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Assessing the impact of key Marine Invasive Non-Native Species on Welsh MPA habitat features, fisheries and aquaculture.

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Marine Biological Association of the UK

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Experts within the project team reviewed *Rapana venosa* (Jack Sewell); *Watersipora subatra* (Christine Wood) and *Asterocarpa humilis* and *Didemnum vexillum* (John Bishop).

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Erys agweddau ansicr allweddol, ac maent wedi’u crynhoi. Mae asesu'r risg a grëir gan ffactorau sy'n achosi straen mewn systemau cymhleth yn heriol, ac mae'r adroddiad hwn yn amlinellu’r cafeatau a’r cyfyngiadau ynglŷn â chreu asesiadau risg a’u gweithredu.

Executive Summary

Marine Invasive Non-Native Species (INNS) are a significant threat to global biodiversity and can have detrimental socio-economic impacts on activities such as fishing, shipping and aquaculture. To address information gaps NRW commissioned the Marine Biological Association of the UK to conduct an evidence assessment for 16 INNS species that are either present or likely to arrive and may cause medium to high risk to marine ecosystems. The project assessed the risk to 41 Welsh Marine Protected Area habitat features that are of particular importance to conservation and fishery and aquaculture operations and target species.

To assess the impacts from INNS two established methods of ranking impacts were adopted. Potential impacts on MPA features were evaluated using the Environmental Impact Classification of Alien Taxa (EICAT) methodology. This approach assesses impacts based on twelve impact pathways on species and habitats (e.g. competition, predation and structural impacts on ecosystem). The impact scores were combined to provide an overall impact score for each INNS X MPA habitat feature. To assess impacts on aquaculture and fisheries, the socio-economic impact classification of alien taxa (SEICAT) method was adopted. The core characteristic of this approach is that it uses changes in people's activities as a common measure for impact. Both assign impacts to the categories: Massive; Major; Moderate, Minor or Minimal concern. If there was not enough evidence to assess impact this was recorded as ‘Data deficient’.

A targeted literature review was undertaken to collate evidence to assess the impacts on MPA features, fisheries and aquaculture. The evidence review was supported by previous in-house work and GBNNSS Risk Assessments. Relevant experts for each INNS were identified by the project team and contacted to request a review of the final factsheet and Excel spreadsheets to quality assure the outputs and provide additional evidence.

The project outputs consist of this report and two summary Excel workbooks that present the EICAT and SEICAT risk assessment scores and confidence. The supporting evidence for each assessed INNS are presented in the annexes of this report.

The results show that MPA features are likely to be exposed to a range of INNS with all assessed species occurring in a range of habitats, although the number of features exposed varied for each INNS. Impacts on MPA features and fishery and aquaculture operations and target species varied. Species capable of altering habitats and biofouling were considered most likely to lead to higher risks. Predatory gastropods could lead to significant impacts on bivalve species. Biogenic habitats comprised of molluscs and on-substrate cultivation of molluscs were at high risk from INNS. Contact with most INNS was not considered likely to lead to direct health and safety impacts, although lifting risks from added biofouling weight and cuts from shells or injuries from clawed crustaceans were identified as potential impacts.

Key uncertainties remain and are summarised. Assessing the risk from stressors in complex systems is challenging and this report outlines caveats and limitations in creating and applying risk assessments.

1 Introduction

1.1. Background

Marine Invasive Non-Native Species (INNS) are a significant threat to global biodiversity and can have detrimental socio-economic impacts on activities such as fishing, shipping and aquaculture. Biosecurity measures can prevent the spread of INNS and limit the likelihood of a species entering an area in the first place; such measures are particularly important for marine ecosystems, where eradication and control techniques have been shown to be less effective. A previous study (Macleod *et al*. 2016) assessed the environmental (with a particular focus on MPA features) and socioeconomic impacts of 8 key marine INNS.

Natural Resources Wales are currently working with stakeholders to develop a cost-effective biosecurity plan for the Pen Llyn a’r Sarnau Special Area of Conservation. This will address the main pathways of introduction and spread of invasive non-native species and minimise the risks to protected features and marine wild fisheries. The project is funded by the European Maritime and Fisheries Fund and is a pilot for developing cost-effective biosecurity planning for Wales’ network of Marine Protected Areas.

Unfortunately, existing impact data for the majority of marine INNS can be scarce and/or impacts have not been quantitatively or experimentally studied over sufficiently long temporal and broad spatial scales (Ojaveer *et al*., 2015). This presents a challenge when prioritising the conservation aims and objectives for a biosecurity plan.

To address information gaps NRW commissioned the Marine Biological Association of the UK to conduct an evidence assessment for 16 INNS either present and presenting high to medium risk to marine ecosystems or considered likely to arrive and cause impacts in the near future. This report details the evidence gathering and risk assessment methodology used to identify the potential impacts of INNS on Welsh MPA features, fisheries and aquaculture. This piece of work does not replicate previous work by Macleod et al. (2016) as it considers additional INNS with no overlap in species between the two reports.

1.2. Aims

The purpose of this contract is to assess potential impacts of 16 INNS (see methods section for species) on Welsh MPA features, commercial fisheries and aquaculture to address knowledge gaps for biosecurity planning.

1.3. Project Outputs

The project outputs consist of this report and two summary Excel workbooks that present the assessment matrices that identify the impact/risk level associated with the INNS with confidence scores on MPA features and socio-economic impacts with a focus on aquaculture and fisheries.

1.4. Report Structure

This report consists of this introductory section and methods, results, discussion and conclusion sections. Detailed technical information for each INNS is presented in the accompanying annexes of this report (Annex 4-19). The information in the annexes underpins the matrix assessments for both the EICAT and SEICAT assessments (provided in separate Excel workbooks).

2 Methods

2.1. Non-native species evaluated by this project

Under the EU Marine Strategy Framework Directive (MSFD) a UK Monitoring and Surveillance list for marine INNS has been developed by Stebbing *et al*. (2015) to focus efforts on priority species and identify those that do or could have high environmental impact.

The MSFD list has been adopted by NRW and the associated information has been used to develop a “Priority Monitoring and Surveillance list for Wales”, which is more specific to the INNS present and absent in Welsh waters. A scaled down list of INNS (Table 1) has been taken from the list for this project. Although the assessment process for compiling these lists has identified the INNS that may have a high impact, there remains a lack of knowledge about their specific impacts on habitats, native species and economic activities.

Table 1. List of INNS taken from the Priority Monitoring and Surveillance List for Wales[1] assessed by this project. More technical detail for each species is presented in the annexes accompanying this report.

Level of risk	Scientific name*	Common name	Annex
High risk species for monitoring	*Asterocarpa humilis*	Compass sea squirt	4
	Crepidula fornicata	American slipper limpet	5
	Didemnum vexillum	Carpet sea squirt	6
	Eriocheir sinensis	Chinese mitten crab	7
	Watersipora subatra	Red ripple bryozoan	8
Medium risk species for monitoring	*Bonnemaisonia hamifera*	Bonnemaison's hook weed	9
	Caprella mutica	Japanese skeleton shrimp	10
	Crassostrea gigas (*Magallana gigas*)	Pacific oyster	11
	Gracilaria vermiculophylla (*Agarophyton vermiculophyllum*)	A red seaweed (no common name)	12
	Diadumene lineata	Orange-striped anemone	13
	Ensis leei	American jackknife clam	14
Species for surveillance	*Rapana venosa*	Asian rapa whelk	15
	Urosalpinx cinerea	American oyster drill	16
	Homarus americanus	American lobster	17
	Mnemiopsis leidyi	American comb jelly	18
	Ocenebra inornata (*Ocinebrellus inornatus*)	Asian/Japanese oyster drill	19
*Updated scientific names are shown in brackets, these are used throughout the report			

2.2. MPA features evaluated by this project

Welsh Marine Protected Areas (MPAs) have been designated to protect a range of features including mobile species and habitats. This project focusses on MPA seabed habitat features. The full list of assessed features is presented in Annex 1. In summary the project assessed the risk to 41 MPA habitats that are of particular importance to conservation defined under various legislative instruments (see Table 2 below). The MPA features were aligned to biotope and sub-biotope features using the JNCC Correlation Matrix that is available on-line[2] and shows relationships between EUNIS (2004 and 2007 versions), the Marine Habitat Classification for Britain and Ireland (v15.03) and habitats listed for protection.

Assessing the impacts of INNS for every species that occurs within an MPA feature would be extremely challenging, not least because the composition of species is likely to vary between the same MPA feature in different locations. Many species, particularly those that are rare, small and without commercial or conservation value are poorly studied with little information available on ecology and life history. To address this, for each MPA feature characterising species were identified that were important to provide physical structure, key functions or that characterised the feature, such that their loss would result in the loss of the MPA feature. For MPA

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[1] https://gov.wales/sites/default/files/publications/2018-02/invasive-aquatic-species-priority-marine-species.pdf

[2] https://mhc.jncc.gov.uk/resources/

features that are broadscale habitats the characteristic species within underlying biotopes were considered.

For MPA features that are defined by the presence of species, such as oyster and mussel beds and fragile anthozoan communities identifying key species is straightforward. For broadscale habitats this is more challenging as a range of different species may be present and work to identify typical species to support management is ongoing. The loss of a characteristic species was considered by this project to represent more of an impact to the feature than a similar decline or loss of a typical species and this is taken into consideration in the assessment methodology and impact categories (see Section 4.7 and Annex 2).

Table 2. Summary of MPA Features for which the risk of INNS establishment and impact was assessed by this project. (For full list see Annex 1).

Source/relevant legislative instrument	Number of features
EUNIS[3] Level 3 (substratum and energy level) and identified as MPA features under ENG Guidance[4]	22
OSPAR 'Threatened and declining habitats'[5]	4
Habitats of Principal Importance, identified under section 7 of the Environment (Wales) Act 2016[6]	13
Joint NERC/OSPAR features	2

2.3. Assessing the likelihood of MPA feature exposure

The starting point for risk assessments was to identify whether the MPA feature (for EICAT based assessments) was likely to be exposed to the INNS. The likelihood of presence in MPA features for each INNS was assessed using a five point scale (see Table 3). We did not assess whether the species had been recorded in that MPA feature in Wales or the likelihood of establishment, but these considerations did inform the assessment of suitability and increased confidence in the assessment.

A number of MPA features, particularly broad-scale habitat features, may occur across a wide range of conditions (such as a range of salinities) with a variety of characteristic species present reflecting differences in habitat. Where possible the likelihood of exposure was assessed for constituent biotopes within a feature to support application of the assessments at a site specific level. The impact assessments, however, consider only the worst-case level of impact for the MPA feature.

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[3] https://eunis.eea.europa.eu/about

[4] Natural England and JNCC (2010)

[5] https://www.ospar.org/work-areas/bdc/species-habitats/list-of-threatened-declining-species-habitats

[6] Environment (Wales) Act 2016 replaces section 42 of the Natural Environment and Rural Communities Act (2006)

Table 3. Assessment categories used to characterise presence of INNS within MPA features.

Presence Category	Definition
Suitable	The MPA feature (or some constituent habitats) is documented either in peer-reviewed literature or reports as suitable, or proxy evidence from similar habitats around the world indicates the habitat is suitable.
Potential	Based on proxy evidence or similar habitat types the habitat is judged likely to be suitable, but the evidence base or density indicates that the habitat is in some way sub-optimal or there are uncertainties around the evidence.
Unlikely	There is considerable uncertainty around suitability and there are some indications based on evidence or proxy information that the MPA feature is unlikely to be suitable for the species. If established this is considered likely to be at low densities.
Not suitable	There are strong indications, based on evidence or proxy information that the MPA feature is not suitable for this species.
No evidence	No evidence was found to assess feature suitability and proxy information was not available or suitable.

2.4. Fisheries and aquaculture activities assessed by this project

The risk assessment considers direct impacts on fisheries and aquaculture via interactions which affect deployment and operation of gear and installations and impacts on target or cultivated species. It does not consider indirect impacts or ancillary activities (for example, impacts on gaining access or impacts on bait).

Key fishery and aquaculture operations (Table 4) were identified from studies on fishing techniques known to occur, or to have recently been pursued, in Wales (Hall *et al*., 2008; ABPmer, 2016).

Target species (Table 5) were identified from UK Sea Fisheries Statistics (MMO, 2018). In 2018 Welsh vessels caught 1 % of the UK landings and 3 % of the value. The main target species were based on landing information for Welsh Ports (2014-2018) and cross-checked against ICES sub-rectangle catches reported for main target species. The final list of fisheries and aquaculture types was agreed with NRW staff. While other fish species are reared in land-based fish farms in Wales, including lump fish, these were disregarded as they are outside of the scope of this project. The project did evaluate both commercial hand-gathering and social/recreational gathering in the SEICAT assessments as these activities may overlap.

Table 4. Main fishery and aquaculture activities likely to occur in Wales. INNS assessments are based on the broader activity categories not sub-activities.

Activity category	Sub-activities within category
Fisheries	
Towed (demersal)	Beam trawl (shrimp)
	Beam trawl (whitefish) (mixed demersal fish, epifauna)
	Multi-rig trawls (mixed demersal fish, epifauna)
	Light otter trawl (mixed demersal fish, epifauna)
Dredges (towed)	Scallops (King)
	Scallops (Queen)
	Mussels, clams, oysters
Static - pots/traps	Pots/creels (crustacea/ gastropods, whelks, peeler crab traps)
Static - fixed nets	Gill nets (mixed fish, crustacea)
	Trammels (mixed fish, crustacea)
	Entangling (mixed fish, crustacea)
Passive - nets	Drift nets (demersal) (mixed fish, crustacea)
Lines	Longlines (demersal) (mixed fish)
Seine nets and other	Beach seines/ring nets (mixed fish)
Commercial hand-gathering	Commercial diving; Commercial hand gathering in the intertidal: winkles, mussels, cockles, lugworm (bait), *Ensis*, peeler-crab (bait), ragworm (bait)
Aquaculture	
Finfish	Cages: Atlantic salmon, sea trout, sea bass, sea bream
Shellfish - off bottom	Trestles (oysters)
	Rope cultivation: mussels
Shellfish - on bottom	Ground lays: mussels, Pacific oysters, Native oysters, clams

Table 5. Summary of commercially targeted or cultivated species for which the risk of INNS establishment and impact was assessed by this project. Unless otherwise indicated the information is from the MMO landings report (MMO, 2018).

Demersal species	Pelagic species	Shellfish
Bass (*Dicentrarchus labrax*)[3]	Herring[1] (*Clupea harengus*)	Brown crab[3] (*Cancer pagurus*); Spider crabs (*Maja squinado*)[3]
Cod (*Gadus morhua*)	Salmon[2] (*Salmo salar*)	Lobsters (*Homarus gammarus*)[2,3]
Hake (*Merluccius merluccius*)		Nephrops (*Nephrops norvegicus*)
Megrim (*Lepidorhombus whiffiagonis*)		Scallops[2] (*Pecten maximus*, *Aequipecten opercularis*)
Monkfish (*Lophius piscatorius*)		Whelks (*Buccinum undatum*)[2,3]
Skates and Rays		Cockle (*Cerastoderma edule*)[2]
Witch (*Glyptocephalus cynoglossus*)		Mussel (*Mytilus edulis*)[2]
Haddock[1] (*Melanogrammus aeglefinus*)		Common prawn (*Palaemon serratus*)[2,3]
Plaice[1] (*Pleuronectes platessa*)		Native oyster (*Ostrea edulis*)
Sole[1] (*Solea solea*)		
Whiting[1] (*Merlangius merlangus*)		Pacific oyster (*Magallana gigas*)
Sea trout[2] (*Salmo trutta trutta*)		
[1]Caught in ICES area but not landed (MMO, 2018);[2] information from Seafish[7];[3] information from Pantin *et al*. (2015).		

2.5. Assigning target species to functional groups

To allow the assessments to be applicable if target species changed in the future and to simplify reporting, target species were assigned to functional groups (Table 6 and Table 7), based on feeding group, reproduction and larval types. The feeding groups for fish were based on diet following the Greenstreet *et al*. (1997) classification.

The pelagic planktivore guild includes not only pelagic fish, but also all fish for which plankton taxa constitute >80% of their diet by weight over their whole life. Demersal benthivores are fish that feed almost exclusively on benthic invertebrates, so the guild includes plaice, but not whiting (*Merlangius merlangus*) or cod, which consume benthos but also have a high proportion of fish in their diet. Piscivores are those species for which other fish constitute more than approximately one-fifth of the diet by weight. No piscivore species feed exclusively on fish, however, so the distinction between pelagic and demersal piscivores does not simply relate to the typical position in the water column. The key distinction is between species that are fundamentally piscivorous planktivores (pelagic piscivores, e.g. mackerels and tuna), and those that are piscivorous benthivores (demersal piscivores, e.g. cod and haddock, *Melanogrammus aeglefinus*).

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[7] https://www.seafish.org/article/introduction-to-the-wales-seafood-industry

Table 6. Summary of functional group information for adult feeding groups, reproduction and larval information for fish species. This information supported assessments of likely impacts of the assessed INNS.

Species	**Feeding group**	**Reproduction**	**Larval information**
Bass (*Dicentrarchus labrax*)	Demersal piscivore	Pelagic eggs[3]	Pelagic larvae[3]
Cod (*Gadus morhua*)	Demersal piscivore	Pelagic eggs[2]	Pelagic larvae[2]
Hake (*Merluccius merluccius*)	Demersal piscivore	Pelagic eggs[2]	Pelagic larvae[2]
Megrim (*Lepidorhombus whiffiagonis*)	Demersal piscivore	Pelagic eggs[3]	
Monkfish (*Lophius piscatorius*)	Demersal piscivore	Pelagic eggs (in gelatinous ribbon)[2]	Pelagic larvae[2]
Skates and Rays	Demersal benthivore	Benthic egg cases[2]	
Witch (*Glyptocephalus cynoglossus*)	Demersal benthivore	Pelagic eggs[3]	Pelagic larvae[3]
Haddock(*Melanogrammus aeglefinus*)	Demersal piscivore	Pelagic eggs[2]	
Plaice (*Pleuronectes platessa*)	Demersal benthivore	Pelagic eggs[2]	Pelagic larvae, demersal at 13mm[2]
Sole[1] (*Solea solea*)	Demersal benthivore	Pelagic eggs[2]	Pelagic larvae, demersal at 7mm[2]
Whiting[1] (*Merlangius merlangus*)	Demersal piscivore	Pelagic eggs[2]	Pelagic larvae[2]
Herring[1] (*Clupea harengus*)	Pelagic planktivores	Demersal eggs preference for gravel[2]	Pelagic larval and post-larval stages [2]
Salmon (*Salmo salar*)	Pelagic piscivore	Not relevant (spawns in rivers)	
Sea trout (*Salmo trutta trutta*)	Demersal piscivore	Not relevant (spawns in rivers)	
Sources [1]Greenstreet *et al*. (1997); [2]Information from Ellis *et al.* 2012; [3]FishBase			

Table 7. Summary of feeding groups, reproduction and larval information for commercially targeted invertebrate species. This information supported assessments of likely impacts of the assessed INNS.

Species	Feeding group	Reproduction	Larval type
Brown crab (*Cancer pagurus*)	[1]Omnivore-crustaceans, including the dog whelk *Nucella lapillus*, the winkle *Littorina littorea*, razor shells *Ensis* spp., the mussel *Mytilus edulis*, the cockle *Cerastoderma edule* and the oyster *Ostrea edulis*.	[1]Adult females carry eggs	[1]Pelagic larvae
Lobsters (*Homarus gammarus*)	[1]Omnivore- crabs, molluscs, urchins, starfish and polychaete worms, but may also include some fish, algae and zooplankton.	[1]Adult females carry eggs	[1]Pelagic larvae
Nephrops norvegicus	[1]Predator/scavenger-crustaceans but also molluscs and to a lesser extent polychaetes and echinoderms.	[1]Adult females carry eggs	[1]Pelagic larvae
Scallops (*Pecten maximus*, *Aequipecten opercularis*)	[1]Suspension feeder	[1]Spawner	[1]Pelagic larvae
Whelks (*Buccinum undatum*)	[1]Predator/scavenger	[1]Benthic juveniles	
Cockle (*Cerastoderma edule*) (intertidal)	[1]Suspension feeder	[1]Spawner	[1]Pelagic larvae
Mussel (*Mytilus edulis*)	[1]Suspension feeder	[1]Spawner	[1]Pelagic larvae
Spider crabs (*Maja squinado*)	[1]Omnivore, scavenger; algae	[1]Adult females carry eggs	[1]Pelagic larvae
Common prawn (*Palaemon serratus*)	[2]Omnivore, algae and small invertebrates	[1]Adult females carry eggs	[1]Pelagic larvae
Sources : [1] BIOTIC; [2] Haig *et al*. (2014)			

2.6. Risk assessment overview

The risk assessment method adopted by the project categorises an MPA feature or socio-economic activity as ‘at risk’ (vulnerable) to an INNS if it could be exposed to an INNS (habitat is suitable for INNS, see Section 4.3) and is sensitive to (impacted by) the pressures produced by these species (such as predation or habitat change). The level of risk is dependent on both the level of exposure (refer back to Section 4.3 above for assessment methodology) and the level of sensitivity (the magnitude of impact). Figure 1 below outlines the risk assessment process for the EICAT methodology (see Section 4.7), but the concept is also applicable to the SEICAT assessments. The pressures produced by INNS for the purposes of this project were defined as ‘impact pathways’ (see Section 4.6). If the MPA feature is not likely to be exposed to the INNS in the future because it is considered unsuitable and there was no evidence for occurrence elsewhere in the native or invaded range, then the impact was not assessed.

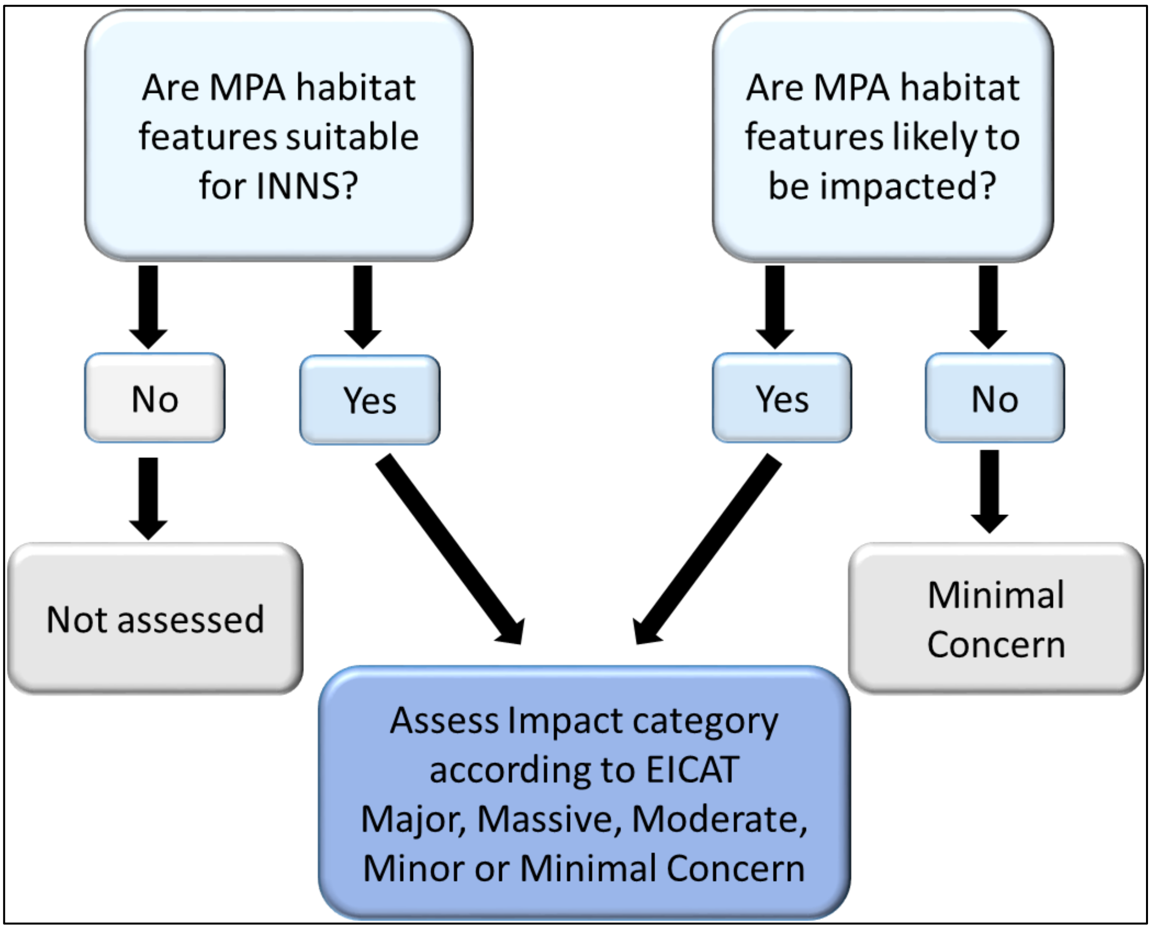

Figure 1. Risk assessment methodology overview. The level of exposure based on habitat suitability and the likely level of impact were considered when developing EICAT assessments.

2.7. Risk assessment methodology: EICAT assessments

The impact assessment criteria (impact pathways) and impact categories were adopted from the Environmental Impact Classification of Alien Taxa (EICAT) project (IUCN 2019), as these are well established, peer-reviewed and supported internationally by experts. The EICAT methodology identifies twelve impact mechanisms (impact pathways) by which alien taxa may cause deleterious impacts in areas to which they have been introduced (Table 8 below). These are based on previous work and aligned with those identified in the International Union for Conservation of Nature (IUCN) Global Invasive Species Database (GISD). For each pathway, there are five guidance criteria against which INNS are evaluated, to assign the level of impact caused under that mechanism. The project adopted these impact mechanisms to assess the level of current or potential impact these may have on MPA habitat features and fisheries and aquaculture target species. Fuller guidance to assess impacts is presented in Annex 2.

The EICAT risk assessments consider the likely impact on characteristic species, those that are key structural or functional species/assemblages and/or those that characterize the biotope groups that define the MPA features or associated sub-features. Changes to characterizing species may result in loss or reclassification of the feature. Where biotopes that might be lost were of particular conservation interest e.g. *Ostrea edulis* beds within the mixed sediment broad-scale habitat this was flagged. The scores from the risk assessments are presented in the summary Excel workbook that accompanies this report and the evidence and scores are provided in Annex 4-22 of this report.

Table 8. Impact mechanisms identified by the EICAT project and adopted by the current project to assess impacts on MPA habitat features and target species for fisheries and aquaculture.

Impact mechanisms	**Definition for impact on MPA habitat feature**
1. Competition	The alien taxon competes with native taxa for resources (e.g. food, water, space), leading to deleterious impact on native taxa.
2. Predation	The alien taxon predates on native taxa, leading to deleterious impact on native taxa.
3. Hybridisation	The alien taxon hybridises with native taxa, leading to deleterious impact on native taxa.
4. Transmission of disease	The alien taxon transmits diseases to native taxa, leading to deleterious impact on native taxa.
5. Parasitism	The alien taxon parasitises native taxa, leading to deleterious impact on native taxa.
6. Poisoning/ toxicity	The alien taxon is toxic, or allergenic by ingestion, inhalation or contact to wildlife, or allelopathic to plants, leading to deleterious impact on native taxa.
7. Bio-fouling	The accumulation of individuals of the alien taxon on the surface of a native taxon (i.e. bio-fouling), leads to deleterious impact on native taxa.
8. Grazing/ herbivory/ browsing	Grazing, herbivory or browsing by the alien taxon leads to deleterious impact on native taxa.
9. Chemical impact on ecosystem	The alien taxon causes changes to the chemical characteristics of the native environment (e.g. pH, nutrient and/or water cycling), leading to deleterious impact on native taxa.
10. Physical impact on ecosystem	The alien taxon causes changes to the physical characteristics of the native environment (e.g. disturbance or light regimes), leading to deleterious impact on native taxa.
11. Structural impact on ecosystem	The alien taxon causes changes to the structural biotope characteristics of the native environment (e.g. changes in architecture or complexity), leading to deleterious impact on native taxa.
12. Indirect impacts through interactions with other species	The alien taxon interacts with other native or alien taxa (through any mechanism, including pollination, seed dispersal, habitat modification, apparent competition, meso-predator release), facilitating indirect deleterious impact on native taxa.

2.8. EICAT Impact categories for MPA features: species and habitats

The EICAT impact categories adopted by this project are described in Table 9. In order to capture habitat feature impacts, the project team added qualifiers for habitat level impacts to the EICAT descriptors, these are shown in italics in Table 9 (below).

For each INNS x MPA feature combination an overall impact (risk assessment) score was developed, based on the combined worst-case impact assessment (for any impact mechanism).

MPA features which were not considered suitable, or for which there was no evidence to assess suitability, were 'Not assessed'. This category reflects either lack of exposure (MPA feature is not at risk) or underlying uncertainty in impact.

The EICAT assessment impact scores for each pathway and the overall combined impact score are provided in the EICAT Excel workbook that accompanies this report. The supporting evidence and summary scores are provided in the species factsheets in this report (Annexes 4-19).

Table 9. Impact categories and definitions adopted from the EICAT risk assessment methodology. Habitat impact qualifiers were added (in italics).

Impact category	**Definition for impact on MPA habitat feature**
Massive	Irreversible local, or global extinction of a native taxon (i.e. change in community structure) *and/or irreversible change to habitat character, e.g. loss of biogenic habitat or substratum type change, e.g. sediment to biogenic habitat structured by INNS*.
Major	Native taxon local extinction (i.e. change in community structure), *and/or change to habitat character, e.g. loss of biogenic habitat or substratum type change, e.g. sediment to biogenic habitat structured by INNS which is reversible.*
Moderate	Native taxon population decline *and/or alteration to key habitat features but habitat is still recognisable.*
Minor	Performance of individuals reduced, but no decrease in population size *and/or some alteration to habitat but not to degree that would impact key characterising species or habitat categorisation, structure or functioning.*
Minimal Concern	Negligible impacts, and no reduction in performance of native taxas' individuals, *negligible impacts on habitat.*
Data deficient	No evidence to assess.

2.9. Assessing socio-economic impacts on aquaculture and fisheries (SEICAT)

The socio-economic impact classification of alien taxa (SEICAT) approach to assessing the socio-economic impacts of non-native species on human welfare was proposed by Bacher *et al*. (2018). This approach assesses the impact on human capabilities. INNS can impact people's opportunities through changes in environmental factors, economic settings or social context. For the current project assessments of socio-economic impact focused on aquaculture and commercial fishing operations which largely map to the category of material assets (see Table 10). Some other impacts on human capabilities associated with these activities were assessed under Health and Safety. Recreational and commercial hand-gathering are partially overlapping activities. Recreational gathering was assessed under Social, spiritual and cultural relationships. Impacts were noted where evidence was found, for example the sharp shells of *Magallana gigas* may impact on access to fishing grounds and impact collection of bait and other target species. The impact categories are defined in Table 11.

Table 10. Relevant constituents of human well-being and examples of subcategories based on those reported in Bacher *et al*. (2018).

Constituents of human well-being	Examples
Safety (combined with health for the SEICAT assessment)	Personal safety e.g. safe handling of by-catch, impacts on safe access or safe operations.
Material and immaterial assets	Adequate livelihoods - Direct and indirect impacts on target species resulting in economic loss. Access to goods - Indirect effects on accessibility of target species or habitats of target species. Impacts on infrastructure and operations. Impacts on farmed species.
Health (combined with safety for the SEICAT assessment)	Impacts on health.
Social, spiritual and cultural relations	Recreational fishing and hand gathering.

Table 11. SEICAT Impact categories

Impact category	Definition for impact on human well-being
Massive	Local disappearance of an activity from all or part of the area invaded by the alien taxon. Change is likely to be permanent and irreversible for at least a decade after removal of the alien taxon, due to fundamental structural changes of socio-economic community or environmental conditions (“regime shift”).
Major	Local disappearance of an activity from all or part of the area invaded by the alien taxon. Collapse of the specific social activity, switch to other activities, or abandonment of activity without replacement, or emigration from region. Change is likely to be reversible within a decade after removal or control of the alien taxon. “Local disappearance” does not necessarily imply the disappearance of activities from the entire region assessed, but refers to the typical spatial scale over which social communities in the region are characterised (e.g. a human settlement).
Moderate	Negative effects on well-being leading to changes in activity size, fewer people participating in an activity, but the activity is still carried out. Reductions in activity size can be due to various reasons, e.g. moving the activity to regions without the alien taxon or to other parts of the area less invaded by the alien taxon; partial abandonment of an activity without replacement by other activities; or switch to other activities while staying in the same area invaded by the alien taxon. Also, spatial displacement, abandonment or switch of activities does not increase human well-being compared to levels before the alien taxon invaded the region (no increase in opportunities due to the alien taxon).
Minor	Negative effect on peoples’ well-being, such that the alien taxon makes it difficult for people to participate in their normal activities. Individual people in an activity suffer in at least one constituent of well-being (i.e. health; safety; assets; and social and cultural relations). Reductions of well- being can be detected through, e.g. income loss, health problems, higher effort or expenses to participate in activities, increased difficulty in accessing goods, disruption of social activities, induction of fear, but no change in activity size is reported, i.e. the number of people participating in that activity remains the same.
Minimal Concern	No deleterious impacts reported despite availability of relevant studies with regard to its impact on human well-being.

2.10. Confidence assessment

Confidence in MPA exposure and the EICAT and SEICAT impact categories was assessed using the categories shown below in Table 12. It should be noted that the confidence refers to the confidence in the impact assessment, i.e. the impact score, not the impact pathway itself. For example, *Rapana venosa* is definitely a predator, however when assessing predation we are not assessing the certainty that predation will arise, but rather the level of confidence in the impact.

Table 12. Confidence levels assigned to establishment and impact assessments.

Confidence Category	Definition of confidence level
High	There is a good understanding of the habitat suitability (for establishment assessments) or impact mechanism. The assessment is based on known impacts in similar habitats and/or similar characterising species, as evidenced by peer-reviewed or other high-quality evidence. There is consensus among sources and little variability in evidence of establishment or impact.
Medium	Whilst there is an understanding of the habitat requirements (for exposure assessments) or impact mechanism, there are some uncertainties over the level of establishment or impact and this may be based on limited evidence and/or proxy information such as species traits or impacts on habitats and characterising species that are relatively dissimilar. There is a majority agreement between experts; but conflicting evidence/opposing views exist or there is some variation in reports on the level of establishment or impact.
Low	There is limited or no understanding of the habitat suitability (for establishment assessments) or impact mechanism. Experts may disagree or there is little or no evidence to support the assessment. There may be a wide variation in the evidence for establishment and level of impact.

2.11. Overall risk and confidence scoring methodology for EICAT

The final impact and confidence scores for ‘Habitat’ and ‘Characterising species’ impact pathways were used in combination to come to an overall impact (risk) score. The overall risk from an INNS for each MPA feature is based on the highest (worst-case) impact score from any impact pathway. The confidence associated with that impact is the confidence in the risk assessment. For example: If ‘Moderate’ was the highest impact and it has an associated confidence score of ‘Low’ then the final confidence score was ‘Low’. Where there were two or more impacts of the same level then the highest confidence score was used. For example: ‘Moderate’ with low confidence and ‘Moderate’ with high confidence, then high was the confidence score that was used.

2.12. Evidence review methodology

The evidence review to develop the EICAT and SEICAT assessments was undertaken in three stages:

Stage 1: Collate available data from previous in-house projects undertaken by the team and GBNNSS Risk Assessments[8].

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[8] Available on-line at: http://www.nonnativespecies.org/index.cfm?pageid=143

Stage 2: Conduct a literature review to identify MPA features, fisheries and aquaculture services that could be impacted by each INNS. To determine which MPA features, fisheries and aquaculture may be impacted by the INNS, the project team conducted an initial high-level screening for overlap, based on evidence for INNS distribution, environmental tolerances and habitat preferences (subsequently updated throughout the literature review). This initial screening exercise identified evidence gaps for the more detailed targeted review and for follow up and expert consultation where necessary. A wide range of literature was sourced, including peer-reviewed evidence and reports using Google and Google Scholar. Access to material was supported by the holdings of the National Marine Biological Library which includes subscriptions to a wide range of journals.

For each INNS, the evidence on species ecology, suitability of MPA features as habitat and relevant impacts was collated in a draft factsheet. The final factsheets provide the evidence that was used to develop the EICAT and SEICAT scores and are presented as Annexes 4-19 in this report.

Stage 3: Expert consultation methodology. For each INNS, relevant experts were identified by the project team and contacted. Experts were asked to review the relevant species factsheet (presented as Annexes 4-19 in this report) and the accompanying EICAT and SEICAT Excel spreadsheets. Experts were invited to provide comments in any form, including comments that could be retained in the species factsheet in the form of personal communications. To support experts the EICAT methodology and the EUNIS correlation matrix that matches the EUNIS and JNCC habitat classification was provided. The project team offered to support experts by phone if required to discuss the methodology or any other aspects. The expert opinion was used to confirm the information within these documents and to add any additional information found to be missing, or alternatively to point out any inaccuracies. Finally, if unable to review the factsheet, experts were invited to suggest suitable individuals that might be able to do this. Unfortunately for some species despite efforts we were unable to find a suitable expert to review the factsheet.

3 Results

3.1. Result outputs

The outputs of the assessment process consist of this report with the evidence and summary scores provided in the species factsheets (Annexes 4-19) and two summary Excel workbooks that provide the EICAT and SEICAT assessment matrices that categorise the impact associated with each INNS. The assessments in the Excel matrices are accompanied by confidence scores.

3.2. Review of MPA feature exposure

The suitability of each MPA feature was assessed for each INNS to evaluate the likelihood that the feature would be exposed to the INNS. In some instances, MPA features, particularly those that are broadscale habitats, represent a wide range of biotopes. Where possible the suitability of constituent biotopes for INNS was assessed within MPA features to identify which were most likely to be exposed. This provides more information for managers and stakeholders to apply risk assessments for specific sites. Any information used to develop the assessment was recorded in the species factsheets (see Annexes 4-19).

Figure 2 (below) shows the number of MPA features/sub-features within each suitability category for each INNS. The number of constituent biotopes or other sub-features was not consistent between INNS, as the exposure assessments were based on specific species’ evidence. For example all of the A1.1 High energy littoral rock broadscale habitat was considered unlikely to be suitable for *Caprella mutica* (Annex 10), whereas the Pacific oyster *M. gigas* (Annex 11) was considered to be found in some constituent biotopes but not those that were characterised by macroalgae.

Figure 2 shows clearly that the number of MPA features and constituent biotopes suitable for each species varies, and that some species appear to be found in a wider range of habitats than others. For most species the number of habitats that were considered to be suitable was a relatively small proportion of the assessed features and a greater number were considered to be only potentially suitable or unlikely to be suitable. The distribution between these categories indicates the underlying uncertainty around occurrence in MPA features, with greater uncertainty attached to assessments of potential and unlikely. Overall, the number of MPA features that could not be assessed for each species is generally quite low.

Species for which a large number of MPA features were assessed as unsuitable are those that are restricted to intertidal or subtidal environments such as *R. venosa* (Annex 15) and *Homarus americanus (*subtidal), or were restricted by other factors such as sediment type (*Ensis leei* Annex 14), or a combination of factors such as light availability (depth) and substratum type for invasive macroalgae (*Bonnemaisonia hamifera*, A*garophyton vermiculophyllum*: Annex 9 and Annex 12, respectively).

Epifaunal species such as *Crepidula fornicata* (Annex 5), *Caprella mutica* (Annex 10), *Didemnum vexillum* (Annex 6), *Diadumene lineata* (Annex 13) and *Watersipora* *subatra* (Annex 8) are considered likely to be able to colonise a wide range of habitats. Confidence tended to be low for impacts resulting from the latter four species due to a lack of records of occurrence in natural habitats as these tend to currently be biofoulers of artificial structures in the invaded ranges.

No MPA seabed features were considered suitable for *Mnemiopsis leidyi* (Annex 18) as the ctenophore has an exclusively pelagic life habit. There is much uncertainty regarding the degree to which this species was responsible for observed changes in fish populations in the Black Sea and whether it was a causal factor. While impacts may ramify to benthic invertebrate species if larvae are predated on, there is no data to support an assessment. Larval supply dynamics can be highly site specific and influenced by a range of factors, such that any assessments would be subject to a high degree of uncertainty. This evidence gap is discussed in Annex 18 and the score for each MPA feature was 'data deficient'.

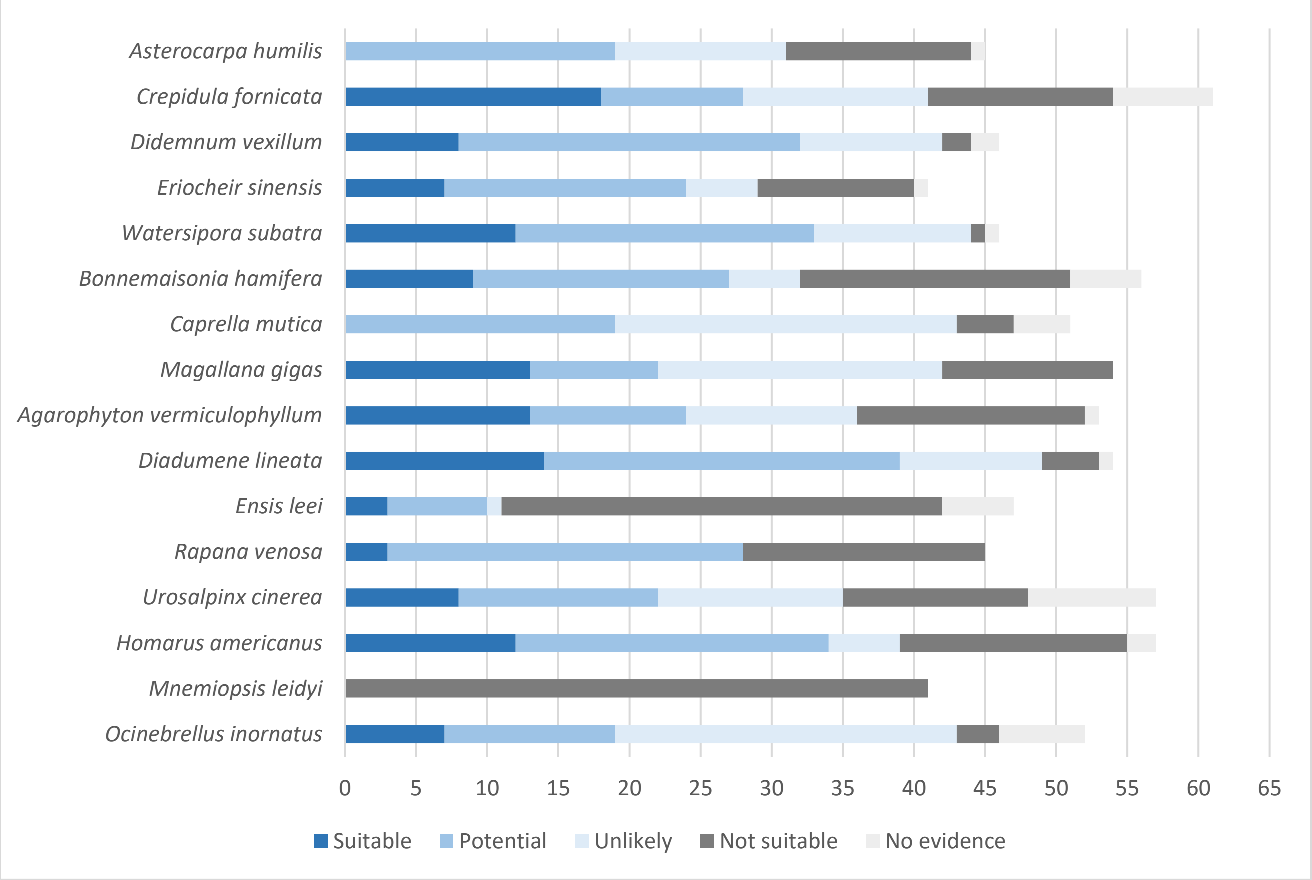

Figure 2. Number of MPA features (sub-features) identified for each exposure category for each INNS. Note the detailed assessments and confidence levels are presented in the EICAT summary table that accompanies this report. Features were assessed where possible to constituent biotopes and the number of assessments do not sum to 41 for each species.

3.3. Review of impacts on characterising species (EICAT)

The EICAT methodology adapted for this project assesses the impacts of INNS on species that characterise MPA features through nine impact pathways. The range of impact scores for each relevant impact pathway are shown in Tables 13 and 14 below. Impacts associated with INNS on characterising species within MPA features ranged from minimal to major. No species was associated with massive impacts on characterising species.

Competition, predation and biofouling were the main pathways by which INNS impact characterising species of MPA features. Hybridisation and transmission of disease were only associated with the American lobster, *H. gammarus*. (Annex 17). No impacts were associated with parasitism, poisoning or toxicity.

Competition for space and structural changes would impact some characterising species, particularly those that are present in high densities and form biogenic habitats. These impacts were assessed through biofouling and the habitat impact pathways (see Section 5.4) in order to avoid double counting impacts.

Table 13. Summary of impacts on MPA features through the species impact pathways: competition, predation, hybridisation, transmission of disease and parasitism.

Scientific name	Competition	Predation	Hybridisation	Transmission of disease	Parasitism
A. humilis	Minimal	Not applicable	Not applicable	Not applicable	Not applicable
C. fornicata	Minor-Minimal	Not applicable	Not applicable	Not applicable	Not applicable
D. vexillum	Minimal-Major	Not applicable	Not applicable	Not applicable	Not applicable
E. sinensis	Minimal-Moderate	Minimal-Major	Not applicable	Not applicable	Not applicable
W. subatra	Moderate	Data deficient	Not applicable	Not applicable	Not applicable
B. hamifera	Minimal-Moderate	Not applicable	Not applicable	Not applicable	Not applicable
C. mutica	Minor	Not applicable	Not applicable	Not applicable	Not applicable
M. gigas	Minimal-Moderate	Not applicable	Not applicable	Not applicable	Not applicable
A. vermiculophyllum	Minimal-Moderate	Not applicable	Not applicable	Not applicable	Not applicable
D. lineata	Minimal	Data deficient	Not applicable	Not applicable	Not applicable
E. leei	Minimal-Moderate	Not applicable	Not applicable	Not applicable	Not applicable
R. venosa	Minimal	Minor-Major	Not applicable	Not applicable	Not applicable
U. cinerea	Minimal	Minimal-Moderate	Not applicable	Not applicable	Not applicable
H. americanus	Minimal	Minimal	Minimal	Minimal-Moderate	Not applicable
M. leidyi	Not applicable	Not applicable	Not applicable	Not applicable	Not applicable
O. inornatus	Minimal	Minimal-Moderate	Not applicable	Not applicable	Not applicable

Table 14. Summary of impacts on MPA features through the species impact pathways: poisoning/toxicity, biofouling, grazing/herbivory/ browsing and indirect impacts.

