FTCs would result in higher fuel costs, which, theoretically, would result in behavioural changes towards using less fuel [19,23]—fishing closer to port or reducing time spent at sea[24]. This suggests that conversely, fuel costs that are lowered via FTCs, may enable fishers to stay out longer and burn more fuel. The continued provision of FTCs may act as a disincentive towards adopting less fuel-intensive fishing methods and/or more fuel-efficient engines. Additionally, given the pervasive nature of FTCs and the level of competition between UK and EU fishing fleets, not least for the same fish stocks but also within the same markets, it seems that any reduction in FTCs may need to be reciprocated across other fishing nations, particularly the EU. As Martini & Innes [25] summarised, "concerns regarding competitiveness have motivated support for reducing fuel costs, under the theory that reducing fuel costs through support are necessary to mitigate any competitive disadvantages of domestic fisheries". This realisation prompted Carvalho & Guillen [2] to go on to argue that the ongoing World Trade Organization negotiations, were therefore "potentially fundamental to avoid the risk of an unlevelled playing field, e.g., in relation to countries with a different fuel tax exemption treatment and their fleets sharing the same fishing grounds or their products competing in the same markets". Ultimately, international harmonization of fuel taxation rates is potentially the best solution to reduce GHG emission derived from fishing [19].

The ability for some of the UK's fishing fleet to be able to remain profitable following an immediate FTC removal appears to be limited, at least in the short term. Consideration should therefore be given on how to provide an appropriate incentive to support the transition away from a reliance on FTCs. In the short term for example this could be through reforming or redirecting the support provided to the industry, provided through the UK Seafood Fund [26]—indeed, the EU is already supporting the energy transition by, for example, supporting changes to more efficient engines through the EMFAF, as it did through the EMFF. [27]. Such government expenditure could be used to update the equipment of the fleet and reduce reliance on fossil fuels. However, a detailed consideration of alternatives and future support mechanisms are outside the scope of this paper, but removing or tapering FTCs, could generate additional public revenue which could be directed back into the fishing sector. Additional revenue could be redeployed to provide the infrastructure to support the electrification of the UK fishing fleet, both at shore and at sea, for example—smaller, less fuel-intensive vessels, could make ideal candidates for transitioning towards alternative forms of power, including electric [28,29]. Or it could be used to support the refitting of fishing vessels with those gears demanding that demand less fuel.

Substantial and persistent fuel price increase through the removal of FTCs would induce greater adjustment possibilities. As such, it is likely that vessels operating in fleet segments currently most reliant on FTC may be displaced into more efficient segments (or novel segments), or there could be a consolidation or shift of quota rights from less to more energy efficient gear [19]. In this scenario, displacement into different fleet segments targeting different species may adversely impact the sustainability of the newly targeted species, particularly if the target species is not managed by quota and could alter the wider environmental impacts of the UK fishing fleet such as increased bycatch. Although a broad move from mobile demersal gear to static or pelagic gear could be a positive outcome in terms of reducing carbon emissions, it is unclear whether those 'viable' fleet segments would remain profitable with a considerable influx of additional vessels and effort. Displacement could also occur spatially, towards areas with greater fish abundance, or where economically viable fleets were able to persist, potentially resulting in increased resource competition and conflict [30]. Alternatively, some fishers may be displaced out of the fishing sector entirely, especially if the transition from low to high fuel costs is abrupt or is done without support. Given this potential outcome, support to the industry during transition should be considered.

Potential displacement or fleet restructuring ramifications following the removal of FTCs and any introduction of alternative sector support mechanisms will require careful consideration. FTC alterations could cause changes in fishing patterns, fleet composition, species targeted, areas fished, and types of gear used. As competition for marine resources, namely space, increases, accurate information on how various sectors, including fisheries, use marine space is paramount to inform an effective and considered marine management decision making regime. It is worth considering whether subsidising businesses via FTCs is in the best interest of the UK taxpayer, as the current benefits, beyond employment opportunities, may not justify the public expenditure, particularly when considering that much of UK quota fished by the UK fleet is held by beneficiaries based overseas [31].

## 5. Concluding remarks

It is recognised that profitability of the fleet is dependent on fuel prices, which in recent years have increased significantly. This work therefore provides a baseline for future analysis.

This paper highlights that revisiting FTCs for the UK commercial fishing sector has the potential to generate both economic and environmental benefits. The removal would likely incentivise a transition towards less fuel-intensive fishing methods, thereby reducing GHG emissions. Less fuel-intensive fishing methods will also lead to a reduction in the ecological impact of fishing on the marine environment, thereby increasing its resilience to the impacts of climate change - in turn benefiting fishers [32].

In light of the objectives of RDR removal for other sectors i.e., delivering carbon net zero by 2050, the persistence of FTCs for fishing is difficult to reconcile. As Martini & Innes [25] explained, "when making the case for reform, it may be more useful to consider the policy objectives motivating support than the impacts of such support". However, the objectives of current UK FTC policy, as applied to fishing, is unclear. As support for FTCs becomes untenable (against a backdrop whereby carbon emission taxes are being considered), there is the opportunity to reimagine how support for the fishing sector could be provided to deliver the transformational change required. Recognising that UK fishers are in competition with other nations' fishers, the challenge will therefore be to ensure they are able to compete economically and are supported to reduce their emissions and ecological footprint. If the objective is improving competitiveness, in the long run the continuation of FTCs can be counterproductive as it delays adjustment and masks structural problems [25]. The consideration of carbon border taxes [33] may also provide economic opportunities in the longer term if the UK is successful in decarbonising its fishing fleet.

If the FTC for the fishing sector is modified, it is unclear what the future UK fishing fleet will consist of, thus it is important to begin discussions and preparations in advance. It is therefore anticipated that this paper will generate debate about how support could be provided to the fishing sector to ensure that it is truly sustainable when viewed from economic, social and environmental perspectives.

Having an accurate value of both the whole fleet FTC and the fleet segments that are most dependant on FTCs is essential if future challenge to the policy arises. This may be especially poignant in light of growing pressure to revisit fossil fuel reliefs from Non-Governmental Organisations [34], other nations [2], scientists [35] and even the fishing industry [36]. Additionally, such challenges are likely to be raised through debates on meeting the target of carbon net zero by 2050 (a 78% reduction by 2030 compared to 1990 levels[4]) [37–39] and reducing the release of blue carbon from seabed disturbance [22] by reducing demersal gear impacts to help deliver the UK Marine Strategy [40]. If appropriately modified, FTCs could help achieve Good Environmental Status [41], the sustainability and climate change objectives in the Fisheries Act 2020 [42], and would globally be in support of the United Nations Sustainable Development Goal 14 [43].

## CRediT authorship contribution statement

DV and DS: conceptualisation, design of the study. JD statistical analysis. All authors contributed to manuscript drafting and revision,

[4] emissions from fishing vessels fall under the definition of domestic transport

and read and approved the submitted version.

## Declaratiom of Competing Interest

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

## Data Availability

Data will be made available on request.

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## Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.marpol.2023.105763.

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