Scientific name	Poisoning/ toxicity	Bio-fouling	Grazing/ herbivory/ browsing	Indirect impacts - interactions with other species
A. humilis	Not applicable	Minimal	Not applicable	Data deficient
C. fornicata	Not applicable	Minor- Massive	Not applicable	Data deficient
D. vexillum	Not applicable	Minimal-Major	Not applicable	Minimal-Moderate
E. sinensis	Not applicable	Not applicable	Minimal-Major	Data deficient-Moderate
W. subatra	Not applicable	Minor	Not applicable	Moderate
B. hamifera	Not applicable	Not applicable	Not applicable	Data deficient
C. mutica	Not applicable	Minimal	Not applicable	Major
M. gigas	Not applicable	Not applicable	Not applicable	Not applicable
A. vermiculophyllum	Not applicable	Minimal-Moderate	Not applicable	Minimal
D. lineata	Not applicable	Minimal	Not applicable	Minimal
E. leei	Not applicable	Not applicable	Not applicable	Data deficient
R. venosa	Not applicable	Not applicable	Not applicable	Data deficient-Moderate
U. cinerea	Not applicable	Not applicable	Not applicable	Not applicable
H. americanus	Not applicable	Not applicable	Not applicable	Data deficient
M. leidyi	Not applicable	Minimal	Not applicable	Major
O. inornatus	Not applicable	Not applicable	Not applicable	Not applicable

3.4. Review of habitat impacts on MPA features (EICAT)

The EICAT methodology assesses impacts of INNS on habitats/ecosystem through three impact pathways: chemical, physical and structural (Table 15). In order to avoid double counting impacts, some physical impacts that could also have been assessed as structural were clearly separated and assessed as physical. The summary scores for the EICAT assessments are presented in the Excel workbook that accompanies this report and the supporting evidence and summary scores are outlined in the annexes to report (Annex 4-19).

There was little evidence to assess chemical impacts for most species. All organisms will modify the chemistry of their immediate environment through nutrient cycling associated with ingestion, respiration and excretion. Indirectly the burrowing activities (bioturbation) of some infaunal species will also alter sediment oxygenation and chemistry. Because chemical impacts would be expected to be density dependent and additive, only organisms present in large abundances such as filter-feeding bivalves, *M. gigas* (Annex 11) and the gastropod *C. fornicata* (Annex 5) were considered capable of influencing local conditions.

Physical impacts such as sediment modifications, changes in hydrodynamics, and changes in light regime by smothering surfaces are more readily observable than chemical impacts. These impacts are associated with abundant or dense epifauna, typically dense beds of molluscs, mats of algae and colonial organisms such as the tunicate *D. vexillum* (Annex 6). Infaunal organisms such as *E. leei* (Annex 14) may alter sediment characteristics through burrowing activities. Filter-feeding molluscs deposit faeces and pseudofaeces and these can alter the physical properties of sediments and their chemistry (although physical impacts are more frequently reported).

Smaller and/or mobile species, such as *H*. *americanus* (Annex 17), predatory gastropods (*R*. *venosa*, *Urosalpinx cinerea* and *Ocinebrellus inornatus*, Annexes 15, 16 and 19) were not considered to lead to direct physical impacts. However the removal of reef-forming species through predation would alter the habitat structure of MPA features. Similarly the presence of INNS that form epifaunal reefs or mats was a key impact associated with a number of species (Table 15).

Table 15. Summary of impacts on MPA features through the ecosystem impact pathways, showing range of assessed impacts.

Scientific name	Chemical impact on ecosystem	Physical impact on ecosystem	Structural impact on ecosystem
A. humilis	Not applicable	Data deficient	Minimal
C. fornicata	Minor	Minimal-Massive	Major-Massive
D. vexillum	Not applicable	Minimal-Major	Minimal-Major
E. sinensis	-	Data deficient-Major	Data deficient-Major
W. subatra	-	Data deficient	Minimal
B. hamifera	Data deficient	Data deficient	Data deficient
C. mutica	-	-	-
M. gigas	Minimal-Moderate	Minimal-Moderate	Minor-Massive
A. vermiculophyllum	Data deficient	Minimal-Moderate	Data deficient-Minimal
D. lineata	Minimal	Data deficient	Minimal
E. leei	Minor	Minor-Major	Data deficient
R. venosa	Minimal	Minimal	Major
U. cinerea	-	-	Minimal-Moderate
H. americanus	-	-	Minimal
M. leidyi	-	-	-
O. inornatus	-	-	Minimal-Moderate

3.5. Summary of MPA impact scores

The assessment scores for each INNS are displayed below in Figure 3 and Table 16. From these it can be seen that the highest risk INNS that could lead to massive impacts on some MPA features were the reef-forming molluscs *M. gigas* (Annex 11) and *C. fornicata* (Annex 5). The impacts of these species on intertidal and subtidal habitats respectively are well documented. Other INNS that occur at high densities such as *E. leei* (Annex 14) may also have significant impacts on habitats, although these are less studied.

Predators may have major to moderate impacts on MPA features where the prey species characterise the MPA feature. Through impacts on biogenic reefs of bivalves, the predatory gastropods, Rapa whelk (*R. venosa* Annex 15), *U. cinerea* (Annex 16) and *O. inornatus* (Annex 19) were considered to have Major or Moderate impacts on a number of MPA features. Uncertainties around the long-term impacts on features were identified (see Section 6) for these species, as although impacts from initial introduction on the existing habitat may be minimal, over time these could become more severe through predation on juveniles. The resultant changes in recruitment could reduce long-term maintenance of the habitat. Where predated species are long-lived, such impacts may not be readily discernible in the short-term. Such habitats would become increasingly vulnerable to other pressures as the potential to recover through replacement of lost individuals is reduced.

Species with minimal impacts on MPA features were the American lobster (*H. gammarus*), *Diadumene lineata* (Annex 13), *C. mutica* and *Asterocarpa humilis* (Annex 4). To date *D. lineata, C. mutica* and *A. humilis* (Annex 4) are species mainly found fouling artificial structures such as sea walls and pilings rather than natural habitats, resulting in little evidence for impacts on MPA features.

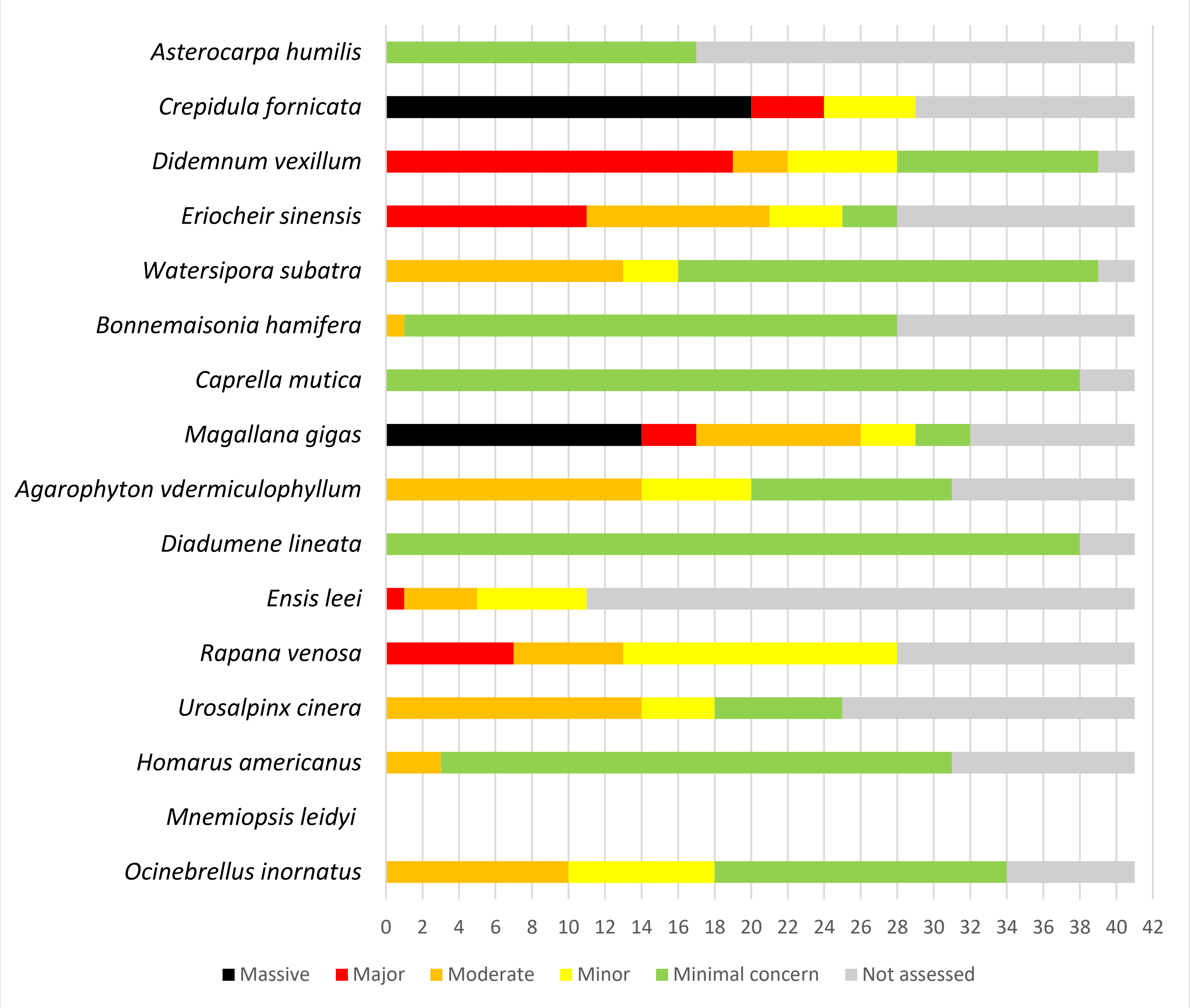

Figure 3. Chart showing the risk assessment (overall scores) for MPA features by each INNS.

Table 16. Summary of overall EICAT scores for each INNS showing the number of MPA features for each ranked assessment score: Massive, Major, Moderate, Minor and Minimal Concern or Not assessed.

Scientific name	Massive	Major	Moderate	Minor	Minimal concern	Not assessed
C. fornicata	19	7	0	0	0	15
M. gigas	14	3	9	3	3	9
D. vexillum	0	19	3	6	11	2
E. sinensis	0	9	11	5	3	12
R. venosa	0	7	6	15	0	13
E. leei	0	1	4	6	0	30
A. vermiculophyllum	0	0	14	6	11	10
W. subatra	0	0	13	3	23	2
U. cinerea	0	0	13	4	8	16
O. inornatus	0	0	10	8	16	7
H. americanus	0	0	3	0	28	10
B. hamifera	0	0	1	0	26	14
D. lineata	0	0	0	0	38	3
C. mutica	0	0	0	0	36	5
A. humilis	0	0	0	0	17	24
M. leidyi: Not included as all habitat impacts have been assessed as 'Data deficient'.						

3.6. MPA Vulnerability

An assessment of MPA vulnerability was undertaken to identify if some MPA features were more at risk than others. Figure 4 (below) shows that the impact pattern varies across MPA features with some more at risk than others. The assessment is relatively basic and does not consider the impact categories weighted by the number of INNS that could result in each impact category. Instead MPA features were ranked by the EICAT impact categories (see Table 17 below). More than half of the assessed MPA features were at massive risk from at least one INNS (Figure 4 and Table 17). Impacted MPA features occurred across a range of habitats including littoral and sublittoral, rock and sediments. Carbonate reefs were considered to be at low risk although this may reflect evidence gaps as there was little information on habitat parameters to support assessments.

The assessment shows that beds of the blue mussel were likely to be at risk of massive impacts from 2 INNS and major impacts from 3 other species. Other biogenic habitats (*Ostrea edulis* beds, *Sabellaria alveolata* reefs, *Modiolus modiolus* beds and *Musculus discors* beds) littoral and sublittoral mixed sediments including sheltered muddy gravels and littoral muds and sands, were also at risk of massive to moderate impacts based on the EICAT scoring. These risks are driven largely by the suitability of these habitats for the habitat altering species, *M. gigas* and *C. fornicata*.

Littoral coarse sediments were least at risk, reflecting the mobility of the substratum and height on the shore (based on JNCC/EUNIS biotope descriptions). This habitat is characterised by the presence of few species, reflecting the general harshness of this environment for marine and coastal species.

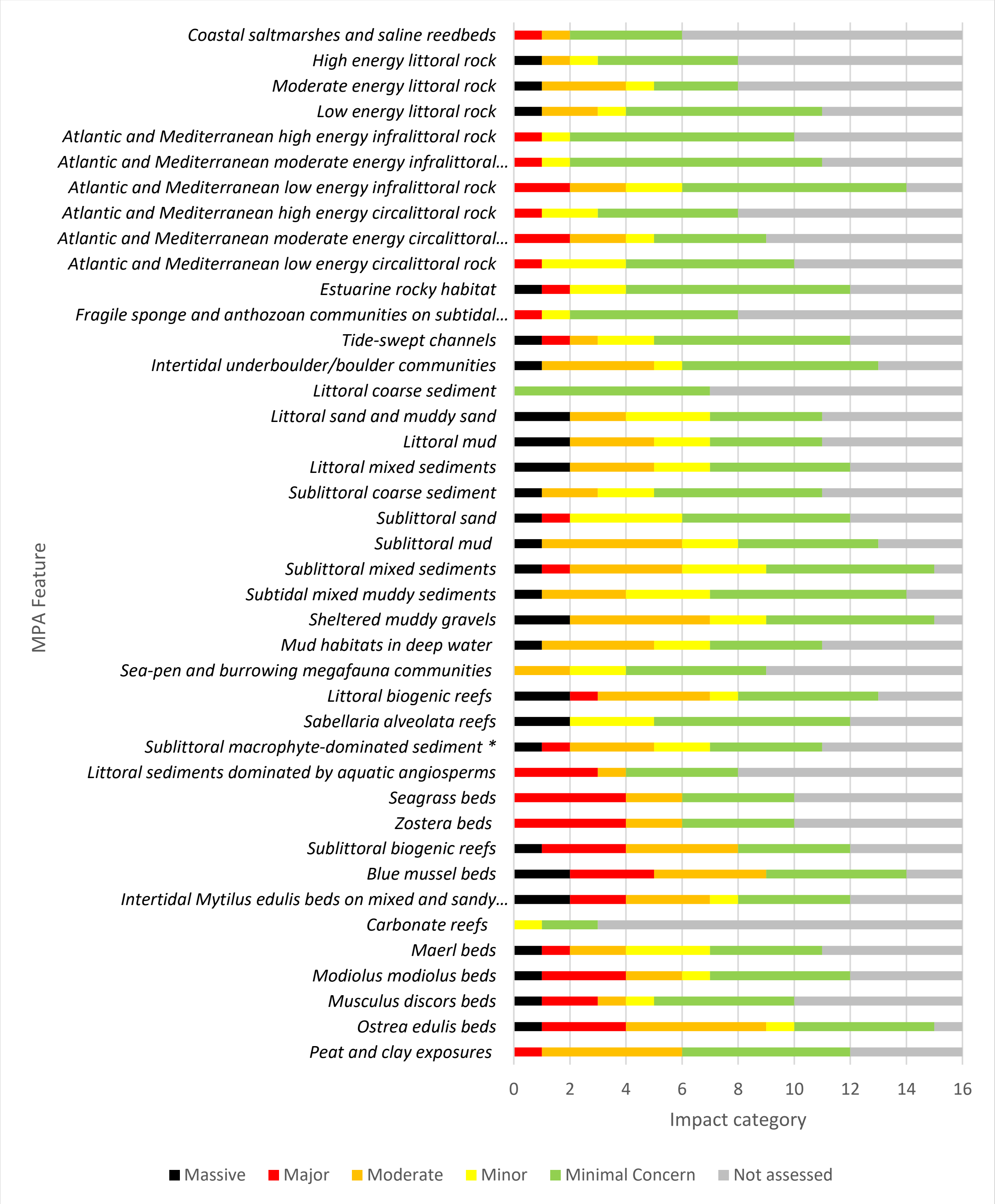

Figure 4. Chart summarising MPA feature risk assessments from INNS.

Table 17. Scores for each MPA ranked by severity of impact and showing the number of INNS species that were considered to exert each ranked assessment score: Massive, Major, Moderate, Minor and Minimal Concern or Not assessed.

Scientific name	Massive	Major	Moderate	Minor	Minimal concern	Not assessed
Blue mussel beds	2	3	4	0	5	2
Intertidal *Mytilus edulis* beds on mixed and sandy sediments	2	2	3	1	4	4
Littoral biogenic reefs	2	1	4	1	5	3
Sheltered muddy gravels	2	0	5	2	6	1
Littoral mixed sediments	2	0	3	2	5	4
Littoral mud	2	0	3	2	4	5
Littoral sand and muddy sand	2	0	2	3	4	5
Sabellaria alveolata reefs	2	0	0	3	7	4
Ostrea edulis beds	1	3	5	1	5	1
Sublittoral biogenic reefs	1	3	4	0	4	4
Modiolus modiolus beds	1	3	2	1	5	4
Musculus discors beds	1	2	1	1	5	6
Sublittoral mixed sediments	1	1	4	3	6	1
Sublittoral macrophyte-dominated sediment *	1	1	3	2	4	5
Maerl beds	1	1	2	3	4	5
Tide-swept channels	1	1	1	2	7	4
Sublittoral sand	1	1	0	4	6	4
Estuarine rocky habitat	1	1	0	2	8	4
Sublittoral mud	1	0	5	2	5	3
Mud habitats in deep water	1	0	4	2	4	5
Intertidal underboulder/boulder communities	1	0	4	1	7	3
Subtidal mixed muddy sediments	1	0	3	3	7	2
Moderate energy littoral rock	1	0	3	1	3	8
Sublittoral coarse sediment	1	0	2	2	6	5
Low energy littoral rock	1	0	2	1	7	5
High energy littoral rock	1	0	1	1	5	8
Zostera beds	0	4	2	0	4	6
Seagrass beds	0	4	2	0	4	6
Littoral sediments dominated by aquatic angiosperms	0	3	1	0	4	8
Atlantic and Mediterranean low energy infralittoral rock	0	2	2	2	8	2

Atlantic and Mediterranean moderate energy circalittoral rock	0	2	2	1	4	7
Peat and clay exposures	0	1	5	0	6	4
Coastal saltmarshes and saline reedbeds	0	1	1	0	4	10
Atlantic and Mediterranean low energy circalittoral rock	0	1	0	3	6	6
Atlantic and Mediterranean high energy circalittoral rock	0	1	0	2	5	8
Atlantic and Mediterranean moderate energy infralittoral rock	0	1	0	1	9	5
Atlantic and Mediterranean high energy infralittoral rock	0	1	0	1	8	6
Fragile sponge and anthozoan communities on subtidal rocky habitats	0	1	0	1	6	8
Sea-pen and burrowing megafauna communities	0	0	2	2	5	7
Carbonate reefs	0	0	0	1	2	13
Littoral coarse sediment	0	0	0	0	7	9

3.7. SEICAT Assessment Results: Health and Safety

Overall few species were considered to lead to health and safety impacts and INNS are not anticipated to cause massive or major impacts on activities through effects on health and safety. Two bivalves, *E*. *leei* (Annex 14) and *M*. *gigas* (11) have sharp edges and have been reported as causing injuries to gatherers and recreational users (assessed as Minor and Moderate risk, respectively). The crustaceans, *H*. *americanus* (17) and *Eriocheir sinensis* (Annex 7) both have pincers and could cause injuries if handled, risk from both of these was assessed as Minor. Biofouling INNS can increase the weight of gear increasing the risk of lifting and handling injuries. This has been reported as a concern for *R*. *venosa* egg capsules.

3.8. SEICAT assessments: Material Assets (Infrastructure and Operations)

The majority of the assessed INNS were not considered to cause direct risks to fisheries and aquaculture operations (see Table 18 and Figure 5). Fishing activities were considered at lower risk than aquaculture operations, with no commercial hand gathering, mobile or static gear operations considered to be at massive or major risk from INNS as most species were not considered to prevent gear being deployed.

INNS that might interfere with operations were those considered likely to clog or damage gears deployed on the bottom. *D*. *vexillum* (Annex 6) was considered likely to be captured by mobile gears, interfering with hauling and sorting and aggregations of washed out *E. leei* shells (Annex 14) might clog lighter towed gears. Drifting accumulations of the algae *A*. *vermiculophyllum* (Annex 12) have been recorded as fouling gear, including boat propellers and this species was therefore considered to potentially impact all operations.

The sharp shells of *M. gigas* (Annex 11) could interfere with beach seining operations and access to fishing grounds by damaging nets. This species is restricted to the shallow subtidal and reefs were not considered likely to affect other fishing operations using mobile gears that are typically deployed in deeper waters from boats.

The crab *E. sinensis* (Annex 7) could interfere with deployment of static gear through damaging or consuming catch.

Aquaculture operations were considered to be at higher risk from INNS with potential massive or major risks from biofoulers. These could alter the suitability of sediments for shellfish culture on sediments (on bottom) or could disrupt operations by growing on cultivated species and infrastructure. Examples include *D. vexillum* (Annex 6), *C. fornicata* (Annex 5) and *M. gigas* (Annex 11) which is itself a cultivated species that can have adverse effects on cultivated individuals. (Predation risks are assessed below for targeted and cultivated species and are another significant risk pathway.

Table 18 . Heat map showing assessed level of impact (risk) of each INNS for 11 types of fishing/aquaculture activity. Key to impact ranks: MC=Minimal Concern; Mr= Minor; Md=Moderate; Mj=Major; Ms=Massive; DD = Data Deficient.

Scientific name	1) Towed (demersal)	2) Dredges (towed)	3) Static - pots/traps	4) Static - fixed nets	5) Passive - nets	6) Lines	7) Seine nets and other	8) Commercial hand gathering	9) Aquaculture: Finfish	10) Shellfish- off bottom	11) Shellfish- on bottom
A. humilis	MC	MC	MC	MC	MC	MC	MC	MC	Md	Md	Md
C. fornicata	Mr	Mr	MC	MC	MC	MC	Mr	MC	MC	Md	Ms
D. vexillum	Md	Md	MC	MC	MC	MC	MC	Mr	Mj	Mj	Mj
E. sinensis	Mr	MC	Md	Md	Mr	Mr	Mr	Md	MC	MC	Mj
W. subatra	MC	MC	MC	MC	MC	MC	MC	MC	Md	Md	Md
B. hamifera	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC
C. mutica	MC	MC	MC	MC	MC	MC	MC	MC	Mj	Mj	Mj
M. gigas	MC	MC	MC	MC	MC	MC	Mr	Md	MC	Md	Ms
A. vermiculophyllum	Md	Md	Md	Md	Md	Md	Md	Md	Md	Md	Md
D. lineata	MC	MC	MC	MC	MC	MC	MC	MC	MC	DD	DD
E. leei	Mr	MC	MC	MC	MC	MC	MC	MC	MC	Mr	DD
R. venosa	MC	MC	Mr	Mr	MC	MC	MC	MC	MC	Md	Mj
U. cinerea	MC	MC	MC	MC	MC	MC	MC	MC	MC	Md	Md
H. americanus	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC
M. leidyi	MC	MC	MC	MC	MC	MC	MC	DD	MC	MC	MC
O. inornatus	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	Md

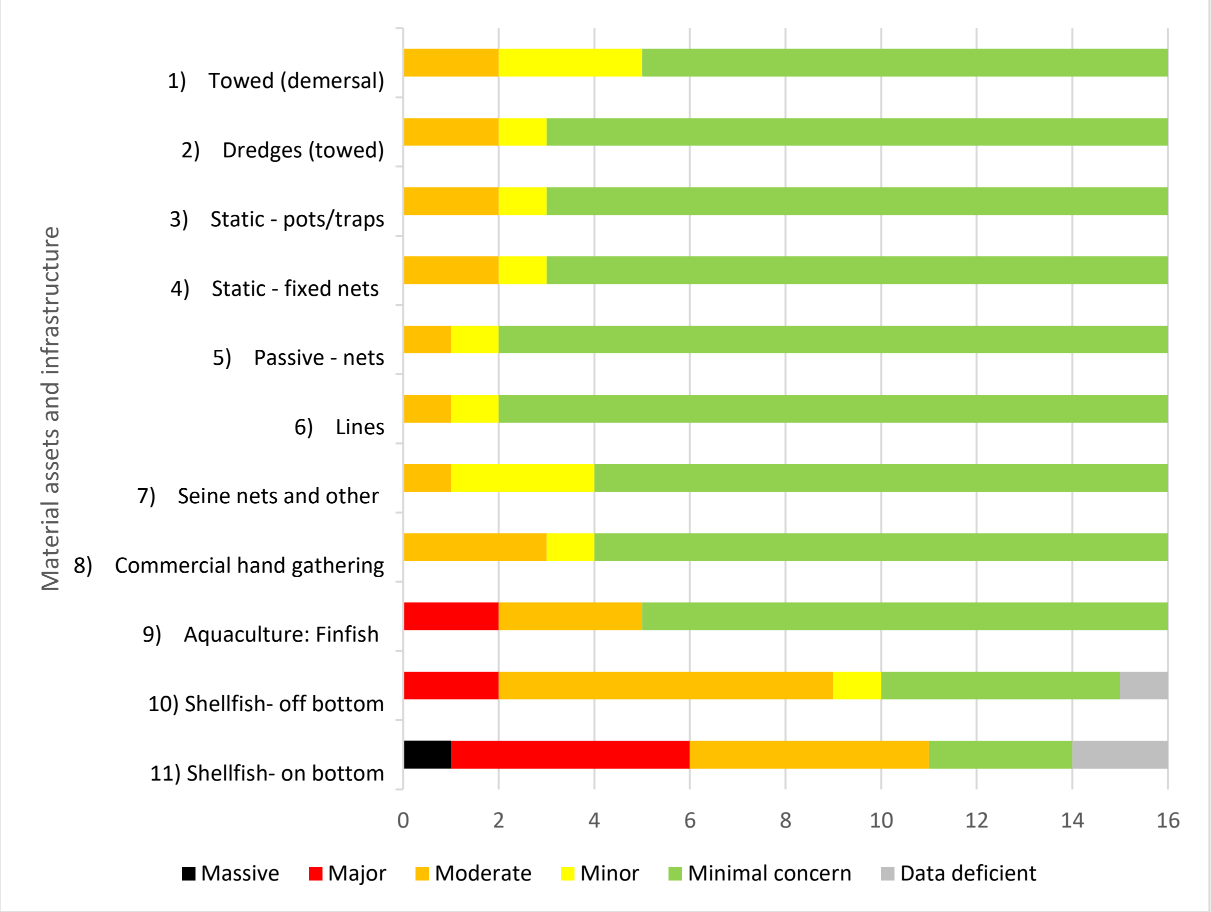

Figure 5. Chart showing risk assessment categories for each assessed operation type.

3.9. SEICAT Aquaculture and Fisheries targeted species

In general INNS were not considered to directly impact most fish stocks and most impact assessments were ‘Minimal Concern’ (see Table 19 below and Figure 6). Fish species that have pelagic eggs, pelagic larvae and/or largely feed on fish were considered unlikely to be directly impacted by INNS inhabiting benthic habitats.

Species of concern were the ctenophore *M. leidyi* (Annex 18), which may compete with or predate on fish eggs and larvae. The impact assessments for this species were highly uncertain (see Section 6.2 below). Epifaunal mat or reef-forming species such as *C. fornicata* (Annex 5) and *M. gigas* (Annex 11) can alter habitat for species that lay eggs or egg cases on the bottom and may interfere with feeding for fish that predate on the benthos. *E. sinenis* (Annex 7) was considered to impact sea trout and salmon through egg predation and habitat modification in the freshwater habitats these use for breeding.

Table 19. Summary of INNS impact assessments on fisheries target species. Key to impact ranks: MC=Minimal Concern; Mr= Minor; Md=Moderate; Mj=Major; Ms=Massive; DD = Data Deficient.

	Bass	Cod	Hake	Megrim	Monkfish	Skates and Rays	Witch	Haddock	Plaice	Sole	Whiting	Herring	Salmon	Sea trout
A. humilis	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC
C. fornicata	MC	MC	MC	MC	MC	Md	MC	MC	Md	Md	MC	MC	MC	MC
D. vexillum	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC
E. sinensis	Md	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	Mj	Mj
W. subatra	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC
B. hamifera	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC
C. mutica	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC
M. gigas	MC	MC	MC	MC	MC	Mn	MC	MC	Mn	Mn	MC	MC	MC	MC
A. vermiculophyllum	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC
D. lineata	DD	DD	DD	DD	DD	MC	DD	DD	DD	DD	DD	DD	MC	DD
E. leei	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC
R. venosa	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC
U. cinerea	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC
H. americanus	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC
M. leidyi	Maj	Maj	Mj	Mj	Mj	MC	Mj	Mj	Mj	Mj	Mj	Mn	MC	MC
O. inornatus	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC

Table 20 summarises INNS impact assessments for target invertebrate species. Commercially targeted crustaceans were generally considered to be un-impacted by INNS, although *H. americanus* was of concern due to the risk of disease transmission to other decapod crustaceans. Assessing risk is subject to inherent uncertainty regarding prevalence, spread and mortality rate from diseases and confidence is low in the assessment (see Annex 14).

Consumption of pelagic larvae by *M. leidyi* (Annex 18) was considered to be a risk for all species with a potentially significant major impact through predation or competition with larvae. However, there is little evidence to assess impact and confidence in the assessment was low.

The species assessed as most at risk were shellfish. These were assessed as being at risk from habitat modification and competition from epifaunal INNS and/or predation. The biofouling INNS *C. fornicata* (Annex 5) and *D. vexillum* (Annex 6) and *M. gigas* (Annex 11) were all considered likely to impact molluscs through habitat changes or biofouling.

The blue mussel, *Mytilus edulis* was considered to be at risk from a number of species including the aforementioned epifauna as well as predatory gastropods, *R. venosa*, *O. inornatus and U. cinerea* (Annex 15, 19 and 16 respectively)*.* These species were considered to have lower impacts on cockles based on their infaunal habitat and on scallops due to their ability to escape predators.

Overall, *M. edulis* and cultivated oysters were considered the most at risk species from INNS with risk from predation, biofouling and competition (*C. mutica*, Annex 10).

Table 20. Summary of INNS impacts on invertebrate species targeted by fisheries. Key to impact ranks: MC=Minimal Concern; Mr= Minor; Md=Moderate; Mj=Major; Ms=Massive; DD = Data Deficient.

Scientific name	**Brown crab**	**Lobsters**	**Nephrops norvegicus**	**Spider crabs**	**Common prawn**	**Scallops**	**Whelks**	**Cockle**	**Mussel**	**Oysters**
A. humilis	MC	MC	MC	MC	MC	MC	MC	MC	MC	Mi
C. fornicata	MC	MC	MC	MC	Md	Ms	Mr	Md	Mj	Mj
D. vexillum	MC	MC	MC	MC	MC	Mj	MC	MC	Mj	Mj
E. sinensis	Md	MC	MC	MC	MC	MC	MC	Md	Mr	Mj
W. subatra	MC	MC	MC	MC	MC	MC	MC	MC	Md	Md
B.hamifera	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC
C. mutica	MC	MC	MC	MC	MC	MC	MC	MC	Mj	Mj
M. gigas	MC	MC	MC	MC	MC	MC	MC	Md	Ms	Ms
A. vermiculophyllum	MC	MC	MC	MC	MC	MC	MC	MC	MC	MC
D. lineata	DD	DD	DD	DD	DD	DD	MC	DD	DD	DD
E. leei	MC	MC	MC	MC	MC	MC	MC	Md	Mr	Mr
R. venosa	MC	MC	MC	MC	MC	Md	MC	Md	Mj	Mj
U. cinerea	MC	MC	MC	MC	MC	MC	MC	MC	Md	Md
H. americanus	Md	Md	Md	Md	Md	MC	MC	MC	MC	MC
M. leidyi	Mj	Mj	Mr	Mj	Mj	Mj	MC	Mj	Mj	Mj
O. inornatus	MC	MC	MC	MC	MC	MC	MC	MC	Mj	Mj

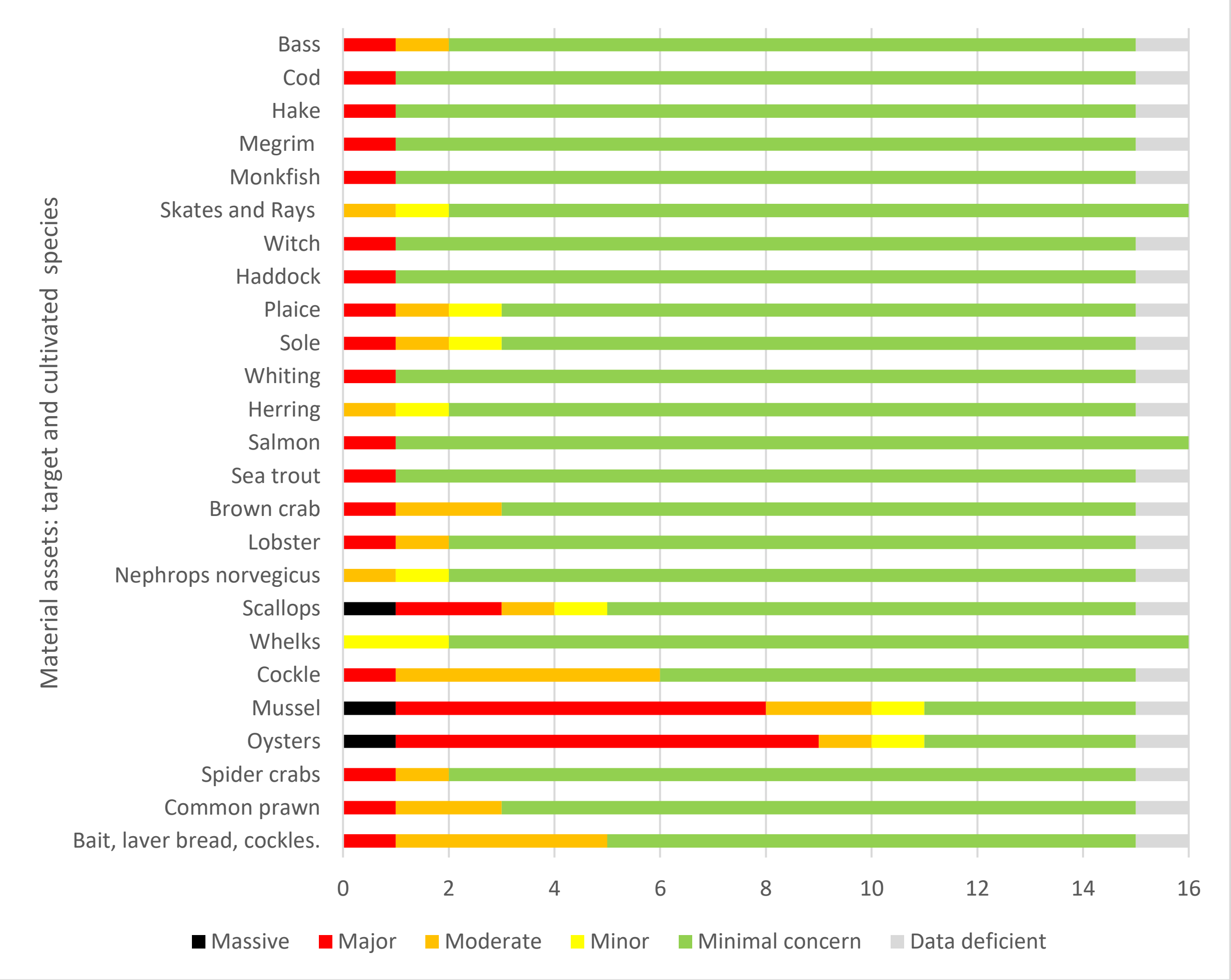

Figure 6. Chart showing risk assessment categories for each assessed targeted or cultivated species.

3.10. SEICAT assessment: Social, spiritual and cultural relations

Table 21 (below) provides the summary assessments for impacts on recreational gathering and collecting. Potential risks to this activity (Moderate impact) results from *M. gigas* (Annex 11) and *E. leei* as these could impact activities through accumulation of living reefs or accumulation of shell debris respectively where these co-occur with activities. The shells of both species can inflict cuts but impacts from *M. gigas* were assessed as a greater risk as the shells are more robust and have sharper edges. Although not assessed, both species could also result in impacts on general recreational users.

Cockle harvesting in the lower subtidal or subtidally could be affected by competition and habitat modification by *E. leei*, while *E. sinensis* (Annex 7) was considered likely to have a moderate impact on the collection of soft-shelled crabs for bait through crab-tiling as it could outcompete these species.

The invasive red algae (*B. hamifera* and *A. vermiculophyllum*: Annex 9 and 12) could alter the composition of intertidal algal communities and out-compete species collected to make laver bread or make these more difficult to find. There was little evidence to support these assessments and the impact was considered to be generally low (Minimal Concern).

Softer bodied biofouling species (*C. mutica*, *D. vexillum*, Annex 10 and Annex 6) and large predators (*R. venosa, H. americanus*) were generally not present in the intertidal. Smaller predatory gastropods (*O. ornatus*, Annex 19 and *U. cinerea*, Annex 16) that occur in intertidal habitats were not considered to predate significantly on cockles and thus reduce stocks.

Table 21. Summary table of impacts on recreational gathering, including bait collection, crab tiling, cockles and laver bread.

Scientific name	Bait collection, laver bread, cockles.
A. humilis	No evidence, interaction considered unlikely.
C. fornicata	Moderate impact. Intertidal bait and seaweed collection is unlikely to be affected, some impacts on cockle harvesting may occur through competition and replacement.
D. vexillum	No evidence, interaction considered unlikely.
E. sinensis	Moderate - Crab tiling likely to be impacted - also potential predator of bivalves and worms.
W. subatra	No evidence, interaction considered unlikely.
B. hamifera	Minimal Concern: It may be more difficult to find and collect the algae (laver) required for the making of laver bread if *B. hamifera* becomes too abundant.
C. mutica	No evidence, interaction considered unlikely.
M. gigas	Moderate: *M. gigas* has sharp edges and may impede hand collecting or gathering and access.
A. vermiculophyllum	Minimal Concern: It may be more difficult to find and collect the algae (laver) required for the making of laver bread if *A. vermiculophyllum* becomes too abundant.
D. lineata	No evidence, interaction considered unlikely.
E. leei	Moderate. Based on impacts on cockles, potential impacts from cuts and accumulation of shell debris.
R. venosa	Minimal Concern. No evidence, interaction considered unlikely.
U. cinerea	Minimal Concern. No evidence, interaction considered unlikely.
H. americanus	No evidence.
M. leidyi	No evidence, interaction considered unlikely.
O. inornatus	Minimal Concern. No evidence, interaction considered unlikely.

4 Discussion

All the assessed species were considered likely to occur in some of the assessed MPA features although the range of suitable habitats and likely exposure differed between species. Where there was information to support assessments of suitability, or evidence that high densities/abundances may occur this was recorded in the factsheets. The relatively greater number of exposure assessments in the potential and unlikely categories compared with suitable and not suitable reflects underlying uncertainty in assessing habitat suitability for INNS.

The chosen EICAT and SEICAT methodologies provide a systematic approach for the collation of existing evidence and balance uncertainty and gaps in that evidence, in order to provide a basis for decision making. Nevertheless, the evidence base itself is incomplete, both in relation to the assessed MPA features, and target and cultivated species, and understanding of the effects of INNS. Hence, a degree of expert judgement was often required by the project team to make assessments. Where there was particular uncertainty this was flagged to expert reviewers with a request for clarification if possible.

The main sources of uncertainty and gaps in knowledge were:

- Lack of detailed ecological evidence for INNS interactions and effects on species, and lifecycle and population dynamics;
- Lack of evidence and understanding on the response of species, communities and habitats to INNS;
- Lack of detailed evidence compared to the EICAT and SEICAT benchmarks and level of effects of INNS; and
- Lack of understanding of the biology, life history and population dynamics of species, the wider indirect links between species, and how those influence the indirect effects of INNS.

Within the project we did not assess in-combination impacts from INNS. INNS may lead to additive, synergistic or antagonistic impacts on other INNS and MPA features. For example, evidence indicates that *R. venosa* as an invasive species may compete with *U. cinerea* (Annex 16) in its native range (Mann and Harding, 2003). These types of assessment were outside the scope of this project. However, from the available project evidence in the EICAT assessment matrix, it would be possible to assess which MPA features might be at greater risk due to potential exposure to multiple INNS.

4.1. Evidence Gaps: Exposure, EICAT and SEICAT impacts

Evidence gaps were a key issue for assessing both the likely exposure of MPA features to INNS and to ascertain impacts on the features and socio-economic factors.

Where possible, information on native species ranges was used as a proxy to identify suitable habitats. However, uncertainties remain as to whether these are directly analogous with UK habitats. Aligning native habitat information and other evidence from invaded ranges with the UK and EUNIS biotope classifications is not straightforward. For constituent biotopes within a feature that were not suitable for an INNS the risk assessment was ‘Not assessed’.

In some instances assessments were based on species that were not yet present in the UK (e.g. *R. venosa*) or where the species’ native range was not well known and there were few proxy records to assess suitable habitats.

The quality of evidence available to assess exposure varies between species and between MPA features for a single species. For most species it was possible to identify at least some MPA features that were considered suitable based on clear evidence (sometimes numerous examples of directly relevant habitats from the UK). Conversely, it was also usually possible to identify MPA features that were unsuitable based on one or more factors. However, for most assessed species there were a number of features where exposure was challenging to assess. Our approach was to capture uncertainty through a five point exposure scale (Suitable- No evidence). This exposure scale had the flexibility to discriminate clearly between MPA features.

The confidence assessments made throughout the sensitivity assessment process were designed to demonstrate the source of the uncertainty in the evidence and the degree of expert judgement and interpretation required to make an assessment. For example, ‘High’ quality evidence may still not be directly applicable to the assessment, and excellent evidence may disagree.

An additional uncertainty, is that where the assessment is based on a few single species’ records it is not clear whether the species has established a sustainable population or whether individuals are present as temporary vagrants traversing unsuitable habitat (for mobile species) or displaced from suitable habitats.

Species’ ranges may also change over time, for example biofouling species that are currently restricted to artificial structures, typically in marinas and harbours or other artificial structures, such as *A. humilis* (Annex 4) and *Watersipora subatra* (Annex 8) may spread over time to natural habitats. In some cases, in invaded ranges, the number of colonised habitat types is increasing over time and current distribution restrictions may alter as species acclimate to prevailing conditions, or climate change increases habitat favourability.

Descriptions of INNS in natural habitats and their impacts typically include some information on habitat types (typically sediment/substratum information), this was rarely sufficient to confidently assess the habitat described to a biotope and in some instances even relating evidence to broad-scale habit types was challenging. For example, a study might describe a species as occupying ‘muddy’ sediments. From this statement it is not clear whether the described habitat is pure mud, sandy mud, muddy sand or mixed sediment. Typically, from this description it would be assumed that all these habitat types would be ‘potential’ habitats and yet there can be key differences between these habitats.

The project team found that evidence for wave exposure and current tolerances was a specific evidence gap that led to uncertainty in assessing likely exposure. The lack of information for these habitat characteristics is likely to reflect inherent variability in wave and current conditions and the lack of easily applied measurements without specialised instruments and/or lack of widely accepted and applied scales.

Intertidal habitats are more accessible for researchers than subtidal and are better studied. Thus sampling and monitoring bias, rather than patterns in habitat suitability, may underlie some assessments. For example, intertidal colonisation by *M. gigas* Annex 10 is well studied, but it is unclear whether the lack of information for subtidal reefs of *M. gigas* reflects the absence of reefs or of observations. For subtidal habitats there are considerable variations in the available evidence. There was little information available to describe the MPA feature carbonate reefs and species that might be associated with these. This particular evidence gap could be addressed in the future.

There is a lack of basic biological knowledge about many common and ecologically important benthic invertebrates (Tyler *et al*., 2011). Commercial, charismatic and experimental model species have been better studied (e.g. oysters, mussels, shrimp, crabs, corals), and yet little is known about otherwise common species (e.g. many polychaetes, cnidarians, sponges, echinoderms), where an understanding of their biology is often inferred from a relatively small number of the species in the group. Information on population dynamics and life history characteristics—vital for assessing impacts—are lacking.

Finally, it should be noted that the exposure assessments are generic assessments of suitability and cannot predict if a specific site will be colonised or not. Habitats that are apparently suitable may fail to support populations. Multiple environmental variables and other factors will influence whether INNS reach a site and become established. Studies on native invertebrate species have found that many apparently suitable habitats are not occupied, which is attributed to larval supply and settlement failures and juvenile mortality (Armonies and Reise, 2003).

The assessments of impacts on MPA features were subject to similar uncertainties. Assessments focussed on characteristic species of MPA features, rather than typical species. This focussed the assessments on species that were considered key structural, key functional or key to defining the biotope s(e.g. seapens define seapen biotopes). This approach is pragmatic as it is not possible to assess all species likely to be present and typical species haven’t been defined for all MPA features. The EICAT assessments are therefore largely based on characterising species and may not address impacts on typical species found within the habitats which could be substantial. For example, the loss of the European lobster from reef habitats and replacement by American lobster, changes the character of the lobster population present but this impact would be down weighted by the EICAT assessment methodology. Application of the assessments should consider site-specific conservation objectives and potential impacts on typical species.

To support SEICAT assessments evidence was collated on recorded impacts on aquaculture and fishing operations and target or cultivated species. In some instances there was clear evidence of impacts. As with the EICAT assessments more evidence was available for readily observable impacts from biofouling and predation. Information was readily available for biofouling of aquaculture gear from species such as *C. mutica* (Annex 10) and for bivalve predation by predatory gastropods such as *R. venosa* (Annex 15), *U. cinera* (Annex 16) and *O. inornatus* (Annex 19). The occurrence of these clear impact pathways are readily predictable although the scale and the resulting impact on aquaculture is less uncertain as these again will be site and operation specific.

Assessments for fish stocks and for fishing operations was more scant. For indirect impacts such as predation on larvae (*M. leidyi*, Annex 18) and impacts on fouling of spawning grounds by species such as *D. vexillum* (Annex 6) are subject to far greater uncertainty around habitat exposure and the likely level of impact. The importance of different types of benthic habitat to fish spawning, juvenile nursery grounds and recruitment to adult population are uncertain. Without this information assessing the likely impact for fish stocks was challenging and the majority of assessments have low confidence. For target species that are closely associated with benthic habitats such as scallops, evidence for impacts was largely based on changes in habitat or predation. This provided an evidence base against which to assess likely impacts. However, even where there are a number of studies the resultant impacts can be unclear. For *H. americanus* (Annex 17), the impacts on stocks are unknown, there could be disease, competition and hybridisation impacts although ultimately stocks may be able to co-exist. Previous reports have found that expert opinion is divided on whether *H. americanus* establishment will result in total losses of Homarid lobsters or replacement of one species by another without a change to the total stock (SwAM, 2016). This example highlights the inherent uncertainty in predicting impacts in complex systems with multiple factors operating over time and in different areas (e.g. interaction with habitat, larval supply, recruitment and individual stock behaviours and ecology).

4.2. Species specific Evidence gaps

***Asterocarpa humilis* (Annex 4):** The literature mentions that it could negatively impact sessile, suspension feeding organisms through competition for space and food resources. This could potentially have big implications on commercially valuable species like mussels. However, data gaps exist around these issues and studies are needed to understand the mechanisms and any impacts.

***Crepidula fornicata* (Annex 5):** Impacts on rock habitats were a clear evidence gap.

***Didemnum vexillum* (Annex 6):** Scientists suggest that this species could have far reaching implications on native communities. However, there is a lack of understanding regarding its impacts by altering benthic community structure and decreasing benthic foraging ability of larger, mobile predators. Further studies are needed to understand any implications this may have on fisheries species. This species is not widely established in the UK on natural habitats and there are key uncertainties in the extent and magnitude of colonisation.

***Eriocheir sinensis* (Annex 7):** A key evidence gap was impacts on seagrass beds through grazing which could be a key potential threat. The Chinese mitten crab is known to feed on fish eggs the impact which this predation may have on species laying eggs in river systems (particularly salmonids) is not fully understood, but may be significant.

Watersipora subatra **(Annex 8):** Little evidence was found to support assessments of physical impacts on habitats. Its ability to overgrow and occupy space increases sediment trapping and is likely to have implications with regards to water flow through the turf community. No studies relating to these issues were found.

Bonnemaisonia hamifera **(Annex 9):** Key uncertainties relate to habitat level impacts. It is speculated that mats induce sediment anoxia and other environmental modifications where it occurs in its '*Trailliella*' phase. However, no evidence was found to assess potential smothering impacts from mats. Furthermore, experiments have found that it may alter biological assemblages through changing grazer behaviour. There are possibilities that this may ramify to higher tropic levels. However, there is a lack of evidence from the field to further understand how these interactions may work.

Caprella mutica **(Annex 10):** There is much in the literature regarding the propensity for this species to foul aquaculture structures in large numbers but there are key uncertainties regarding establishment in natural habitats and likely resultant impacts. Some studies have found that it competes with mussels for food to the detriment of the mussels. However, feeding studies on this species are limited and are largely from the laboratory. There is a lack of field data to fully examine any impacts this species may have with regards to competition for food with shellfish in the aquaculture industries. There are also limited studies to suggest it has the ability to reduce recruitment. The mechanisms for this are not understood and more studies are needed.

Magallana gigas **(Annex 11):** Key uncertainties relate to the extent and magnitude of subtidal colonisation with most studies restricted to intertidal habitats and impacts.

Agarophyton vermiculophyllum **(Annex 12):** Much of the literature contains conflicting information with regards to its benefits of providing habitat, versus its negative impacts through means of altering trophic dynamics and nutrient cycling. There is evidence to suggest it may negatively impact seagrasses, although there are data gaps with regards to its long-term effects on these habitats. Since seagrass beds remain nationally scarce in the UK and their recovery from impacts can be slow it remains crucially important to have a better understanding on any impacts *A. vermiculophyllum* may have on UK populations.

Diadumene lineata **(Annex 13):** There is an evidence gap with regards to any effects this species may have on commercially valuable species. *D. lineata* is known to feed upon larvae of oysters. It can also form large clonal aggregations and is often found in areas close to valuable shellfish populations. Its impact on these fisheries has not been studied and is poorly understood.

Ensis leei**:** Impacts on native *Ensis* species and effects on infaunal communities are unclear and require further investigation. Assessing indirect interactions such as changes in trophic links and their effects are challenging to assess, yet it is apparent that this species may be of concern due to competition with native bivalves. Mass die-offs and the accumulation of shell debris occur but the long-term implications of this for subtidal and intertidal habitats and the human activities that take place, particularly fishing are unclear.

Rapana venosa (Annex 15)**:** This species is not yet established in the UK. Predation impacts on commercial species are relatively well known although the magnitude of these over longer timescales is uncertain. A key challenge for assessing impacts was the degree to which predation on bivalves found within MPA features would alter the status and classification of features, this represents more of a conceptual difficulty in assessing risk to natural habitat features.

Urosalpinx cinerea (Annex 16) and ***Ocinebrellus inornatus*** (Annex 19)**:** Temporal aspects of impacts were challenging to assess for these species and were highlighted as a concern. Over time the distribution or abundance of an INNS may change, requiring updating of risk assessments. Particular challenges were identified in the long-term aspect of predation by *U. cinerea* and *O. inornatus* on biogenic bivalve reefs. When introduced to an area many adult bivalves may have already reached size refugia from predation. However, if recruitment of juveniles is severely restricted then in the long-term reefs will reach senescence and not be replaced. In the species' native range, predators may have restricted populations of other smaller predators, and this predation release in introduced ranges may mean that impacts between native and colonised habitats are different.

Homarus americanus**:** There are key uncertainties around the impacts on the native lobster *Homarus gammarus,* the level of hybridization that is likely to occur between the two species, and what implications this may have on native species, ecosystems and lobster fisheries.

Mnemiopsis leidyi**:** There are key uncertainties around this species' impacts on fish stocks and benthic species with pelagic eggs and/or larval stages. The key drivers for impacts (predation versus competition are unresolved) and where changes in fish stocks have been observed this was coupled with overfishing and eutrophication pressures. Within the time constraints it was not possible to review the extensive literature, also key uncertainties seem not to have been resolved by scientists between invaded regions. Impacts on benthic habitats through competition and predation on larvae were not assessed for MPA features due to the high level of uncertainty. Based on information for fish stocks and some crustaceans, impacts on target and cultivated species were assessed as Minor. Confidence is low.

4.3. Risk Assessment Application

A number of general limitations regarding the impact assessments should be considered in their application:

- The impact assessments are generic and NOT site-specific. They are based on the likely effects of an INNS pressure on a 'hypothetical' MPA feature/ biotope based on general characteristics of these features.
- The impact assessments are general assessments that indicate the likely effects of a given INNS (likely to arise from one or more impact pathways) on MPA features or aquaculture and fisheries operations as represented in Wales.

- The assessments have attempted where possible to take account of spatial or temporal scales, e.g. seasonal variation in *R. venosa* egg laying, scale of biogenic reefs in relation to *M. gigas* reefs.
- The significance of impacts arising from pressures also needs to take account of the scale of the features.
- There are limitations in the scientific evidence on the biology and ecology of MPA features and their responses to INNS on which the impact assessments have been based.
- The EICAT impact assessment methodology takes account of the level of impact and the likely recovery to differentiate between the two most severe impact ranks (massive and major). Eradication of INNS is difficult and recovery will generally only occur where management measures are implemented. There is the potential, therefore for all major impacts to become massive.
- As a general rule, where the impact is ranked as higher, the need for management measures is greater.
- A rank of ‘Minor or Minimal Concern does not mean that no impact is possible from a particular ‘INNS x feature’ combination, only that a limited impact was judged to be likely based on the available evidence.

5 Conclusions

To address information gaps NRW commissioned the Marine Biological Association of the UK to conduct an evidence assessment for 16 INNS species, either present and presenting high to medium risk to marine ecosystems, or considered likely to arrive and cause impacts in the near future. The project assessed the risk to 41 Welsh Marine Protected Area habitat features and characteristic species that are of particular importance to conservation. The project also assessed the risk to fishery and aquaculture operations in Wales and the associated targeted or cultivated species.

To assess impacts from INNS two established methods of ranking impacts were adopted. MPA feature impacts were assessed using the Environmental Impact Classification of Alien Taxa (EICAT) methodology. This approach assesses impacts based on a number of impact pathways on species and habitats (e.g. competition, structural impacts on ecosystem). To assess impacts on aquaculture and fisheries the Socio-economic impact classification of alien taxa (SEICAT) method was adopted The core characteristic of this approach is that it uses changes in people's activities as a common measure for impact

For each assessed species a factsheet (Annexes 4-19) was created to record the evidence and basis for the assessments. Evidence on the effects of each INNS on MPA features, fisheries and aquaculture was reviewed. A range of experts were approached to review the factsheets and the Excel workbooks produced by this project. Overall there were few changes to EICAT and SEICAT scores suggested by the experts but the review was extremely valuable to provide additional quality assurance to the factsheets and to identify additional evidence or expert opinion.

Each assessment was accompanied by an assessment of the quality of the evidence, its applicability to the assessment and the degree of concordance (agreement) between the items of evidence to create an assessment of the overall confidence in the impact score.

Species capable of altering habitats and biofouling were considered most likely to lead to the greatest impacts on MPA features and aquaculture operations: the Pacific oyster (*M. gigas*) and the slipper limpet (*C. fornicata*) were assessed as likely to cause severe impacts to a range of intertidal and shallow subtidal MPA features and to damage aquaculture operations through impacts on substrates used for cultivation and by biofouling individuals. These species are both present in Wales and likely to expand further. Other species likely to have major effects on MPA features are the tunicate *D. vexillum* which is currently found mostly on artificial structures but has the potential to colonise natural habitats.

Predatory gastropods such as the Rapa whelk, *R. venosa* (Annex 15) could lead to significant impacts on bivalve species and the biogenic habitats these create and to impact these where they are cultivated species for aquaculture. *R. venosa* is currently not established in Wales but presents a potential threat. The oyster drills *U. cinerea* and *O. inornatus* could also pose a threat in the long-term to MPA features characterised by bivalves through effects on recruitment. Aquaculture operations could also be affected where oysters and mussels are predated upon by these species.

More than half of the assessed MPA features were at massive risk from at least one INNS. Reflecting the range of INNS distributions, impacted MPA features occurred across a range of habitats including littoral and sublittoral rock and sediments. A simple ranking assessment showed that beds of blue mussel were the feature most likely to be severely impacted by multiple INNS. They were considered to be at risk of massive impacts from two INNS (*C. fornicata* and *M. gigas*) and major impacts from three other species (*D. vexillum*, *E. sinensis* and *R*. *venosa*).

The higher levels of impact risks from INNS to MPA features were driven largely by the suitability of the habitat for *M. gigas* and *C. fornicata* and the likelihood and severity of habitat changes resulting from these species. MPA features at risk from these species include biogenic habitats (*Ostrea edulis* beds, *Sabellaria alveolata* reefs, *Modiolus modiolus* beds and *Musculus discors* beds) and littoral and shallow sublittoral sediments and rock features.

Little evidence was found for impacts on fish stocks and fishing operations. In general INNS are unlikely to prevent the operation of mobile gears although these are concerns that gears could be clogged by species that are present in large quantities such as aggregations of *M. leidyi* or large colonies of *D. vexillum*. The razorfish *E. leei* could also disrupt mobile gears where large numbers of empty shells are caught. These species could impose sorting costs. Biofoulers that grow on target species such as *C. fornicata* could impose cleaning costs for catch such as scallops.

Indirect effects through competition and changes in supply and habitat suitability and food supply are challenging to access. There is little information of key nursery grounds for many fish species and interaction between INNS and juveniles has rarely been assessed. Biofouling species and predators are most likely to affect habitat suitability and food supply but there are few studies to support this and assessments for impacts on fish stocks are low.

Aquaculture operations were considered to be at higher risk from INNS with potential massive or major risks from biofoulers. The species assessed as most at risk were shellfish. These were assessed as being at risk from habitat modification and competition from epifaunal INNS and/or predation. The biofouling INNS *C. fornicata* (Annex 5) and *D. vexillum* (Annex 6) and *M. gigas* (Annex 11) were all considered likely to impact molluscs through habitat changes or biofouling.

The blue mussel, *Mytilus edulis* was considered to be at risk from a number of species including the aforementioned epifauna as well as predatory gastropods, *R. venosa*, *O. inornatus and U. cinerea.* Overall, *M. edulis* and cultivated oysters were considered the most at risk species from INNS with risk from predation, biofouling and competition.

Contact with most INNS was not considered likely to lead to direct health and safety impacts, although lifting risks from added biofouling weight and cuts from shells or injuries from clawed crustaceans were identified as potential impacts. Although not specifically considered, consumption of INNS can also lead to health risks. Uptake of toxic dinoflagellates by *M. gigas* could result in severe health risks to humans but these risks are dependent on exposure to toxins and are not specific to *M. gigas* and were not assessed.

Throughout the report we have emphasised that the impact assessments are associated with a number of uncertainties and limitations. The results, while useful, should be interpreted with caution, particularly with regard to inherent uncertainties around sensitivity of habitats and species and the exposure to INNS which are influenced by numerous variables. The lack of evidence for impacts on fisheries in particular was identified as a key limitation of the study that strongly affects the results of the risk assessment. Limitations in the methodology and the application of sensitivity assessments were outlined. There was limited evidence to assess the effects of INNS on most MPA features and the assessed activities, particularly on fish stocks.

In summary, this project has increased our understanding of the pressures that may arise from INNS on MPA seabed habitats and species and has developed a number of impact assessments for aquaculture and fisheries operations to support management although these should be used cautiously due to the identified evidence gaps, uncertainties and limitations identified.

We recommend that future work should be undertaken to address evidence gaps and that assessments should be focussed at the local MPA site level and habitat level to further refine understanding of impacts for local sites and socio-economic activities.

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Annex 1 MPA habitat features

Table 1.1. Welsh MPA Broad-scale Habitat features (EUNIS Level 3) as identified within Ecological Network Guidance.

Broadscale habitats	EUNIS Code
High energy littoral rock	A1.1
Moderate energy littoral rock	A1.2
Low energy littoral rock	A1.3
Littoral coarse sediment	A2.1
Littoral sand and muddy sand	A2.2
Littoral mud	A2.3
Littoral mixed sediments	A2.4
Coastal saltmarshes and saline reedbeds	A2.5
Littoral sediments dominated by aquatic angiosperms	A2.6
Littoral biogenic reefs	A2.7
Atlantic and Mediterranean high energy infralittoral rock	A3.1
Atlantic and Mediterranean moderate energy infralittoral rock	A3.2
Atlantic and Mediterranean low energy infralittoral rock	A3.3
Atlantic and Mediterranean high energy circalittoral rock	A4.1
Atlantic and Mediterranean moderate energy circalittoral rock	A4.2
Atlantic and Mediterranean low energy circalittoral rock	A4.3
Sublittoral coarse sediment	A5.1
Sublittoral sand	A5.2
Sublittoral mud	A5.3
Sublittoral mixed sediments	A5.4
Sublittoral macrophyte-dominated sediment	A5.5
Sublittoral biogenic reefs	A5.6

Table 1.2. Welsh MPA Habitat features as identified within Ecological Network Guidance from the OSPAR Threatened and Declining List and Habitats of Principal Importance (Section 42 NERC, 2006).

Feature name	OSPAR or Habitat of Principal Importance
Blue mussel beds	NERC (all)
Carbonate reefs	NERC (Wales only)
Estuarine rocky habitat	NERC (all)
Fragile sponge and anthozoan communities on subtidal rocky habitats	NERC (all)
Intertidal *Mytilus edulis* beds on mixed and sandy sediments	OSPAR
Intertidal underboulder/boulder communities	NERC (all)
Maerl beds	OSPAR / NERC (all)
Modiolus modiolus beds	OSPAR / NERC (all)
Mud habitats in deep water	NERC (all)
Musculus discors beds	NERC (W)
Ostrea edulis beds	OSPAR
Peat and clay exposures	NERC (E, W & NI)
Sabellaria alveolata reefs	NERC (all)
Seagrass beds	NERC (all)
Sea-pen and burrowing megafauna communities	OSPAR
Sheltered muddy gravels	NERC (all)
Subtidal mixed muddy sediments	NERC (W)
Tide-swept channels	NERC (all)
Zostera beds	OSPAR

Annex 2 Environment impact categories based on EICAT

	Massive	**Major**	**Moderate**	**Minor**	**Minimal Concern**
Categories	Causes local extinction of characterising species (i.e., taxa vanish from communities at sites where they occurred before the alien arrived), OR characteristic habitat features would be reclassified, i.e. the habitat feature (or sub-features) is lost., which is irreversible; even if the alien taxon is no longer present the native taxon cannot recolonize the area.	Causes local or subpopulation extinction of at least one characterising species (i.e., taxa vanish from communities at sites where they occurred before the alien arrived); which is reversible if the alien taxon is no longer present OR characteristic habitat features would be reclassified, i.e. the habitat feature (or sub-features) is lost. A number of species may be lost from the wider biological assemblage.	Causes population declines in at least one characterising species, but no local population extinctions. Modification of characteristic features of habitats would be noticeable but would not result in reclassification of habitat features (or sub-features) but characterising species may be impacted and characterising species would decline. There may be wider impacts on the typical biological assemblage and some taxa may be lost.	Causes reductions in individual performance (e.g., growth, reproduction, defence, immune-competence) of characteristic species but no declines in local native population sizes. Although there may be some minor modification of characteristic features of habitats these do not result in reclassification or loss of suitability of characterising species such that these would be lost. Other typical species within the biological assemblage may decline or be lost.	Negligible level of impacts; no reduction in performance (e.g., growth, reproduction, defence, immune-competence) of individuals of native taxa and no changes to key characteristic features of habitat OR the impact mechanisms is not applicable.

(1) Competition	Competition resulting in replacement or local extinction of one or several characterising species resulting in reclassification and loss of the habitat feature (or loss of sub-features); changes are irreversible.	Competition resulting in local population extinction of at least one characterising species resulting in loss of the habitat (or loss of sub-features) but changes are reversible when the alien taxon is no longer present.	Competition resulting in a decline of population size of at least one characterising species, but no local population extinction. Other species within the biological assemblage may be more severely impacted or lost.	Competition affects performance of native characterising species without decline of their populations. Other typical species within the biological assemblage may decline or be lost.	Negligible level of competition with native taxa; reduction of native characterising species is not detectable or there is no competition. There may be some declines in other species within the typical biological assemblage.
(2) Predation	Predation results in local extinction of one or several characterising species resulting in reclassification and loss of the habitat feature (or loss of sub-features); changes are irreversible.	Predation results in local population extinction of at least one characterising species resulting in loss of the habitat feature (or loss of sub-features); but changes are reversible when the alien taxon is no longer present.	Predation results in a decline of population size of at least one characterising species, but no local population extinction. Other species within the biological assemblage may be more severely impacted or lost.	The alien taxon preys on native characterising species, without leading to a decline in their populations. Other typical species within the biological assemblage may decline or be lost.	Not applicable; predation on native taxa is classified at least as MN. Reduction of native characterising species is not detectable or there is no competition. There may be some declines in other species within the typical biological assemblage. If the species is not a predator then the assessment is ‘Not applicable’.

(3) Hybridisation	Hybridisation between the alien taxon and native characterising taxa leading to the loss of at least one pure native population (genomic extinction) resulting in reclassification and loss of the habitat feature (or loss of sub-features); changes are irreversible, pure native taxa cannot be recovered even if the alien and hybrids are no longer present.	Hybridisation between the alien taxon and native taxa leading to the loss of at least one pure native population (genomic extinction), resulting in loss of the habitat (or loss of sub-features); but changes are reversible when the alien taxon is no longer present.	Hybridisation between the alien taxon and native non-characterising taxa is regularly observed in the wild; there may be a local decline of populations of at least one characterising native taxon, but pure native taxa persist. Non characterising species within the biological assemblage may be more severely impacted or lost.	Hybridisation between the alien taxon and native taxa is observed in the wild, but rare; no decline of pure local native populations leading to habitat feature or sub-feature classification. Other species within the biological assemblage may be hybridised.	No hybridisation between the alien taxon and native characterising species observed in the wild (prezygotic barriers), hybridisation with a native taxon is possible in captivity. There may be some declines in other species within the typical biological assemblage. If there is no evidence for hybridisation then the assessment is 'Not applicable'.
(4) Transmission of disease or parasites	Transmission of disease or parasites to native taxa resulting in local extinction of one or several characterising species resulting in reclassification and loss of the habitat feature (or loss of sub-features). Changes are irreversible.	Transmission of disease or parasites to native taxa resulting in local extinction of at least one at least one characterising species resulting in loss of the habitat (or loss of sub-features); but changes are reversible when the alien taxon is no longer present.	Transmission of disease or parasites to native taxa resulting in a decline of at least one characterising species but no local extinction. Disease may be more severely affecting other species typical within the biological assemblage, including loss of some populations and it has been found in native and alien co-occurring	Transmission of disease or parasites to native taxa affects performance of one or more characterising species without leading to a decline of their populations; alien taxon is a host of a disease which has also been detected in native taxa. Other typical species within the biological assemblage may decline or be lost.	The alien taxon is a host or vector of a disease or parasites transmissible to native taxa but disease not detected in native taxa; reduction in performance of native individuals is not detectable.

			individuals (same time and space.		
(5) Parasitism This impact mechanisms is restricted to species that are parasites	Parasitism by the alien taxa will directly result in local extinction of one or several characterising species resulting in reclassification and loss of the habitat feature (or loss of sub-features). Changes are irreversible.	Parasitism by the alien taxa will directly result in local population extinction of at least one characterising species resulting in loss of the habitat feature (or loss of sub-features); but changes are reversible when the alien taxon is no longer present.	Parasitism by the alien taxa will directly result in a decline of population size of at least one characterising species, but no local population extinction. Other species within the biological assemblage may be more severely impacted or lost.	Parasitism by the alien taxa will directly affect performance of characterising species without decline of their populations. Other typical species within the biological assemblage may decline or be lost.	Negligible level of parasitism or disease incidence (pathogens) on native taxa, reduction in performance of native individuals is not detectable. If the species is not a parasite then the assessment should be ‘Not applicable’.

(6) Poisoning/ toxicity	The alien taxon is toxic/allergenic by ingestion, inhalation, or contact to wildlife or allelopathic to plants, resulting in local extinction of at least one or several characterising species resulting in reclassification and loss of the habitat feature (or loss of sub-features); changes are irreversible.	The alien taxon is toxic/allergenic by ingestion, inhalation, or contact to wildlife or allelopathic to plants, resulting in local population extinction of at least one characterising species resulting in loss of the habitat (or loss of sub-features), but changes are reversible when the alien taxon is removed.	The alien taxon is toxic/allergenic by ingestion, inhalation, or contact to wildlife or allelopathic to plants, resulting in a decline of population size of at least one characterising species, but no local population extinction. Other species within the biological assemblage may be more severely impacted or lost.	The alien taxon is toxic/allergenic by ingestion, inhalation, or contact to wildlife or allelopathic to plants, affecting performance of characterising species, without decline of their populations. Other typical species within the biological assemblage may decline or be lost.	The alien taxon is toxic/allergenic/ allelopathic, but the level is very low, reduction of performance of native individuals is not detectable. If there is no evidence for poisoning/toxicity then the assessment is 'Not applicable'.
(7) Bio-fouling or other direct physical disturbance	Bio-fouling or other direct physical disturbance resulting in local extinction of at least one or several characterising species resulting in reclassification and loss of the habitat feature (or loss of sub-features); changes are irreversible.	Bio-fouling or other direct physical disturbance resulting in local population extinction of at least one characterising species resulting in loss of the habitat (or loss of sub-features), but changes are reversible when the alien taxon is no longer present.	Bio-fouling or other direct physical disturbance resulting in a decline of population size of at least one characterising species, but no local population extinction. Other species within the biological assemblage may be more severely impacted or lost.	Bio-fouling or other direct physical disturbance affects performance of characterising species without decline of their populations. Other typical species within the biological assemblage may decline or be lost.	Negligible level of biofouling or direct physical disturbance on native taxa; reduction in performance of native individuals is not detectable. If there is no evidence for biofouling, i.e. mobile species, those that attach to substratum only etc, then the assessment is 'Not applicable'.

(8) Grazing/ herbivory/ browsing	Herbivory/grazing/ browsing resulting in local extinction of at least one or several characterising species resulting in reclassification and loss of the habitat feature (or loss of sub-features); changes are irreversible.	Herbivory/grazing/ browsing resulting in local population extinction of at least one characterising species resulting in reclassification and loss of the habitat (or loss of sub-features), but changes are reversible when the alien taxon is no longer present.	Herbivory/grazing/ browsing resulting in a decline of population size of at least one characterising species, but no local population extinction. Other species within the biological assemblage may be more severely impacted or lost.	Herbivory/grazing/ browsing affects performance of individuals of characterising species without decline of their populations. Other typical species within the biological assemblage may decline or be lost.	Negligible level of herbivory/grazing/ browsing on native taxa, reduction in performance of characterising species is not detectable. There may be some declines in other species within the typical biological assemblage. If the species is not a grazer then the assessment should be ‘Not applicable’.
(9) Chemical impact on ecosystems	Changes in chemical ecosystem characteristics (e.g., changes in nutrient cycling, pH) resulting in local extinction of at least one or several characterising species resulting in reclassification and loss of the habitat feature (or loss of sub-features); changes are irreversible.	Changes in chemical ecosystem characteristics (e.g., changes in nutrient cycling, pH) resulting in local population extinction of at least one characterising species resulting in loss of the habitat (or loss of sub-features), but changes are reversible when the alien taxon is no longer present.	Changes in chemical ecosystem characteristics (e.g., changes in nutrient cycling, pH) resulting in a decline of population size of at least one characterising species, but no local population extinction. Other species within the biological assemblage may be more severely impacted or lost.	Changes in chemical ecosystem characteristics (e.g., changes in nutrient cycling, pH) affecting performance of characterising species without decline of their populations. Other typical species within the biological assemblage may decline or be lost.	Small changes in chemical ecosystem characteristics detectable (e.g., changes in nutrient cycling, pH), but no reduction in characterising species. There may be some declines in other species within the typical biological assemblage.
10) Physical impact on ecosystems.	Changes in physical ecosystem characteristics (e.g.,	Changes in physical ecosystem characteristics (e.g.,	Changes in physical ecosystem characteristics (e.g.,	Changes in physical ecosystem characteristics (e.g.,	Small changes in physical ecosystem characteristics

Note: changes in biogenic habitat or substratum are assessed through structural impact)	changes in temperature, light regime or wave exposure) resulting in loss of habitat feature (or sub-features); changes are irreversible.	changes in temperature, light regime or wave exposure) resulting in loss of the habitat (or loss of sub-features), but changes are reversible when the alien taxon is no longer present.	changes in temperature, light regime or wave exposure) resulting in a decline of population size of at least one native taxon, but no local population extinction.	changes in temperature, light regime or wave exposure) affecting performance of native individuals without decline of their populations. Other typical species within the biological assemblage may decline or be lost.	detectable (e.g., changes in temperature, light regime or wave exposure), but no reduction in performance of characterising species detectable. There may be some declines in other species within the typical biological assemblage.
(11) Structural impact on ecosystems	Changes in structural ecosystem characteristics (e.g., changes in biogenic habitat, architecture or complexity) resulting in habitat reclassification and concomitant loss of characterising species and typical biological assemblage, changes are irreversible.	Changes in structural ecosystem characteristics (e.g., changes in biogenic habitat, architecture or complexity) resulting in habitat reclassification and concomitant loss of characterising species and typical biological assemblage, but changes are reversible when the alien taxon is no longer present	Changes in structural ecosystem characteristics (e.g., changes in biogenic habitat, architecture or complexity) which are not severe enough to result in habitat feature (or sub-feature reclassification) but which do result in a decline of population size of at least one characterising species or typical biological assemblage, but no local population extinction.	Changes in structural ecosystem characteristics (e.g., changes in biogenic habitat, architecture or complexity) with no change in habitat classification but some alteration to microhabitats that may affect performance of native individuals without decline of characterising species. The typical biological assemblage may be more severely affected by habitat changes and some species may decline or be lost.	Small changes in structural ecosystem characteristics detectable (e.g., changes in architecture or complexity), but no reduction in performance of native characterising species and the habitat (or sub-features) remain recognisable. There may be some declines in other species within the typical biological assemblage.
(12) Indirect impacts	Interaction of an alien taxon with	Interaction of an alien taxon with other taxa	Interaction of an alien taxon with other taxa	Interaction of an alien taxon with other taxa	Interaction of an alien taxon with

through interaction with other species	other taxa leading to indirect impacts (e.g., gamete dispersal, habitat modification, apparent competition, filter-feeding on propagules) causing local extinction of one or several native taxa, leading to irreversible changes that would not have occurred in the absence of the alien taxon.	leading to indirect (e.g., gamete dispersal, habitat modification, apparent competition, filter-feeding on propagules) causing local population extinction of at least one native taxon; changes are reversible but would not have occurred in the absence of the alien taxon.	leading to indirect impacts (e.g., gamete dispersal, habitat modification, apparent competition, filter-feeding on propagules) causing a decline of population size of at least one native taxon, but no local population extinction; impacts would not have occurred in the absence of the alien taxon.	leading to indirect impacts (e.g., gamete dispersal, habitat modification, apparent competition, filter-feeding on propagules) affecting performance of characterising species without decline of their populations; impacts would not have occurred in the absence of the alien taxon. Other species within the biological assemblage may decline or be lost.	characterising species leading to indirect (e.g., gamete dispersal, habitat modification, apparent competition, filter-feeding on propagules) but reduction in performance of native individuals is not detectable. There may be some declines in other species within the typical biological assemblage.

Annex 3 Socio-economic impact categories based on SEICAT

	Massive	**Major**	**Moderate**	**Minor**	**Minimal Concern**
Social and economic impact on activities through, health, safety, assets and social and relations	Local disappearance of an activity from all or part of the area invaded by the alien taxon. Change is likely to be permanent and irreversible for at least a decade after removal of the alien taxon, due to fundamental structural changes of socio-economic community or environmental conditions ("regime shift").	Local disappearance of an activity from all or part of the area invaded by the alien taxon. Collapse of the specific social activity, switch to other activities, or abandonment of activity without replacement, or emigration from region. Change is likely to be reversible within a decade after removal or control of the alien taxon. "Local disappearance" does not necessarily imply the disappearance of activities from the entire region assessed, but refers to the typical spatial scale over which social communities in the region are characterised (e.g. a human settlement).	Negative effects on well-being leading to changes in activity size, fewer people participating in an activity, but the activity is still carried out. Reductions in activity size can be due to various reasons, e.g. moving the activity to regions without the alien taxon or to other parts of the area less invaded by the alien taxon; partial abandonment of an activity without replacement by other activities; or switch to other activities while staying in the same area invaded by the alien taxon. Also, spatial displacement, abandonment or switch of activities does not increase human well-being compared to levels before the alien taxon invaded the region (no increase in opportunities due to the alien taxon).	Negative effect on peoples' well-being, such that the alien taxon makes it difficult for people to participate in their normal activities. Individual people in an activity suffer in at least one constituent of well-being (i.e. health, safety; assets; and social and cultural relations). Reductions of well- being can be detected through e.g. income loss, health problems, higher effort or expenses to participate in activities, increased difficulty in accessing goods, disruption of social activities, induction of fear, but no change in activity size is reported, i.e. the number of people participating in that activity remains the same.	No deleterious impacts reported with regard to its impact on human well-being.

Annex 4 Compass sea squirt: *Asterocarpa humilis*

Common name(s): Compass sea squirt.

Synonyms: *Asterocarpa cerea; Cnemidocarpa asymmetra; Cnemidocarpa auklandica; Cnemidocarpa cerea; Cnemidocarpa gregaria; Cnemidocarpa robinsoni; Dendrodoa gregaria; Pandocia gregaria; Styela asymmetra; Styela cerea; Styela humilis; Tethyum asymmetron* (Shenkar *et al*., 2019).

Domain: Phylum: Chordata, Class: Ascidiacea, Order: Stolidobranchia, Family: Styelidae, Genus/species: *Asterocarpa humilis* (Shenkar *et al*., 2019).

Description: A solitary sea squirt up to 4 cm in length. Orange-red in colour (Wood *et al.,* 2017). When the siphons are open four cream-white prominent lines are visible, interspersed with thinner lines resembling a compass face, hence its common name. The four prominent lines are still visible in partly closed siphons. This solitary (or unitary) ascidian can form clumps by growing attached to one another (Bishop, 2017; Wood *et al.,* 2017). The tunic may be encrusted with sponges, hydroids and algae (Page *et al.,* 2016).

(© John Bishop, MBA).

Figure 4.1. Compass sea squirt: *Asterocarpa humilis*

Asterocarpa humilis: Habitat

Native range: In its presumed native range of New Zealand (Bishop *et al.,* 2013) *Asterocarpa humilis* occurs subtidally under boulders, on wharf piles and fouling bivalves (Page *et al.,* 2016) and was listed by Morton & Miller (1973) as one of the 'big five' species of sessile animal on piles and wharves and as feature of the natural sublittoral fringe in the Dunedin area of South Island.

Substratum type: In its introduced range, *A. humilis* is mainly found in marinas and harbours, although it has also been found on aquaculture installations (Clarke & Castilla, 2000). The species has the apparent potential to colonize natural low-intertidal and subtidal habitats in its introduced range (Bishop, 2017; Wood *et al.,* 2017) and occurs occasionally on the shore in the Plymouth, UK, area (J. Sewell, C Wood & J Bishop, unpubl. observations).

Salinity: It is found in fully marine to low estuarine areas (Bishop *et al.,* 2013; Shenkar *et al.*, 2019; Wood *et al.,* 2017).

Depth: Low-intertidal (Bishop, 2017) – 26 m (Millar, 1982). In its native ranges of New Zealand it is found < 18 m (Page *et al.,* 2016).

Wave exposure: Seemingly favours sheltered sites, e.g. harbours and marinas, in its introduced ranges (Bishop, 2017).

Asterocarpa humilis: Establishment in Wales as of 2020

A. humilis has been documented in Holyhead Marina, north Wales, in 2011 and 2014 (Bishop *et al*., 2013), and was also found at two marinas in Milford Haven (SW Wales), but no other Welsh Marinas, surveyed in 2014 (Wood *et al*, 2015).

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Asterocarpa humilis: Impact pathways based on species traits, biology and ecology

Competition: *A. humilis* attaches to bivalves (Page *et al*., 2016) and associated substrates and is a possible competitor for food and space resources (Bishop, 2017). A survey in France found it on artificial structures in an oyster farm and nearby on natural habitat occurring with oysters (*Ostrea edulis),* scallops (*Pecten maximus*) and slipper limpets (*Crepidula fornicata*) (Bishop *et al.*, 2013). *A. humilis* could negatively affect other shallow-water suspension feeding sessile organisms. It may compete for resources and could impact on native species abundance (Bishop, 2017). Little is known at present about any impacts, like local species extinctions, that it may cause (Bishop, 2017). This pathway has been assessed as ‘Minimal concern’ with medium confidence.

Predation: Tunicates are suspension feeders using a mucus net to capture particulate matter (Petersen, 2007). This pathway has been assessed as ‘Not applicable’.

Hybridisation: *A. humilis* is a hermaphrodite that may be capable of self-fertilizations; it retains its eggs and broods its young in the atrial cavity (Bishop *et al*., 2013). There is no evidence of hybridization with other species, and no species that appears particularly closely related to *A. humilis* occurs regularly in UK waters. This pathway has been assessed as ‘Not applicable’.

Transmission of disease: *A. humilis* is not known to transmit diseases. This pathway has been assessed as ‘Not applicable’.

Parasitism: *A. humilis* is not known to be parasitic. This pathway has been assessed as ‘Not applicable’.

Poisoning/toxicity: *A. humilis* is not known to be toxic. This pathway has been assessed as ‘Not applicable’.

Bio-fouling: *A. humilis* is a bio-fouler. It can form clumps following aggregated settlement of larvae. This gives it the potential to significantly foul oyster and mussel gear, possibly compete for food and potentially smother farmed bivalves (Bishop, 2017). This pathway has been assessed as ‘Minimal concern’ with medium confidence.

Grazing/herbivory/browsing: Ascidians are suspension feeders (see previous section). This pathway has been assessed as ‘Not applicable’.

Indirect impacts through interactions with other species: No indirect interactions with other species have been reported. This pathway has been assessed as ‘Data deficient’.

Asterocarpa humilis: Impact pathways – Habitats

Chemical impact on ecosystem: No chemical impacts have been reported. This pathway has been assessed as ‘Not applicable’.

Physical impact on ecosystem: No physical impacts have been reported. This pathway has been assessed as ‘Data deficient’.

Structural impact on ecosystem: ‘Minimal concern’ with high confidence. *A. humilis* can form clumps, single ascidians attaching to each other, following aggregated larval settlement (Bishop, 2017). This behaviour could modify habitat structure altering habitat occupancy and species abundance. However, no impacts have been reported or are anticipated (Bishop, 2017).

Asterocarpa humilis: Interactions with MPA Features

MPA features that provide suitable habitat

Mainly found on artificial structures to date (Bishop, 2017), although it has started to be recorded in natural habitats (Wood *et al.,* 2017). Until more data is available from further surveys none of the MPA habitats have been assessed as suitable habitat.

MPA features considered potentially suitable for Asterocarpa humilis

The MPA features below provide potentially suitable habitat for *A. humilis* based on suitable attachment substrate.

- A1.3 Low energy littoral rock (medium confidence);
- A3.3 Atlantic and Mediterranean low energy infralittoral rock (medium confidence);
- A4.3 Atlantic and Mediterranean low energy circalittoral rock (low confidence);
- Intertidal underboulder/boulder communities (medium confidence);
- Estuarine rocky habitat (low confidence);
- A5.4 Sublittoral mixed sediments (low confidence);
- Subtidal mixed muddy sediments (low confidence);
- Sheltered muddy gravels:
- A2.42, A5.43, A5.44 (low confidence);
- A2.7 Littoral biogenic reefs (low confidence);
- *Sabellaria alveolata* reefs (low confidence);
- A5.6 Sublittoral biogenic reefs (low confidence);
- Blue mussel beds (low confidence);
- Intertidal *Mytilus edulis* beds on mixed and sandy sediments (low confidence);
- Maerl beds (low confidence);
- *Modiolus modiolus* beds (low confidence);
- Fragile sponge and anthozoan communities on subtidal rocky habitats (except A4.133 Scottish Sea lochs & A4.211 mostly found in Scotland & Ireland) (medium confidence);
- A5.5 Sublittoral macrophyte-dominated sediment:
    - A5.52 where bivalves occur (medium confidence);
- *Ostrea edulis* beds (low confidence); and
- Peat and clay exposures (low confidence).

MPA features considered unlikely to be suitable for Asterocarpa humilis

The MPA features below are unlikely to provide suitable habitat for *A. humilis* based on their exposed (energy) nature. They have all been scored with low confidence.

- A1.1 High energy littoral rock;
- A1.2 Moderate energy littoral rock;
- A3.1 Atlantic and Mediterranean high energy infralittoral rock;
- A3.2 Atlantic and Mediterranean moderate energy infralittoral rock;
- A4.1 Atlantic and Mediterranean high energy circalittoral rock;
- A4.2 Atlantic and Mediterranean moderate energy circalittoral rock;
- Tide-swept channels; and
- *Musculus discors* beds.

The MPA features below are unlikely to provide suitable habitat based on a lack of data to suggest otherwise. They have all been scored with medium confidence.

- A5.5 Sublittoral macrophyte-dominated sediment:
    - A5.51, A5.53, A5.54 (medium confidence);
- A2.6 Littoral sediments dominated by aquatic angiosperms;
- Seagrass beds; and
- *Zostera* beds.

MPA features unsuitable for Asterocarpa humilis

- A2.5 Coastal saltmarshes and saline reedbeds are unsuitable due to their position high up the shore (high confidence).

The MPA features below are unsuitable based on a lack of attachment substrate and/or the mobile nature of the substrate:

- A2.1 Littoral coarse sediment (medium confidence);
- A2.2 Littoral sand and muddy sand (low confidence);
- A2.3 Littoral mud (low confidence);
- A2.4 Littoral mixed sediments (low confidence);
- A5.1 Sublittoral coarse sediment (medium confidence);
- A5.2 Sublittoral sand (low confidence);
- A5.3 Sublittoral mud (low confidence);
- Sheltered muddy gravels:
    - A2.41 (low confidence);
- Mud habitats in deep water (low confidence); and
- Sea-pen and burrowing megafauna communities (low confidence).

No evidence for *Asterocarpa humilis* in these MPA habitats

- A5.7 Carbonate reefs.

Summary of key impacts on MPA features

Asterocarpa humilis is only found on artificial structures in its UK introduced range (Bishop, 2017) apart from occasional records on the shore in the Plymouth area (J. Sewell, C. Wood & J. Bishop, unpublished observations). It may have negative impacts on other shallow water, sessile filter-feeders in regards to habitat occupancy and abundance, although it is not understood if this could lead to any native species extinctions (Bishop, 2017). Since it is mainly found on artificial structures to date this it is unlikely to cause any significant impacts on any of the MPA features.

Asterocarpa humilis Socio-Economic Impacts

Human health and safety: No impacts are known.

Aquaculture operations: *A. humilis* has possibly spread to the UK through the importation of commercial bivalves such as oysters. It is also thought to be capable of translocation via attachment on boat hulls (Bishop *et al.,* 2013; Pinochet *et al.,* 2017). There is the risk that vessels working between aquaculture sites could enable the spread of this species between sites. It is possible that clumps of *A. humilis*, which form when they attach together, could clog pipes and marine infrastructure including aquaculture gear (Bishop, 2017). The potential abundance and clump formation means it has the ability to become a significant fouler within this industry. In Chile a dozen have been recorded along aquaculture bivalve longlines (Pinochet, 2017). Aquaculture gear could become clogged and cumbersome (Bishop, 2017) if significantly fouled. There is little evidence to suggest that this will be the case so these possible impacts have been assessed as 'Minimal concern' but with low confidence.

Cultivated species- Mussels and Oysters: *A. humilis* is a possible competitor for food and space with sessile filter-feeders (Bishop, 2017) possibly impacting shellfisheries. There is little evidence to suggest that this will be the case so these possible impacts have been assessed as 'Minimal concern' but with low confidence.

Fisheries operations: See aquaculture operations above. Currently there is no evidence to assess impacts but it is considered unlikely to affect operations or target fish or shellfish species.

Asterocarpa humilis: References

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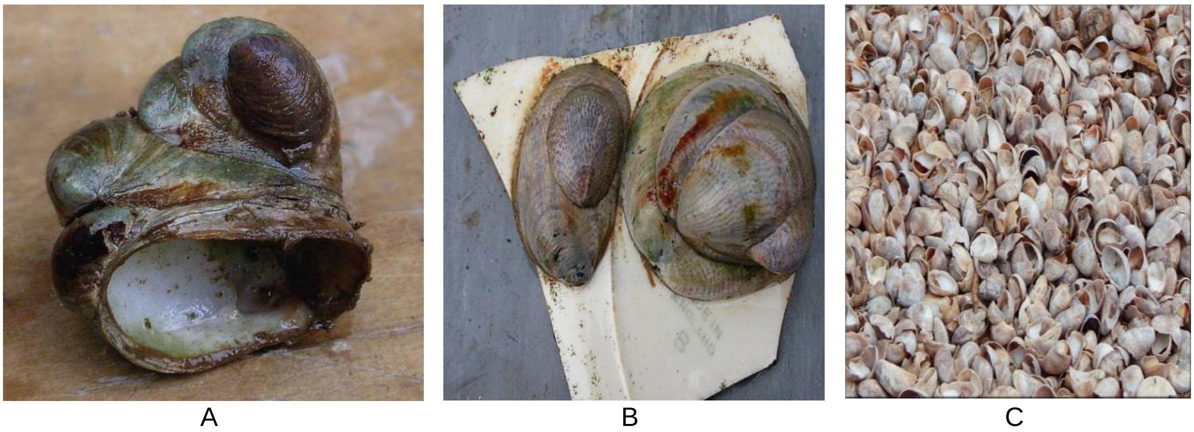
Annex 5 Slipper limpet : *Crepidula fornicata*

Common name(s): American slipper limpet; Slipper limpet; Atlantic slippersnail

Synonyms: *Crepidula densata; Crepidula virginica; Crypta nautarum; Patella fornicata* (Gofas, 2004).

Domain: Phyla: Mollusca, Class: Gastropoda, Order: Littorinimorpha , Family: Calyptraeidae , Genus/species: *Crepidula fornicata* (Gofas, 2004).

Description: The shell is oval and flattened with the spire being much reduced. It is smooth and mottled in colours of cream, yellow pink and red. Growth lines are irregular. The shells large aperture has a shelf that extends to half of the length. Shells can be up to 5 cm length in males and 10 cm in females (Rayment, 2008). *Crepidula fornicata* are often found growing on top of each other forming stacks with the largest on the bottom with each individual getting progressively smaller towards the top of the chain (GISD, 2019; Rayment, 2008).



(A & B by © J Bishop, MBA; C by Lori Schroder/www.jaxshells.org)

Figure 5.1 Slipper limpet. Crepidula *fornicata*

Crepidula fornicata: Habitat

Native range*:* *C. fornicata* (L.), which was unintentionally introduced to Europe in the 1870s with oysters imported for farming purposes from the Atlantic coast of North America (Blanchard 1997). Walne (1956) described its native geographical area as ranging from Escuminac point (47°N) on the Canadian coastline to the Caribbean islands.

Substratum type: *C. fornicata typically* inhabit shallow, sheltered bays, lagoons, estuaries and sheltered sides of islands. The species is found on a variety of substrates including rock, gravel, sand, mud and artificial substrates (GISD, 2019, Hinz et al., 2011). However, they are more abundant on muddy or mixed muddy habitats (Blanchard, 1997). Their larvae need a hard substrate on which to settle and metamorphose which is usually sand or gravel. Once established their numbers quickly grow and their shell stacks become traps for sediment and suspended matter. This coupled with their production of biodeposits (pseudofaeces) turn the substratum increasingly muddy and anoxic. Their large numbers of shells can also alter the biotope by forming a hard substrate. It is on mud or mixed muddy habitats where *C. fornicata* are more often found (De Mountaudouin, 1999; GISD, 2019), typically attached to shells and stones around the low water mark and the shallow sublittoral (Rayment, 2008). They are found attached to *M. edulis* (mussels) and *Ostrea edulis* (oysters) shells among others (Rayment, 2008) and often mentioned in the literature inhabiting typical oyster habitat (Blanchard, 2009). Additionally, they are able to tolerate high turbidity and low water quality (Blanchard, 2009).

Salinity: Variable 20- 40 ppt is tolerated with 30 ppt being the optimum (Blanchard, 2009).

Depth: Wave action may displace some individuals and stacks higher up the shore from lower shore populations but intertidal habitats above mid-shore are generally a stressful environment for *C. fornicata* (Bohn, 2014). Areas of tidal flats with extended and regular periods of emersion are not favourable (Thieltges *et al*., 2003). Where the sediment is suitable *C. fornicata* is typically found in the sublittoral fringe and lower shore (Blanchard, 2009) where it may be abundant (Bohn *et al*., 2015; Thieltges *et al*., 2003). Maximum recorded subtidal depth is 100 m in the Atlantic (Blanchard, 2009).

Wave exposure: *C. fornicata* typically reaches highest abundances in sheltered to very sheltered conditions (Blanchard, 2009). However, Hinz et al. (2011) recorded this species in rough ground with high current velocities off the Isle of Wight. It may be that the oscillatory water movements associated with wave action are less tolerated than current velocities which are unidirectional but no evidence was found to address this. Although attached individuals may be able to survive in wave exposed habits due to the low profile of the shell, wave action may limit the size of stacks that can be formed and thus reduce reproductive success, although no evidence was found to assess this

Crepidula fornicata: Establishment in Wales in 2020

C. fornicata spread rapidly within Welsh coastal waters since it was first recorded in the Milford Haven Waterway (MHW) in 1953 (Cole, 1953). Although it is widely established in South and South West Wales, there has been little indication of a northwards range extension of the species through natural processes (e.g. larval dispersal); it seems to remain absent from areas north of the Milford Haven Waterway (Bohn, 2014).

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Crepidula fornicata: Impact pathways based on species traits, biology and ecology

Competition Established populations of *C. fornicata*, with high densities of several thousand individuals per m^2 (De Montaudouin *et al.* 1999a) could compete with other filter feeders, for food and space (Blanchard, 2009). *C. fornicata* was introduced into the Bay of Brest, France in the 1970s and quickly spread throughout the basin (Thieltges *et al*., 2006b;Thouzeau, 1989;Thouzeau *et al*., 2000). Dramatic changes to the trophic structure were reported: phytoplankton composition changed which over time changed the food web structure of the basin (Chauvaud *et al*., 2000).

Thieltges (2005), experimented with artificial *C. fornicata* stacks on *M. edulis* shells and found the mortality rates of *M. edulis* to be the same as real *C. fornicata* stacks (see biofouling below). These results imply that competition for food is not important (Thieltges, 2005). Riera *et al*., (2002) found that both *M. edulis* and *C. fornicata* feed upon similar food sources suggesting potential food competition if supply was limited. However, there is no evidence for trophic competition between the two species (Thieltges *et al*., 2006b).

Site specific factors, particularly food supply will alter the level of impact from dense beds. The impact from this pathway was assessed as minor for all MPA features characterized by filter feeders with low confidence. For MPA features where establishment was likely to be at low densities and/or the feature was not characterized by dense filter feeders the impact was assessed as minimal concern with low confidence.

Note: competition for space is assessed through the structural impact pathway below.

Predation: *C. fornicata* is a filter-feeder feeding on pelagic algae, detritus and bacterial material (GISD, 2019) and this pathway is ‘Not applicable’.

Hybridisation: There is no evidence in the literature to suggest that *C. fornicata* hybridizes with other species (Laverty *et al*., 2015). As such this pathway is considered ‘Not applicable’.

Transmission of disease: In Rhode Island, USA, *C. fornicata* was found not to be host of any trematode larvae which is unusual since it is commonly assumed that marine gastropods are all trematode hosts (Pechenik, *et al*., 2012). Thieltges *et al*., (2008) found that the presence of *C. fornicata* can reduce trematode parasite load on nearby *M. edulis* by up to 77% (Thieltges *et al*., 2006b; Thieltges *et al*., 2008). No evidence was found that this species transmits disease (Laverty *et al*., 2015) only that it acts as a sink for larval trematodes (Thieltges *et al*., 2008). This pathway is considered ‘Not applicable’.

Parasitism: It has been found to be a host for the shell boring polychaete *Polydora ciliata* in the Wadden Sea (eastern North Sea) (Thieltges *et al*., 2006a). However, there is no evidence in the literature to suggest that *C. fornicata* is a parasite itself and this pathway is considered ‘Not applicable’.

Poisoning/toxicity: *C. fornicata* is not toxic or poisonous (Rayment, 2008) so this pathway is ‘Not applicable’.

Bio-fouling: This invasive species is often found attached to bivalve shells such as mussels *M*. *edulis* (mussels) and oysters *O. edulis* (Rayment, 2008; Thieltges *et al*., 2003). Thieltges (2005) experimentally tested in the field the effects of *C. fornicata* as an epibiont on the mussel *M. edulis*. Mussels were shown to have a 28-30 % mortality rate when fouled with *C. fornicata* and those that did survive had a reduced shell growth. When compared with unfouled mussels, fouled mussels had a 3-5 times lower growth rate. Field experiments found that *C. fornicata* stacks that were fouling *M*. *edulis* shells caused changes in small scale hydrodynamics through increased drag forces (Thieltges, 2005). This enhanced drag caused considerable more energy expenditure for the mussel in the form of byssus thread production to prevent dislodgement and may impact overall fitness of the mussel (Thieltges, 2005).

This impact pathway refers to the growth of *C. fornicata* on characterising species, competition for space and structural changes to the habitat resulting from the replacement of hard or mobile substratum by dense beds of *C. fornicata* are assessed through the structural change pathway (below). For biogenic features both biofouling and structural change are inseparable and the assessments are the same for both these impacts.

Grazing/ herbivory/ browsing: *C. fornicata* is largely a suspension feeder but may graze biofilms (Martin *et al*., 2006) as it does not consume macrophytes this pathway is not applicable and no impacts have been recorded (Laverty *et al*., 2015).

Indirect impacts through interactions with other species: Indirect impacts from *C. fornicata* are challenging to disentangle from the direct effects of smothering resulting from the build-up of faeces and pseudofaeces and reduction in hydrodynamic energy.

Dense beds of filter feeders have the potential to alter food web pathways including microbial activity, resulting in altered nutrient and matter cycling with associated indirect impacts on species and changes to pelagic and benthic production (see chemical impacts below). *C. fornicata* has lower filtration rates than some other invasive bivalves such as zebra mussels and in the Bay of Brest where dense populations have established, its impact on primary production in the water column appears moderate with no changes in chlorophyll biomass (Martin *et al*., 2006).

Selective feeding by *C. fornicata* probably has a major influence on microorganisms at the water–sediment interface and may change species composition and dynamics in overlying water and in the sediments and favours bacteria. Increased bacterial abundances coupled with enhanced biodeposition affect the microbial food web.

Associated carbon and nitrogen cycling processes coupled with the excretion and respiration of *C. fornicata* enhance nitrogen-regeneration and carbon-release at the water–sediment. Ammonium regeneration by dense populations of *C. fornicata* may regulate primary production and secondary productivity in enclosed bays and might increase eutrophication and support phytoplanktonic blooms lead to shifts in phytoplankton-community composition (Martin *et al*., 2006 and references therein).

As impacts will be density dependent and site specific they are challenging to assess and this has not been attempted for MPA features. Direct changes to MPA habitat features through biodeposition are more readily assessed and are likely to outweigh the indirect effects.

Crepidula fornicata: Impact pathways – Habitats

Chemical impact on ecosystem:

Dense beds of filter feeders can alter food webs and nutrient cycling within the ecosystem. Martin *et al*., (2006) has suggested that *C. fornicata* has the ability to elevate water nitrate levels through their high level of filter feeding that is implicated in a reduction of primary productivity and an increased frequency of algal blooms (dinoflagellate).

It has also been suggested that this species can influence geochemical cycling (carbon, nitrogen and silicon) when occurring as dense beds. It is possible that their rapid deposition of pseudofaeces, which contains carbon, could give them the ability to fix large amounts of carbon, since the sediments they produce do not get re-suspended. Over time this process could provide a significant carbon sink through natural sequestration (Martin *et al*., 2006).

Assessing the impact of these changes for MPA seabed features is challenging as impacts will be density dependent and site specific with factors such as tidal flushing mediating changes. In general, macrophyte dominated biotopes may benefit from enhanced nutrient cycling and increased water column nitrate levels. For all MPA features the impact is assessed as ‘Minor’. Although dense populations may alter food web dynamics and nutrient cycling impacts ramify to the water column rather than seabed habitats (Martin *et al*., 2006). Increased bacterial activity and stimulation of microbial food webs may enhance benthic production. Confidence in the assessment is low due to uncertainties and lack of evidence for benthic habitats.

Physical impact on ecosystem: Field experiments monitoring the effects of *C. fornicata* biofouling *M. edulis* shells found there were small scale changes in hydrodynamics due to the stacks formed by many *C. fornicata* individuals. The stacks extend up into the water column resulting in enhanced drag forces. This causes added energy expenditure on the mussel by producing extra byssus threads to prevent being dislodged (Thieltges, 2005). Ehrhold *et al*., (1998) also found that this stacking behaviour caused changes in benthic sediments and near-bottom currents.

Mussel dominated MPA features are assessed as minor concern based on additional energetic expenditure without reducing population size. There is no evidence that hydrodynamic impacts alone affect other MPA features and this impact pathway is assessed as ‘Minimal concern’. Increased sediment deposition is assessed through the structural change pathway below. Confidence for all assessments is low.

Dense beds of filter feeders capture large amounts of suspended particles and can reduce water turbidity resulting in increased light penetration. This may be beneficial for adjacent macrophyte dominated biotopes. Smothering by *C. fornicata* will reduce light availability to the smothered habitats. Impacts on light availability are assessed for maerl beds (impact assessed as massive with high confidence, see below) but not MPA features characterised by erect macroalgae.

Structural impact on ecosystem: This invasive species has smothered the seabed substrate in shallow bays along the Channel-Atlantic coast of France with its sheer numbers. It can create beds of several thousand individuals per m^2 (De Montaudouin *et al.*, 1999a), creating a separate biotope with its own characteristic community (Thouzeau *et al*., 2000).

Dense beds of *C. fornicata* filter large quantities of suspended particles and produce faeces and pseudofaeces which are deposited and trapped within the beds, altering habitat structure (Blanchard, 2009; FitzGerald, 2007; De Mountaudouin, 1999a; GISD, 2019) and creating anoxic, muddy sediments. Sediment type is a key factor influencing benthic assemblages and changes in sediment may therefore lead to the replacement and loss of biotopes within MPA broadscale habitat features.

The effect of *C. fornicata* on benthic communities differs according to the habitat they colonize. In muddy sediment, with or without the slipper limpet, the community is strongly dominated by deposit-feeders, but species richness and abundance are higher when *C. fornicata* is present. In coarser sediment, suspension-feeders may be abundant, but only in the absence of *C. fornicata* (De Montaudouin, 1999b). Sediment changes may also impact species through sediment preferences. When in the presence of an established *C. fornicata* community mysids have been reported to decrease in abundance (Vallet *et al*., 2001). The abundance levels are linked to the formerly soft substrate becoming a hard shell substrate (Vallet *et al*., 2001).

Where *C. fornicata* is likely to smother and significantly alter the character of the MPA feature the impact was typically assessed as ‘Massive’, as eradication is unlikely. Confidence is high where evidence has identified such impacts in the same or very similar features (see impacts on key MPA features below). Where the habitats are likely to be less suitable and dense populations were unlikely to establish but some colonisation and changes were likely then the impact was assessed as ‘Major’, and confidence was moderated according to the evidence base and establishment. Impacts on rock habitats were a clear evidence gap, as *C. fornicata* was unlikely to achieve high densities in these habitats: the impact was assessed as ‘Minor’ but at low confidence.

Crepidula fornicata: Interactions with MPA Features

MPA features that provide suitable habitat for Crepidula fornicata

Based on presence of *C. fornicata* in A4.2511 in the JNCC biological comparative tables and surveys reported by Hinz et al. (2011) that recorded presence of *C. fornicata* in rough ground (including gravel) subject to high current velocities. Confidence is medium based on uncertainties around substratum, wave exposure and density.

- A4.1 Atlantic and Mediterranean high energy circalittoral rock.
- A4.2 Atlantic and Mediterranean moderate energy circalittoral rock
- A4.3 Atlantic and Mediterranean low energy circalittoral rock
- Tide-swept channels (A4.25) based on presence in A4.2511 in the JNCC biological comparative tables

Biogenic habitats

There is a particular association with mussels which provide suitable surfaces for attachment (Bohn *et al*., 2015, Thieltges *et al*., 2003) see also section below. MPA features considered suitable are:

- A2.7 Littoral biogenic reefs (note not A2.71 Littoral *Sabellaria* reefs):
- A2.72 Littoral mussel beds on sediment, have been assessed as providing suitable habitat for *C. fornicata* because of the association with *M. edulis* (GISD, 2019; Rayment, 2008; Bohn *et al*., 2015; Thieltges *et al*., 2003) (High confidence);
- Intertidal *Mytilus edulis* beds on mixed and sandy sediments (High confidence);
- Blue mussel beds (with the exception of the strandline biotope (A2.212);
- Maerl beds provide suitable habitats, *C. fornicata* have been found to smother maerl beds in France (Thouzeau *et al*., 2000) (High confidence):
- Maerl beds;
- A5.5 Sublittoral macrophyte-dominated sediment Level 4 biotope A5.51 (Maerl beds);
- *Ostrea edulis* beds (Thieltges, *et al*., 2003 and references therein);
- Sublittoral biogenic reefs (note no evidence for A5.63 Circalittoral coral reefs):
  - A5.61 Sublittoral polychaete worm reefs on sediment based on observed occurrence (Pearce *et al*., 2007), (High confidence);
  - A5.62 Sublittoral mussel beds on sediment (High confidence).

Sediments:

Intertidal sediments with limited tidal emersion and subtidal sediments that are characteristic of habitats sheltered from wave and tidal currents are generally considered suitable for *C. fornicata* where suitable attachment surfaces such as stones and shells occur. The following MPA features are considered suitable (confidence is high, unless otherwise indicated):

- A2.4 Littoral mixed sediments (based on observations of *C. fornicata* in south west Wales on this sediment type, generally in the lower intertidal (Bohn *et al*., 2015) (High confidence);
- A5.1 Sublittoral coarse sediment: Hinz et al. (2011) recorded C. fornicata in rough ground (gravels and sands) with high current velocities off the Isle of Wight (H).
- A5.2 Sublittoral Sand:
  - A5.21 is a lagoon feature and oyster ponds and lagoons have provided suitable habitat (Blanchard, 1997), (Low confidence);
- A5.4 Sublittoral mixed sediment: Biotopes A5.41, A5.42, A5.43 within the broadscale habitat, were considered suitable. Some of the constituent biotopes are given a high confidence, as *C. fornicata* either characterises the biotope or have been recorded in the habitat e.g. oyster beds. (Medium confidence);
  - A5.431 and A5.432 *C. fornicata* is a characteristic species of these biotopes (Connor, 2014);
  - A5.435 Based on habitat preference (oyster beds) (Rayment, 2008; (Blanchard, 2009);
  - A5.44 Based on habitat preference (GISD, 2019);
- A5.3 Sublittoral mud (see below for A5.32, A5.33, A5.34, A5.36 and A5.37):
  - A5.31 is a lagoon feature and oyster ponds and lagoons have provided suitable habitat (Blanchard, 1997) (Low confidence);
  - A5.35 Circalittoral sandy mud suitable, some biotopes contain scallops for attachment or are characterised by shell debris (Medium confidence);
- Sheltered muddy gravels: based on substratum and shelter (High)
- Mud habitats in deep water:
  - A5.35 Circalittoral sandy mud suitable, some biotopes contain scallops for attachment or are characterised by shell debris (note A5.36 and A5.37 are considered unlikely to be suitable based on substratum and depth, see below). (Medium confidence).

MPA features considered potentially suitable for Crepidula fornicata

A wide range of substratum types where wave exposure and/or tidal currents are low may be suitable for *C. fornicata*. Circalittoral habitats may be more suitable than the algal dominated biotopes of the infralittoral. Potentially suitable biotopes include the following rock, biogenic and sediment habitats:

Rock habitats:

- A3.3 Atlantic and Mediterranean low energy infralittoral rock, A3.36 based on presence of M. edulis and mixed substrata in some constituent sub-biotopes. (Medium confidence).

Biogenic habitats:

- *Sabellaria alveolata* reefs
    - A5.612 *Sabellaria alveolata* on variable salinity sublittoral mixed sediment (not A2.71 Littoral *Sabellaria* reefs, see below), (High confidence);
- A2.2 Littoral sand and muddy sand; A2.24 is considered likely to be suitable, but not A2.1, A2.2 and A2.3;
- *Modiolus modiolus* beds (based on association of *C. fornicata* with *M. edulis*, confidence is low as no evidence was found); and
- *Musculus discors* beds (based on association of *C. fornicata* with *M. edulis*, confidence is low as no evidence was found).

Sediment habitats

- A5.4 Subtidal mixed muddy sediment, the biotopes A5.44 and A5.45 are considered potentially suitable but not the A5.42 sub-biotopes (see below). (Medium confidence);
- A5.2 Sublittoral sand;
    - A5.23 Infralittoral fine sand and A5.25 Circalittoral fine sand. Based on suitable habitat where sheltered and attachment surfaces are present. (Low confidence);
    - A5.24 Infralittoral muddy sand and A5.26 Circalittoral muddy sand; based on substratum (Low confidence);
- A2.3 Littoral mud may provide suitable habitat where this occurs in sheltered conditions and low on the shore and there are suitable attachment surfaces such as shell debris (based on habitat preferences described by Blanchard, 2009; GISD, 2019; Thieltges *et al*., 2003). (High confidence);
    - A2.31 Polychaete/bivalve-dominated mid estuarine mud shores;
    - A2.32 Polychaete/oligochaete-dominated upper estuarine mud shores;
- A5.3 Sublittoral mud:
    - A5.32; A5.33 A5.34, potentially suitable where substratum contains attachment surfaces such as living molluscs or shell debris. (Low confidence).

MPA features considered unlikely to be suitable for Crepidula fornicata

C. fornicata is present in higher densities in sites that are on the intertidal/subtidal fringe or fully subtidal, that are sheltered with lower levels of wave exposure and tidal currents (Blanchard, 1997, Thieltges *et al*., 2003). Emersion, sediment mobility, predation and wave action may all contribute to unsuitability (Thieltges *et al*., 2003, Bohn *et al*., 2015). The following habitats were considered unlikely to be suitable for *C. fornicata*:

A2.71 Littoral *Sabellaria* reefs, based on exposure to waves and/or tidal currents and free draining of biogenic reef (Low confidence).
Intertidal boulder communities (based on presence of macroalgae and observations by Bohn *et al*. (2015) that C. fornicata avoided boulder areas. (Confidence is low due to uncertainties around relevance of description to the MPA feature).

Limited establishment of *C. fornicata* has been observed in seagrass beds in Arcachon bay (France) (De Montaudoüin *et al*., 2001). The mechanism underlying the exclusion was not studied but if establishment is linked to sediment and the sweeping action of fronds, then other MPA features that include macrophyte dominated habitats are considered unsuitable:

A2.6 Seagrass/Zostera habitats including littoral sediments dominated by aquatic angiosperms (Medium confidence due to the lack of further evidence);
Sublittoral seagrass beds (Medium confidence due to the lack of further evidence);

- A5.5 Sublittoral macrophyte-dominated sediment (Low confidence):
    - A5.53 Sublittoral macrophyte-dominated sediment;
    - A5.54 Angiosperm communities in reduced salinity;
    - A5.52 Kelp and seaweed communities on sublittoral sediment (also a constituent of the Subtidal mixed muddy sediments MPA feature); and
- A3.3 Atlantic and Mediterranean low energy infralittoral rock:
    - A3.31, A3.32 and A3.34 based on the presence of macroalgae (Medium confidence).

Over time *C. fornicata* create a muddy sediment by trapping finer particles and deposition of faeces and pseudofaeces (biodeposits). However, fine sediments may not provide suitable habitat for establishment where there is a lack of suitable attachment surfaces. The following sediment habitats were considered unlikely to be suitable (note overlap between broadscale habitats and other MPA features):

- A5.2 Sublittoral sand:
    - A5.22 and A5.25 due to mobility of substratum. (Medium confidence);
    - A5.27 Deep circalittoral sand: due to depth, (Medium confidence);
- A5.3 Sublittoral mud:
    - A5.36 Circalittoral fine mud and A5.37 Deep circalittoral mud are considered unlikely to be suitable based on substratum and depth (Blanchard, 2009) (Medium confidence);
- Mud habitats in deep water:
    - A5.36 Circalittoral fine mud and A5.37 Deep circalittoral mud are considered unlikely to be suitable based on substratum and depth (Blanchard, 2009) (Medium confidence); and
- Sea-pen and burrowing megafauna communities. The constituent A5.36 biotopes are considered unlikely to be suitable based on substratum and depth (Blanchard, 2009) (Medium confidence).

MPA features considered unsuitable for Crepidula fornicata

Intertidal habitats that are restricted to higher shore heights (above mid-shore) and/or are coupled with high levels of wave action or tidal currents are stressful environments for *C. fornicata* (Bohn, 2014). Macroalgae may also limit establishment while wave exposure may reduce the size of stacks that can be formed.

Vegetated habitats:

- A2.5 Coastal saltmarshes and saline reedbeds, due to shore height (Blanchard 2009). (High confidence).

Rock habitats:

- A2.1 Littoral coarse sediments are likely to be too mobile and exposed to wave action to support C. fornicata (High confidence);
- A1.1 High energy littoral rock; unsuitable due to height on exposure (Bohn, 2014, Blanchard, 2009), (High confidence).
- A1.2 Moderate energy littoral rock (based on exposure and macroalgae, (Medium confidence).
- Peat and clay exposures (A1.127 and A1.223) (Low confidence).
- A3.1 Atlantic and Mediterranean high energy infralittoral rock (based on wave and tidal energy and presence of macroalgae). (Medium confidence).
- A3.2 Atlantic and Mediterranean moderate energy infralittoral rock (based on wave and tidal energy and presence of macroalgae). (Medium confidence).
- Estuarine rocky habitat (based on shore height and/or wave and tidal energy and presence of macroalgae) (Medium confidence).
- A3.3 Atlantic and Mediterranean low energy infralittoral rock (based on macroalgae):
- Tide-swept channels:
    - A1.15; A3.22; based on wave exposure (Blanchard, 2009) and macroalgae (Medium confidence).

Sediment habitats:

- A1.3 Low energy littoral rock, (based on macroalgae), (Medium confidence); and
- A2.2 Littoral sand and muddy sand, biotopes A2.21 Strandline; A2.22 due to mobility of substratum and A2.23 due to fine mobile sand, Note A2.24 considered potentially suitable see above). Confidence is medium based on Thieltges et al., (2003).

MPA features for which there is no evidence

- A5.6 Sublittoral biogenic reefs: Level 4 biotope; A5.63 Lophelia.
- Fragile sponge and anthozoan communities on subtidal rocky habitats; and
- A5.7 Carbonate reefs.

Summary of impacts on key MPA features.

Maerl beds: *C. fornicata* have smothered maerl beds in France (Hall-Spencer *et al*., 2003, Thouzeau *et al*., 2000) as the beds cover the maerl thalli which then become clogged with silt (faeces and pseudofaeces). Consequently the maerl thalli die which has a dramatic impact on the associated maerl community. Densities of 400 individuals per m^2 were found on maerl beds in the Bay of Saint-Brieuc (Thouzeau, 1989) while more recently Hamon and Blanchard (1994) found a 4 km^2 area of maerl bed had been smothered.

C. fornicata are present in maerl beds in Milford Haven and have dramatically increased since 2005 and likely to increase further. The increased silty fraction of the seabed sediment around the maerl beds in Milford Haven is thought to be caused by *C. fornicata* (European Community Directive, 2018) through increased sedimentation rates and silt trapping and reduced particle re-suspension (Barnes *et al*., 1973; De Mountaudouin, 1999; GISD, 2019). Since maerl beds are extremely sensitive to siltation the presence of *C. fornicata* is both a high pressure and a threat (JNCC 2019). The physical impact has been assessed as massive due to the slow recovery rates of maerl, (Perry and Tyler-Walters, 2018).

***Sabellaria spinulosa* reefs:** *C. fornicata* have been found associated with *S. spinulosa* reefs at Hastings Shingle Bank, SE coast of UK, with up to 66 individuals found per grab sample taken (Pearce *et al*., 2007) and in lower numbers in the East coast REC area with no more than 4 per grab sample (Pearce *et al*., 2011a and b). While the relationship between these two species has not been investigated to date it is possible that potential impacts on *S. spinosa* reefs could occur through changes in substrate suitability or other interactions (Gibb *et al*., 2014).

Mussel and Oyster beds: Mussel and Oyster beds provide suitable habitat for *C. fornicata* which is an epibiont of shells and have been found successfully living in both mussel and oyster beds throughout Europe (GISD, 2019; Thieltges, 2005). See aquaculture: target species, below for more information.

Crepidula fornicata: Socio-Economic Impacts:

Health and safety: No impacts are known (Laverty *et al*., 2015).

Aquaculture Operations: C. fornicata could alter the substratum in areas of on-bottom cultivation reducing suitability and requiring clearance or management. In addition, fouling of cultivated species results in processing costs. The impact on on-bottom culture has been assessed as ‘Major’ at high confidence.

Aquaculture cultivated species:

Oysters – *Magallana gigas***:** *C. fornicata* has been introduced and spread through the movement of oysters for farming. It has been found that trawling and dredging has further contributed to its spread (Blanchard, 2009). Once introduced to oyster beds they become competitors for food and space. *C. fornicata* also deposits pseudofaeces upon the oysters which can suffocate and prevent oyster spat settlement (Blanchard, 2009). De Montaudouin *et al.* (1999a) found that competition of *C. fornicata* on oyster growth was minor when compared with intraspecific competition between oysters.

Mussels – *M. edulis*: *C. fornicata* is an epibiont and is often found fouling shells. Field experiments by Thieltges (2005) looked at the effect *C. fornicata* stacks may have on *M. edulis* when fouling them. Results showed a 28 -30 % mortality rate in mussels with *C. fornicata* stacks growing on them and a 3-5 times lower shell growth rate when compared to unfouled mussels. These results show that the presence of *C. fornicata* amongst mussel beds could cause a significant threat to this fishery where mussel spat is collected. The same study found that the stacks caused added hydrological drag upon the mussel which in turn caused added energy expenditure in the production of byssus threads to prevent being dislodged (Thieltges, 2005). It is possible that this added drag may cause mussel clumps to be removed by water flow. This study concluded that *C. fornicata* is potentially an important mortality factor for *M. edulis*. Thieltges *et al*., (2003) reported that *C. fornicata* was abundant on mussel beds in the intertidal to subtidal transition zone in the northern Wadden Sea in the year 2000. Thieltges (2005) also observed mussel beds in the shallow subtidal infested with high abundances of *C. fornicata,* with almost no living mussels, along the shore of the List tidal basin, northern Wadden Sea. The impact to on-bottom culture of mussels has been assessed as major at high confidence.

Fisheries Operations: Where mobile gears retain large amounts of *C. fornicata* there would be additional sorting costs. Fitzgerald (2007) notes that in small scale Native oyster fisheries the presence of *C. fornicata* means that dredging becomes more difficult due to the added weight, the applicability of this to more robust mobile gears is unclear. For all mobile gears the impact is assessed as Minor at low confidence. For all static gears the impact is assessed as minimal concern. The main impacts are likely to result from direct and indirect impacts on target species (see below).

Fisheries target species:

Flatfish. *C. fornicata* cover reduces juvenile flatfish survival by preventing burial in sediments and increasing vulnerability to predators (Kostecki *et al*., 2011). In Mont Saint-Michel Bay (France) flatfish distributions are significantly reduced and almost no flatfish are found at the highest density of slipper-limpet; a similar impact was considered likely for skates although not demonstrated (Kostecki *et al*., 2011).
Sole, *Solea solea:* Le Pape *et al.* (2004) found *C. fornicata* to have a negative impact on the density of young-of-the-year sole, *Solea solea,* in the Bay of Biscay

(France) coastal nursery areas. The invasive mollusc did not have an impact on the extent of the nursey however the density of juvenile sole was significantly less in areas where *C. fornicata* had become established. Le Pape *et al.,* (2004) suggests that this gastropod reduces the available soft sediment required for the juvenile sole to feed on and bury in.

The impact on this fishery was assessed as Moderate with low confidence due to the lack of quantitative data for the impact on adult fish stocks (Le Pape *et al*. 2004, Kostecki *et al*., 2011).

Native oysters: Where *C. fornicata* occur with Native oysters, *Ostrea edulis* there are negative impacts for the fisheries operations. Economic impacts occur to oyster fishermen through all aspects of handling: high abundance of *C. fornicata* increases the sorting time and knocking these epibionts off the oysters takes time and damages oyster stock (FitzGerald, 2007).

Scallops: *C. fornicata* have smothered maerl beds in France affecting scallop dredging. While scallops grow well amongst maerl beds their densities are much lower on *C. fornicata* beds. Maerl beds are also known to be feeding areas for juvenile fish and are a good habitat for brood stock of commercial bivalve species (Hall-Spencer *et al*., 2003). This may change if maerl beds are smothered and replaced with *C. fornicata* beds.

Scallops bury themselves in soft sediment to evade predators but can also escape if attacked (FitzGerald, 2007). However, *C. fornicata* can impact this escape defence by adding extra weight when attached to scallops. It may also be difficult for scallops to find space to hide if the bottom sediment is covered with *C. fornicata* shells which will leave them exposed to predators. As such their habitat is reduced and their densities drop (FitzGerald, 2007). Negative impacts to the scallop fisheries include: Increased sorting time; reduced quality of catch (reportedly ~10kg of C. fornicata to every 50kg of scallops); time and expense needed for end of sorting disposal and loss of harvestable area (FitzGerald, 2007). Fresard *et al*., (2006) reported an estimated 25% loss of harvestable area per year which represents a loss of 97 % harvestable area in 12 years. Based on this evidence the impact on scallop fisheries is assessed as massive at high confidence.

***Mussel-M. edulis*, spat collection for on-growing and harvesting:** See above for impacts on mussel beds. Impact on this fishery is assessed as massive, where dense beds establish at medium confidence.

Other commercial mollusc species: *C. fornicata* has also been recorded as an epibiont on common whelk (*Buccinum undatum*), scallops and cockles (*Cerastoderma edule*) (Thieltges *et al*., 2003). There will be trophic competition for resources such as space, food between these species and *C. fornicata.* Its ability to alter habitats would also have a negative impact on these molluscs (De Mountaudouin, 1999; GISD, 2019).

Impacts on whelk fisheries are assessed as minor at low confidence as no evidence was found for fouling rates although it is assumed there may be some impacts on the population through reduced growth due to increased energetic costs and increased handling times to allow for clearing. Fouling may also prevent whelks entering baited pots.

Impacts on cockle fisheries are assessed as moderate at low confidence given the paucity of evidence for establishment and density in these habitats.

Crepidula fornicata: References

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Annex 6 Carpet sea squirt: *Didemnum vexillum*

Common name(s): Carpet sea squirt; marine vomit (Dijkstra, 2009).

Synonyms: *Didemnum vestitum; Didemnum vestum* (Shenkar *et al*., 2019).

Domain: Phyla: Chordata, Class: Ascidiacea, Order: Aplousobranchia, Family: Didemnidae, Genus/species: *Didemnum vexillum* (Shenkar *et al*., 2019).

Description: Colonial sea squirt that forms extensive sheets (2-5 mm thick) as well as long, pendulous outgrowths or tendrils. Colour can range from pale orange to cream or beige and can be marbled in appearance. It is firm and leathery in texture. It is interspersed with large water exits and many, smaller pores that close when disturbed turning to white spots (Bishop, 2010; Dijkstra, 2009).

(A & B by © John Bishop, MBA; image C by © Gordon King.)

Figure 6.1. Carpet sea squirt: *Didemnum vexillum*

Didemnum vexillum: Habitat

Native range: Over the past 40 years, an increasing number of previously unrecorded populations of a colonial ascidian, recently identified as *D. vexillum*, have been documented in most temperate coastal regions of the world, impacting aquaculture operations, natural rocky habitats, cobble/gravel substrates, and eelgrass beds. The earliest sample thought to be *D. vexillum* was collected in Mutsu Bay, Japan in 1926, but was not identified to species at the time (Lambert, 2009). Because of incomplete historical records and the numerous mis-identifications of this species, the native range of *D. vexillum* has not been conclusively known. Genetic diversity in Japan is greater than in any other region and in agreement with the sparse historical data, the molecular evidence suggests that Japan lies within the native range of *D. vexillum* (Stefaniak *et al*., 2012).

Substratum type: Initially recorded in the UK from marinas and adjacent, shallow man-made structures, but also present on natural shores in N. Kent (Hitchen, 2012) and the Solent (J Bishop unpublished observations) and on the open seabed (N Kent;[9]). In other areas of introduction it can also be found on natural pebble, cobble

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[9] https://www.youtube.com/watch?v=eHrwWFaUF4Y

and gravel seabeds, tidal pools, seagrass beds, mussels (Bishop, 2010), tidal lagoons and estuaries and on bivalve and salmon aquaculture facilities (Dijkstra *et al*., 2007) and on macroalgae (Dijkstra *et al*., 2007, Vercaemer *et al*., 2015). Its success as a colonizer is partly due to its environmental tolerances*. Didemnum vexillum (D. vexillum)* tolerates a wide range of salinities (Gröner *et al*., 2011), temperatures (0-28ºC) (Dijkstra *et al*., 2007) and nutrients (Carman *et al*., 2007). It is able to colonize both vertical and horizontal surfaces of fouling and benthic communities (Dijkstra *et al*., 2007). It its invasive ranges it is often found fouling artificial structures in harbours but, unlike many invasive species that stay restricted to artificial substratum, it can quickly move on to foul natural, healthy benthic substrates (Dijkstra, 2009; Kleeman, 2009). It can be found in shallow, intertidal rockpools (Kleeman, 2009), on subtidal rocky substrates (Dijkstra, 2009) and fouling a variety of different substrates to depths of 30-80m (Kleeman, 2009). It may be found deeper where surveys are more limited (Kleeman, 2009). It has been recorded offshore, on the east coast of USA forming a monoculture over vast areas of the seabed, more than 230 km^2, fouling benthic substrates and sessile communities. Its spread has been limited in this area by shifting sandy substrates. It prefers some sort of epibenthos to attach to rather than a barren substrate (Kleeman, 2009) and seems to thrive best on the shaded underside of floating objects like pontoons and boat hulls (Kleeman, 2009).

Salinity: ~ 20-45 ppt tolerated. Common in marine waters ~33 ppt and have been recorded to grow fastest in high salinity conditions of 26-30 ppt. *D. vexillum* has been found in estuarine conditions (Dijkstra, 2009; Dijkstra *et al*., 2007) though colony die-offs have been noted in salinities < 20 psu (Bullard & Whitlatch, 2009).

Depth: <1- 65 m (Valentine *et al*., 2007; Kleeman, 2009) / 80 m (Bishop, 2010).

Wave exposure: *D. vexillum* has been recorded overgrowing areas of pebble/cobble substratum in New England, USA that experience high degrees of physical disturbance from currents and wave action (Mercer *et al*., 2009). The tidal currents are often >1ms^{-1}(1-2 knots) with high sedimentation (Valentine *et al*., 2007). Investigations found that *D. vexillum* has a high tensile strength when compared with other colonial sea squirts and a greater elasticity (Reinhardt *et al*., 2012). Mckenzie *et al*. (2017) state that *D. vexillum* is usually found in areas where the colony is protected from wave action and sedimentation.

Didemnum vexillum: Establishment in Wales as of 2020

First discovered in Holyhead Marina (2008) covering algae, mussels and manmade substrata including pontoons and ropes (Holt et al., 2009), but not found during surveys of other Welsh marinas (Wood et al., 2014). In the UK also found on W. coast of Scotland (Clyde, and Lochs Creran (Cottier-Cook et al., 2019) and Fyne (https://www.peelports.com/media/3135/carpet-sea-squirt-in-loch-fyne-information-heet.pdf), south Devon and the Solent (J Bishop, unpublished data), N. Kent (Hitchin, 2012), Essex (Wood *et al*., 2016) and Suffolk (C Ashelby, unpublished data). Eradication attempted in Wales (Holyhead Marina) in 2009/2010 (Sambrook *et al*., 2014), but ultimately unsuccessful.

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Didemnum vexillum: Impact pathways based on species traits, biology and ecology

Competition: *D. vexillum* competes with other sessile organisms for space and food (Dijkstra, 2009) whilst at the same time preventing epibenthic larvae from settling on it by lowering its surface pH (Bullard *et al*., 2013; Morris *et al*., 2009). It smothers sessile communities and has a tendency to monopolize resources like space and food through its ability to rapidly colonise areas (Dijkstra, 2009) and its seeming lack of predators (Valentine *et al*., 2007). Its traits of being a habitat generalist that pioneers disturbed areas, being shade tolerant, fast growing and highly reproductive with the ability to asexually reproduce have led to its invasive success (Dijkstra, 2009). The assessment for this pathway has been based on likelihood of establishment and the native species within these habitats (see EICAT Excel spreadsheet that accompanies this report for detail). Impacts are assessed as major where sessile organisms and algae may be overgrown and smothered and where competition is therefore focussed on space occupation (at medium confidence). For less suitable habitats where colonisation and spread may be limited and/or where species are largely deposit feeding infauna and competition was for food rather than space, impacts were assessed as minimal concern at high confidence.

Predation: Adult stages of *D. vexillum* are filter feeders of phytoplankton, suspended bacteria and detritus in the water column. The short duration larva stage gains its nutrients from its egg yolk (Mckenzie *et al*., 2017 and references therein). These feeding habits are unlikely to have a deleterious effect on taxa inhabiting MPA features so this pathway is considered ‘Not applicable’.

Hybridisation: There is little in the literature to suggest that *D. vexillum* will hybridise with other native sea squirts. The internal fertilization mechanism of this group of colonial ascidians makes hybridisation less likely than in external fertilising groups (J. Bishop pers. comm.) therefore, this pathway has been assessed as ‘Not applicable’.

Transmission of disease: There has not been a disease found in its invasive range that can limit its success (Gittenberger, 2010). There is also nothing in the literature to suggest that *D. vexillum* transmits any diseases to native taxa. As such this pathway is considered ‘Not applicable’.

Parasitism: While this species has been recorded carrying parasites in its guts (Reuckert *et al*., 2015) it is not a parasite itself. Therefore this pathway is considered ‘Not applicable’.

Poisoning/toxicity: It uses chemical defences which result in a lower surface pH of 1-2 (Bullard *et al*., 2013). This is used to prevent the settlement of unwelcome larvae.

However, these chemicals are deterrents not poisons so this pathway has been assessed as ‘Not applicable’.

Bio-fouling: *D. vexillum* is a successful biofouler. It has been found as an epibiont on kelp (Dijkstra *et al*., 2007), mussels (Bishop, 2010), eelgrass (Carman & Grunden, 2010); aquaculture gear (Morris *et al*., 2009) and hard surface sessile communities (Bishop, 2010; Dijkstra, 2009) including pebble and cobble substrates (Mercer *et al.,* 2009). It has the ability to form monocultures (Dijkstra, 2009) of large areas of seabed (Morris *et al*., 2009) growing over many different substrates and sessile communities (Carman *et al*., 2009; Dijkstra, 2009) and artificial structures in its path (Carman *et al*., 2009). It has fouled aquaculture gear in New England (USA) (Morris *et al*., 2009) and is capable of encapsulating and smothering bivalves (Carman *et al*., 2009; Valentine *et al*., 2007). This pathway has been assessed based on likelihood of establishment and to what extent it may establish within the MPA features and ranges from major to minimal concern and with confidence varying according to information for potential to occur within the habitat and form dense colonies (see EICAT Excel spreadsheet that accompanies this report for detail).

Grazing/herbivory/browsing: *D. vexillum* is a filter feeder of plankton so this pathway is categorised ‘Not applicable’.

Indirect impacts through interactions with other species: *D. vexillum* has a tendency to overgrow substrate and sessile communities (Dijkstra, 2009). This ability with its generalist traits of pioneering disturbed areas, being shade tolerant, fast growing and being highly reproductive with the ability to asexually reproduce has enabled it to establish quickly and turn vast areas into a monoculture of colonial *D. vexillum* (Dijkstra, 2009). This has negative impacts on the habitat and the species that rely on this habitat for food and shelter. It can inhibit the feeding of sessile organisms that have been overgrown, inhibit the feeding of predators such as fish and crustaceans by growing over their prey (Morris *et al*., 2009; Reinhardt *et al*., 2012), prevent settlement by taking up space and by inhibiting settlement on itself by lowering its surface pH (Bullard *et al*., 2013) and covering habitat that would otherwise be used by mobile organisms such as lobsters and crabs (Morris *et al.,* 2009; Reinhardt *et al*., 2012). The assessment for this pathway of minimal concern or moderate has been based on likelihood of establishment and the habitat community within these habitats (see EICAT Excel spreadsheet that accompanies this report for detail). Some of the MPA features have been assessed through the ‘competition pathway’ above.

Didemnum vexillum: Impact pathways – Habitats

Chemical impact on ecosystem: There is nothing in the literature to suggest that *D. vexillum* is poisonous or toxic leading to deleterious effects on the surrounding native taxa. As such this pathway is considered ‘Not applicable’.

Physical impact on ecosystem: *D. vexillum* has a tendency to overgrow habitats turning them into a monoculture (Bullard *et al.*, 2007; Dijkstra, 2009). This prevents light and food from reaching the sessile community underneath it. The barrier it forms between the substratum below and everything else above also prevents predators from feeding on the bottom and larvae and spat from settling (Dijkstra, 2009; Morris *et al*., 2009). *D. vexillum* has been recorded growing on eelgrass in NE USA (Carman & Grunden, 2010) which could lead to reduced growth due to a reduction of light availability. Subsequently in California, Long & Grosholz (2015) demonstrated that overgrowth of Z*ostera marina* blades by *D. vexillum* reduced the above-ground growth of the seagrass. A reduction in eelgrass growth, due to light reduction, has been observed in the past as a result of fouling by other invasive ascidians (Wong & Vercaemer, 2012).

The impact assessments of the MPA features for this pathway range from 'minimal concern' to 'major concern' depending on the area within the habitats that *D. vexillum* is likely to foul. The amount of fouling will determine the physical impacts. Where there has been uncertainty on these assessments they have been scored with low confidence (see EICAT Excel spreadsheet that accompanies this report for detail).

Structural impact on ecosystem: *D. vexillum* is a highly successful biofouler. Its ability to grow fast, reproduce asexually, inhabit a wide variety of habitats, pioneer disturbed habitats and tolerate shade allows it to spread into many areas fast (Dijkstra, 2009). It grows over many substrates including communities of sessile invertebrates changing the habitat to a monoculture of *D. vexillum* (Dijkstra, 2009). Its surface has a low pH which is surmised to be the reason many settling larvae of sessile species don't settle on this invasive tunicate (Morris *et al*., 2009). *D. vexillum* has substantially restructured benthic communities on Georges Bank on the E. Seaboard of N. America (Lengyel *et al*., 2009; Kaplan *et al*., 2018). Impacts on MPA features have been assessed from 'minimal concern' to 'major concern' depending on the likelihood that *D. vexillum* could foul small or large areas within the habitats. Where there has been uncertainty on these assessments they have been scored with low confidence (see EICAT Excel spreadsheet that accompanies this report for detail).

Didemnum vexillum: Interactions with MPA Features

MPA features that provide suitable habitat Didemnum vexillum

Biogenic reefs including mussel and oyster beds, with their multi-faceted nature, provide suitable substratum to adhere to (Dijkstra *et al*., 2007; Carman *et al*., 2009; Valentine *et al*., 2007). There is a lack of evidence suggesting that *D. vexillum* establishes well in the intertidal zone however, it can establish on the shore, preferring downward facing surfaces, like overhangs or the underside of boulders (Hitchin, 2012) presumably to prevent desiccation. High confidence.

Intertidal underboulder/boulder communities;

A2.6 Littoral sediments dominated by aquatic angiosperms;
Seagrass beds;
Zostera beds;
A5.6 Sublittoral biogenic reefs; (medium confidence)
Blue mussel beds (except A2.212);
Intertidal *Mytilus edulis* beds on mixed and sandy sediments; and
Modiolus modiolus beds.

MPA features considered potentially suitable for Didemnum vexillum

All the following MPA features are considered potentially suitable habitat for *D. vexillum* (Bishop, 2010; Carman *et al*., 2009; Dijkstra, 2009; Valentine *et al*., 2007):

Littoral, infralittoral and circalittoral rock are considered potentially suitable based on suitable attachment opportunities and have all been scored medium confidence except 'Estuarine rocky habitats' which are low confidence.

- A1.1 High energy littoral rock;
- A1.2 Moderate energy littoral rock;
- A1.3 Low energy littoral rock;
- A3.1 Atlantic and Mediterranean high energy infralittoral rock;
- A3.2 Atlantic and Mediterranean moderate energy infralittoral rock;
- A4.1 Atlantic and Mediterranean high energy circalittoral rock;
- A4.2 Atlantic and Mediterranean moderate energy circalittoral rock;
- A4.3 Atlantic and Mediterranean low energy circalittoral rock;
- Estuarine rocky habitat (except A1.32):
  - A3.32, A3.36 (low confidence);
- Fragile sponge and anthozoan communities on subtidal rocky habitats (except A4.133 & A4.211 only Scotland & Ireland to date); and
- Tide-swept channels (except A5.5211).

Littoral and sublittoral habitats with suitable surfaces for attachment:

- A5.1 Sublittoral coarse sediment (low confidence);
- A5.4 Sublittoral mixed sediments (medium confidence);
- Subtidal mixed muddy sediments (medium confidence);
- Sheltered muddy gravels (medium confidence);
- Mud habitats in deep water (medium confidence) ;
- Sea-pen and burrowing megafauna communities (medium confidence);
- A5.5 Sublittoral macrophyte-dominated sediment (medium confidence);
- Maerl beds (high confidence);
- *Musculus discors* beds (high confidence);
- *Ostrea edulis* beds (high confidence);
- Peat and clay exposures and (low confidence);
- A2.7 Littoral biogenic reefs:

    - A2.72 (medium confidence).

MPA features considered unlikely to be suitable for Didemnum vexillum

The following sediments are unlikely to be suitable except where suitable attachment surfaces occur in crevices and overhangs in the littoral zone and/or because of the mobile nature of the sediments. Medium confidence.

- Estuarine rocky habitat:
    - A1.32;
- Tide-swept channels:
    - A5.5211;
- A2.1 Littoral coarse sediment.

The habitats below are considered to lack suitable attachment surfaces. Medium confidence.

- A2.2 Littoral sand and muddy sand;
- A2.3 Littoral mud;
- A2.4 Littoral mixed sediments;
- A5.2 Sublittoral sand; and
- A5.3 Sublittoral mud.

The habitats below are considered too well drained to be suitable for *D. vexillum* to establish in. Medium confidence.

- Littoral biogenic reefs:
    - A2.71;
- *Sabellaria alveolata* reefs:
    - A2.71.

MPA features that are not suitable habitat for Didemnum vexillum

The habitats below are considered too high up on the shore with a lack of tidal flow. High confidence.

- A2.5 Coastal saltmarshes and saline reedbeds;
- Blue mussel beds:
    - A2.212.

No evidence for *Didemnum vexillum* in these MPA habitats

- Carbonate reefs.

Summary of the impacts on the MPA features

D. vexillum is generally found fouling artificial structures in the UK (J. Bishop pers, comm,). MPA features most at risk are likely to be situated near to artificial structures where *D. vexillum* can easily spread out from. MPA features such as rock and gravel with an established sessile community will provide suitable attachment substrate (Bishop, 2010; Coutts & Forrest, 2007; Osman & Whitlatch, 2007) as do biogenic reefs with their many attachment surfaces. It has been found to die when exposed to air for > 6 hours (Laing *et al*., 2010) and is therefore unlikely to establish well in the littoral zone.

Bivalve beds: (Oysters and mussels) *D. vexillum* has been recorded fouling bivalves including mussels, oysters and scallops. These give the colonial sea squirt a multi-faceted structure with which to adhere to. Pacific oysters have been fouled by *D. vexillum* on the west coast of Canada (Valentine *et al*., 2007) and mussels in Ireland (Minchin & Nunn, 2013). Impacts on bivalve reefs are assessed as 'Major' with high confidence.

Tidal swept deep gravel (cobbles and pebbles) habitats: These habitats are considered to be vulnerable due to their suitability for D. vexillum to colonise. At Georges Bank, USA, large swathes of benthic gravel habitat have been colonized by *D. vexillum*. 230 km^2 has been covered and only the presence of mobile sands have put a stop to its further spread (Valentine *et al*., 2007).

Seagrass and Zostrea habitats: *D. vexillum* has been recorded growing on eelgrass habitats in northeast United States (Carman & Grunden, 2010). This may lead to reduced light and therefore growth which has been demonstrated with other invasive tunicate species (Wong & Vercaemer, 2012). Impacts are assessed as 'Major' at high confidence.

Didemnum vexillum: Socio-Economic Impacts

Human health and safety: No impacts are known

Aquaculture Operations: It is highly likely that *D. vexillum* may establish on fish farm cages, nets and other equipment and gear. They are known to establish on vertical, artificial structures like harbour walls, pilings and mussel longlines (Bishop, 2010; Coutts & Forrest, 2007; Osman & Whitlatch, 2007). This would cause a negative, economic impact with regards to clearance maintenance since the weight of this colonial sea squirt would restrict light and water flow through the net and its weight would pull it downwards. Infestations of tunicates, such as *D. vexillum,* could cause the aquaculture industry huge economic losses through loss of shellfish and biofouling of the equipment (Carver *et al.,* 2003). It has been found in association with mussel longline cultivation in Ireland (Minchin & Nunn, 2013) and in Canada fouling mussel cages (Lambert, 2009). Impacts on aquaculture operations are assessed as major at medium confidence.

Aquaculture cultivated species:

Mussel aquaculture: *D. vexillum* has been termed a ‘shellfish pest’ because of its ability to completely encapsulate bivalves, including mussels, and smother them resulting in death (Carman *et al*., 2009; Valentine *et al*., 2007). Partial encapsulation can reduce the bivalve growth rather than causing death but either way this is a negative impact on shellfisheries where this invasive sea squirt has become established. It has to be removed manually or by other methods (Coutts & Forrest, 2007). Aquaculture industries in New Zealand have spent $807,000 attempting to eradicate this ‘pest’ species from economically important areas such as mussel lines where it smothered the green mussel, *Perna canaliculus* (Dijkstra, 2009). It has also been recorded attached to the blue mussel (*Mytilus edulis*) (see Figure 3.1B).Impacts on mussels are assessed as ‘Major’ at medium confidence.

Oyster aquaculture: *D. vexillum* has been termed a ‘shellfish pest’. It has been recorded growing over beds of shellfish often smothering them to death (Carman *et al*., 2009; Valentine *et al*., 2007). It grows rapidly and can overgrow shell clusters in a few weeks resulting in reduced shell growth, misshapen growth and even death. Pacific oysters fouled by *D. vexillum* on the west coast of Canada showed a lower condition index than oysters which had been treated, both chemical and mechanical, to reduce fouling. It is possible that the lowered condition of these fouled oysters was due to a decrease in water flow to the bivalves which limited access whilst also creating competition for food resources (Switzer *et al*., 2011). In Ireland it has been found establishing on oyster trestle tables and on oyster reefs in Canada. It is thought that the oyster farms in Canada have become infested with *D. vexillum* as a result of the movement of contaminated oysters from spawning and settlement bays in British Columbia (Mckenzie *et al*., 2017).

The spread of *D. vexillum* to the shellfish industry in Wales would have serious negative impacts. These impacts have already been seen in countries including Canada and New Zealand with *D. vexillum* fouling oysters and mussels. Impacts have been seen previously when solitary tunicates like *Ciona intestinalis* (vase tunicate) have fouled shellfisheries and as a result increased costs of production and processing whilst also negatively affecting meat yields and growth rates because of increased competition for resources (Mckenzie *et al*., 2016). The build -up of this colonial sea squirt on the outside of aquaculture nets and bags restricts water flow and food availability to the commercially farmed species and causes extra work through removal maintenance of the aquaculture equipment (Mckenzie *et al*., 2016).

Oysters-*Ostrea edulis*, *Magallana gigas*: *D. vexillum* can inhibit oyster growth and feeding rates, impact overall health (Switzer *et al*., 2011) and even smother oysters (Carman *et al*., 2009; Valentine *et al*., 2007). It may also impact spat settlement. See oyster fishery and scallops above. The impact was assessed as ‘Major’ with medium confidence.

Lobster-*Hommarus gammarus*: Should *D. vexillum* become established on lobster holding pens and associated equipment this would also have negative, economic impacts with regards to equipment maintenance.

Fisheries Operations: *D. vexillum* is unlikely to establish on fishing gear that has limited periods underwater such as lobster pots and fishing nets. There may be impacts on fisheries operations using mobile gears where these become clogged and heavier and where favoured fishing grounds are fouled. Impacts are assessed as 'Moderate' at low confidence due to lack of evidence.

Fisheries target species:

Scallop-*Pecten maximus, Aequipecten opercularis*: *D. vexillum* has become established and widespread in New England (USA) waters. It has been found fouling coastal, shellfisheries aquaculture gear which may impact shellfish growth rates and increases costs through extra maintenance (Morris *et al*., 2009). It has colonised vast areas of Eastport, Maine (Valentine *et al*., 2007) and Georges Bank (USA) on shell-gravel substratum including commercial sea scallop (*Placopecten magellanicus*) grounds (Morris *et al*., 2009) and has been recorded completely encapsulating scallops and smothering them to death (Valentine *et al*., 2007). Individuals fouled by *D. vexillum* can have a decreased swimming ability which can in turn limit their escape success from predators and their access to food-rich habitats, which ultimately could affect growth and survival (Dijkstra & Nolan, 2011).
In laboratory experiments it has been found that the larvae of the bay scallop (*Argopecten irradians irradians)* avoided settling on *D. vexillum* which suggests a reduction in suitable settlement area for this scallop in areas of high *D. vexillum* abundance (Morris *et al*., 2009). It has been hypothesised that the larvae avoided this invasive, colonial tunicate due to low pH of *D. vexillum's* surface tissue. These results could have impacts for the wider scallop community where *D. vexillum* is present (Morris *et al*., 2009). Given the predicted impacts on scallop survival and settlement this impact has been assessed as 'Major'.

Finfish: *D. vexillum* has been termed an 'ecosystem engineer' due to its ability to alter habitat complexity and community structure. Observed changes in benthic community structure have included more-deposit feeders and infauna which is possibly the result of decreasing foraging ability of larger, mobile predators. A significant increase in polychaete worms has also been observed in areas of *D. vexillum* establishment. These changes to benthic habitat community have led to speculation that it may negatively impact benthic prey availability for benthic fish species (Lengyel *et al*., 2009). As there is no evidence to date to support these speculations and there appears to be little overlap between mud and sand that are likely fish foraging and nursery grounds, this report has assessed this potential threat as 'Minor', with low confidence.

Atlantic herring (*Clupea harengus*): It is possible that herring spawning grounds could be affected by *D. vexillum* overgrowing gravel habitats where Atlantic herring lay their eggs. These spawning grounds can be localized in shallow, coastal waters (Mckenzie *et al*., 2017). This impact was assessed as 'Moderate'.

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Annex 7 Chinese mitten crab: *Eriocheir sinensis*

Common name(s): Shanghai hairy crab; hairy crab; mitten crab; Chinese mitten crab; Chinese freshwater edible crab

Synonyms:

Domain: Animalia (Kingdom) Arthropoda (Phylum) Crustacea (Subphylum) Multicrustacea (Superclass) Malacostraca (Class) Eumalacostraca (Subclass) Eucarida (Superorder) Decapoda (Order) Pleocyemata (Suborder) Brachyura (Infraorder) Eubrachyura (Section) Thoracotremata (Subsection) Grapsoidea (Superfamily) Varunidae (Family) Varuninae (Subfamily) *Eriocheir* (Genus) *Eriocheir sinensis* (Species) (WORMS 2020)

Description:

A large crab with a maximum carapace (body) length of 56 mm. The carapace is squarish in outline, narrowing towards the front and has four anterolateral teeth on each side, 4th tooth small, and four teeth on the frontal margin. Olive green to light brown in colour with legs paler than carapace. Legs are approximately twice the length of the body. The most obvious distinguishing feature of the Chinese mitten crab is the dense setal mats of ‘wool-like’ ‘hair’ on the claws, present in males and females but denser and covering more area in males. The leading edges of the legs are also setosed (hairy).

Common in the River Thames having first been introduced as larvae in ballast water in 1935 (first UK introduction). Including the Thames, now also established in the Rivers Medway and Dee, with records from many other UK river systems including the Rivers Tyne, Humber, Mersey and Severn. Records also exist from Devon, Dungeness and Southfields Reservoir near Castleford, Yorkshire. Crabs can live for 1-5 years and males and females die after reproducing (Panning 1938). Ovigerous females are able to brood 250,000 – 1million eggs at a time (Cohen and Carlton, 1997).


(Image: Ecomare/Sytske Dijksen (CC BY-SA 4.0)

Figure 7.1 Chinese mitten crab: *Eriocheir sinensis*

Eriocheir sinensis: Habitat

Native range: Originates from The Yellow Sea and East China Sea from Korea to South-East China. Xu *et al*. (2009) describe difficulties regarding species' taxonomy and suggest both historic genetic lineages and hybridization with nearby populations of congeneric species extending Northward to Vladivostok, Russia and south to the Taiwan Strait.

Substratum type: Marine/ Estuarine mud and mixed sediment, especially those where suitable boulders, cobbles and other refuge providing features are present.

Adults usually live in freshwater and make burrows in muddy riverbanks. Aquatic vegetation and marshes may provide an alternative habitat. The ability of *Eriocheir sinensis* to traverse land has enabled colonisation of ponds and lakes and other in-land water bodies.

Salinity and Temperature: Juveniles occur in lower estuaries and marine habitats but immediately migrate upstream (summer and early autumn) into brackish and freshwater systems. Adults migrate downstream during the autumn into deep, open, higher salinity locations of the upper estuary to reproduce. Being catadromous, the species spends the majority of its adult life in fresh water but is obligated to return to higher saline conditions in order to reproduce. Adults are euryhaline and can occur in fresh water to full sea-water conditions. Larvae are released in saline-brackish conditions. Laboratory studies suggest that *E. sinensis* can develop from hatching to full metamorphosis in salinities of 15-35ppt and temperatures from 12 – 18^{o}C, with an optimal level of 15-25ppt. (Anger, 1991). Adult crabs can survive water temperatures ranging from 4° to 31–32°C and has become abundant in river systems with winter

estuary temperatures as low as 5°C and adjacent sea surface water temperatures below 0°C (Cohen and Weinstein, 2001).
Depth: No specific information could be found, but *E. sinensis* is most commonly recorded in estuarine and freshwater habitats, as well as shallow bays and inlets. Areas with submerged vegetation are often preferred, which are limited to shallow areas (Veilleux & De Lafontaine, 2007).

Wave exposure: *E. sinensis* prefers sheltered, bays and estuaries and can most often be found in slow moving water with submerged vegetation (Veilleux & De Lafontaine, 2007), typical of low energy habitats.

Eriocheir sinensis: Establishment in Wales as of 2020

There is currently an established population in the River Dee North Wales (Falkingham et al. 2016). In 2019, a recently cast, intact molt was found washed up on the shore in Swansea Bay, and a live individual was captured on the English side of the Severn Estuary near Bristol. Although likely two isolated records, it is possible that records could indicate a nearby population or populations in the south of Wales (Bristol Channel/Severn Estuary).

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Eriocheir sinensis: Impact pathways based on species traits, biology and ecology

Competition: Both adults and juveniles are capable of defeating the native crab *Carcinus maenas* in competition for refuge space. Occasionally small individuals of both species may share space (Gilbey et al., 2008).

In fresh water, *E. sinensis* has the potential to compete with crayfish (both native and invasive) for space and food resource. Its ability to feed on a wide range of invertebrate species and the eggs of fish means populations of certain fish and invertebrates may be reduced, thus potentially reducing food resources for a range of native predatory fish and bird species.

Impacts were assessed as ‘Moderate’ (high confidence) where native crabs were considered important functional species within the biotope and ‘Minor’ (low confidence) for other habitats based on competition for food.

Predation: *E. sinensis* is an opportunistic omnivore, and in the marine and estuarine environment, prey can include bivalves, soft-bodied invertebrates and small crustaceans. Potential prey species include those such as amphipods - which play important roles in breaking down organic detritus in river systems – and fly larvae, worms, copepods and nymphs (Czerniejewsk et al., 2010). *E. sinensis* is known to consume mussels (e.g., *Mytilus edulis*), an ecologically important species in many parts of Wales, forming habitat and providing an important food source for birds, fish and invertebrate species. Wójcik et al. (2015) found adult crabs were able to consume between 5 and 15 individual mussels from a range of size classes in a 24 hour period. Furthermore, the study showed additional physical damage to mussels not consumed resulting in impaired life functions. The Chinese mitten crab is known to feed on fish eggs (Webster et al. 2015) the impact which this predation may have on species laying eggs in river systems (particularly salmonids) is not fully understood, but may be significant.

Impacts of predation ranged from ‘Major’ at medium confidence where reef forming bivalves could be severely affected to moderate where characterising invertebrates were likely to be predated on to minor and minimal concern for less suitable habitats for *E. sinensis* where abundance would be expected to be low (confidence variable).

Hybridisation: *E. sinensis* does not have any congeners nor other closely related species native to the region and hybridization is therefore considered extremely unlikely.

Transmission of disease: *E. sinensis* is known to transmit the crayfish plague pathogen (Aphanomyces astaci), which is fatal to European native crayfish species. (Schrimpf et al., 2014). Impacts are likely to impact freshwater systems, rather than marine and this impact pathway is assessed as ‘Not applicable’.

Parasitism: *E. sinensis* is not a parasite and this impact is ‘Not applicable’.

Poisoning/toxicity: ‘Not applicable’, *E. sinensis* is not toxic or poisonous.

Bio-fouling: *E.sinensis* a mobile species with a planktonic larval phase. Whilst the species is thought to have been introduced into some parts of Europe as a hitch hiker on boat hulls (Shakirova et al., 2007), it is not considered a bio-fouling species and as such, this impact is ‘Not applicable’.

Grazing/herbivory/browsing: *E. sinensis* is known to graze on a range of vascular plants in fresh water habitats (Czerniejewsk et al., 2010). Grazing on vascular plants, combined with physical damaged caused by grasping and uprooting shoots is likely to lead to decreased vegetation in invaded freshwater systems (Schoelynck et al., 2019). Such grazing is likely in brackish environments too and saltmarsh habitat may be particularly vulnerable. *Zostera* beds may also be vulnerable to similar impacts when occurring in estuarine conditions. Although no studies could be found to verify this assertion, we consider the potential threat worthy of further investigation.

Indirect impacts through interactions with other species:

Potential to impact native diadromous fish species (Atlantic Salmon, Brown trout, smelt) and other diadromous and freshwater species using gravel beds, cobbles and vegetation for spawning by predating eggs (Webster et al., 2015). Removal of key species of fish may have indirect impacts on food webs. Salmonids in particular provide an important food source for piscivorous, mammals and birds and, themselves control freshwater invertebrate populations. Coupled with egg predation, it has been suggested that increased bank erosion caused by burrowing may cause siltation of gravel runs and loss of substrate suitable for egg laying, due to riverbank burrowing and accelerated erosion.

Eriocheir sinensis: Impact pathways – Habitats

Chemical impact on ecosystem: No chemical impacts known. There is nothing in the literature to suggest that *E. sinensis* is poisonous or toxic leading to deleterious effects on the surrounding native taxa. As such this pathway is considered ‘Not applicable’.

Physical impact on ecosystem: Burrowing can occur in extremely high densities sometimes exceeding 3 burrows per m^2 (Panning, 1939) and this activity has the potential to undermine saltmarsh and bank vegetation, altering river/ seabed structure and sedimentary composition. *Zostera* beds may also be vulnerable, particularly in estuarine conditions although no studies could be found to verify this assertion.

Structural impact on ecosystem: The high levels of potential mussel consumption (Wójcik et al., 2015) suggests a potentially high impact on reef structures constructed by mussels. Grazing on vascular plants, combined with physical damaged caused by grasping and uprooting shoots is likely to lead to decreased vegetation in invaded freshwater systems (Schoelynck et al., 2019). Many such plants create important structural complexity with these ecosystems. Such damage is likely in marine and brackish environments too and saltmarsh habitat may be particularly vulnerable if combined with burrowing contributing to erosion.

Eriocheir sinensis: Interactions with MPA Features

MPA features that provide suitable habitat for *Eriocheir sinensis*

- Coastal saltmarshes and saline reed beds
- Estuarine rocky habitat (Not the intertidal biotopes A1.32);
- A5.1 Sublittoral mixed sediments;
- Blue mussel beds;
- A2.2 Littoral sand and muddy sand;
- A2.3 Littoral mud; and
- A2.4 Littoral mixed sediments.

MPA habitats considered potentially suitable for *Eriocheir sinensis*

A5.6 Sublittoral biogenic reefs;

- A5.5 Sublittoral macrophyte-dominated sediment * A5.5

- A2.6 Littoral sediments dominated by aquatic angiosperms A2.6
- Intertidal under boulder/ boulder communities
- Maerl beds; (*Lithothamnion glaciale* only)
- A5.1 Sublittoral coarse sediment;
- A5.3 Sublittoral mud;
- A5.2 Sublittoral sand;
- Subtidal mixed muddy sediments;
- *Modiolus modiolus* beds;
- Sheltered muddy gravels;
- *Ostrea edulis* beds; and
- Peat and clay exposures.

MPA habitats considered unlikely to be suitable for *Eriocheir sinensis*

- A1.2 Moderate energy littoral rock;
- A1.3 Low energy littoral rock;
- A3.3 Atlantic and Mediterranean low energy infralittoral rock;
- A4.3 Atlantic and Mediterranean low energy circalittoral rock;
- Fragile sponge and anthozoan communities on subtidal rocky habitats;
- Tide-swept channels;
- *Musculus discors* beds;
- *Sabellaria alveolata* reefs; and
- A2.1 Littoral coarse sediment.

MPA features unsuitable for *Eriocheir sinensis*

- A4.1 Atlantic and Mediterranean high energy circalittoral rock;
- A4.2 Atlantic and Mediterranean moderate energy circalittoral rock;
- A1.1 High energy littoral rock;
- A3,1 Atlantic and Mediterranean high energy infralittoral rock; and
- A3.2 Atlantic and Mediterranean moderate energy infralittoral rock.

Summary of the impacts on the MPA features.

Key impacts are likely through damage caused by burrowing in sediments and saltmarsh banks. Bivalve reefs may be vulnerable due to high levels of potential mussel consumption. Saltmarsh habitat and perhaps seagrass could be vulnerable to uprooting especially if combined with burrowing contributing to erosion.

Eriocheir sinensis: Socio-Economic Impacts

Health and Safety: The Chinese mitten crab is an aggressive crab and can cause minor injuries if handled incorrectly when processing catches.

Aquaculture Operations: No evidence was found for impacts on aquaculture operations through infrastructure.

Aquaculture cultivated species: Adult crabs are cable of consuming 5-15 individual mussels per day (Wójcik et al., 2015) In heavily infested areas, where mussels are cultivated 'on bottom' this level of predation would likely reduce stock available to commercial harvest. As a generalist predator, it is also likely that other cultivated species (scallops, oysters and clams) might be consumed. Impacts on bottom cultivation of mussels are assessed as 'Major' at medium confidence. *E. sinensis* was not considered likely to affect off-bottom culture and finfish culture.

Fisheries Operations: *E. sinensis* is known to cause damage to fishing gear and damage catches, particularly fish caught in bottom fished static gear or traps (Peters et al., 1933). In addition to damaging gear and catches, large quantities of crabs have been known to fill traps and nets reducing potential catches and in the USA, shrimp fishermen have reported extremely large bycatches of crabs when using mobile trawling gears, increasing processing time and potentially damaging associated shrimp catches (Veldhuizen, 2001). Impacts on mobile gears were assessed as 'Minor' at medium confidence and 'Moderate' at high confidence for static gears.

Fisheries Target Species: *E. sinensis* is capable of foraging and consuming fish eggs, including salmonid eggs (Webster et al., 2015). Such predation, combined with physical impacts on spawning habitat poses a threat to commercially important species, In particular salmon and sea trout breeding in Welsh river systems. Impacts on these species are assessed as 'Major' at medium confidence. Bass may be impacted through changes in the quality of saltmarsh nursery habitats (moderate impact at low confidence). Other target species were assessed as 'Minimal concern' at high confidence.

Consumption of bait species and crabs could affect commercial hand gathering, impacts were assessed as 'Moderate' at high confidence.

Eriocheir sinensis: References

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Annex 8 Red ripple bryozoan: *Watersipora subatra*

Common name(s): Red ripple bryozoan (MBA, 2014).

Synonyms: No, but there has been some confusion regarding taxonomy for this species. The species existing in the UK at present was previously referred to as *Watersipora subtorquata* and has since been re-identified as *Watersipora subatra* by Vieira *et al*., (2014). Recent DNA testing has also defined *W. subtorquata* and differentiated it from the closely related spp. *W. subovoidea* and *W. edmondsoni* and another unnamed *Watersipora* sp. (Mackie *et al*., 2006). Together these species may have been included in the *W. subtorquata* complex in the past. With this in mind, caution is advised concerning information about *W. subatra*, particularly regarding distribution data for this species (Tidbury, 2015).

Domain: Phylum: Bryozoa, Class: Gymnolaemata, Order: Cheilostomatida, Family: Watersiporidae, Genus/species: *Watersipora subatra* (WoRMS, 2019).

Description: An orange-red encrusting bryozoan, especially at the growing edges, sometimes can be purple brown, black or grey. The brighter edges are where the younger zooids reside. Its growth form varies with age. The colonies are small, flat and circular in shape when young. As the colony develops it can overgrow itself giving a rippled effect to its surface. Older more established colonies are often leaf-like in appearance (foliaceous) forming lobes and frills that can be erect, sitting up off the substratum (Porter, 2012; Fofonoff *et al*., 2018; MBA, 2014) The zooids are tightly packed in radiating lines and are rectangular and elongate in shape. A large, single orifice with a U-shaped indentation sits in the middle of each zooid (Gordon, 1989; Ryland *et al*., 2009; Viera *et al*., 2014).

(© John Bishop, MBA.)

Figure 8.1: Red ripple bryozoan: *Watersipora subatra*

Watersipora subatra: Habitat

Native range: The native origin of this species remains uncertain (Bishop, 2015) though the gulf of Mexico has been highlighted as a potential origin (Ryland *et al*., 2009). Due to the complexity of its identity uncertainty surrounds the suggested introduced range of *W. subatra* (Bishop, 2015).

Substratum type: *Watersipora subatra* (*W. subatra*) is found colonizing a variety of hard substrates, both natural and man-made. As an early successional species it is efficient at colonizing novel habitats found on artificial structures. It has been found in marinas, on docks, boat hulls, oil platforms, pilings and floating debris as well as natural substrates including seaweeds, shells and rocks and floating substrata (Mackie et al., 2006; Ryland, 2009; Cohen & Zabin, 2009; Porter, 2012; Kuhlenkamp & Kind, 2013; Page *et al*., 2019). Commonly found in the lower intertidal and shallow subtidal it can also be found in depths > 10 m (Porter, 2012).

Salinity: The published salinity tolerances for this species vary widely in the literature, some contradictory. GISD (2020) states it tolerates 25-49 psu, while Fofonoff *et al*. (2018) reports 18-40 psu.

Depth: Most common in lower intertidal and shallow subtidal but can be found down to 10 m and below (Porter, 2012; GISD, 2020).

Wave exposure: *W. subatra* can survive under a wide range of wave exposures, from sheltered sites such as enclosed marinas to exposed rocky shores (Floerl *et al.*, 2004; Bishop *et al*. 2015; Zabin *et al*., 2018).

Watersipora subatra: Establishment in Wales as of 2020

Recorded in Milford Haven and Martin’s Haven, Wales (NBN Atlas, 2017). Also from Dale beach, Pembrokeshire (C. Wood, pers. comm. 2019).

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Watersipora subatra: Impact pathways based on species traits, biology and ecology

Whilst *Watersipora* spp. have been relatively well studied in calm-water environments, where its effects on other species are variable (Sellheim *et al*. 2010; Needles & Wendt 2013). Little is known about its potential ecological impacts in open-coast conditions.

Competition: *W. subatra* has out-competed congeneric species in some of its invasive ranges (Gordon & Mawatari, 1992; Fofonoff *et al*. 2018), and it has been seen to spread rapidly on natural shores (Zabin *et al*., 2018; C. Wood, pers. comm., 2020). This impact pathway is assessed as 'Moderate' with low confidence.

Predation: 'Not applicable'. Bryozoans feed using an organ called a lophophore, a fold in the skin that is surrounded by cilia covered tentacles which sweep the water capturing plankton and bacteria (Bullivant, 1968; Fofonoff *et al.*, 2018; GISD, 2020). Their impact on the surrounding plankton population is unknown due to a lack of data.

Hybridisation: 'Not applicable'. *W. subatra* is not known to hybridize.

Transmission of disease: 'Not applicable'. *W. subatra* is not known to transmit diseases.

Parasitism: 'Not applicable'. *W. subatra* is not known to be parasitic.

Poisoning/toxicity: 'Not applicable'. Floerl *et al*. (2004) report *W. subatra* colonies provide non-toxic points of attachment for other sessile organisms.

Bio-fouling: *W. subatra* is a biofouler with a tolerance for copper and mercury antifouling paints (Floerl *et al*., 2004; Piola & Johnston, 2006; Fofonoff *et al*., 2018). Biofouling more generally is a global issue causing huge economic impacts on maritime industries. McLachlan (2017) suggests billions of dollars worldwide are spent on biofouling annually due to increase in fuel consumption, research, maintenance, and upkeep. *W. subatra* is a highly invasive bryozoan (Vieira, 2014) and known nuisance fouler to artificial marine substrates (Ryland *et al.,* 2009). It has the ability to colonize and overgrow fouling communities (see: 'Indirect impacts through interactions with other species' below) although its impacts on natural habitats are unknown (Tidbury, 2015 and references therein). This pathway is assessed as 'Minor' with low confidence.

Grazing/herbivory/browsing: 'Not applicable'. *W. subatra* is a filter feeder.

Indirect impacts through interactions with other species: *W. subatra* is capable of forming large colonies that can dominate fouling communities influencing their composition through space occupancy and overgrowth (Sellheim *et al*., 2010; Sams & Keough, 2012; Needles & Wendt, 2013). Its ability to modify species composition, increasing species richness and epifaunal motile animal diversity, is thought to be related to increased structural complexity and sediment retention (Sellheim *et al*., 2010). While this species is a common intertidal bryozoan in much of its introduced range until recently it has only occasionally been recorded on natural shores in the UK (Bishop, 2015 and references therein). However, in the Tamar Estuary, where it used to have an inconspicuous presence amongst the fouling community in marinas, its abundance has markedly increased as well as being recorded on natural shores (Wood *et al*., 2017). This also appears to be the case elsewhere in the UK (C. Wood pers.comm, 2020).Therefore, this impact pathway has been assessed as ‘Moderate’ but with low confidence.

Watersipora subatra: Impact pathways – Habitats

Chemical impact on ecosystem: ‘Not applicable’. None reported.

Physical impact on ecosystem: The sediment built up in fouling communities caused by the presence of *W. subatra* is likely to cause physical impacts such as changes in water flow throughout the fouling community structure. There is no evidence to support this, so this impact pathway has been assessed as ‘Data deficient’.

Structural impact on ecosystem: This species can form large colonies overgrowing other sessile and encrusting species. This behaviour has the ability to alter the environment structure. It has been documented dominating fouling communities (Sellheim *et al*., 2010; Sams & Keough, 2012; Needles & Wendt, 2013) increasing habitat complexity with its growth forms and ability to retain sediments. This habitat alteration can have positive effects on species richness and diversity (Sellheim *et al*., 2010; Tidbury, 2015; Fofonoff *et al*., 2018) by providing structurally complex refugia. There is little evidence to suggest the ability of *W. subatra* to modify habitat structure has any negative impacts. This impact pathway has been assessed as ‘Minimal concern’ with low confidence.

Watersipora subatra: Interactions with MPA Features

MPA features that provide suitable habitat *W. subatra*

The habitats below are suitable for *W. subatra* based on suitable attachment opportunities and/or association with mussels and other shellfish. They all have medium confidence apart from ‘Intertidal underboulder/boulder communities’,this has been scored as high confidence because *W. subatra* has been found in this habitat in northern France.

- A1.3 Low energy littoral rock;
- A1.1 High energy littoral rock;
- A1.2 Moderate energy littoral rock;
- Intertidal underboulder/boulder communities (high confidence);
- A2.7 Littoral biogenic reefs:
    - A2.72;
- A5.6 Sublittoral biogenic reefs:
    - A5.62;
- Blue mussel beds;
- Intertidal *Mytilus edulis* beds on mixed and sandy sediments;
- *Ostrea edulis* beds;

- *Musculus discors* beds;
- *Modiolus modiolus* beds; and
- Peat and clay exposures:
    - A1.223.

MPA habitats considered potentially suitable *W. subatra*.

The habitats below are considered potentially suitable for *W. subatra* based on suitable attachment substrates, especially where they occur in the lower littoral or shallow subtidal.

- A3.1 Atlantic and Mediterranean high energy infralittoral rock (medium confidence);
- A3.2 Atlantic and Mediterranean moderate energy infralittoral rock (medium confidence);
- A3.3 Atlantic and Mediterranean low energy infralittoral rock (medium confidence);
- A4.1 Atlantic and Mediterranean high energy circalittoral rock (low confidence);
- A4.2 Atlantic and Mediterranean moderate energy circalittoral rock (except A4.211 found mainly in Scotland & Ireland) (low confidence).
- A4.3 Atlantic and Mediterranean low energy circalittoral rock (low confidence);
- Estuarine rocky habitat (low confidence);
- Fragile sponge and anthozoan communities on subtidal rocky habitats (low confidence);
- Tide-swept channels (low confidence);
- A2.4 Littoral mixed sediments (low confidence);
- A5.1 Sublittoral coarse sediment (low confidence);
- A5.3 Sublittoral mud (low confidence):
    - A5.33, A5.34, A5.35;
- A5.4 Sublittoral mixed sediments (medium confidence);
    - Subtidal mixed muddy sediments (low confidence);
- Sheltered muddy gravels (low confidence);
- A2.7 Littoral biogenic reefs:
    - A2.71 (medium confidence);
- *Sabellaria alveolata* reefs (low confidence);
- A5.5 Sublittoral macrophyte-dominated sediment (low confidence);
- A5.6 Sublittoral biogenic reefs (low confidence):
    - A5.61, A5.63;
- Peat and clay exposures:
    - A1.227 (low confidence); and
- Maerl beds (low confidence).

MPA habitats considered unlikely to be suitable *W. subatra*

The habitats below are unlikely to be suitable for *W. subatra* based a lack of suitable substrate to attach to and/or the mobile nature of the sediments (medium confidence).

- A2.1 Littoral coarse sediment;
- A2.2 Littoral sand and muddy sand;
- A5.3 Sublittoral mud:
    - A5.36, A5.37;
- A5.2 Sublittoral sand;
- Mud habitats in deep water;
- Sea-pen and burrowing megafauna communities.

The seagrass habitats below are considered unlikely to be suitable for *W. subatra* to establish amongst, as there is a lack of evidence (low confidence).

- A2.6 Littoral sediments dominated by aquatic angiosperms;
- Seagrass beds; and
- *Zostera* beds.

These habitats are considered unlikely to be suitable for W. subatra because they occur too far up on the shoreline (medium confidence).

- A2.5 Coastal saltmarshes and saline reedbeds .

MPA features unsuitable for *W. subatra*

- A2.3 Littoral mud – These habitats lack suitable attachment opportunities (high confidence).

No evidence for W. subatra in these MPA habitats

- A5.7 Carbonate reefs.

Watersipora subatra: Socio-Economic Impacts

Human health and safety: No impacts are known.

Aquaculture Operations: It has been recorded as a pest species for biofouling on artificial substrates such as boat hulls and aquaculture equipment (Bishop, 2015 and references therein; Cohen, 2011). Biofouling is thought to have a global economic impact of billions of dollars (McLachlan, 2017) so this impact pathway has been assessed as ‘Moderate’ with low confidence.

Mussels and Oysters: It is reported to often encrust loosely on mussels with the apparent potential to affect the aquaculture industry. It is thought to be spread through the movement of shellfish within the aquaculture industry (Needles *et al*., 2015; Bishop, 2015 and references therein). It is possible that it may foul oysters also. Little is known about its impacts on mussels, and similarly oysters, but it could hinder their development and overall health so this impact pathway has been assessed as ‘Moderate’ with low confidence.

Fisheries Operations: This species is a biofouler that may have negative economic impacts on artificial substrates including fishing equipment. Hull fouling is thought to be a major source of translocation for this species (Bishop, 2015 and references therein; Cohen, 2011; Wilson, 2017). It has also been suggested that the movement of shellfish by the fishing industry may have allowed this species to invade non-native ranges. For example, it has been found growing in locations around the British Isles close to oyster culture operations (Mackie *et al*., 2006; Ryland *et al*., 2009; Bishop, 2015 and references therein; Cohen, 2011).

The negative fouling impacts on boat hulls are well documented (Bishop, 2015 and references therein; Wilson, 2017). Its tolerance of copper and mercury anti-fouling paints makes it a particular nuisance (Piola & Johnston, 2006; Wilson, 2017; Fofonoff *et al*., 2018). Fouling of boat hulls is known to reduce speed, efficiency, upkeep and maintenance (McLachlan, 2017). Its ability to resist some anti-fouling paints exacerbates fouling issues. It becomes the primary fouling organism on anti-fouled structures thus providing attachment substrate for other fouling and motile species (Floerl *et al.,* 2004). This ability to foul boat hulls and facilitate colonisation by other species could also facilitate the spread of other non-native species (Floerl *et al.,* 2004).

It may be that its ability to foul natural substrate could impact local biodiversity and fisheries. Unlike its documented negative impacts upon artificial structures any impacts it may have on natural habitats have still to be investigated (Tidbury, 2015 and references therein). Possible impacts have been assessed as ‘Minimal concern’ with low confidence.

Oysters-*Ostrea edulis*, *Magallana gigas*: See ‘Mussels and Oysters’ above. This impact pathway has been assessed as ‘Moderate’ with low confidence.

Watersipora subatra: References

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Annex 9 Bonnemaison’s hook weed: *Bonnemaisonia hamifera*

Common name(s): Hook weed; Pink cotton wool; Bonnemaison’s hook weed

Synonyms: *Asparagopsis hamifera; Trailliella intricata* (Guiry & Guiry, 2019).

Domain: Phylum: Rhodophyta, Class: Florideophyceae, Order: Bonnemaisoniales, Family: Bonnemaisoniaceae, Genus/species: *Bonnemaisonia hamifera* (Guiry & Guiry, 2019).

Description: *Bonnemaisonia hamifera* is a red macroalga growing up to 30 cm high. It is branched with an erect main axis of 1 mm in diameter. Branches have distinctive curved hooks protruding from them. Branches are arranged opposite (in pairs) and spirally with one branch of each pair being slightly longer than the other. It also has a small (2-3 cm) filamentous, tetrasporophyte phase known as ‘*Trailliella'* (Sweet, 2011).

A

B

(A) © Michiel Vos anbollenessor.com & (B) © John Bishop, MBA)

Figure 9.1 Bonnemaison’s hook weed: *Bonnemaisonia hamifera*

Bonnemaisonia hamifera: Habitat

Native range: *B. hamifera* originates in the Pacific and was probably introduced from Japan (Eno *et al*., 1997).

Substratum type: The adult gametophyte phase of *B. hamifera* occurs in lower littoral tidal pools (Guiry & Guiry, 2020; Morton & Picton, 2016) and shallow subtidal habitats. It grows predominantly epiphytically on macroalgae e.g. *Cystoseira* spp. using its characteristic hooks to attach (Gollasch, 2009; Sweet, 2011; Morton & Picton, 2016), and on maerl beds in Scotland (Bunker *et al*., 2018).

The juvenile tetrasporophyte phase, '*Trailliella',* occurs in the intertidal zone on the lower shore in shaded rock pools (Sweet, 2011). This phase has, in the past, been thought to be a separate species, *Trailliella intricata.* It has been found growing epiphytically on *Corallina* spp. (Guiry & Guiry, 2020), *Zostera marina* and *Ruppia maritima* (Fofonoff *et al*., 2018; Johnson *et al*., 2005), and also on maerl beds in Scotland (Bunker *et al*., 2018) and the NE Atlantic (Pena *et al*., 2014). This phase of *B. hamifera* may also occur in on rocks or commonly as a key element within the 'infralittoral muddy gravel' biotope, widely found in lagoons and sea lochs, where the mud is often gravelly with cobbles and can be black and anoxic close to the surface. There it often forms a continuous mat of '*Trailliella'* that is regularly 10 cm thick and can be as much as 100 cm thick at some sites. The quantity can become so overwhelming that it has the ability to modify the environment (Connor *et al*., 2004).

Gollasch (2009) suggests that it occurs in both its native and non-native ranges on sublittoral rock and other hard substrata equating to EUNIS code A3. Gollasch states that adults are exclusively epiphytic while the '*Trailliella'* form is occasionally found on hard substratum (Fofonoff *et al*., 2018) including artificial (Gollasch, 2009).

Salinity: Leidenburger *et al*. (2015) reported that *B. hamifera* was found at sea surface salinities of between 14.26 and 37.55 psu (based on satellite data and distribution).

Depth: The gametangial phase (*B. hamifera)* is found in shallow sublittoral or occasionally low littoral habitats, especially in tidepools and occurs down to 20 m (Morton & Picton, 2016; Sweet, 2011). The tetrasporangial '*Trailliella*' phase occurs from the lower littoral to sublittoral habitats to 8 m depth (Morton & Picton, 2016).

Wave exposure: The **'***Trailliella' phase* of this alga prefers very sheltered conditions (Connor *et al*., 2004).

Bonnemaisonia hamifera: Establishment in Wales as of 2020

B. hamifera has been recorded from the Gower to Anglesey (NBN Atlas, 2017). The *'Trailliella'* phase was recorded growing on *Zostera* blades (Johnson *et al*., 2005). Its reproduction is restricted by temperature around Britain (Connor *et al*., 2004; Breeman *et al*., 1988) although it is now widely distributed on southern and western coasts of Britain as far north as the Shetland Isles.

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Bonnemaisonia hamifera: Impact pathways based on species traits, biology and ecology

Competition: While *B. hamifera* could potentially compete with other algae and seagrasses (Gollasch, 2009), in experiments *B. hamifera* showed a relatively slow growth rate and did not alter community biomass production rates (Sagerman *et al*., 2014). There is very little evidence in the literature of instances of competition with other algae and this pathway is assessed as ‘Minimal concern’ at low confidence. Maerl, however, has been assessed as of ‘Moderate concern’ at low confidence. The assumption being that maerl is a very slow growing species and any competition with *B. hamifera* could result in overgrowth of the maerl.

Predation: *B. hamifera* is a primary producer so this impact pathway is considered ‘Not applicable’.

Hybridisation: *B. hamifera* is not known to hybridize with other species and this impact pathway is assessed as ‘Not applicable’.

Transmission of disease: *B. hamifera* is not known to transmit diseases therefore this pathway has been assessed as ‘Not applicable’.

Parasitism: *B. hamifera* is not known to be parasitic and this impact pathway has been assessed as ‘Not applicable’.

Poisoning/toxicity: It has been found that *B. hamifera* has natural anti-fouling properties in the form of a secondary metabolite, 1,1,3,3-tetrabromo-2-heptanone, occurring on its surface tissues that inhibits bacterial fouling when compared to surrounding red algae (Nyland *et al*., 2008). Since this substance is a deterrent rather than a poison this pathway is assessed as ‘Not applicable’.

Bio-fouling: *B. hamifera* is not a bio-fouler so this impact pathway is considered ‘Not applicable’.

Grazing/herbivory/browsing: *B. hamifera* is a primary producer so this impact pathway is considered ‘Not applicable’. However, grazing species may be affected by establishment of this species (see Indirect impacts below).

Indirect impacts through interactions with other species: Enge *et al*. (2013) experimentally noted that native herbivores found significantly better refuge from fish predation amongst the invasive *B. hamifera* when compared with native algae based on its unpalatability. The herbivores reduced native algae through grazing while boosting the growth of *B. hamifera.* This study provides evidence of herbivores shifting community structure towards dominance by a well- defended invasive alga by means of a refuge-mediated apparent competition which negatively impacts native algae (Enge *et al*., 2013).

In experiments, the assessment of survival and growth of a native generalist herbivore confirmed that the biomass produced by *B. hamifera* constitutes a very low-quality food (Sagerman, 2014). Establishment of *B. hamifera* may alter the structure of biological assemblages through impacts on grazers that ramify to higher trophic levels. However, since there is limited evidence to support these findings from the wild this impact pathway has been assessed as ‘Data deficient’

Bonnemaisonia hamifera: Impact pathways – Habitats

Chemical impact on ecosystem: *B. hamifera* is not known to have any chemical impacts on the ecosystem. Smothering by mats of the tetrasporophyte phase could induce anoxia in underlying sediments altering sediment chemistry and suitability for infauna resulting in changes to the characteristic community. A relatively recent review of the ecology of *B. hamifera* (Tyler-Walters, 2016) and searches undertaken for this project found no evidence for impacts of *B. hamifera* on native species. This pathway is assessed as ‘Data deficient’.

Physical impact on ecosystem: ‘*Trailliella’,* the tetrasporophyte phase of *B. hamifera* has the ability to form very large, dense loose-lying mats. Connor *et al*. (2004), in the biotope description, explains that these dense mats of up to 100 cm thick, through their sheer abundance, can modify the environment. What modifications can occur and how is not described. Dense mats will prevent light reaching the sediment. These changes would alter habitat suitability for epiflora such as sediment diatoms. A relatively recent review of the ecology of *B. hamifera* (Tyler-Walters, 2016) and searches undertaken for this project found no evidence for impacts of *B. hamifera* on native species. This pathway is assessed as ‘Data deficient’.

Structural impact on ecosystem: Widely found in lagoons and sea lochs in its tetrasporophyte phase, often as a continuous mat of dense loose-lying algae. Mats can be 10-100 cm thick at some sites. These vast quantities can modify the environment (Connor *et al*., 2004). Dijkstra *et al*. (2017) found that introduced algae species to Maine, USA (including *B. hamifera*) were more morphologically complex than native species. This provided more structurally complex habitats for meso-invertebrates which were found in higher abundance on non-native algae when compared to native species. Whilst there is evidence of positive impacts on habitat structure and biodiversity there is no evidence in the literature suggesting any negative effects caused by *B. hamifera* (Dijkstra *et al*., 2017). Therefore, this pathway has been assessed as ‘Data deficient’.

Bonnemaisonia hamifera: Interactions with MPA Features

MPA features that provide suitable habitat *Bonnemaisonia hamifera*

The MPA features and biotopes below are known to provide suitable habitat for *B. hamifera* since it is a characterising species of constituent biotopes (Connor *et al*., 2004) which is why they have all been scored with high confidence.

- A3.3 Atlantic and Mediterranean low energy infralittoral rock;
- Tide-swept channels: A3.223; and

- A5.5 Sublittoral macrophyte-dominated sediment:
    - A5.526 'Trailliella' phase.

The MPA features below have been recorded in Scotland and/or NE Atlantic with *B. hamifera* growing on them which is why they have been scored with a high confidence.

- A2.6 Littoral sediments dominated by aquatic angiosperms;
- Seagrass beds;
- *Zostera* beds; and
- Maerl beds.

The MPA features below have been assessed as suitable for *B. hamifera* based on depth, suitable mixed attachment substrate and reduced salinity lagoon habitat. This assessment has been scored with medium confidence.

- A5.4 Sublittoral mixed sediments:
    - A5.41; A5.43 (except A5.434).

The MPA features below has been assessed as suitable for *B. hamifera* based on depth and the sheltered, anoxic muddy habitat. (medium confidence).

- Mud habitats in deep water:
    - A5.7211.

MPA habitats considered potentially suitable for *Bonnemaisonia hamifera*

The MPA features below provide potentially suitable habitat for *B. hamifera* based on substratum, depth and its association with other macroalgae species. They have all been scored with medium confidence.

- Estuarine rocky habitat;
- Tide-swept channels:
    - A3.22; A1.15;
- A2.7 Littoral biogenic reefs:
    - A2.72; and
- A5.5 Sublittoral macrophyte - dominated sediment:
    - A5.52.

The MPA features and sub-biotopes below provide potentially suitable habitat for *B. hamifera* based on their association with *Corralina* spp. and/or sheltered tidal pools. However, since the evidence is limited as to where *B. hamifera* may establish in these MPA features they have been scored with low confidence.

- A1.1 High energy littoral rock:
    - A1.123; A1.124 ; A1.125; – tidal pools; and
- Intertidal underboulder/boulder communities - *Corralina* spp. and tidal pools.

The MPA features below provide potentially suitable habitat for *B. hamifera* based on suitable attachment substratum in suitable habitat. All these habitats have been scored with medium confidence apart from 'Atlantic and Mediterranean moderate energy infralittoral' which has been scored as high confidence. The latter was scored in this way since one of its biotopes has *B. hamifera* as a characterising species within it and there is good potential for the others.

- A1.3 Low energy littoral rock:
  - A1.31; A1.32 on the lower shore;
- A3.2 Atlantic and Mediterranean moderate energy infralittoral rock;
- A5.1 Sublittoral coarse sediment:
  - A5.13;
- Blue mussel beds;
- Intertidal *Mytilus edulis* beds on mixed and sandy sediments;
- Subtidal mixed muddy sediments:
  - A5.445;
- Sheltered muddy gravels;
- A2.4 Littoral mixed sediments;
- *Ostrea edulis* beds; and
- Peat and clay exposures.

The MPA features below provide potentially suitable habitat for *B. hamifera* based on shallow, muddy habitats. It is difficult to predict weather *B. hamifera* will establish in these habitats which is why the confidence has been scored as medium.

- A5.3 Sublittoral mud:
  - A5.31; A5.32; A5.33; A5.34;
- Mud habitats in deep water:
  - A5.7221; and
- A2.3 Littoral mud.

MPA habitats considered unlikely to be suitable for *Bonnemaisonia hamifera*

The MPA features below are considered unlikely to be suitable based on exposure. These assessments have been scored with medium confidence as the evidence suggests both phases of *B. hamifera* establish in sheltered conditions.

- Tide-swept channels:
  - A5.5211- this biotope has been assessed as unlikely to be suitable habitat based its tide-swept nature;
- A3.1 Atlantic and Mediterranean high energy infralittoral rock;
- A5.4 Sublittoral mixed sediments:
  - A5.434.

The MPA features below are considered unlikely to be suitable based on lack of suitable algal attachment substrate for the adult phase and the mobile nature of these habitats. The assessments below have been scored as low confidence as the non-epiphytic *'Trailliella'* phase may establish here, although this is thought unlikely.

- A5.2 Sublittoral sand:
    - A5.24; A5.22.

MPA features unsuitable for *Bonnemaisonia hamifera*

The MPA features below are considered too deep for *B. hamifera* to establish in. All these habitats have been scored with high confidence.

- A4.1 Atlantic and Mediterranean high energy circalittoral rock;
- A4.2 Atlantic and Mediterranean moderate energy circalittoral rock;
- A4.3 Atlantic and Mediterranean low energy circalittoral rock;
- Fragile sponge and anthozoan communities on subtidal rocky habitats;
- A5.1 Sublittoral coarse sediment:
    - A5.14; A5.15;
- A5.2 Sublittoral sand:
    - A5.25; A5.27;
- A5.6 Sublittoral biogenic reefs:
    - A5.61; A5.62; A5.63;
- *Modiolus modiolus* beds;
- A5.3 Sublittoral mud:
    - A5.35; A5.36;
- *Musculus discors* beds;
- Subtidal mixed muddy sediments:
- A5.45;
- Mud habitats in deep water;
    - A5.35; A5.36; A5.37 (except A5.7221);
- Sea-pen and burrowing megafauna communities;
- Tide-swept channels:
    - A4.251; A4.2511; A4.2512; A4.1122; A4.1121; A4.112; A4.111; and
- A5.4 Sublittoral mixed sediments:
    - A5.45.

The MPA features below are considered too exposed to wave action for *B. hamifera* to establish in. All these habitats have been scored with high confidence.

- A1.1 High energy littoral rock; and
- A1.2 Moderate energy littoral rock.

The MPA features below are considered too mobile in nature and/or too high up on the shore for *B. hamifera.* All these habitats have been scored with high confidence.

- A2.5 Coastal saltmarshes and saline reedbeds;

- A2.1 Littoral coarse sediment; and
- A2.2 Littoral sand and muddy sand.

No evidence for *Bonnemaisonia hamifera* in these MPA features

- A2.7 Littoral biogenic reefs: A2.71; A2.711;
- A5.5 Sublittoral macrophyte-dominated sediment:
    - A5.51; A5.53; A5.54;
- A2.3 Littoral mud:
    - A2,324; A2.325; A2.3251;
- A2.7 Littoral biogenic reefs;
- *Sabellaria alveolata* reefs;
- A5.5 Sublittoral macrophyte-dominated sediment:
    - A5.51; A5.53; A5.54; and
- A5.7 Carbonate reefs.

Summary of the impacts on the MPA features

There is no evidence for impact on MPA features. MPA features most at risk are maerl beds due to their very low growth and recovery rates (Perry *et al*., 2018) marlin). Other features that may be impacted include littoral rock and seagrass beds. MPA features containing macroalgae and seagrass are of minimal concern due to current evidence of co-existence and slow growth rates of *B. hamifera.* Mats of '*Trailliella'* could cause anoxia and other environmental modifications where they occur. However, there is no evidence of this which represents a key uncertainty with this species and any impacts it may have. This is why habitats where it may occur have been assessed as 'Data deficient'.

Bonnemaisonia hamifera: Socio-Economic Impacts

Health and safety: No impacts are known.

Aquaculture Operation: Slow growth (Sagerman *et al.*, 2014) and lack of evidence for establishment on artificial substratum indicates the same would be true for aquaculture operations. No evidence was found for impacts on aquaculture operations.

Aquaculture cultivated species: No evidence was found to suggest any interactions with aquaculture cultivated species.

Fisheries Operation: Intertidal and shallow sublittoral rock, not suitable for mobile gears and static gears are more likely to be deployed at greater depths than *B. hamifera* occurs.

Fisheries Target species: No evidence was found to suggest any interactions with target species.

Hand collection: The collection of seaweed for the purposes of making laver bread could be impacted if *B. hamifera* becomes too abundant. It may be more difficult to find and collect the algae (laver) required for the making of laver bread.

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Annex 10 Japanese skeleton shrimp: *Caprella mutica*

Common name(s): Japanese Skeleton shrimp; ghost shrimp (Cook, 2019).

Synonyms: *Caprella macho* (Horton *et al.,* 2010).

Domain: Phylum: Arthropoda, Class: Malacostraca, Order: Amphipoda, Family: Caprellidae, Genus/species: *Caprella mutica*.

Description: A large skeleton shrimp (a caprellid amphipod) up to 3.5 cm in length with the females being smaller (~1.5 cm) (Oakley, 2006). Adult males have two elongate body segments covered in fine hairs, while there are large dorsal spines on body segments three to seven. Females also have dorsal spines, and have orange spots on their brood pouch; both sexes are bright orange to red when adult (Oakley, 2006) but more translucent as juveniles (GBNNSS, 2011 and references therein).

(Image by Alchetron).

Figure 10.1: Japanese skeleton shrimp: *Caprella mutica*

Caprella mutica: Habitat

Native range: *Caprella mutica* is indigenous to north-east Asia, but has been very widely distributed across biogeographic regions on the Pacific and Atlantic coasts of the USA, Alaska, Canada, across Europe from Belgium to Norway, including the UK and Ireland. It has not been found in the Baltic or the Mediterranean to date (Cook et al. 2007); in Britain there are records from sites throughout the Risk Assessment Area, although it is not found in sites which experience salinities of < 15 psu (GBNSS, 2011).

Substratum type: *C. mutica* is found on a range of substrata (Oakley, 2006) and associates with soft fouling organisms in its invasive ranges (Boos *et al.,* 2011; Coolen *et al*., 2006). Preferred habitats include fine filamentous structures such as hydroids (Ashton, 2006; Cook *et al.*, 2007 and references therein), foliose surfaces of macroalgae and turf- like bryozoans that they can grab hold of rather than hard substrates like bivalves (Boos *et al.,* 2011 and references therein). It is also found attached to drifting algae, in particular *Sargassum muticum* (Oakley, 2006).

C. mutica is often found on artificial structures such as mooring buoys (Kerckhof *et al*., 2007) marinas, aquaculture sites and harbours (Ashton, 2006). It has been found on off-shore windfarms and oil platforms (Coolen *et al*., 2006) while on the west coast of Scotland it has been found living on mussel and salmon farm infrastructure (Ashton, 2006). Its occurrence in exceptionally high densities on artificial structures such as pontoons and aquaculture infrastructure, which are raised off the seabed, enables the species to avoid benthic predation pressure (Cook *et al*., 2016). While the species has been found occurring on natural coastlines subtidally (K. Boos, pers. comm. In: Cook *et al*., 2007) and intertidally (J. Bishop, pers. comm.), they do not seem to become abundant in these habitats.

Salinity: Optimum 18-35 ppt (100% mortality in salinity <16 ppt and high mortalities >40 ppt) (Ashton *et al*., 2007a; Cook, 2019 and references therein). *C. mutica* becomes sluggish < 18 ppt and cannot survive < 16 ppt salinity (Ashton *et al*., 2007a) so will not be found in brackish waters such as estuaries (Ashton, 2006; Ashton *et al*., 2007a).

Depth: The species is found in the littoral zone down to 13 m in its native range (Boos *et al.,* 2011 and references therein) and may occur attached to the submerged underside of floating artificial substrates in the littoral zone in the UK (GBNNSS, 2011 and references therein; Bishop, pers. comm.). It also occurs sublittorally to 20m depth (Boos *et al*., 2011; GBNNSS, 2011; Cook, 2019, Coolen *et al*., 2016).

Wave exposure: In its native range, *C. mutica* is typically reported from sheltered bays while non-native populations of *C. mutica* have been recorded from environments with a variety of flow regimes, including those experiencing strong tidal and wind currents (e.g., exposed fish farms) and those that are more sheltered (e.g., enclosed bays and harbours) (Ashton, 2006; Shucksmith, 2007).

Caprella mutica: Establishment in Wales as of 2020

There are records from Holyhead, Anglesey (NBN Atlas, 2017).

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Caprella mutica: Impact pathways based on species traits, biology and ecology

Competition: *C. mutica* have been recorded living in close proximity to and under the same environmental conditions as native caprellids. It is possible that *C. mutica* will have some niche overlap with the common British native caprellid *Caprella linearis* (Cook, 2019 and references therein).

Aquarium trials show *C. mutica* to be aggressive and competitive towards native skeleton shrimps by displacing them from substrate even at low densities (Shucksmith *et al*., 2009). It is possible their large size gives them a competitive advantage. *C. linearis* and *C. mutica* share the same preference for fine, filamentous habitat like hydroids and have been observed competing for space in the wild as well as in aquaria (Cook, 2019 and references therein). The hostile behavior was only observed in structurally simple habitat. In more structurally diverse habitats *C. linearis* was found to shelter away from *C. mutica* in refuges provided and the two species co-occurred peacefully (Shucksmith *et al*., 2009). Shucksmith *et al*. (2009) found that only low numbers of *C. mutica* were needed to displace *C. linearis*. However, once *C. linearis* reached low density numbers the aggressive, displacement behavior of *C. mutica* was restricted. It is possible that *C. linearis* may not be outcompeted but rather left to co-exist alongside *C. mutica* but in low numbers (Shucksmith *et al*., 2009). However, Coolen *et al*. (2016) modelled habitat preferences in the southern North Sea for both caprellid species and suggests there would be some habitat separation as well as a niche overlap. *C. linearis and C. mutica* niche overlapped in suitable near-surface/intertidal habitats while only *C. linearis* could occupy the deeper subtidal habitats.

Turcotte (2010) found that *C. mutica* competed with mussels for food (microalgae) on mussel lines to the detriment of the mussels (see: ‘Aquaculture cultivated species-Mussels’ below). Both of these examples of caprellid competition have been assessed as ‘Major’ with medium confidence.

Predation: *C. mutica* is an omnivore that can utilise a range of feeding strategies (Cook *et al*., 2007); while zooplankton may form part of its diet it is not an active predator of macrofauna and this impact pathway is considered to be ‘Not applicable’.

Hybridisation: No evidence was found for hybridisation and this impact pathway is assessed as ‘Not applicable’.

Transmission of disease: No records of disease transmission by *C. mutica* were found in the literature and this impact pathway is considered to be ‘Not applicable’.

Parasitism: *C. mutica* is not a parasite and this impact pathway is assessed as ‘Not applicable’.

Poisoning/toxicity: *C. mutica* is not known to be toxic or poisonous and this impact pathway is assessed as ‘Not applicable’.

Bio-fouling: *C. mutica* is found in large numbers on artificial substrates (Boos *et al*., 2011 and references therein) as well as natural substratum like macroalgae and hydroids (Ashton, 2006; Cook *et al.*, 2007). In its invasive range it is mainly associated with artificial substratum, in areas such as marinas, harbours and aquaculture sites (Boos *et al.,* 2011 and references therein). Once established its numbers can explode, reaching up to 300,000 individuals/ m^2 (Ashton, 2006). In high densities like this it has been recorded blocking water pump intakes attached to caged fish feeding systems (Boos *et al.,* 2011 and references therein). It has not been recorded fouling natural substratum around the UK and has been assessed as ‘Minimal concern’ with high confidence.

Grazing/herbivory/browsing: *C. mutica* is an omnivore that can utilise a range of feeding strategies (Cook *et al.,* 2007). It is often termed a filter feeder where phytoplankton and plant detritus may form part of its diet. In the laboratory it has been observed scraping the surface of macroalgae (NOBANIS, no date, and references therein). Since there is nothing in the literature to suggest that its grazing habit has any impact on its environment this pathway is ‘Not applicable’.

Indirect impacts through interactions with other species: *C. mutica* is mainly found on artificial structures at present. Assuming this will not change then the impact on benthic biodiversity and native species will remain low (Cook, 2019 and references therein). High densities, up to 300,000 individuals/ m^2, have been recorded in the summer months and high feeding rates could have an impact on plankton biodiversity (Ashton, 2006) although the scale of this impact at present is not known (Cook, 2019 and references therein). *C. mutica* has also been found to reduce recruitment of the invasive tunicate *Ciona intestinalis* around Prince Island, Canada (Collin & Johnson, 2014). The exact mechanisms behind these interactions are unclear although it is proposed to be through predation, predator avoidance behavior, settlement interference or a mixture of these (Collins & Johnson, 2014). As there is little evidence to suggest these impacts are affecting the natural environment around the UK this impact pathway has been assessed as ‘Not applicable’.

Caprella mutica: Impact pathways – Habitats

Chemical impact on ecosystem: There is no evidence to suggest that *C. mutica* has a chemical impact on the ecosystem and this impact pathway is assessed as ‘Not applicable’.

Physical impact on ecosystem: It is possible that this species could impact light penetration to fouling communities when they attach to artificial substrates in large densities (Ashton, 2006; Cook *et al*., 2007). Their numbers have been recorded up to 300,000 individuals/ m^2 (Ashton, 2006). However, since they are almost exclusively found fouling artificial structures in their non-native ranges (Cook, 2019 and references therein; Boos *et al*., 2011 and references therein; Ashton, 2006), they are unlikely to have a physical impact on the MPA features. Therefore, this impact pathway has been assessed as ‘Not applicable’.

Structural impact on ecosystem: While *C. mutica* can have an impact on artificial substratum by fouling it is unlikely to foul any of the MPA features and this impact pathway is assessed as ‘Not applicable’.

Caprella mutica: Interactions with MPA Features

MPA features that provide suitable habitat for *Caprella mutica*

None assessed as suitable for establishment of reproducing populations. *C. mutica* is found to establish in the UK on artificial structures and has been recorded within natural habitats but it is unclear if individuals have been displaced by water movements or other transport vectors.

MPA habitats considered potentially suitable for *Caprella mutica*

The MPA features below provide potentially suitable habitat for *C. mutica* based on suitable fouling community, presence of macroalgae, suitable attachment structures like hydroids and environmental tolerances. They have been scored with low confidence.

- A3.1 Atlantic and Mediterranean high energy infralittoral rock;
- A3.2 Atlantic and Mediterranean moderate energy infralittoral rock;
- A3.3 Atlantic and Mediterranean low energy infralittoral rock;
- *Musculus discors* beds;
- *Modiolus modiolus* beds:
  - A5.621, A5.623, A5.624;
- Maerl beds;
- A5.5 Sublittoral macrophyte-dominated sediment;
- A5.3 Sublittoral mud:
  - A5.33, A5.34;
- *Sabellaria alveolata* reefs:
  - A5.612; and
- Sheltered muddy gravels:
  - A2.42.

The MPA features below provide potentially suitable habitat for *C. mutica* in the shallower areas of these habitats (< 13m). They have been scored with low confidence.

- A4.1 Atlantic and Mediterranean high energy circalittoral rock (except A4.133 found in Scottish lochs);
- A4.2 Atlantic and Mediterranean moderate energy circalittoral rock (except A4.211 found mainly in Scotland and Ireland);
- A4.3 Atlantic and Mediterranean low energy circalittoral rock;
- Fragile sponge and anthozoan communities on subtidal rocky habitats (except A4.211, A4.2111, A4.2112, A4.133 found mainly in Scotland);
- Tide-swept channels;
- A5.4 Sublittoral mixed sediments;
- A5.6 Sublittoral biogenic reefs:

    - A5.61, A5.62; and
- *Ostrea edulis* beds.

The MPA feature below may provide potentially suitable habitat based on the presence of *Caprella linearis* (native caprellid). *C. mutica* has been recorded living alongside native caprellids. (Confidence is medium).

- Subtidal mixed muddy sediments:
    - A5.421.

MPA habitats considered unlikely to be suitable *Caprella mutica*

The MPA features below are unlikely to provide suitable habitat for *C. mutica* since they are littoral habitats. *C. mutica* is unlikely to establish in the littoral zone unless on submerged surfaces of artificial structures like pontoons and mooring buoys. These have been scored medium confidence.

- A1.1 High energy littoral rock;
- A1.2 Moderate energy littoral rock;
- A1.3 Low energy littoral rock;
- Intertidal underboulder/boulder communities;
- A2.1 Littoral coarse sediment;
- A2.4 Littoral mixed sediments;
- Littoral sediments dominated by aquatic angiosperms:
    - A2.711;
- A2.2 Littoral sand and muddy sand;
- A2.3 Littoral mud;
- A2.7 Littoral biogenic reefs;
- *Sabellaria alveolata* reefs:
    - A2.711;
- Intertidal *Mytilus edulis* beds on mixed and sandy sediments;
- Blue mussel beds; and
- Peat and clay exposures.

The MPA features below are unlikely to provide suitable habitat for *C. mutica* based on either fluctuating salinities; low salinities; depth; mobile/scouring environment and sparse/lack of suitable turf community and attachment opportunities.

- A5.6 Sublittoral biogenic reefs:
    - A5.63 (medium confidence);
- Sea-pen and burrowing megafauna communities (low confidence);
- Mud habitats in deep water (low confidence);
- A5.1 Sublittoral coarse sediment (low confidence);
- A5.2 Sublittoral sand (low confidence);

- A5.3 Sublittoral mud (low confidence);
- Sheltered muddy gravels:
    - A2.41(medium confidence); and
- Subtidal mixed muddy sediments (except A5.421) (low confidence).

MPA features unsuitable for *Caprella mutica*

The MPA features below are unsuitable habitat for *C. mutica* based on salinity tolerances and/or depth. (Medium confidence).

- A2.5 Coastal saltmarshes and saline reedbeds;
- Estuarine rocky habitat; and
- *Modiolus modiolus* beds:
    - A5.622.

No evidence for *Caprella mutica* in these MPA features

- A5.7 Carbonate reefs;
- A2.6 Littoral sediments dominated by aquatic angiosperms;
- Seagrass beds; and
- *Zostera* beds.

Summary of the impacts on the MPA features

C. mutica inhabits artificial structures in its UK non-native range (Ashton, 2006; Boos *et al*., 2011 and references therein). This may include aquaculture gear, pontoons, buoys, marinas and pilings (Ashton, 2006). *C. mutica* has been assessed as unlikely to impact any of the MPA features, impacts are assessed as 'Minimal concern' at high confidence.

Caprella mutica: Socio-Economic Impacts

Human health and safety: No impacts are known

Aquaculture operations: *C. mutica* has been recorded in high densities, up to 300,000 individuals/ m^2 (Ashton, 2006) in the summer months of May-September (Boos *et al*., 2011 and references therein) blocking water pump intakes attached to caged fish feeding systems causing economic impacts. Their numbers reportedly die back again in the colder winter months (Fedotov, 1991) although juveniles have been reported on fish cages in Scotland throughout the year (Ashton, 2006). Impacts have been assessed as 'Major' at high confidence for caged aquaculture operations.

C. mutica are recorded settling on mussel lines taking up valuable space for mussel spat (Turcotte, 2010). This fouling behavior has an economic cost associated with the removal of this species as well as any loss of utility (Boos *et al*., 2011 and references therein). It has been reported to be transported between aquaculture sites via stock movements of oysters and on the boats that shuttle between sites (Ashton *et al*., 2007b; Cook *et al*., 2007). Impacts have been assessed as 'Major' at medium confidence for shellfish aquaculture operations.

Aquaculture cultivated species- Mussels and Oysters: Turcotte (2010) demonstrated experimentally that *C. mutica* on mussel spat collectors in Canada reduced the nutritional status of the *Mytilis* spat, explaining observed reductions in their growth. Turcotte (2010) hypothesized the reduction in mussel growth was related to reduced feeding by the mussels, and suggested two possible mechanisms, which were not mutually exclusive. One was competition with caprellids for the common food resource, phytoplankton, where *C. mutica* had advantage by its positioning on top of (upstream in the feeing current of) the mussels. The second suggested mechanism was interference with mussel feeding by the caprellids trampling over the mussels causing them to close up and stop feeding. It is possible that *C. mutica* could compete with oyster spat in a similar way. Impacts have been assessed as 'Major' at medium confidence.

Fisheries operations: No specific evidence was found for direct impacts on fishing operations. *C. mutica* are unlikely to be retained in mobile gear meshes due to small size. Impacts have been assessed as 'Major' at medium confidence.

Target Species- Finfish: High densities of *C.mutica* are known to have high feeding rates, climaxing in the summer months (Fedotov, 1991). Impact on plankton is unknown (Cook, 2019 and references therein) although it is possible they may impact fish larvae, and therefore fisheries, especially since their feeding rate remains constant day and night (Cook, 2019 and references therein). Caprellids are opportunistic feeders, and *C. mutica* is no different being shown to consume non-living fish farm-derived particulate material, phytoplankton and bacteria (Cook *et al*., 2007). Impacts on finfish are assessed as 'Minimal concern' at low confidence.

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Annex 11 Pacific Oyster: *Magallana gigas*

Common name(s): Pacific Oyster

Synonyms: *Crassostrea gigas* (alternate representation)
There is currently some debate over whether or not *Magallana angulata* (the Portuguese oyster) is in fact the same species. Whilst the two species are often considered synonomous (e.g. Reece *et al*., 2008), other research suggests that both are separate species of Asian origin (Gagnaire *et al*., 2018).

Domain: Phyla: Mollusca, Class: Bivalvia, Order: Ostreoida, Family: Ostreidae, Genus/species: *Magallana gigas.*

Description: In wild non-native populations the shell length can exceed 20 cm. An off-white to yellow or bluish grey in colour, the shell often has deep purple patches (see Fig. 8.1). The left valve is deeply cupped with 6 or 7 bold ribs making the shell margin rough. The right valve is flat or slightly convex and has ribs corresponding to channels of left valve (Hughes, 2008).

(Image © C. Wood and J. Bishop)

Figure 11.1. Pacific oyster: *Magallana gigas* on rock and mixed sediments in the river Yealm, Devon.

Magallana gigas: Habitat

Native range: The native range of distribution of *M. gigas* extends from Cape Mariya (Russia) to Hong Kong (China). Wild populations of the species are present in locations with predicted sea surface temperatures between.14.0 and 28.6C for the warmest month of the year, and between -1.9 and 19.8C for the coldest month (Carrasco and Barón, 2010).

Substratum type: Larvae require some hard substrate but can settle on small items such as shells in otherwise fine sediments. *M. gigas* is typically found on rock, concrete artificial structures or shells and stone (Spencer *et al*., 1994; Wrange *et al*., 2010; Kochmann *et al*., 2013).

Intertidal mudflats, sandflats, intertidal biogenic reefs (*Mytilus edulis; Sabellaria alveolata*) and intertidal rock (GBNNSS, 2019) are commonly colonised. However, a survey of Poole Harbour found that *M. gigas* were absent from sandflats and areas where there was strong competition with macroalgae and intertidal areas that were relatively steep in gradient (McKinstry and Jensen, 2013). Preferred habitats were areas with shallow gradients and mixed substrate containing mud, shingle and shell (McKinstry and Jensen, 2013). Intertidal width appears to be an additional factor determining suitability. All of the Pacific oysters found during the Poole Harbour surveys were located on shores with a shallow gradient and so relatively expansive intertidal area. Kochman *et al*. (2013) found a positive correlation between the width of the intertidal area and oyster presence, with a greater likelihood of oyster presence being recorded on extensive, intertidal shores (> 50 m). Similarly, in the Solway Firth *M. gigas* were all found at sites where the intertidal width was greater than 40 m (Cook *et al*., 2014).

Exposed bedrock and large boulders which were devoid of macroalgae, appeared to provide suitable habitat for wild *M. gigas* in the Solway Firth region. Substrate with dense macroalgal coverage was deemed unsuitable for the settlement and/or establishment of wild *M. gigas* (Cook *et al*., 2014) and surveys rarely found *M. gigas* under macroalgae cover (Kochman *et al*., 2013).

Artificial hard substratum such as groynes and walls and oyster growing racks may be colonised (McKinstry and Jensen, 2013; Cognie *et al*., 2006; Fey *et al*., 2010).

Salinity: *M. gigas* is tolerant of a wide range of salinity (Chu *et al*., 1996) but highest densities of *M. gigas* are found at salinities >20psu. Larvae can survive salinities between 19-35 psu (Troost, 2010 and references therein).

Depth: *M. gigas* are typically found in the intertidal and shallow subtidal (Wrange *et al*., 2010). Most studies have focused on intertidal populations of *M. gigas* and the extent of recruitment and colonisation of sublittoral habitats is currently unclear (Dolmer *et al*., 2014; Herbert *et al*., 2016). *M. gigas* may be extending its range subtidally in invaded areas (Dolmer *et al*., 2014) and Herbert *et al*. (2016) cite examples of *M. gigas* found at 10 m depth in the Oosterschelde (Holland) where fishermen have reported settlement on adult Pacific oysters. In the Wadden Sea, settlement has been recorded at 10 m below low water on sediments (Herbert *et al*., 2012 and references therein).

Currently, recruitment of *M. gigas* appears to be significantly higher in the intertidal than the shallow subtidal, and the survival of adult oysters or mussels in the subtidal could be limited by predation. Kochman *et al*. (2013) reported that *M. gigas* were present on a subtidal mussel bed that was accessed during an exceptionally low tide. A survey of Lim Bay on the eastern Adriatic only found *M. gigas* at the sublittoral edge and not at 3 or 6m depth (Stagličić *et al*., 2018). Herbert *et al*. (2012) report that on rocky seabed habitats *M. gigas* have not been recorded below extreme low water.

Temperature: Adult oysters can survive water temperatures up to 40°C and at low tide, air temperatures as low as -5°C or lower depending on the salinity of water enclosed in their shells (Troost, 2010 and references therein). Larvae can survive temperatures between 16-30°C.

Wave exposure: No specific evidence was found for wave exposure tolerance, however, *M. gigas* reaches highest abundances on sheltered shores (Cook *et al*., 2014).

Magallana gigas: Establishment in Wales as of 2020

Syvret *et al*. (2008) undertook analysis of risk of natural recruitment of *M. gigas* for regions of the British Isles. Based on his results, Wales is considered moderate risk. Spatfalls of *M. gigas* from aquaculture sources were recorded at mid-shore level mostly attached to mussel shell at Menai strait in 1993 (Spencer *et al*., 1994). There has also been settlement in the Milford Haven, genetic studies have shown that these specimens are “wild” as opposed to the source being from nearby/relic oyster farms (K. Griffiths, NRW pers. comm.).

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Magallana gigas: Impact pathways based on species traits, biology and ecology

Competition: *M. gigas* is a trophic competitor for other bivalves and other filter feeders (Decottignies *et al*., 2007) and dense reefs would likely impact populations of native bivalve species, including mussels and native oyster and other filter feeders such as *Sabellaria alveolata* (Cognie *et al*., 2006). Field observations of increases in *Lanice conchilega* in areas with *M. gigas* cultivation (Ropert and Goulletquer, 2000) suggest that while these are trophic competitors the impact is of minimal concern, perhaps due to *L. conchilegas* ability to feed on a wider range of particle sizes and deposit feed (Ropert and Goulletquer, 2000). (See also aquaculture cultivated species below). *M. gigas* will also compete for space (assessed separately, see habitat impacts below).

Impacts on MPA features characterized by dense beds of filter feeders are assessed as ‘Moderate’. Impacts on characterizing species in other MPA features, which may include some filter feeders, but not at high densities are assessed as ‘Minimal concern’ with low confidence.

Predation: Not applicable; *M. gigas* is a filter feeder not a predator.

Hybridisation: There is no evidence that *M. gigas* can hybridise with *Ostrea edulis* or other native bivalves and this impact pathway is currently considered ‘Not applicable’.

Transmission of disease: The movement of Pacific oysters to France (from the US) appears to have resulted in the introduction of the pathogen *Haplosporidium nelsoni* but with no apparent consequences to date. There is no evidence that *H. nelsoni* is present in the UK, but its introduction might be possible if oysters are directly transferred from France to the British Isles for aquaculture (GBNNSS, 2019). No evidence was found for impacts arising from disease transmission by *M. gigas* and this pathway is assessed as ‘Data deficient’.

Parasitism: *M. gigas* is not a parasite and this impact pathway is considered ‘Not applicable’. It should be noted, however, that copepod parasites of *M. gigas*, *Mytilicola orientalis* and *Myicola ostreae* were introduced with imports of *M. gigas* from France into Ireland. Although no causal relationship was established the introduction coincided with summer mortality of farmed *M. gigas* (Herbert *et al*., 2012 and references therein). *M. orientalis* causes loss of gonadal mass in *M. edulis* but hardly affects *M. gigas* (Troost, 2010 and references therein). *M. gigas* also appears to be affected by a range of native parasites but at lower levels than native species (Troost, 2010 and references therein).

Poisoning/toxicity: Not applicable*. M. gigas* by itself is not poisonous or toxic. Toxic dinoflagellate blooms may lead to uptake of toxins which could result in poisoning if ingested but this indirect pathway is not assessed.

Bio-fouling: *M. gigas* may overgrow mussel beds and *Sabellaria* reefs (Dubois *et al*., 2006; Fey *et al*., 2010). Biofouling and the conversion of biogenic reefs is therefore an impact associated with this species (see MPA features below for further information), this pathway was assessed through the ‘structural impacts’ pathway below for habitats, to avoid double counting. No reports were found of *M. gigas* attached to macroalgae or seagrass.

Grazing/herbivory/browsing: Not applicable, *M. gigas* is a filter feeder.

Indirect impacts through interactions with other species: *M. gigas* are efficient filter feeders, and increased clearance rates in the Oosterschelde estuary has altered phytoplankton abundance and may result in further trophic effects. *M. gigas* may also consume planktonic larvae including those of other bivalves such as *C. edule*, *M. edulis* (Troost, 2010) and *Ostrea edulis* (Stagličić *et al*., 2018).

Magallana gigas: Impact pathways – Habitats

Chemical impact on ecosystem: In experiments the addition of *M. gigas* to sediments, changed sediment characteristics, including levels of porewater ammonium (Wagner *et al*., 2014). Reefs may also affect nutrient cycling through particle and solute transport (Troost, 2010 and references therein).
No evidence for deleterious effects was found, although combined changes in chemical and physical factors may result in changes in species richness. Impacts were assessed as ‘Minimal’ for rock habitats and ‘Moderate’ for sedimentary habitats based on changes in sediment characteristics and possible impacts on native species. Confidence was low for both assessments due to the lack of evidence. Impacts on biogenic habitats were assessed as ‘Data deficient’.

Physical impact on ecosystem: *M. gigas* excrete large amounts of faeces and pseudofaeces, altering sediment characteristics (Troost, 2010). In experiments the addition of *M. gigas* to sediments, changed sediment characteristics, increasing organic content, silt:sand ratios, and levels of porewater ammonium (Wagner *et al*., 2012). Reefs of *M. gigas* alter water turbulence and reduce water flow increasing

settlement of food particles. Changes in sediment characteristics can alter population structure of characterising species, for sedimentary habitats this impact pathway is assessed as ‘Moderate’ at low confidence due to the lack of evidence.
Reefs reduce heat and water loss and on dark rock shores in British Columbia, low to moderate densities of *M. gigas* cooled rock by solar reflectance and enhanced the survival of limpets (Padilla, 2010). As the assessment is likely to be highly site-specific and beneficial this pathway was not assessed.

Structural impact on ecosystem: Pacific oysters can form large reefs of individuals cemented together and can overgrow and transform sedimentary and biogenic habitats (see MPA feature impacts below) resulting in the loss of natural habitats (Troost, 2010). Typically, patches of *M. gigas* form on areas of hard substratum and shell beds e.g. mussels, new spat settle on older shells and eventually reefs of cemented individuals form (Fey *et al.,* 2010). Reefs are persistent and following mass mortality are likely to remain in place allowing recolonisation if extensive spatfall from another stock allows (Troost, 2010).

Structural changes may provide additional habitat heterogeneity increasing biodiversity but also alter feeding opportunities for other native species. Loss of mudflat, mussel beds and other habitat, exacerbated by the expansion of *M. gigas* reefs may impact wider ecosystems by reducing feeding sites for fish, birds and other benthic predators.

This is a key pathway by which *M. gigas* impacts habitats and characterising species, for habitats where *M. gigas* has been recorded as reef forming the impact is assessed as ‘Massive’ with the level of confidence varying according to habitat suitability. Where habitat is potentially suitable but no records of reef formation were found, the impact was assessed as ‘Moderate’. For habitats that are unlikely to be suitable impact was assessed as ‘Minor’. Confidence in moderate and minor assessments was low.

Magallana gigas: Interactions with MPA Features

MPA features that provide suitable habitat.

Intertidal sediments: *M. gigas* may establish on the following intertidal sediment habitats (Herbert *et al*., 2012) (High confidence):

- A2.2 Littoral sand and muddy sand:
  - A2.23; A2.24; (where suitable attachment surfaces, such as shell debris are present);
- A2.3 Littoral mud (where suitable attachment surfaces, such as shell debris are present);
- A2.4 Littoral mixed sediments. In Poole Harbour *M. gigas* favour mixed substrates (McKinstry and Jensen, 2013); and
- Sheltered muddy gravels: A2.41 and A2.42.

Littoral Rock (A1.1; A1.2; A1.3) *M. gigas* is recorded on littoral rock (Kent, Southern England, (Herbert *et al*., 2012); other records include presence on low, moderately exposed and high energy littoral rock in the Bay of Brest, France (Lejart and Hily 2011, cited from Herbert *et al*., 2016). In the Pacific North West, *M. gigas* is common on low energy littoral rock shores, whereas abundances are much lower (<10% cover) on high energy littoral rock (Ruesink, 2007, cited from Herbert *et al*., 2012). In Scotland wild *M. gigas* were typically found in the lower intertidal, inhabiting bedrock, bedrock encrusted with barnacles, large and small boulders and crevices found in the bedrock (Cook *et al*., 2014) they were not found on macroalgae or in areas overgrown with macroalgae (Cook *et al*., 2014). Estuarine rocky habitats are mostly sheltered and shallow and were therefore considered potential habitats for *M. gigas*. Based on available evidence, confidence is High.

Intertidal underboulder communities: *M. gigas* are unlikely to be found under boulders as they require exposure to the water column in order to feed, they are found on intertidal boulders where macroalgae cover is low (Cook *et al*., 2014; Smith *et al*., 2015). (High confidence).

Littoral biogenic reefs: formed of *Mytilus edulis* or *Sabellaria alveolata* may be colonised by *M. gigas* (Herbert *et al*., 2012; Cognie *et al*., 2006; Dubois *et al*., 2006) although mussels and worms may persist (see below: Impacts on MPA features). (High confidence).

MPA habitats considered potentially suitable for Magallana gigas

M. gigas has been reported on sediments down to 10 m but reports are limited (Herbert *et al*., 2016 and references therein). The following MPA features were considered potentially suitable:

- A5.2 Sublittoral sand:
  - A5.21; A5.24; A5.26; A5.27 biotopes (Medium confidence);
- A5.3 Sublittoral mud:
  - A5.31; A5.32; A5.33; A5.34 biotopes (Medium confidence);
- A5.4 Sublittoral mixed sediments:
  - A5.41; A5.42; Shallow examples of A5.43 and A5.44, based on substratum and exposure (Medium confidence);
- Subtidal mixed muddy sediments:
  - A5.43 and A5.44 biotopes (Medium confidence);
- Sheltered muddy gravels:
  - A5.43 and A5.44 where these occur in shallow areas (Medium confidence);
- A5.6 Sublittoral biogenic reefs
  - A5.61; A5.62. Shallow examples may be colonised based on depth and exposure;
- *Modiolus modiolus* beds (Low confidence);
- *Ostrea edulis* beds (High confidence); and
- Peat and clay exposures (Medium confidence).

MPA habitats considered unlikely to be suitable for *Magallana gigas*

MPA features that occur above the mid-shore and/or are highly mobile were considered unsuitable for *M. gigas*:

- A2.5 Coastal saltmarshes and saline reedbeds: based on shore height, (Herbert *et al*., 2012) (High confidence);
- A2.1 Littoral coarse sediment (Medium confidence); and

- A2.2 Littoral sand and muddy sand
    - A2.21 height on shore (High confidence); and
    - A2.22 sediment mobility and exposure (High confidence).

All Atlantic and Mediterranean infralittoral rock were considered unlikely to provide potential habitat as *M. gigas* is considered to be an intertidal species on rocky shores not recorded below extreme low water (Herbert *et al*., 2012). Evidence on depth and wave/exposure water currents is lacking and confidence was assessed as low. MPA features and/or constituent biotopes that are characterised by dense algae cover were considered unlikely to be suitable based on Cook *et al*. (2014):

- A1.1 High energy littoral rock:
    - A1.12, A1.15 biotopes (Medium confidence);
- A1.2 Moderate energy littoral rock, examples with dense macroaglae (Medium confidence);
- A1.3 Low energy littoral rock, examples with dense macroaglae (Medium confidence);
- A3.1 Atlantic and Mediterranean high energy infralittoral rock (Low confidence);
- A3.2 Atlantic and Mediterranean moderate energy infralittoral rock (Low confidence);
- Estuarine rocky habitat: biotopes A3.32; A3.36 (Low confidence; and
- Tide-swept channels: biotopes A1.15; A3.21; A3.22; A4.11; A4.25 (Low confidence).
- A5.5 Sublittoral macrophyte-dominated sediment:
    - A5.53; A5.52; A5.51 biotopes. Assessment based on substratum and depth. Some colonisation of shallow habitats where macroalgal cover is sparse and on maerl beds may occur (Low confidence);
- A2.6 Littoral sediments dominated by aquatic angiosperms;
- Seagrass beds; and
- *Zostera* beds.

M. gigas has been reported on sediments down to 10 m but reports are limited (Herbert *et al*., 2016 and references therein). Fine mud sediments, with no suitable attachments and deeper sediment habitats are considered unlikely to be suitable for *M. gigas*:

- A5.3 Sublittoral mud:
    - A5.35; A5.36 unlikely based on substratum and depth (Medium confidence);
- Subtidal mixed muddy sediments: A5.4 biotopes based on depth and A5.52 based on macroalgal cover in biotopes (Low confidence);
- Sheltered muddy gravels; A5.4 biotopes based on depth (Medium confidence);
- Mud habitats in deep water: A5.35; A5.36 biotopes, based on substratum and depth (Medium confidence);

- Sea-pen and burrowing megafauna communities: based on substratum and depth (Medium confidence);
- Maerl beds. No records, assessment based on substratum and depth, some colonisation of shallow beds may occur (Low confidence); and
- *Musculus discors* beds: Unlikely based on depth (Low confidence).

M. gigas is largely considered an intertidal/shallow subtidal species and habitats that predominantly occur at depths >10 m were considered unsuitable (confidence is Low unless otherwise indicated, based on lack of evidence for subtidal establishment and lack of information on wave exposure and tidal currents):

- A4.1 Atlantic and Mediterranean high energy circalittoral rock;
- A4.2 Atlantic and Mediterranean moderate energy circalittoral rock;
- A4.3 Atlantic and Mediterranean low energy circalittoral rock;
- Fragile sponge and anthozoan communities on subtidal rocky habitats (Medium confidence);
- A5.1 Sublittoral coarse sediment;
- A5.2 Sublittoral sand:
  - A5.22; A5.23; A5.25. Not suitable based on sediment mobility and/or depth (Medium confidence);
- A5.4 Sublittoral mixed sediments:
  - A5.45: (offshore biotope in deep water);
- Mud habitats in deep water:
  - A5.72 anoxic and not suitable (Medium confidence);
  - A5.37 unsuitable based on depth (Medium confidence); and
- Carbonate reefs, assessment based on substratum and depth (Low confidence).

Summary of key impacts on MPA features.
MPA features most at risk are intertidal mudflats and sandflats and intertidal biogenic reefs (*Sabellaria alveolata*). In Sylt, Wadden Sea mussel beds and mudflats have now been transformed to *M. gigas* reefs within the intertidal. It is likely that similar habitats in Wales will be affected should *M. gigas* spread.

Seagrass A study in British Columbia found that while oysters and eelgrass coexist at a regional scale, eelgrass is typically absent directly seaward of oyster beds, it is not clear if this was due to tidal level or exclusion by the oysters. If a causal link exists between oyster presence and eelgrass absence, then expansion of feral and farmed oyster beds may result in further eelgrass loss (Kelly & Volpe, 2007). Recent studies in the USA (Wall *et al*., 2008) suggest that the presence of filter feeding bivalves may increase eelgrass productivity. However, the study was undertaken using native species to the area under specific environmental conditions and whether similar benefits would occur in UK waters with the introduction of *M. gigas* is not clear. Impacts on seagrass have been assessed as 'Major' with medium confidence.

Blue Mussel beds (also relevant to the MPA features 'Intertidal *Mytilus edulis* beds on mixed and sandy sediments' and 'littoral biogenic reefs': *M. gigas* can out-compete *Mytilus edulis* (Padilla, 2010). In the Wadden Sea and North Sea, *M.*

gigas has overgrown mussel beds in the intertidal zone (Diederich, 2005; 2006; Kochmann *et al*., 2008, Wrange *et al*., 2010; Fey *et al*., 2010), although they did show a preference for settling on conspecifics before the mussels and struggled to settle on mussels with a fucoid covering. In Sweden, Wrange *et al*. (2010) found the highest densities of *M. gigas* were always associated with shallow mussel beds and the highest abundances of oysters were found above 1 m depth.

Diederich (2005; 2006) examined settlement, recruitment and growth of *M. gigas* and *M. edulis* in the northern Wadden Sea. *M. gigas* recruitment success was dependent on temperature, and in the northern Wadden Sea, only occurred in six of the 18 years since *M. gigas* was first introduced. Survival of juveniles is higher in mild than in cold winter and survival of both juveniles and adults on mussel beds is higher than that of the mussels themselves. *M. gigas* also grows faster than *Mytilus edulis* in the intertidal and can reach 2-3 times the length of mussels within one year. In addition, growth rates in *M. gigas* were independent of tidal level, emergence regime, substratum, *Fucus* cover and barnacle epifauna (growing on both mussels and oysters); while growth rate of *M. edulis* was decreased by these factors. The faster growth rate could make *M. gigas* more competitive than *M. edulis* where space or food is limiting. As oyster reefs form on former mussel beds, the available habitat for *M. edulis* can become restricted (Diederich, 2006) although mussels persist but at lower depths. In the northern Wadden Sea the change to mixed oyster and mussel bed is considered permanent (Reise *et al*., 2017).

It has been observed that mussel beds in the Wadden Sea that were adjacent to oyster farms were quickly converted to oyster beds (Kochmann *et al*., 2008). Padilla (2010) predicted that *M. gigas* could either displace or overgrow mussels on rocky and sedimentary habitats of low or high energy. Kent and Essex Inshore Fisheries and Conservation Authority (IFCA) (cited in Herbert *et al*., 2012) reported that *M. gigas* had developed a significant stock on mussel beds on the Southend foreshore and that, by 2012, there were few mussels left in the affected area, but made no conclusions as to the reason for the decline in mussels (Kent and Essex IFCA pers. comm. cited in Herbert *et al*., 2012). In the Wadden Sea declines in *M. edulis* have been attributed to climate change rather than competition with *M. gigas* (Nehls *et al*., 2006).

Impacts on mussel beds have been assessed as ‘Massive’ at high confidence, where conditions allow reef formation to occur.

Sabellaria alveolata: Reefs in the shallow subtidal may be colonised by *M. gigas*, in the Bay of Mont Saint-Michel, France, Dubois *et al*. (2006) found that *M. gigas* had escaped from adjacent aquaculture facilities and were growing on *S. alveolata* reefs. Similarly, in Bourgneuf Bay, also in France, wild *M. gigas* were observed in areas usually colonised by *S. alveolata* (Cognie *et al*., 2006). Diversity of associated species was highest on the reefs with oysters. There were also some differences in the age structure of these reefs suggesting that there may have been negative effects on *S. alveolata* recruitment. Studies have suggested that *M. gigas* could have the potentially beneficial effect of increasing the probability of interception of *S. alveolata* larvae sinking or swimming down the water column, as demonstrated by flume settlement experiments and models (Soniat *et al*., 2004. Green and Crowe (2013) conducted manipulative experiments in the intertidal where live and dead *M. gigas* were attached to boulders and observed that the presence of living and dead shells reduced settlement of *S. alveolata* in comparison with control boulders. The oysters may smother *S. alveolata* by growing over the tube ends and could out-compete the larvae, juveniles, and adults for space. In addition, *M. gigas* and *S. alveolata* are both suspension feeders, and they ingest food particles in the same size range resulting in trophic competition (Dubois *et al*., 2003). Impacts on this feature have been assessed as 'Massive' at high confidence, where conditions allow reef formation by *M. gigas* to occur.

Magallana gigas: Socio-Economic Impacts

Health and safety: In terms of consumer demand for recreational activities in coastal and marine areas, ICES (Miossec *et al*., 2009) suggest that the presence of *M. gigas* can affect recreational activities in positive (through harvesting) and negative ways and that its sharp shells make it a nuisance to many recreational activities and lead to injury (GBNNSS, 2019). Impacts are assessed as 'moderate' at low confidence.

Aquaculture Operation: Wild *M. gigas* can overgrow aquaculture infrastructure, including trestles used for growing farmed *M. gigas* (Cognie *et al*., 2006) increasing maintenance costs. At sites in France, feral Pacific oysters are trophic competitors of farmed oysters (e.g. Cognie *et al*., 2006) and in the UK, settlement of spat on farmed oysters and aquaculture gear creates additional operational costs and may lead to reduced product quality.

Impacts have been assessed as 'Massive' for on bottom cultivation of mussels and oysters due to potential for habitat change, impacts on growth through competition and processing costs, at high confidence.

Aquaculture cultivated species: Wild *M. gigas* compete for food resulting in growth reductions in farmed individuals (Cognie *et al*., 2006). Impacts on mussels are also likely.

Fisheries Operations: *M. gigas* appears to be restricted to intertidal and shallow subtidal habitats and most fisheries operations occur outside these habitats. Impacts are assessed as 'Minimal concern' at high confidence based on lack of exposure for most fishery operations. However, *M. gigas* reefs could prevent foot access to fishing vessels or impede boat launching from shore and Troost (2010) suggests that fishermen avoid reefs because they can damage netting although no further detail is provided.

The presence of reefs of *M. gigas* may impact on beach seine operations by restricting gear operation and potentially damaging gear. Impacts are assessed as 'Minor' at low confidence, impacts on demersal anchor and fly shoot seines would be considered to be of minimal concern as these are used in deeper waters where reef formation is less likely.

Reefs of *M. gigas* may also restrict hand gathering operations both through changes in the distribution and abundance of target species resulting from habitat change and by making operations more difficult due to the risk of cuts from the sharp shells. Impacts are assessed as 'Moderate' with high confidence for this activity.
Although reports are contradictory in the ability of *M. gigas* to substantially alter the environment through **competition**, or to minimise commercial stocks, potential impacts from substratum changes are greater. There is a risk that mussel seedbeds will become difficult to fish because of the presence of oysters. Similarly, *Ostrea edulis* beds could be impacted by a mixing of the two types of oysters. The presence of *M. gigas* could affect the supply and collection of native oyster spat. See 'Aquaculture cultivated species' above. In the Wadden Sea it has been thought to affect mussel and cockle beds, resulting in losses in these commercial fisheries (Diederich, 2006).

Fisheries Target species: No evidence was found for direct interaction with and impacts on fish target species that are demersal and pelagic piscivores and pelagic planktivores.

Demersal benthivores: Skates and rays, plaice (*Pleuronectes platessa*) and Sole (*Solea solea*) may be impacted where *M. gigas* colonises areas where these feed on sediment infauna. *M. gigas* is limited to the intertidal and shallow subtidal and therefore is unlikely to impact on species found in deeper waters and offshore such as witch (*Glyptocephalus cynoglossus*) and haddock (*Melanogrammus aeglefinus*) although some impacts may occur in inshore feeding areas. The replacement of sediment habitats by *M. gigas* reefs would reduce food supply to species that feed on infauna in sheltered areas. No evidence was found for such impacts and generally these were considered to be of 'Minimal concern' at high confidence.

In the Wadden Sea, *M. gigas* has affected mussel, native oyster and cockle beds, resulting in many studies to ascertain the economic impacts. Cockles (*Cerastoderma edule*) are considered to be more resilient due to mobility but some evidence has been found that *M. gigas* alters both reefs and substrate (Diederich, 2006). In the Wadden sea competition for space between *M. gigas* and *C. edule* occurs where there is overlap although *C. edule* are typically found higher on the shore (Diederich, 2006).

It is possible that Pacific oyster beds increase settlement opportunities for mussels although the extent to which this will benefit the mussel industry is unclear (Troost, 2010). Feeding interactions and competition with native, commercially important bivalves is likely to be complex. It is likely that the feeding mechanisms of *M. gigas* and structure will interfere with the feeding success of native bivalve species of commercial importance (Troost 2010). Again the possible economic significance of such impacts is unclear. Given the current value of UK wild mussel fisheries of £2.0 million, native oysters of £0.1 million, and cockles of £10.1 million (all values for wild harvest, 2004 (shellfish.org.uk)), economic loss could represent £12.2 million per year in an absolute scenario (cited from GBNNSS, 2019). Complete losses are unlikely but *M. gigas* may overgrow some beds resulting in additional costs in cleaning and processing and may impact harvesting of seed mussels for relaying (Diederich, 2006). Impacts on cockles are assessed as 'Moderate', impacts on native oysters and mussels are assessed as 'Massive' (all assessments have medium confidence).

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Annex 12 A red seaweed: Agarophyton vermiculophyllum

Common name(s): Worm wart weed; Black wart weed (Maggs & Magill, 2014 and references therein).

Synonyms: *Gracilaria vermiculophylla* (Gurgel *et al*., 2018; Guiry & Guiry, 2020)

Domain: Phylum: Rhodophyta, Class: Florideophyceae, Order: Gracilariales, Family: Gracilariaceae, Genus/species: *Agarophyton vermiculophyllum* (Ohmi) Gurgel, J.N.Norris & Fredericq, 2018

Description: A type of red seaweed *Agarophyton vermiculophyllum (A. vermiculophyllum))* can be dark red to almost black. Its branched fronds are elongated and slender growing to around 2 m in length. Branches are irregular, often wrinkled with longitudinal grooves, and feel elastic due to the lower parts being hollow. Their holdfast is a small disc. Male plants tend to be bushier than females which can appear more straggly. It can be found loose-lying or attached to small stones or shells (Wood, 2019 and references therein; GISD, 2015).

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Figure 12.1. *Agarophyton vermiculophyllum.*

Agarophyton vermiculophyllum: Habitat

Native range: Maggs and Magill (2014), provide a description and references for the native range and subsequent spread of *A. vermiculophyllum*. It is native to the NW Pacific (Japan, Korea, China and Vietnam) and was described from Hokkaido by Ohmi (1956) it was first reported outside this range in the Eastern Pacific, Mexico and California (Bellorin, 2004) and later in British Columbia (Saunders, 2009).

Substratum type: Temperate to sub-tropical in its native range it is generally found in rocky habitats. In its UK invasive range it favours soft-bottomed, muddy, sheltered bays, inlets, harbours, lagoons and estuaries which are rich with nutrients (Nyberg & Wallentinus, 2009; Thomsen *et al*., 2007; Freshwater *et al*., 2006). It can survive being buried in sediment for several weeks (Thomsen & McGlathery, 2006) and experimentally has survived darkness for more than 5 months (Nyberg & Wallentinus, 2009). It is also tolerant of other stresses such as desiccation, grazing and low nutrients (Rueness, 2005). Populations that have established are often found in association with habitat building benthic invertebrates. It has been found assimilated into the tubes of *Diopatra cuprea* (plumed/decorator worm) (Thomsen & McGlathery, 2006), lugworms and other tube-building worms, cockles, snails, and growing on the byssal threads of mussels (Thomsen *et al*., 2007). In Virginia, USA, within intertidal saltmarshes, it is found to inhabit both the mudflat areas in association with tube-worms as well as further up on the saltmarsh edges, and growing amongst the *Spartina alterniflora* (saltmarsh cordgrass) (Thomsen *et al.,* 2009).

Salinity: 5-60 PSU, although optimum salinity for it is 10-45 (Rueness, 2005).

Depth: The available evidence suggests that *A. vermiculophyllum* inhabits shallow habitats (Nyberg & Wallentinus, 2009; Maggs & Magill, 2014 and references therein).

Wave exposure: *A. vermiculophyllum* is found in sheltered (Wood, 2019 and references therein), low energy estuaries and bays (Freshwater *et al*., 2006).

Agarophyton vermiculophyllum: Establishment in Wales as of 2020

There has been a putative report from Porthmadog, Gwynedd (Krueger-Hadfield *et al*., 2017b). It has been recorded on the Dwyryd Estuary forming mats in saltmarsh pans (JNCC, 2019).

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Agarophyton vermiculophyllum: Impact pathways based on species traits biology and ecology

Competition: It is possible that it could compete with native seagrass beds for light and oxygen. It can form dense mats that could outcompete or smother seagrasses in intertidal saltmarshes (Thomsen *et al*. 2009). It has been shown to negatively impact net leaf photosynthesis and survival rates of the seagrass *Zostera marina* (Martinez-Lüscher and Holmer, 2010). However, these large mats are often unattached and the patches they create can be ephemeral (Abreu *et al*., 2011) so it is not understood if there is any long term impact on seagrass beds (Maggs & Magill, 2014 and references therein).

Once established, *A. vermiculophyllum* can attain large biomass and these large populations could displace native macroalgae species such as fucoids. It is possible that it hinders native settlement by preventing substratum availability, and may cause mortality in larval stages by reducing oxygen and light availability (Maggs & Magill, 2014 and references therein; GISD, 2015; Hammann *et al*., 2013). However, in its invasive range in England and Ireland there is little evidence that it has had any negative effects. It tends to establish in muddy areas where there are few other algal species and it has not reached large enough biomass levels to adversely affect oxygen levels or water current movements. In fact in Brownsea Island lagoon, UK, it has provided suitable habitat for the lagoon species *Corophium insidiosum* (Maggs & Magill, 2014 and references therein).

In MPA features where macroalgae, seagrass and cordgrass occur this pathway has been assessed as ‘Moderate’ with low confidence. In habitats where reef-building invertebrates and bivalve beds occur, this pathway has been assessed as a ‘Minor concern’ with low confidence. In all other habitats it has been assessed as ‘Minimal concern’ with low confidence.

Predation: *A. vermiculophyllum* is an alga and as such, this pathway has been assessed as ‘Not applicable’.

Hybridisation: There is nothing in the literature to suggest that *A. vermiculophyllum* hybridizes. This pathway has been assessed as ‘Not applicable’.

Transmission of disease: *A. vermiculophyllum* is not known to transmit diseases therefore this pathway has been assessed as ‘Not applicable’.

Parasitism: *A. vermiculophyllum* is not known as a parasite therefore, this pathway has been assessed as ‘Not applicable’.

Poisoning/toxicity: *A. vermiculophyllum* has been shown to have more resistance to epiphytes than native conspecifics partly due to secondary metabolites (Wang *et al*., 2016). Since these substances are deterrents rather than a poison this pathway is assessed as ‘Not applicable’.

Bio-fouling: *A. vermiculophyllum* is known to foul fishing nets, trawls, boat propellers and power plant cooling intakes (Maggs & Magill, 2014 and references therein). In these instances, it has been assessed as of moderate concern with a medium confidence. It also forms dense mats that can cover seabed sediment (Thomsen & McGlathery*,* 2006). As there is no evidence that this species has overgrown any habitats around the UK this impact pathway has been assessed as of minimal concern but with low confidence. The exception to this is with *maerl* beds, which has been assessed as ‘Moderate’ impact with low confidence. The assumption being that *maerl* beds are very slow growing species and would be highly impacted by *A. vermiculophyllum* overgrowth.

Grazing/herbivory/browsing: *A. vermiculophyllum* is an alga and as such, this pathway has been assessed as ‘Not applicable’.

Indirect impacts through interactions with other species: *A. vermiculophyllum* has been recorded growing amongst beds of *Zostera marina* (seagrass) in Denmark (Thomsen*,* 2010). Studies have recorded a positive impact on invertebrate species richness and abundance possibly through the structural complexity created by both species growing together creating protection from predators, and attachment space for spat and food for grazers (Thomsen, 2010). Drift *A. vermiculophyllum* may impact seagrass via competition for light and nutrients and by increasing the levels of anoxia, ammonia and sulphide in the water column or sediment pore-water (Thomsen, 2010). These effects could result in long-term reduction of habitat stability, ultimately destroying seagrass habitat (Hauxwell *et al*., 2003) and any associated community (Thomsen, 2010). While this is a possibility, there is no actual evidence, and this impact pathway has been assessed as ‘Minimal concern’ with low confidence.

Agarophyton vermiculophyllum: Impact pathways – Habitats

Chemical impact on ecosystem: *A. vermiculophyllum* is likely to have important implications for nutrient cycling and trophic dynamics in its invasive range due to its movement, accumulation and decomposition (Thomsen & McGlathery, 2006). However, there is little in the literature regarding any chemical impacts on the ecosystem so this has been assessed as ‘Data deficient’.

Physical impact on ecosystem: Its ability to form large, dense mats once established (Thomsen & McGlathery, 2006) creates shading and encourages sediment accretion which can effect erect native algae and seagrasses (Maggs & Magill, 2014 and references therein). Negative impacts by *A. vermiculophyllum* on *Zostera* spp. above ground biomass has been observed and thought to be caused by shading, oxygen and nutrient depletion as well as reduced water currents (Maggs & Magill, 2014 and references therein). Since *A. vermiculophyllum* patches can be ephemeral (Abreu *et al*., 2011) their long-term threat to seagrass beds is uncertain (Maggs & Magill, 2014 and references therein).

The increased likelihood of anoxia, change in water patterns and the resultant altered sedimentation rate caused by *A. vermiculophyllum* algal mats may negatively impact food availability for deposit feeders (Nyberg & Wallentinus, 2009).

Structural impact on ecosystem: It has been observed to form large, dense mats once established. These mats could grow over macroalgae beds and seagrasses (Thomsen & McGlathery, 2006; Thomsen, 2010) which would create a very different structural environment (GISD, 2015). Weinberger *et al*. (2008) noted that in feeding trials grazers preferred *Fucus vesiculosus* rather than *A. vermiculophyllum*. Other studies have found that native grazers preferred to consume native algae to *A. vermiculophyllum* when offered a choice (Nejrup *et al*. (2012). This preference by grazers could encourage the spread of *A. vermiculophyllum (*Weinberger *et al*., 2008).

Maggs & Magill (2014 and references therein) noted while this species can dominate algal assemblages it could increase the structural complexity of soft-bottom shores supporting grazers and epibiota such as red algae, and provide refugia for mobile invertebrates and demersal fish species. Thomsen (2010) found that where *A. vermiculophyllum* was growing amongst *Zostera marina* (seagrass) beds it had a positive effect on most invertebrate groups. Together the seagrass and the algae provided more structurally complex habitats that supported a greater invertebrate abundance and species richness. As the literature tends to suggest a positive impact on structural complexity when *A. vermiculophyllum* occurs in habitats, this pathway has been assessed as ‘Minimal concern’ with low confidence.

Agarophyton vermiculophyllum: Interactions with MPA Features

MPA features that provide suitable habitat for Agarophyton vermiculophyllum
The MPA features below are considered suitable for *A. vermiculophyllum* to establish in based on their shallow, low energy, soft-bottomed features. They have all been scored with high confidence.

- A2.3 Littoral mud;
- A2.4 Littoral mixed sediments;
- A5.2 Sublittoral sand:
- A5.24;
- A5.3 Sublittoral mud:
    - A5.31, A5.32, A5.33, A5.34; and
- Sheltered muddy gravels.

The MPA features below are considered suitable for *A. vermiculophyllum* based on the biogenic reefs. *A. vermiculophyllum* is associated with biogenic reefs in shallow, low energy, soft-bottomed habitats. They have all been scored as high confidence with the exception of *Maerl* beds and Peat and clay exposures that have been scored with medium confidence as there is less understanding of their associations with *A. vermiculophyllum.*

- A2.7 Littoral biogenic reefs;
- Blue mussel beds;
- *Maerl* beds:
    - A5.513, A5.514;
- *Ostrea edulis* beds; and
- Peat and clay exposures.

The MPA features below are considered suitable for *A. vermiculophyllum* to establish in based on its association with seagrass beds and *Spartina alterniflora*. They have been assessed with high confidence.

- Seagrass beds:
- *Zostera* beds;
- A2.5 Coastal saltmarshes and saline reedbeds:
    - A2.5542.

MPA habitats considered potentially suitable for *Agarophyton vermiculophyllum*

The MPA features below are considered potentially suitable for *A. vermiculophyllum* to establish in based habitat preference of shallow, soft-bottom sediments and association with biogenic reefs.

- A2.2 Littoral sand and muddy sand:
    - A2.24 (high confidence);
- A5.2 Sublittoral sand:
    - A5.21, A5.22, A5.23 (medium confidence);
- A5.4 Sublittoral mixed sediments:
    - A5.41, A5.42, A5.43 (medium confidence);
- Subtidal mixed muddy sediments:

    - A5.445 (low confidence);
- *Sabellaria alveolata* reefs:
    - A2.71 (high confidence);
- Intertidal *Mytilus edulis* beds on mixed and sandy sediments; (medium confidence); and
- *Maerl* beds:
    - Except A5.513, A5.514, A5.512 (medium confidence).

The MPA features below are considered potentially suitable for *A. vermiculophyllum* to establish in based on their association with macroalgae and reedbeds. They have all been assessed with low confidence as there are other factors that may make them unsuitable such as hard sediments or too far up on the shore.

- A2.5 Coastal saltmarshes and saline reedbeds:
    - A2.54, A2.55;
- Intertidal underboulder/boulder communities;
- A5.5 Sublittoral macrophyte-dominated sediment; and
- Littoral sediments dominated by aquatic angiosperms.

MPA habitats considered unlikely to be suitable for *Agarophyton vermiculophyllum*

The MPA features below are considered unlikely to be suitable for *A. vermiculophyllum* to establish in because they are too rocky. This is based on the evidence that this species has only established in muddy, soft-bottom bays in the UK (Maggs & Magill, 2014 and references therein). They have all be scored as low confidence because this species is known to establish in rocky habitats in its native ranges (Wood, 2019 and references therein).

- A1.1 High energy littoral rock;
- A1.2 Moderate energy littoral rock;
- A1.3 Low energy littoral rock;
- A3.1 Atlantic and Mediterranean high energy infralittoral rock;
- A3.2 Atlantic and Mediterranean moderate energy infralittoral rock;
- A3.3 Atlantic and Mediterranean low energy infralittoral rock; and
- Estuarine rocky habitat.

The MPA features below are considered unlikely to be suitable for *A. vermiculophyllum* to establish in because they are too far up on the shore. However, *A. vermiculophyllum* has a wide range of environmental tolerances so these have been scored as low confidence.

- A2.2 Littoral sand and muddy sand:
    - A2.21, A2.22; and
- A2.5 Coastal saltmarshes and saline reedbeds:
    - A2.51, A2.52, A2.53.

The MPA features below are considered unlikely to be suitable for *A. vermiculophyllum* to establish in because of their course, hard bottom sediment.

- Tide-swept channels:

    - A1.15, A3.22 (medium confidence);
- A2.1 Littoral coarse sediment (low confidence); and
- A5.1 Sublittoral coarse sediment (low confidence).

MPA features unsuitable for *Agarophyton vermiculophyllum*
The MPA features below are considered unsuitable for *A. vermiculophyllum* to establish in because they are too deep. They have all be scored as high confidence because *A. vermiculophyllum* has only established in shallow bays in the UK (Maggs & Magill, 2014 and references therein).

- Fragile sponge and anthozoan communities on subtidal rocky habitats;
- A5.2 Sublittoral sand:
    - A5.25, A5.26, A5.27;
- A5.3 Sublittoral mud:
    - A5.35, A5.36, A5.37;
- A5.4 Sublittoral mixed sediments:
    - A5.44, A5.45;
- Subtidal mixed muddy sediments:
    - A5.45;
- Sea-pen and burrowing megafauna communities;
- A4.1 Atlantic and Mediterranean high energy circalittoral rock;
- A4.2 Atlantic and Mediterranean moderate energy circalittoral rock;
- A4.3 Atlantic and Mediterranean low energy circalittoral rock;
- Mud habitats in deep water;
- A5.6 Sublittoral biogenic reefs;
- *Modiolus modiolus* beds;
- *Musculus discors* beds;
- Tide-swept channels:
    - A4.11; A4.25; and
- *Sabellaria alveolata* reefs:
    - A5.612.

The MPA feature below is considered unsuitable for *A. vermiculophyllum* to establish in because it is too exposed. *A. vermiculophyllum* establishes in low energy habitats and has been scored with medium confidence.

- *Maerl* beds:
    - A5.512.

No evidence for *Agarophyton vermiculophyllum* in these MPA habitats

- A5.7 Carbonate reefs.

Summary of key impacts on MPA features

MPA features most at risk are sheltered, low energy, soft-bottomed bays and estuaries, in particular muddy sediments with biogenic reefs, seagrass, macroalgae and *Spartina alterniflora* for permanent attachment. In Dorset this species has established in Christchurch Harbour and around Brownsea Island Lagoon, and in Ireland in Carlingford Lough and Dundrum Bay. In both Dorset and Ireland it has established in muddy areas where there is little to compete with in regards to species such as macroalgae and there is no evidence of any negative impacts to date (Maggs & Magill, 2014 and references therein).

Seagrass, cordgrass (*Spartina alterniflora)* and macroalgae (relevant MPA features: Coastal saltmarshes and saline reedbeds; Sublittoral macrophyte-dominated sediment; Littoral sediments dominated by aquatic angiosperms; Seagrass beds and *Zostera* beds): *A. vermiculophyllum* has been observed reducing the amount of *Zostera* spp. above ground biomass when co-occurring together. This is thought to be caused by shading, oxygen and nutrient depletion as well as reduced water currents arising when it forms dense mats that shade out light and promote sediment accretion (Maggs & Magill, 2014 and references therein). However, these dense patches of algae can be ephemeral (Abreu *et al*., 2011) so their long-term impacts are unknown. Impacts are assessed as ‘Moderate’ at low confidence.

Biogenic reefs (relevant MPA features: Littoral biogenic reefs; *Sabellaria alveolata* reefs; Blue mussel beds; Intertidal *Mytilus edulis* beds on mixed and sandy sediments; *Maerl* beds; *Ostrea edulis* beds and Peat and clay exposures): *A. vermiculophyllum* is known to attach to mussel byssal threads, oysters and to associate with habitat building benthic invertebrates (Maggs & Magill, 2014 and references therein). Maggs & Magill (2014) state that bays with biogenic reefs are most at risk because an attached population of *A. vermiculophyllum* is more likely to establish and persist than an unattached population. Once established it may prevent settlement of spat by fouling the available substrate (Wood, 2019 and references therein). Impacts are assessed as ‘Moderate’ at low confidence.

Agarophyton vermiculophyllum: Socio-Economic Impacts

Health and safety: No impacts are known in the UK. Hammann *et al*. (2016) found that toxic compounds, including Prostaglandin $E_{2,}$ produced by *A. vermiculophyllum* as grazing deterrents have increased in concentration by up to 390% in non-native ranges. It is assumed the extra protection is needed in its new environment so individuals with higher concentrations are selected for. This species is grown for human consumption in its native range of Asia. There is now the worry that it may translocate back to Asia from its invaded territories causing an increase in human intoxication (Hammann *et al*, 2016).

Aquaculture operations: This species could impact aquaculture if it establishes and forms dense mats (Thomsen & McGlathery, 2006). In one of its earliest recorded sites, Hog Island Bay, USA, it now accounts for ~80% of the total algal biomass (Thomsen & McGlathery, 2007). It is known to grow on small pebbles and shells (GISD, 2015) and has been documented dominating algal assemblages (Maggs & Magill, 2014 and references therein) which could have a negative impact on algal farmers. It can become abundant once established and can attain high biomass (Maggs & Magill, 2014 and references therein). Its long, stringy thalli can easily foul nets, propellers and other marine machinery (Maggs & Magill, 2014 and references therein). Krueger-Hadfield *et al*. (2017b) documented that *A. vermiculophyllum* population sites around the British Isles are also shellfish aquaculture sites giving weight to the evidence that shellfish is a major source of spread of this invasive species, in particular oyster aquaculture (Krueger-Hadfield *et al*., 2017a).

Fouling aquaculture gear and shellfish: *A. vermiculophyllum* fouls artificial marine structures and shellfish (Thomsen *et al*., 2007) this may lead to operational difficulties and potentially related economic losses. This impact pathway has been assessed as moderate with low confidence.

Fisheries operations: At high abundances, *A. vermiculophyllum* can foul nets (Freshwater *et al*., 2000), boat propellers and other machinery with its long, stringy thalli (Maggs & Magill, 2014 and references therein). These issues could have a negative economic impact on fisheries through damaged nets, extra haulage and cleaning and disposing of the fouling alga. Fisheries would also have the added impact of transporting *A. vermiculophyllum* fragments in its nets and associated marine gear (Thomsen *et al*., 2007). Impacts are assessed as ‘Moderate’ at low confidence.

Fouling fishing gear: *A. vermiculophyllum* is known to foul fishing and associated gear, including nets and boat propellers (Maggs & Magill, 2014 and references therein). While there is little evidence to suggest this has been the case in the UK since the arrival of *A. Vermiculophyllum*, it could happen in the future should it establish in abundance. Impacts are assessed as ‘Moderate’ at low confidence.

Fisheries Target species: Dense mats of *A. vermiculophyllum* on substratum will make habitats less accessible (Maggs & Magill, 2014 and references therein). This can impact bottom feeding fish that may change their foraging behaviours in response, as was seen with red mullet in response to the invasive algae *Caulerpa taxifolia* (Levi and Francour, 2004).

Demersal benthivores: Skates and Rays; Witch (*Glyptocephalus cynoglossus*); Haddock (*Melanogrammus aeglefinus*); Plaice (*Pleuronectes platessa*) and Sole (*Solea solea*) may be impacted if *A. vermiculophyllum* fouls benthic feeding sediment making it inaccessible to these demersal benthivores. It may change their foraging behaviour and displace populations although there is little evidence for overlap with key feeding grounds and the impact is assessed as ‘Minimal concern’ with medium confidence.

A. vermiculophyllum fouls shellfish potentially reducing shellfish health and creating gathering and sorting difficulties for fishing operations and possible economic losses. The impact is assessed as is assessed as ‘Minimal concern’ with medium confidence.

Hand collection: The collection of seaweed for the purposes of making laver bread could be impacted if *A. vermiculophyllum* becomes too abundant. It may be more difficult to find and collect the algae (laver) required for the making of laver bread. This pathway has been assessed as ‘Minimal concern’ with low confidence.

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Annex 13. Orange striped anemone: *Diadumene lineata*

Common name(s): Orange-striped anemone (Bilewitch, 2009).

Synonyms: *Sagartia lineata* (original name)*; Aiptasiomorpha luciae; Diadumene luciae; Haliplanella lineata; Haliplanella luciae; Sagartia davisi; Sagartia luciae* (Daly & Fautin, 2019*).*

Domain: Phylum: Cnidaria, Class: Anthozoa, Order: Actinaria, Family: Diadumenidae, Genus/species: *Diadumene lineata* (Daly & Fautin, 2019).

Description: *Diadumene lineata (D. lineata)* is a small, delicate anemone with a smooth column up to 20 mm in diameter in GB (but larger in its native range). Generally, it is olive green or brown with contrasting orange vertical stripes (less often yellow or white). It has 25-100 slender, smooth tentacles, which are all of one type and typically of uniform yellow, white, or grey colour (see Figure 13.1). Defensive thread-like tentacles (acontia) can extend through pores in the column. When fully extended the top part of the column (capitulum) is distinctly divided from the lower part by a parapet (Wood, 2020).

(© John Bishop, MBA).

Figure 13.1. Orange striped anemone: *Diadumene lineata*

Diadumene lineata: Habitat

Substratum type: *Diadumene lineata* is found in brackish inshore waters including bays, estuaries, harbours and marinas where it fouls hard substrate (Bilewitch, 2009). Found on mudflats and brackish marsh channels, it is only limited by the need for hard substratum for attachment (Cohen, 2011). Often found attached to shells, associated with mussels and oysters, as well as rocks, boulders, jetties, sea walls, buoys, pillings and sometimes seaweeds (Bilewitch, 2009; Cohen, 2011). It has also been found in saltmarshes associated with *Spartina alterniflora* (Molina *et al*., 2009).

Salinity: *D. lineata* can survive for at least two weeks in salinities from 0.5 to 35 ppt, although growth and fission are reduced below 24ppt, and salinities below 7 ppt are ultimately lethal (Shick & Lamb, 1977; Molina *et al*., 2009; Podbielski *et al*., 2016; Ryan & Miller, 2019).

Depth: Shallow down to a few hundred metres (Cohen, 2011).

Wave exposure: Sheltered waters (Fofonoff *et al.,* 2003).

Diadumene lineata: Establishment in Wales as of 2020

There are a few scattered reports from Cardiff, Milford Haven, Abereiddy and subtidally off Anglesey (NBNAtlas, 2019).

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Diadumene lineata: Impact pathways based on species traits, biology and ecology

Competition: *D. lineata* has been recorded in large clonal aggregations that could out-compete some native species (Podbielski *et al*. 2016), aggregations of over 4,000 individuals per square metre have been observed along from the Atlantic coast in the USA (Shick & Lamb, 1977). There is no evidence of any competitive impact from this species so this pathway has been assessed as of ‘Minimal concern’ with low confidence.

Predation: ‘Data deficient’. Anemones are known predators on the larvae of commercially valuable species like oysters (Steinberg & Kennedy, 1970), using tentacles to catch and/or sting their prey with nematocysts. They are generally not considered a threat and their impact upon commercial species has not been studied. Due to their abundance in places and the commercial value of shellfish populations their ecological impact should not be assumed to be minor and should be classified as unknown (Baker *et al*., 2004).

Hybridisation: ‘Not applicable’. In its native range in Japan, *Diadumene lineata* reproduces sexually by releasing eggs and sperm into the water, and asexually by longitudinal fission and pedal laceration (Atoda, 1973). Studied populations in its introduced range have been historically assumed to reproduce only asexually (Baker *et al*., 2004; Fofonoff *et al.,* 2003), however, recently, populations with fertile males and females have been reported from the Pacific and Atlantic Coasts of North America (Newcomer *et al.* 2019; Ryan & Miller, 2019) supporting potential sexual reproduction in non-native populations. However, there is no evidence suggesting hybridization with a separate species could occur.

Transmission of disease: ‘Not applicable’.

Parasitism: ‘Not applicable’. *D. lineata* is not known to be a parasite.

There is no evidence of *D. lineata* transmitting any diseases (Fofonoff *et al.,* 2003).

Poisoning/toxicity: Not applicable’. Although *D. lineata* can sting using nematocysts, this is described under predation. It is not otherwise toxic or poisonous.

Bio-fouling: This species has been described as a ship-fouling organism (Gollasch & Riemann-Zürneck, 1996; Zabin *et al*., 2004). It is considered an introduced species in many parts of the world (Beneti *et al*., 2015) helped by its ability to spread via fouling methods (Cohen, 2011). It can withstand extreme physiological tolerances enabling it to colonize areas that other fouling species cannot. It can coat itself in mucus (encystment) to protect itself from extreme conditions. This characteristic behaviour probably enables it to survive long-term while being transported on ship hulls (Carlton, 2003; Cohen, 2011) or on shellfish such as mussels and oysters (Bilewitch, 2009). However, there is no evidence to suggest any fouling impacts from this species so this pathway has been assessed as of ‘Minimal concern’ with low confidence.

Grazing/herbivory/browsing: ‘Not applicable’. *D. lineata* feeds by trapping epibenthic animals and zooplankton on its tentacles (Fofonoff *et al.,* 2003), it does not feed by grazing.

Indirect impacts through interactions with other species: No impacts have been recorded for this species (Fofonoff *et al.,* 2003). This pathway has been assessed as of ‘Minimal concern’ with low confidence.

Diadumene lineata Impact pathways – Habitats

Chemical impact on ecosystem: Podbielski *et al*. (2016) suggests that there may be an impact on biochemical fluxes of the benthic ecosystem when they aggregate in large numbers. This pathway has been assessed as of ‘Minimal concern’ with low confidence.

Physical impact on ecosystem: None reported – ‘Data deficient’.

Structural impact on ecosystem: None reported. However, when large aggregations form they may compete for food and space resources with native benthic encrusting and bio-fouling species (Podbielski *et al*., 2016). This impact pathway has been assessed as of 'Minimal concern' with low confidence.

Diadumene lineata: Interactions with MPA Features

MPA features that provide suitable habitat for *Diadumene lineata*

The MPA features below provide suitable habitat for *D. lineata* based on its association with mussels and oysters. They have all been scored with high confidence.

- A5.6 Sublittoral biogenic reefs:
  - A5.62;
- A2.7 Littoral biogenic reefs:
  - A2.72;
- Blue mussel beds;
- Intertidal *Mytilus edulis* beds on mixed and sandy sediments;
- *Modiolus modiolus* beds;
- *Musculus discors* beds;
- Peat and clay exposures:
  - A1.1223; and
- Ostrea edulis beds.

The MPA features below provide suitable attachment opportunities for *D. lineata* and/or its associated community. They have all been scored with high confidence.

- A2.5 Coastal saltmarshes and saline reedbeds:
  - A2.553, A2.554 (associated with *Spartina alterniflora*);
- A1.3 Low energy littoral rock;
- A3.2 Atlantic and Mediterranean moderate energy infralittoral rock;
- A3.3 Atlantic and Mediterranean low energy infralittoral rock;
- A4.3 Atlantic and Mediterranean low energy circalittoral rock; and
- Sheltered muddy gravels:
  - A5.431, A5.432, A5.435.

MPA habitats considered potentially suitable for *Diadumene lineata*

The MPA features below have fluctuating salinities that are tolerated by *D. lineata* whilst also providing suitable attachment opportunities.

- A2.5 Coastal saltmarshes and saline reedbeds (except A2.553, A2.554) (low confidence);

- Seagrass beds (medium confidence);
- A2.6 Littoral sediments dominated by aquatic angiosperms (medium confidence);
- *Zostera* beds (medium confidence); and
- Estuarine rocky habitat (medium confidence).

The MPA features below all have potentially suitable attachment opportunities for *D. lineata* and environmental conditions (including suitable muddy sediments).

- A1.2 Moderate energy littoral rock (low confidence);
- A1.1 High energy littoral rock (low confidence);
- A3.1 Atlantic and Mediterranean high energy infralittoral rock:
    - A3.11 (medium confidence);
- A4.1 Atlantic and Mediterranean high energy circalittoral rock:
    - A4.11, A4.111 (low confidence);
- A4.2 Atlantic and Mediterranean moderate energy circalittoral rock (medium confidence);
- Tide-swept channels:
    - A3.22, A4.25 (low confidence);
- Intertidal underboulder/boulder communities:
    - A3.2112 (low confidence);
- A2.4 Littoral mixed sediments:
    - A2.43 (medium confidence);
- A2.2 Littoral sand and muddy sand:
    - A2.24 (low confidence);
- A5.4 Sublittoral mixed sediments (medium confidence);
- Subtidal mixed muddy sediments (low confidence);
- A5.3 Sublittoral mud (low confidence);
    - A5.354;
- Mud habitats in deep water (medium confidence);
- Sea-pen and burrowing megafauna communities (medium confidence);
- A2.7 Littoral biogenic reefs:
    - A2.71 (low confidence);
- *Sabellaria alveolata* reefs (low confidence);
- Peat and clay exposures:
    - A1.127 (low confidence); and
- A5.5 Sublittoral macrophyte-dominated sediment (low confidence).

The MPA features below are considered potentially suitable habitats for *D. lineata* based on the associated community within them. They have been scored with medium confidence.

- A5.6 Sublittoral biogenic reefs:
    - A5.61, A5.63; and

- Maerl beds.

MPA habitats considered unlikely to be suitable *Diadumene lineata*

The MPA features below are considered unlikely to be suitable habitats for *D. lineata* based on a lack of suitable attachment substratum (low confidence).

- A2.3 Littoral mud;
- Sheltered muddy gravels: except
    - A5.431, A5.432, A5.435.

The MPA features below are considered unlikely to be suitable habitats for *D. lineata* as they are likely to be too high-energy environments. They have all been scored with low confidence:

- Tide-swept channels:
    - A4.11, A1.15;
- Intertidal underboulder/boulder communities:
    - A1.2142;
- A3.1 Atlantic and Mediterranean high energy infralittoral rock:
    - A3.111, A3.112, A3113, A3.114, A3.115, A3.116, A3.117;
- A4.1 Atlantic and Mediterranean high energy circalittoral rock:
    - A4.112, A4.1121, A4.1122, A4.13; and
- Fragile sponge and anthozoan communities on subtidal rocky habitats except:
    - A4.133, A4.211, A4.211, A4.2112 - which occur mainly in Scotland and Ireland.

The MPA features below are considered unlikely to be suitable habitats for *D. lineata* as they are likely to be too high-energy environments (some strandline habitats), and/or lack suitable attachment substrata and/or due to the mobile nature of some sediments. They have all been scored with low confidence.

- A2.1 Littoral coarse sediment;
- A2.2 Littoral sand and muddy sand:
    - A2.21, A2.22, A2.23; and
- A2.4 Littoral mixed sediments:
    - A2.41, A2.42.

MPA features unsuitable for *Diadumene lineata*

The habitats below are considered unsuitable habitats for *D. lineata* based on a lack of suitable attachment substratum.

- A5.2 Sublittoral sand (medium confidence);
- A5.3 Sublittoral mud (All except A5.354) (low confidence).

The ‘Sublittoral course sediment’ (A5.1) habitats have been assessed as unsuitable based on their mobile nature (medium confidence).

No evidence for *Diadumene lineata* in these MPA habitats

- A5.7 Carbonate reefs.

Summary of the impacts on key MPA features

Diadumene lineata is a global invader but there have been no economic or ecological impacts reported to date (Fofonoff *et al.,* 2003).

Diadumene lineata: Socio-Economic Impacts

Health and safety: No impacts are known.

Aquaculture Operations: No direct impacts on aquaculture operations were found in the literature. The socio-economic impact could not be assessed due to lack of evidence and is therefore, ‘Data deficient’.

Cultivated Species - Oysters, mussels and scallops: ‘Data deficient’. Anemones are known predators of oyster larvae and other commercially valuable species (Steinberg & Kennedy, 1970), although their impact as predators has not be studied (Baker *et al*., 2004). In addition, their ability to form large aggregations (Shick & Lambe, 1977) could have a negative effect on these commercially valuable populations. Little is known about the impacts due to a lack of data.

Fisheries Operations: The Impact was assessed as ‘Minimal concern’ at low confidence for fisheries using mobile and static gears as *D. lineata* was considered unlikely to alter the level of activity.

Target Species- Oysters, mussels and scallops: See ‘Cultivated species’ above.

Finfish with pelagic larvae: See ‘Cultivated species’ above.

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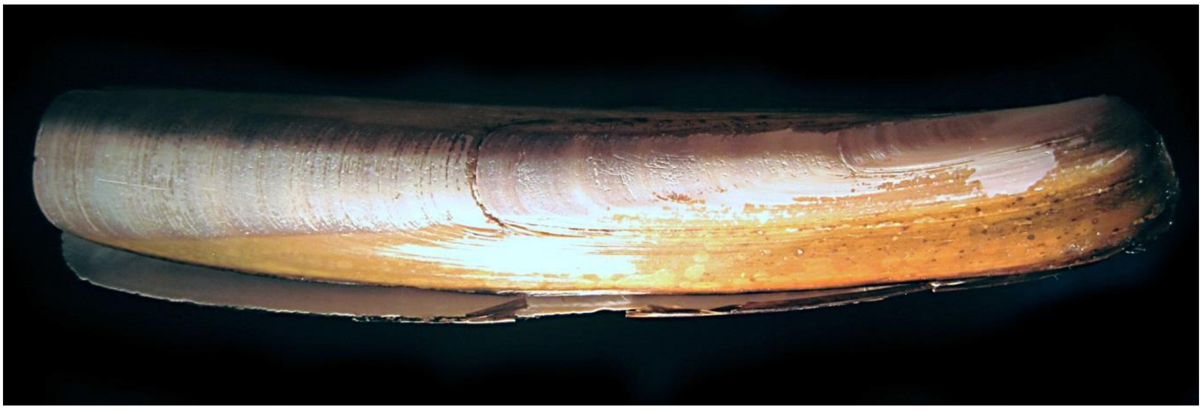
Annex 14. American jack knife clam: *Ensis leei*

Common name(s): American jack knife clam; Bamboo clam; razor clam; American sword sheath.

Synonyms: *Ensis americanus; Ensis arcuatus var. directus; Ensis directus*; *Solen directus* (MolluscaBase, 2019).

Domain: Phyla: Mollusca, Class: Bivalvia, Order: Adapedonta, Family: Pharidae, Genus/species: *Ensis leei* (MolluscaBase, 2019).

Description: A thin, elongated bivalve with a maximum length of approximately 20 cm. The length is around six times the width with both narrow ends of the same width (Sweet, 2010).



(© by Jack Sewell).

Figure 14.1. American jack knife clam: *Ensis leei*

Ensis leei: Habitat

Native range: E*. leei* has its native distribution in the north-west Atlantic from the Gulf of St. Lawrence along the whole US coast to the Gulf of Mexico where it occupies mainly shallow, sandy subtidal areas down to a water depth of 37 m (Theroux and Wigley, 1983; Leavitt, 2010).

Substratum type: As a burrowing, infaunal species, *E. leei* is restricted to sedimentary habitats. In the western part of the Wadden Sea, early establishment of *E. leei* (reported as *E. directus*) was at low shore levels in sandy sediments, with low species diversity and abundances (Beukema and Dekker, 1995). Its habitat range has subsequently expanded. The habitat of *E. leei* in the Wadden Sea has been modelled by Schwemmer *et al.*, (2019) based on distribution (sediment and hydrodynamic data). The model showed that sediment preferences varied: *E. leei* preferred coarse sand sediments in the northern sub-area but coarse sand and muddy sediments in the southern sub-area (Schwemmer *et al*., 2019).

In its introduced range *E. leei* is able to inhabit a variety of substrates from fine sand and course sand, to silt and almost pure mud on the banks of channels in The Wash (Palmer, 2003). It has a high tolerance for high silt concentrations (Kamermans *et al*., 2013; Witbaard *et al*., 2015).

Salinity: *E. leei* seems to prefer more estuarine conditions than the native *Ensis* and is found in areas influenced by estuarine outflows (Desroy *et al*., 2002; Jensen, 2015; Sweet, 2010). Maurer *et al*., (1974), report that the salinity range in native habitats was 7-32 ppt.

Depth: *E. leei* is intolerant of long periods of emersion and in Europe is found on the lower shore to fully subtidal areas where intertidal exposure is lower than the mid-upper shore levels (Schwemmer *et al*., 2019, Beukema and Dekker 1995). In Canada this species has a preference for depths of 5-8 m where currents are low-moderate (Kenchington *et al*., 1998) although they can be found as deep as 100 m (Theroux & Wigley, 1983).

Wave exposure: *E. leei* prefers areas with moderately high bed shear stress (Schwemmer *et al*., 2019; Leavitt, 2010).

Ensis leei: Establishment in Wales as of 2020

Recorded locally around Milford Haven, South Wales (Sweet 2010).

E. leei can establish in high densities. It has been recorded having a settling density of about 150 individuals m^2 (Beukema & Dekker, 1995), 440 individuals m^2 (Mühlenhardt-Siegel *et al*., 1983) and more than 3000 individuals m^2 (Dauwe *et al*., 1998) in the Wadden Sea. In a few years following establishment their numbers can be 10 - 100 times higher than local mollusc species (Severijins, 2004).

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Ensis leei: Impact pathways based on species traits, biology and ecology

Competition: At high densities, *E. leei* may compete for both food and space with native species (Severijns, 2004, Gollash *et al*., 2015 and references therein). Dense populations of *E. leei* may decrease food availability for other filter feeders such as cockles and mussels (Armonies and Reise 1998).

Along the Belgium coast in the local *Abra abra* community the most common species now found is *E. leei* (Gollash *et al*., 2015 and references therein). Declines in other bivalve species, *Mactra stultorum* and *Cerastoderma edule*, and tellinids, have also been observed since the introduction of *E. leei*. Similarly in the Netherlands and Belgium there was a reported reduction in *Spisula subtruncata* abundance which coincided with an increase in *E. leei* numbers. While there are no official reports of local extinctions of native species, *E*. *minor* seems to have disappeared along the Belgium coast in areas of *E. leei* and *E*. *magnus* has been largely replaced by *E. leei* in coastal areas of France and Belgium with *E*. *magnus* displaced to areas offshore (Gollash *et al*., 2015 and references therein).

Observed changes in infaunal community composition may be due to competitive exclusion of native species by *E. leei*, or result from a number of factors including associated sediment changes (see physical change below). Given the inherent uncertainties confidence in assessments was low. Impact on MPA features characterised by infaunal bivalve filter feeders, such as cockles was assessed as ‘Moderate’ based on likely competition for food and space. Impacts on biotopes characterised by mobile burrowing infauna, predatory or deposit feeding species were assessed as ‘Minor’ as competition is reduced for space and resources. Impacts on epifaunal filter feeders such as, oysters and scallops were assessed as ‘Minor’ based on reductions in growth but not loss of populations.

Bio-fouling: *E. leei* is a free-living infaunal species and is not a bio-fouler, this impact pathway is assessed as ‘Not applicable’.

Predation: *E. leei* is not a predator and this impact pathway is assessed as ‘Not applicable’.

Grazing/herbivory/browsing: Not applicable: *E. leei* is a filter feeder.

Hybridisation: No records of hybridisation with native species by *E. leei* were found in the literature and this impact pathway is considered ‘Not applicable’.

Parasitism: *E. leei* is not a parasite and this impact pathway is assessed as ‘Not applicable’. While no known parasites have been introduced with *E. leei*, they have become hosts for some native parasites. This may affect the natural life cycles of these parasites although it may also lessen the parasite burden of the native hosts (Krakau, *et al.,* 2006).

Transmission of disease: No records of disease transmission by *E. leei* were found in the literature and this impact pathway is considered to be ‘Not applicable’.

Poisoning/toxicity: *E. leei* is not poisonous or toxic and this impact pathway is assessed as ‘Not applicable’.

Indirect impacts through interactions with other species: *E. leei* has the ability to modify its surrounding environment by its rapid growth, high abundance, burrowing behaviour and its ability to trap silt. This could impact the habitat and the native community that lives there (Gollash *et al*., 2015 and references therein; Severijns, 2004).

Tulp *et al*., (2010) based on fish and bird stomach analyses, concluded that the establishment of *E. leei* must have caused a major change in trophic relationships in the Dutch coastal zone (Tulp *et al*., 2010). Changes in fish and bird distribution and predation may result in indirect impacts on seabed habitats but there was no evidence to assess this impact and ‘Data deficient’ is recorded in the EICAT spreadsheet.

Ensis leei: Impact pathways – Habitats

Physical impact on ecosystem: Sediment containing dense populations *of E. leei* tended to become enriched with faecal material although it is not known whether this material was released by *E. leei* within or above the sediment, or only accumulated in the beds (Armonies and Reise, 1998). Sediment enriched with faecal material will is more cohesive and less susceptible to erosion. Armonies and Reise (1998), observed that in *E. leei* beds the percentage of very fine sand increased and became more densely packed. Sediment characteristics are important in determining habitat suitability for infauna: changes in sediments are likely to result in changes to the species present. Increased settlement of deposit feeder species in the presence of *E. leei* have been recorded and changes in species present from those found in coarser sands, such as the filter feeding *Lanice conchilega*, to species typical of finer sediments such as the deposit feeder *Owenia fusiformis* (Armonies and Reise 1998).

Physical impacts on sediments that already contain a significant mud fraction were assessed as ‘Minor’ as characterising and associated species are likely to tolerate an increase in finer sediments, although there may be some changes in population structure.

Structural impact on ecosystem: High abundances of *E. leei*, coupled with regular mass mortalities results in a large amount of shell debris. Millions of shells and dying specimens are now frequently observed washed onto Belgian beaches (Houziaux *et al*., 2011). Accumulated shell material could lead to the creation of new structural habitats (Kerckhof *et al*., 2007) and the build-up of silt (Gollash *et al*., 2015 and references therein) which could, in turn, lead to a negative effect on local biodiversity (Kerckhof *et al*., 2007).

Impacts will depend on proximity to *E. leei* dominated habitats, the population size of *E. leei*, the frequency and intensity of mass mortality events and other factors such as local hydrodynamics, substratum types and shore profiles. Due to these inherent uncertainties this impact pathway was assessed as ‘Data deficient’.

Chemical impact on ecosystem: Dense beds of filter feeders can alter food webs and nutrient cycling within the ecosystem (Martin *et al*., 2006). Assessing the impact of these changes for MPA seabed features is challenging as impacts will be density dependent and site specific with factors such as tidal flushing mediating changes. For all MPA features the impact is assessed as ‘Minor’. Although dense populations may alter food web dynamics and nutrient cycling, impacts ramify to the water column rather than seabed habitats. Increased bacterial activity and stimulation of microbial food webs through accumulation of faecal material and fine sediments may enhance benthic production. Confidence in the assessment is low due to uncertainties and lack of evidence for benthic habitats.

Ensis leei: Interactions with MPA Features

MPA features that provide suitable habitat for *E. leei*

- A5.2 Sublittoral sand(Moderate confidence):
    - A5.23; A5.24; A5.25; A5.26 Based on reported habitat preferences (Beukema & Dekker, 1995; Gollasch *et al*., 2015);
- A5.3 Sublittoral mud. Based on sediment preferences for sandy muds (Schwemmer *et al*., 2019) (High confidence):
    - A5.33 Infralittoral sandy mud,
- A5.35 Circalittoral sandy mud; and
- Mud habitats in deep water (High confidence):
- A5.35 Circalittoral sandy mud. Based on sediment preferences (Schwemmer *et al*., 2019) (High confidence).

MPA features considered potentially suitable for *E. leei*

- A2.2 Littoral sand and muddy sand:
- A2.22; A2.23; A2.24 potential habitat where these are on the lower shore (Schwemmer *et al*., 2019; Beukema and Dekker, 1995). (High confidence);
- A2.4 Littoral mixed sediments: Potential suitable habitat as *E. leei* is found in muds and sands) (Palmer, 2003, Schwemmer *et al*., 2019). (Medium confidence);
- A5.1 Sublittoral coarse sediment
- A5.133-A5.137; A5.142-A5.145; A5.15. Potential suitability inferred from presence of coarse sand and bivalves in shallow habitats (Schwemmer *et al*., 2019) (Low confidence);
- A5.4 Sublittoral mixed sediments
- A5.42 ; A5.43; A5.44; A5.45;A5.46 Based on habitat preferences- muds/clean sands (Palmer, 2003; Beukema and Dekker, 1995; Schwemmer *et al*., 2019) (Low confidence);
- Subtidal mixed muddy sediments: Based on habitat preferences- muds/clean sands (Palmer, 2003; Beukema and Dekker, 1995; Schwemmer *et al*., 2019) (Low confidence);
- Sheltered muddy gravels A5.43; A5.44. Based on habitat preferences- muds/clean sands (Palmer, 2003; Beukema and Dekker, 1995; Schwemmer *et al*., 2019) (Low confidence); and
- *Ostrea edulis* beds: Based on potentially suitable depths and substratum (Schwemmer *et al*., 2019) (Medium confidence).

MPA features unlikely to be unsuitable for *E. leei*

E. leei has been recorded from intertidal mudflats (Schwemmer *et al*., 2019). However, A2.3 Littoral mud biotopes A2.31 and A2.32 that occur in estuaries were considered unsuitable habitat, due to regular and prolonged emersion, (Schwemmer *et al*., 2019). (Medium confidence).

MPA features unsuitable for *E. leei*

E. leei is an infaunal species requiring sediment habitats to burrow into. All rock/firm and very coarse substratum habitats were considered unsuitable (High confidence):

- A1.1 High energy littoral rock;
- A1.2 Moderate energy littoral rock;
- A1.3 Low energy littoral rock;
- A3.1 Atlantic and Mediterranean high energy infralittoral rock;
- A3.2 Atlantic and Mediterranean moderate energy infralittoral rock;
- A3.3 Atlantic and Mediterranean low energy infralittoral rock;
- A4.1 Atlantic and Mediterranean high energy circalittoral rock;
- A4.2 Atlantic and Mediterranean moderate energy circalittoral rock;
- A4.3 Atlantic and Mediterranean low energy circalittoral rock;
- Estuarine rocky habitat;
- Fragile sponge and anthozoan communities on subtidal rocky habitats;
- Intertidal underboulder/boulder communities; and
- Peat and clay exposures.

The following biogenic habitats were also considered unsuitable based on substratum (High confidence):

- A2.7 Littoral biogenic reefs;
- A5.6 Sublittoral biogenic reefs;
- Sabellaria alveolata reefs;
- Blue mussel beds;
- Intertidal Mytilus edulis beds on mixed and sandy sediments;
- Maerl beds;
- Modiolus modiolus beds; and
- Musculus discors beds.

E. leei has been recorded from intertidal mudflats (Schwemmer *et al*., 2019) and subtidal muddy sand sediments. No evidence was found that this species inhabits offshore, subtidal muds and the following mud habitats were considered unsuitable:

- A5.2 Sublittoral sand (Low confidence):
    - A5.21; A5.22 based on salinity;
- A5.24 based on subtidal sediment preferences (Schwemmer et al., 2019); A5.27 depth;
- A5.3 Sublittoral mud (Medium confidence):
    - A5.31; A5.34; A5.36; A5.37 based on sediment (Schwemmer et al., 2019);
- Mud habitats in deep water
    - A5.36; A5.37 based on sediment (Schwemmer et al., 2019) and reported depth preferences (Jensen, 2015). (Medium confidence)
    - A5.7211- unsuitable based on anoxia (Medium confidence):

- Sea-pen and burrowing megafauna communities (Medium confidence).

E. leei is found on the lower shore, the following habitats were considered unsuitable based on shore height (and where indicated other factors):

- A5.2 Coastal saltmarshes and saline reedbeds (High confidence);
- A2.1 Littoral coarse sediment (High confidence): Based on unsuitable sediment and shore height (Schwemmer et al., 2019; Sweet, 2010).
- A2.2 Littoral sand and muddy sand (Medium confidence):
    - A2.21 & A2.22 Based on shore height and sediment (Schwemmer et al., 2019; Sweet, 2010).

MPA features that could not be assessed (no evidence)

No evidence was found for presence in the following habitats and no suitable proxy evidence was identified to support an assessment:

- A5.5 Sublittoral macrophyte-dominated sediment;
- A2.6 Littoral sediments dominated by aquatic angiosperms;
- Seagrass beds;
- *Zostera* beds; and
- A5.7 Carbonate reefs.

Ensis leei: Socio-Economic Impacts

Health and safety: Shells of *E. leei* can be sharp and could cause cuts to recreational users of beaches, including those engaged in activities such as cockle harvesting or bait digging (Gollasch *et al*., 2015).

Aquaculture Operations: No direct impacts on aquaculture operations were found in the literature. It is possible that dense populations of *E. leei* may reduce the suitability of on-substratum growing areas through changes in sediment type and disturbance to newly laid juveniles. Mass mortalities of dense populations of *E. leei* may cause impacts on shell-fish on-growing areas through accumulation of shells. The socio-economic impact could not be assessed due to lack of evidence and is therefore, ‘Data deficient’.

Aquaculture: Cultivated Species: *E*. *leei* may compete for food with *Mytilus edulis* grown on lines. No evidence was found to assess this potential impact. As competition may reduce growth rather than prevent aquaculture, the impact was assessed as ‘Minor’ (with low confidence).

Fisheries Operations: Empty shells of *E. leei* may accumulate in large numbers and may reduce catching efficiency of mobile gears and may also cause damage to nets (VLIZ, 2011; Gollasch *et al*., 2015). The source material for this assertion could not be checked within the project timescale and assessing impact is difficult. Impact was assessed as ‘Minor’ at low confidence for fisheries using mobile gears, as it was considered unlikely to alter the level of activity.

Fisheries target species: *E. leei* may be exploited commercially in a similar way to other native species (Addison *et al*., 2006; Breen *et al*., 2011). Socio-economic benefits or losses and environmental impacts from this fishery have not been assessed as these are outside the project scope.

E. leei was not considered to directly impact finfish that are pelagic or demersal piscivores and was assessed as ‘Minimal concern’. Impacts on demersal benthivores may occur indirectly through habitat change and/or competition with other benthic species that alters infaunal assemblages and changes in the availability of favoured prey species.

Diet studies show that on the Dutch coast, *E. directus* makes up a significant contribution (20-100% of the total wet weight in fish stomachs) to the diet of plaice, sole, dab, flounder and dragonet (Tulp *et al*., 2010). The proportion *E. directus* in the diet increases with fish length (Tulp *et al*., 2010). Because of its great burying depth the species is not easily accessible. Fish either profit from massive die-offs that regularly occur, or they extract individuals from the sediment, it is quite likely that only smaller individuals that are closer to the surface can be accessed (Tulp *et al*., 2010). As *E. leei* provides a potential food source to demersal benthivores, it was assessed as ‘Minimal concern’ to acknowledge that changes in benthic community structure and food availability may be associated with this species. Confidence is assessed as low. Spatial competition with other bivalves would reduce the supply of other species but increases in sediment silts associated with dense beds, may favour polychaetes that could provide an additional food source, offsetting losses of bivalves.

Whelk (*Buccinum undatum*) and commercially targeted crustaceans consume a range of food sources and scavenge on carrion. Mass mortalities of *E. leei* may provide an occasional food source, so that impact was assessed as ‘Minimal concern’.

Oysters-*Ostrea edulis*, *Magallana gigas*: E*. leei* may compete with oysters for food. No evidence was found to assess this potential impact. As competition may reduce growth the impact was assessed as ‘Minor’ (with low confidence).

Native *Ensis spp*., In France and Belgium native Ensis species have been reported to have disappeared or been displaced offshore when *E. leei* becomes established in inshore coastal areas (Gollash *et al*., 2015 and references therein). Impact on these species was assessed as ‘Major’ with medium confidence. (Note these species are not included in the SEICAT assessment).

Ensis leei: References

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Annex 15: Asian rapa whelk: *Rapana venosa*

Common name(s): Asian rapa whelk; Veined rapana whelk (Zenetos, 2016); Purple whelk (Sealife base, 2020).

Synonyms: *Purpura venosa; Rapana marginata; Rapana pechiliensis; Rapana pontica; Rapana thomasiana* (MolluscaBase, 2019); *Rapana thomasiana thomasiana* (ICES, 2004).

Domain: Phyla: Mollusca, Class: Gastropoda, Order: Neogastropoda, Family: Muricidae, Genus/species: *Rapana venosa* (MolluscaBase, 2019).

Description: *Rapana venosa* is a large, predatory gastropod mollusc up to18 cm in length. Its shell is rounded, bumpy with darker veins running around it. There are two colour morphs found with predominantly white/grey individuals found on pale sediments and likewise mainly dark brown shells found on hard, rocky structures (ICES, 2004). The shell opening is wide and toothed on the outer lip with a short, open siphon canal. Internally the shell is orange-peach and glossy (Kerckhof *et al*., 2006; Sewell & Sweet, 2011).

Image supplied by C. Wood and J. Bishop.

Figure 15.1 Asian rapa whelk: *Rapana venosa*

Rapana venosa: Habitat

Native range: The large Asian gastropod mollusc *R. venosa* is native to the Sea of Japan, Yellow Sea, Bohai Sea, and the East China Sea to Taiwan. This species has been introduced to the Black Sea with subsequent range expansion to the Adriatic Sea and Aegean Sea, the Chesapeake Bay on the East Coast of the United States, and the Rio de la Plata between Uruguay and Argentina (Mann *et al*., 2004).

Substratum type: *R. venosa* is a habitat generalist (Zenetos, 2016) that can be found colonizing hard and mixed substrates either natural like rocky outcrops and *Zostera* beds (Culha *et al*., 2009) or artificial structures like jetty legs. In Chesapeake Bay *R. venosa* occupies shallow hard-substrate habitats until reaching shell lengths in excess of 70 mm and then migrates into deeper habitats with sand or mud substrates where they forage on infaunal bivalves (ICES, 2004). It is often found residing on and under soft sediment (Sewell & Sweet, 2011) and muddy bottoms

(Gilberto *et al*., 2006; Capitulo *et al*., 2002). Being nocturnal they spend most of the day buried under sediment (Harding & Mann, 1999), seemingly favouring compact sandy bottoms (Zenetos, 2016). In the Río de la Plata estuary *R. venosa* was found in shallow sandy and muddy sediments (Giberto *et al*., 2006).

Salinity: 16-35 ppt. This species has been found living in mixohaline (0.5-30 ppt), estuarine waters in South America (Gilberto *et al*., 2006). Mann and Harding (2003) found that salinity tolerance is the dominant response controlling their potential dispersal into estuarine conditions. All larval stages of *R. venosa* were found to survive for 48 hours in salinities of 15 ppt with minimal mortalities while *R. venosa* veliger survival was significantly less at < 7 ppt. There was no difference in percentage survival at 16 ppt (Mann & Harding, 2003). *R. venosa* have become established in full salinity in the North Sea (Kerckhof *et al*., 2006), Mediterranean and Black Sea (ICES, 2004) as well as in its native ranges (ICES, 2004; Mann & Harding, 2003).

Depth: This species is confined to subtidal habitats and occurs from 3-90 m (Sewell & Sweet, 2011). In the Black Sea *R. venosa* is found on rocky and sandy bottoms to 40 m (ICES, 2004) with some individuals found at the sublittoral margin (0-0.5m) with optimal depth based on abundance at 15m (Culha *et al*., 2009). In Chesapeake Bay, USA evidence suggests it occupies shallow, hard substrate habitats until it reaches 70 mm in length when it migrates to deeper habitats (ICES, 2004).

Wave exposure: No evidence found.

Rapana venosa: Establishment in Wales as of 2020

R. venosa has not been recorded in Wales to date. It has been identified in the North Sea on three separate occasions with more reports of it being fished in the North Sea. It is reasonable to presume there is a small, established population in the North Sea (Nieweg, 2005). Wales has suitable habitat and prey however it may be on the edge of *R. venosa* temperature range. Literature suggests (ICES, 2004) that it needs extended periods at 18 °C to reproduce successfully (Harding *et al*., 2008) although it demonstrates large temperature tolerances (4-27°C) in its' native range of Korea (Chung *et al*., 1993). Temperature limitations may be the reason this species does not yet seem to have become established. However as sea temperatures become warmer, particularly in shallow inshore areas, there is a risk that conditions will become suitable for the whelk to become established by the 2020s along the south coast of England and further north by the 2050s (Cook *et al*., 2013).

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Rapana venosa: Impact pathways based on species traits, biology and ecology

Competition: Laverty *et al*., (2015) categorised *R. venosa* as competing with several species for food and/or space with, at least one native species declining. It could compete with the native common whelk, *Buccinum undatum* (*B*. *undatum)* (Kerckhof *et al*., 2006) as both species fall into the same niche of living in soft sediment and preying on burying bivalves (Sewell & Sweet, 2011). Other predatory gastropods will also compete with *R. venosa*, but no information on competitive interactions was found.

Although competition is likely, the whelk and other bivalve predators are not characteristic species of MPA features and this impact pathway is considered to be of minimal concern with respect to seabed habitats, although socio-economic considerations are assessed (see below).

Predation: *R. venosa* preys on a number of marine invertebrates including soft sediment burying bivalves, other mollusc and decapods. In large numbers this species has the potential to decimate bivalve fisheries (Sewell & Sweet, 2011). *R. venosa* is reportedly capable of consuming an 8 cm clam in under an hour (Sewell & Sweet, 2011). They are able to smother a clam with their body triggering it to open and then inserting their proboscis, thereby predating upon the clam without any drilling (Zenetos, 2016).

In the Black Sea R. venosa can reach high densities with up to 1500 individuals caught per one 15 min. trawl in some sites on the Bulgarian shelf (Zolotarev, 1996). At these high densities it has resulted in the loss of edible, native bivalves *Ostrea edulis*, *Pecten ponticus*, and *Mytilus galloprovincialis* (Zolotarev, 1996). These species are either present in Wales (O. edulis) or have ecologically and commercially important congeners in Wales that would be likely to incur the same fate.

Savini and Occhipinti-Ambrogi (2006) found that *R. venosa* showed an average consumption of 1 bivalve per day (or 1.2 g wet weight). They were selective regarding prey size and species. Experimentally *R. venosa* were offered three species of bivalves as potential prey, two Mediterranean local fisheries species Mytilus galloprovincialis and Tapes philippinarum plus the introduced clam *Anadara* (*Scapharca*) *inaequivalvis*. Savini and Occhipinti-Ambrogi (2006) found that prey selection was size and species selective towards the small individuals of *A. inaequivalvis* with less predation on the commercial species (Savini & Occhipinti-Ambrogi, 2006).

This selective predation could influence community structure or create competition with native species (Savini & Occhipinti-Ambrogi, 2006).

Their wide range in dietary preferences and salinity tolerances suggest many bivalve species, including *Mytilus edulis* and *Mya arenaria*, would be at risk from predation should R. venosa become established and present in high numbers (ICES, 2004). Other recorded species that are consumed include the piddock *Pholas dactylus* in the Romanian Black Sea (Micu 2007).

Impacts from predation are density dependent and it is unclear whether water temperatures will allow dense population of R. venosa to develop in UK waters. For MPA features that are characterized by dense beds of molluscs e.g. Blue mussel beds, the impact is assessed as 'Major' at moderate confidence, the severity of impact would be greater if dense populations develop. The impact on MPA features that are not characterized by bivalves, e.g. macrolagal dominated rock, the impact is assessed as minor, as predation will not result in loss or reclassification of the feature.

Hybridisation: This species is not known to hybridise (Zenetos, 2016, Laverty et al., 2015) and the impact pathway is 'not applicable'.

Transmission of disease: No known examples (Laverty *et al*., 2015), this impact pathway is 'not applicable'.

Parasitism: *R. venosa* is not a parasite and this pathway is 'not applicable'.

Poisoning/toxicity: Not applicable, *R. venosa* is not toxic or poisonous.

Bio-fouling: *R. venosa* is a mobile mollusc and as such this impact pathway is 'not applicable'. It can however, lay large quantities of egg capsules on gear, equipment and living animals (potentially including commercially important ones, which could be considered fouling). The impact is not strictly considered to be biofouling and the impact pathway is considered to be 'not applicable'.

Grazing/herbivory/browsing: *R. venosa* is not an herbivore and this impact pathway 'not applicable' (Laverty *et al*., 2015).

Indirect impacts through interactions with other species: An availability of larger shells, from *R. venosa,* has led to larger striped hermit crabs (*Cilbanarius vittatus*) in Chesapeake Bay, USA whilst also enabling them to expand their range. This larger size of hermit crab has demonstrated an ability to consume significant amounts of oyster spat (ICES, 2001). There was 'No evidence' to assess this impact, for MPA features, as hermit crabs are habitat generalists this score was provided in the accompanying EICAT assessment table where this is likely to be the only indirect impact.

Loss of filter feeders through predation by *R. venosa* could have indirect effects on other species and the ecosystem. Dense beds of filter feeders capture large amounts of suspended particles and can reduce water turbidity resulting in increased light penetration. This may be beneficial for adjacent macrophyte dominated biotopes such as seagrass beds (Wall *et al*., 2008). At high abundances filter feeders can also alter food webs and nutrient cycling within the ecosystem Martin *et al*., (2006). The impact from this pathway will be density dependent and site specific, depending on factors such as background turbidity and eutrophication. The impact has been assessed as ‘Moderate’ for seagrass at low confidence in order to identify potential issues.

Rapana venosa: Impact pathways - Habitats

Chemical impact on ecosystem: No direct chemical impacts on ecosystem are recorded, bioturbation through burrowing actions may result in some changes in sediment chemistry but these are not considered to lead to impacts on MPA features and are assessed as ‘minimal concern’. This impact pathway is not relevant (not applicable) to hard substratum habitats. Loss of filter feeding bivalves is assessed through the impact pathway ‘indirect impacts through interactions with other species’.

Physical impact on ecosystem: No physical impacts are known, there may be limited changes in sediment characteristics due to burrowing and bulldozing movements similar to bioturbation by urchins (Lohrer *et al*., 2005) but sediment reworking is not considered to lead to impacts on MPA features and are assessed as ‘minimal concern’. This impact pathway is not relevant (not applicable) to hard substratum habitats. Loss of biogenic structure is assessed as a structural impact (below).

Structural impact on ecosystem: *R. venosa* is a predator of bivalves (Sewell & Sweet, 2011; Zenetos, 2016). A loss in habitat forming bivalves could impact habitat structure and therefore refugia for a diversity of marine creatures (Sewell & Sweet, 2011). Savini and Occhipinti-Ambrogi (2006) found that *R. venosa* showed selective predation when offered three separate bivalve species as prey items. This prey selectivity could alter local community structure amongst filter feeding bivalves which may have long-term ecological effects (Savini & Occhipinti-Ambrogi, 2006). Loss of piddocks would alter the structure of clay and soft rock habitats by reducing boring. The small holes created by piddocks provide shelter for other species and increase species diversity but also result in erosion and loss of the habitat.

This impact pathway is considered relevant only to MPA features that are characterized by prey items that create biogenic habitats, for all other features the assessment was ‘Not applicable’.

Rapana venosa: Interactions with MPA Features

MPA features that provide suitable habitat for *Rapana venosa*

- A5.2 Sublittoral sands are a favoured habitat (Sewell & Sweet, 2011) (Moderate confidence)

MPA habitats considered potentially suitable for *Rapana venosa*

R. venosa is a habitat generalist and is found on subtidal rock and sediment habitats. Little information was found to assess wave exposure, tidal current tolerances and the majority of MPA features were identified as potentially suitable rather than definitely suitable (confidence is Medium).

Rock habitats

- A3.1 Atlantic and Mediterranean high energy infralittoral rock;
- A3.2 Atlantic and Mediterranean moderate energy infralittoral rock;
- A3.3 Atlantic and Mediterranean low energy infralittoral rock;
- A4.1 Atlantic and Mediterranean high energy circalittoral rock;
- A4.2 Atlantic and Mediterranean moderate energy circalittoral rock ;
- A4.3 Atlantic and Mediterranean low energy circalittoral rock;
- Estuarine rocky habitat (Not the intertidal biotopes A1.32);
- Fragile sponge and anthozoan communities on subtidal rocky habitats;
- Tide-swept channels; and
- Carbonate reefs.

Sediment:

- A5.1 Sublittoral coarse sediment;
- A5.3 Sublittoral mud;
- A5.4 Sublittoral mixed sediments;
- Subtidal mixed muddy sediments;
- Sheltered muddy gravels;
- Mud habitats in deep water;
- Sea-pen and burrowing megafauna communities; and
- *Sabellaria alveolata* reefs.

Biogenic and Vegetated habitats:

- A5.5 Sublittoral macrophyte-dominated sediment;
- Seagrass beds (Not the intertidal biotope A2.61);
- *Zostera* beds (Not the intertidal biotope A2.61);
- A5.6 Sublittoral biogenic reefs;
- Blue mussel beds (Not the intertidal biotopes A2.212; A2.72);
- Maerl beds;
- *Modiolus modiolus* beds;
- *Musculus discors* beds; and
- Ostrea edulis beds.

MPA habitats considered unlikely to be suitable for *Rapana venosa*

- A3.2 Atlantic and Mediterranean moderate energy infralittoral rock

MPA features unsuitable for *Rapana venosa*

All intertidal MPA features were considered unsuitable for *R. venosa*, although storms may displace individuals into the intertidal (ICES, 2004). Confidence is high

Summary of the impacts on the MPA features.

Subtidal MPA features characterised by prey species were considered to be most at risk from *R. venosa*. Predation could result in the loss of mussel beds (M. edulis and *M. modiolus*) and the structure and functions these provide to the marine ecosystem such as role in filter feeding and providing habitat. *R. venosa* could also reduce the abundances of characterising and typical bivalve species in other habitats altering the structure of the biological assemblage and reducing availability of prey for other species such as fish and crabs. Recovery of impacted habitats would rely on removal of *R. venosa*.

Rapana venosa: Socio-Economic Impacts

Health and Safety: Not applicable. *R. venosa* does not pose any direct threats to humans. In some Risk Assessments fouling of nets by type of impact has been considered a possible health risk due to hazards associated with increased lifting such as back injury, danger of small vessels capsizing etc. (Jack Sewell, pers. comm.).

Aquaculture Operations: This species has a significant impact on squid fisheries in the Adriatic. *R. venosa* uses the squid nets as spawning substratum by crawling inside. They tend to occupy all the available net space and create considerable extra load and weight (ISSG, 2007). This tendency to use nets as spawning substrate would cause aquaculture operators considerable time, energy and monetary loss for clearance. There are also health and safety risks associated with the extra weight when lifting gear (see health and safety below). Egg laying is seasonal, reportedly lasting 10-12 weeks (Saglam and Duzgunes, 2007; Harding *et al*., 2007) to longer periods (March-September, ICES, 2004). Impacts from fouling are assessed as 'Moderate' at low confidence.

Aquaculture cultivated species: *R. venosa* has caused significant effects on the ecosystem in its introduced ranges (ISSG, 2007). *R. venosa* is a known predator of bivalves (Zenetos, 2016; Zolotarev, 1996) and as such could have a negative impact on mussel, *Mytilus edulis*, and oyster, *Crassostrea gigas* (Kerckhof *et al*., 2006), and *Ostrea edulis* (Zenetos, 2016) aquaculture activities. They are known to have negative impacts on commercial shellfish both in their native and non-native ranges (Harding *et al*., 2008). They predate upon bivalve spat as well as preying upon the adults (ICES, 2004; Zenetos, 2016). While mussels cultivated on long-lines that don't touch the bottom (Savini and Occhipinti-Ambrogi, 2006), structures such as trestles could be accessible (based on presence in artificial habitats). Consumption of cultivated molluscs by *R. venosa* would result in losses and impose costs to target and remove individuals. The impact is assessed as ‘Moderate’ for off-bottom cultivation and ‘Major’ for on-substrate cultivation of *M. edulis*. Confidence is low due to uncertainty around the magnitude of impact.

Oysters-*Ostrea edulis*, *Magallana gigas*: Impacts are mediated by density of *R. venosa* could impact oyster beds through predation and over time or at higher densities impacts may be severe. The impact was assessed as ‘Major’ with medium confidence.

In Chesapeake Bay, USA the striped hermit crab, *Cilbanarius vittatus,* now takes refuge in the large, empty shells of *R. venosa*. This has allowed them to expand their range and to become larger due to the large shell size now at their disposal (ICES, 2004). The larger than usual, striped hermit crabs have demonstrated the ability to consume a significant amount of oyster spat (ICES, 2004). If this were to happen around the UK with the native hermit crab then there could be a significant impact on local oyster populations both wild and farmed.

Fisheries Operations: No negative impacts on static or mobile gears were reported although it seems likely that *R. venosa* may be caught by mobile gears and may enter static pots. In the reproductive season, pots and gears may be fouled by egg cases, as this impact is seasonal and may be limited depending on length of deployment impacts are assessed as ‘Minor’ at low confidence.

Fisheries Target Species: As a bivalve predator, at high densities *R. venosa* could impact mollusc fisheries such as cockles *Cerastoderma edule* (Kerckhoff *et al*., 2006) and clams, scallop through predation and whelks (through competition). Impacts on scallops and cockles are assessed as ‘Moderate’ (at low confidence) due to scallop escape mechanisms and burial of cockles. Subtidal *M*. *edulis* beds are likely to be more vulnerable as they are epifaunal, occur at high densities and are sessile, impacts on this species was assessed as ‘Major’.

R. venosa is acclimated to estuarine/brackish waters of coastal regions where intensive bivalve harvesting usually takes place and as such is considered a serious pest to bivalve fisheries (Savini *et al*., 2007).

On the Turkish coast of the Black Sea a fishery has begun to exploit this new, invasive species. Aydin *et al*. (2006) reported 207 fishing vessels fishing for whelks.

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Annex 16. American oyster drill: *Urosalpinx cinerea*

Common name(s): American oyster drill; American (whelk) tingle; Atlantic oyster drill; Eastern (oyster) drill[10].

Synonyms: *Urosalpinx cinerea var. follyensis*; *Fusus cinereus* (Worms); *Buccinum plicosum*; *Fusus cinereus*; *Fusus imbricatus*; *Urosalpinx aitkinae*; *Urosalpinx follyensis*; *Urosalpinx cinereus* (Fofonoff *et al*, 2018).

Domain: Phyla: Mollusca, Class: Gastropoda, Order: Neogastropoda, Family: Muricidae, Genus/species: *Urosalpinx cinerea* (WoRMS[11])

Description: The shell is up to 4 cm high and 2 cm broad. It is conical, with 7-8 whorls and has a pointed spire. The shell is cream or grey sometimes with brown markings (Oakley, 2006).

(© Jack Sewell)

Figure 16.1. American oyster drill: *Urosalpinx cinerea*

Urosalpinx cinerea: Habitat

Native range: The native range of *U. cinerea* is the east (Atlantic) coast of North America, northwards to Newfoundland and southwards to northeast Florida (Abbott, 1974, cited from Faasse, 2011). *U. cinerea* has been introduced to the west (Pacific) coast of North America, where it has been reported from San Francisco Bay in 1890 (Cohen, 2005). It introduced to the UK in or prior to 1920 with Atlantic oysters, *Crassostrea virginica*. (Cole, 1942; Carriker, 1955).

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[10] https://www.cabi.org/isc/datasheet/60187

[11] World Register of Marine Species http://www.marinespecies.org/aphia.php?p=taxdetails&id=140429

Substratum type: Found inhabiting shallow subtidal and mid-lower intertidal habitats in bays and estuaries (Cole 1942, Carriker 1955), intertidal mudflats and the sublittoral (Faasse, 2011 and authors therein). When inhabiting estuaries they prefer the shallow, muddy creeks rather than the main estuary channels or the gravelly sand substratum (Cole, 1942).

They are also associated with other bivalves including mussels (Sweet, 2011) and artificial structures such as marinas and docks (Faasse, 2011 and authors therein). In their native range they inhabit rocky and shell bed areas below the mid-tidal line (Williams *et al*., 1983) and are known to tolerate very turbid conditions (Faasse, 2011 and authors therein). They have been found to spawn on the underside of boulders, which provide microhabitats suitable for young *U. cinerea* (Faasse, 2011), as juveniles are able to prey upon oyster spat and barnacles as soon as they emerge from their egg capsules (Carriker 1955).

Salinity: *Urosalpinx cinerea* is tolerant of low salinity. It can be found in estuarine conditions and is typically found in areas closer to rivers than estuary mouths (Buhle *et al*., 2009, Cole 1942). Reported range is 13-40 PSU (Cohen, 2011; Sweet, 2011; Fofonoff *et al*, 2018).

Depth: *U. cinerea* is found from the mid intertidal to 36m depth (Carriker, 1955 and references therein). In Georgia, they were common in the lower portions of intertidal oyster beds, but rarely reached the upper regions (Walker 1971).

Wave exposure: *U. cinerea* are found in sheltered areas away from wave exposure (Franz, 1971).

Urosalpinx cinerea: Establishment in Wales as of 2020

U. cinerea has not been recorded in Wales to date.

Aquaculture is the key vector for dispersal of *U*. *cinerea* (Carriker 1957, Cole, 1942, Faasse and Ligthart, 2009). *U. cinerea* does not have a free swimming larval phase, local populations increase rapidly as dispersal is limited (Sweet, 2011). Their numbers can be high in their native range (Hancock, 1954) and can be much higher when introduced outside of their native range, possibly due to a lack of predators and parasites in non-native ranges (Cole, 1942).

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Urosalpinx cinerea: Impact pathways based on species traits, biology and ecology

Competition: The American oyster drill may compete with native molluscs such as the dog whelk *Nucella lapillus* and *Ocenebra erinacea* (Cole, 1942; Orton and Lewis,

1931) but may be predated on by native *Polinices* spp. (Flowers, 1954). No native oyster drills or other carnivorous snails characterize MPA features, therefore the replacement of these with the functionally similar *U. cinerea* is not considered likely to lead to the loss or reclassification of MPA features through competition. This pathway is assessed as ‘minimal concern’ for all MPA features. Note: predation on native fauna and aquaculture species is assessed separately below.

Predation: *U. cinerea* is an active predator. Recorded prey species include at least 20 species of bivalves (including oysters and mussels), gastropods, barnacles, bryozoans, and small decapod crustaceans identified with prey preferences varying between populations (Carriker, 1955; Williams *et al*., 1983; Fasse, 2011 and authors therein; Fasse & Lighthart, 2007).

U. cinerea can begin to consume bivalve spat upon hatching (Pope, 1910, cited from Carriker, 1955) and large numbers of newly settled individuals may be consumed (Cole, 1942). Eventually, surviving bivalves will reach a size refugia from predation as they mature and the shells thicken (Carriker, 1955; Mackenzie, 1977; Lord and Whitlatch, 1991). *U. cinerea* has a negative effect on native oysters and commercial oyster beds (Carriker, 1955; Oakley, 2006); each individual could consume about 40-60 oyster spat per year (Cole, 1942; Hancock 1954) leading to significant losses (Carriker, 1955).

Predation impacts were considered to be of minimal concern in MPA features where bivalves or other prey such as barnacles were not characterising species and/or establishment was considered unlikely. Confidence was assessed as medium for these biotopes as little evidence was available to assess establishment.

Predation impacts were considered to be minor where some loss of native species was possible but these are not preferred prey (for example barnacles, burrowing bivalves) or where densities of *U. cinerea* are likely to be low. Confidence is low due to lack of specific evidence.

Predation impacts were considered to be moderate for MPA features characterized by barnacles, *Mytilus edulis* and *Ostrea edulis* as many adults would be expected to have reached a size where predation is limited (see below: impacts on key MPA features and socio-economic impacts for further information). However, at high densities and over longer time periods *U. cinerea* may reduce recruitment and impacts could be more severe. In Willapa Bay, Washington, USA, predation by *U. cinerea* is a factor inhibiting restoration of the native Olympia oyster (Buhle and Ruesink, 2009). The confidence assessment of low for these MPA features reflects uncertainty in the severity of the long-term impact and the suitability of the habitat for the species to become established in high densities.

Hybridisation: No evidence was found for hybridisation with native species and this impact pathway is assessed as ‘Not applicable’.

Transmission of disease: No records of disease transmission by *U. cinerea* were found in the literature and this impact pathway is considered to be ‘Not applicable’.

Parasitism: *U. cinerea* is not a parasite and this impact pathway is assessed as ‘Not applicable’. *U.cinerea* has been found to host the flatworm trematode larvae *Cercaria sensifera.* It has been found in the reproductive and digestive glands of individuals in the UK as well as in its native range. This parasite is also known to live in the native gastropod, *Nucella lappilus* (Stunkard & Shaw, 1931). For this reason is it unlikely to have any negative impact on any native species since it is already found in *N. lappilus*. Other parasites found to live inside *U.cinerea* are *Hoploplana inquilina* (a flatworm) and (Fasse, 2011 and authors therein) and an unidentified, highly specialised arthropod (probably an isopod) living in the liver region (Cole, 1942). There is no evidence that these parasites could have any impacts on native species in Welsh MPAs.

Poisoning/toxicity: *U. cinerea* predates on organisms by drilling with a radula and secreting acids to aid drilling through shells. It is not poisonous or toxic and this impact pathway is assessed as ‘Not applicable’.

Bio-fouling: *U. cinerea* is a mobile epifaunal species and is not a bio-fouler, this impact pathway is assessed as ‘Not applicable’.

Grazing/herbivory/browsing: *U. cinerea* is a predator not a herbivore and this impact pathway is assessed as ‘Not applicable’.

Indirect impacts through interactions with other species: Through its consumption of reef forming bivalves *U. cinerea* may indirectly alter the biogenic structure of habitats if it becomes established in mussel and oyster habitats. It is known to feed on mussels and oysters and in its invasive range has been documented in high concentrations. This impact is assessed through the ‘Structural impact on ecosystem’ pathway below.

Urosalpinx cinerea: Impact pathways – Habitats

Chemical impact on ecosystem: Changes in filter feeder density through predation by *U. cinerea* may result in indirect impacts on biogeochemical cycling within the ecosystem. Such impacts would be density dependent and site-specific. No direct impacts on the ecosystem were found in the literature and this impact pathway is considered ‘Not applicable’.

Physical impact on ecosystem: As a small, mobile, epifaunal species *U. cinerea* is not anticipated to cause direct physical impacts on the ecosystem and this pathway is ‘Not applicable’.

Structural impact on ecosystem: As a small, mobile, epifaunal species *U. cinerea* is not anticipated to cause direct structural impacts. Indirectly this species may alter habitat structure through the removal of biogenic reef forming species (mussels and oysters). Predation was not considered to result in the loss of the entire oyster reef or mussel bed and impacts were assessed as ‘Moderate’ at low confidence.

Urosalpinx cinerea: Interactions with MPA Features

MPA features that provide suitable habitat for *Urosalpinx cinerea*

The following MPA features were considered suitable for *U. cinerea* based on reports in similar habitats. Confidence is high for all assessments:

Intertidal boulder communities: Known to inhabit underside of boulders which is their preferred microhabitat for spawning (Faasse, 2011 and authors therein);

- A5.4 Sublittoral mixed sediments:
- A5.435 Based on association with oysters and habitat preferences (Faasse, 2011 and authors therein);
- Sheltered muddy gravels
- A5.435 Based on association with oysters and habitat preferences (Faasse, 2011 and authors therein);
- A2.7 Littoral biogenic reefs:
- A2.72 Littoral mussel beds on sediment. Based on association with mussels and habitat preferences Fasse & Lighthart, 2007;Sweet, 2011);
- Blue mussel beds;
- Intertidal *Mytilus edulis* beds on mixed and sandy sediments;
- *Ostrea edulis* beds; and
- Peat and clay exposures.

MPA habitats considered potentially suitable for *Urosalpinx cinerea*

The following MPA features or constituent biotopes where these are listed, were considered potentially suitable for *U. cinerea*. Assessments were typically based on exposure, sediment or substratum and the presence of suitable prey, however, there was some uncertainty and confidence is indicated for each feature:

- A2.5 Coastal saltmarshes and saline reedbeds. Found in riverine influenced creeks, where infested oysters have been laid (Cole, 1942) (Medium confidence);
- A1.2 Moderate energy littoral rock. Based on presence intertidally on rocks and association with mussels (Faasse, 2011 and authors therein). (High confidence);
- A1.3 Low energy littoral rock. Based on presence intertidally on rocks, in sheltered areas (Faasse, 2011 and authors therein). (High confidence);
- A4.2 Atlantic and Mediterranean moderate energy circalittoral rock.
    - A4.24 and A4.25 in estuaries or bays with shallow, variable salinity habitats where prey items are available (Faasse, 2011). (Medium confidence).
- Estuarine rocky habitat:

  - A1.3 Based on presence intertidally on rocks, in sheltered areas (Faasse, 2011 and authors therein) (High confidence);
  - A3.361 is tide swept but contains prey. (Low confidence);
- Tide-swept channels:
  - A4.25 as prey items are available (Faasse, 2011). (Medium confidence).
- A2.2 Littoral sand and muddy sand.
  - A2.24 Polychaete/bivalve-dominated muddy sand shores: where these occur in estuaries. (Low confidence).
- A2.3 Littoral mud.
  - A2.31 Based on habitat (estuarine and littoral sediment) and prey preferences (Sweet, 2011). (Medium confidence).
- A2.4 Littoral mixed sediments. Based on preferences for sheltered habitats and availability of prey. (Low confidence).
- A5.2 Sublittoral sand.
- A5.21 Based on habitat preferences (Faasse, 2011 and authors therein). (Low confidence).
- Sheltered muddy gravels:
  - A2.42 Based on habitat and prey preferences (Faasse, 2011 and authors therein). (Low confidence);
- A5.6 Sublittoral biogenic reefs:
  - A5.625 and A5.624 could be suitable biotopes for U. cinerea due to its dense mussel beds and its sheltered muddy shell bottoms (Cole, 1942). (Low confidence);
- Modiolus modiolus beds.
  - A5.624 could be a suitable biotope for U.cinerea due to its dense mussel beds and its sheltered muddy shell bottoms (Cole, 1942). (Low confidence); and
- Peat and clay exposures :
  - A1.127 Based on habitat preferences (Sweet, 2011). (Medium confidence).

MPA habitats considered unlikely to be suitable for *Urosalpinx cinerea*

Based on depth, wave exposure and/or water currents, macroalgal cover and a lack of prey, the following MPA features were considered unlikely to be suitable (at low confidence):

- A3.2 Atlantic and Mediterranean moderate energy infralittoral;
- A3.3 Atlantic and Mediterranean low energy infralittoral rock;
- Estuarine rocky habitat;
- Fragile sponge and anthozoan communities on subtidal rocky habitats; and
- Tide-swept channels.

Based on habitat factors such as sediment substratum, depth and a lack of prey items, the following sediment MPA features were considered unlikely to be suitable:

- A2.4 Littoral mixed sediments (Low confidence);
- A5.2 Sublittoral sand:
    - A5.22; A5.23; A5.24; A5.25; A5.26; A5.27. (Medium confidence);
- A5.4 Sublittoral mixed sediments:
    - A5.41; A5.42 A5.43; A5.44. (Medium confidence);
- Sheltered muddy gravels:
    - A2.41; A5.432; A2.421. (Low confidence);
- Sea-pen and burrowing megafauna communities (Low confidence);
- A5.6 Sublittoral biogenic reefs:
    - A5.622 too deep and found in open waters (Sweet, 2011). A5.623 is found mainly in Scotland; A5.621 may have currents that are too strong (Low confidence);
- Modiolus modiolus beds:
    - A5.622 too deep and found in open waters (Sweet, 2011); A5.623 is found mainly in Scotland; A5.621 may have currents that are too strong; and
- Musculus discors beds (Low confidence).

MPA features unsuitable for *Urosalpinx cinerea*

Wave and current exposed, offshore and deep MPA features are considered unsuitable for this species which is largely restricted to estuaries and sheltered bays. The following habitats were considered unsuitable:

- A1.1 High energy littoral rock (Medium confidence);
- A3.2 Atlantic and Mediterranean high energy infralittoral rock (Medium confidence);
- A4.1 Atlantic and Mediterranean high energy circalittoral rock (Low confidence);
- A4.2 Atlantic and Mediterranean moderate energy circalittoral rock;
    - A4.21, A4.22 more typical of open sea situations, A4.23 lack of suitable prey;
- A4.3 Atlantic and Mediterranean low energy circalittoral rock (Low confidence);
- Tide-swept channels:
    - A1.15 (Medium confidence);
    - A4.1; A5.5211 (Low confidence);
- A2.3 Littoral mud:
    - A2.32 Polychaete/oligochaete-dominated upper estuarine mud shores, lack of suitable prey (Low confidence)
- A5.1 Sublittoral coarse sediment. (Medium confidence);
- A5.3 Sublittoral mud (Medium confidence);
- Subtidal mixed muddy sediments; and

- Mud habitats in deep water (Low confidence).

MPA features that occur high on the shore, lack suitable prey or are highly mobile were considered unsuitable:

- A2.1 Littoral coarse sediment (High confidence);
- A2.2 Littoral sand and muddy sand:
    - A2.21 Strandline-height on shore. (Medium confidence);
    - A2.22 Barren or amphipod-dominated mobile sand shores. (High confidence); and
    - A2.23 more typical of fully marine habitats (Medium confidence).

MPA features with no evidence

No evidence or suitable proxies were identified to assess whether the following MPA features were suitable habitats or not:

- *Sabellaria alveolata* reefs;
- A5.5 Sublittoral macrophyte-dominated sediment;
- A2.6 Littoral sediments dominated by aquatic angiosperms;
- Seagrass beds;
- Zostera beds;
- A5.7 Carbonate reefs;
- Maerl beds;
- Littoral biogenic reefs:
    - A2.71 Littoral *Sabellaria* reefs;
- A5.6 Sublittoral biogenic reefs:
    - A5.61 Circalittoral coral reefs, A5.61 Sublittoral polychaete worm reefs on sediment.

Summary of the impacts on the MPA features.

The main ecological concerns regard predation of oysters and mussels, with intertidal and shallow subtidal bivalve reefs potentially impacted. For habitats where mussels and oysters are not predated or are predated at low rates due to the size of mature individuals, consumption of juveniles preventing natural recruitment may over time result in a loss of reefs.

Urosalpinx cinerea: Socio-Economic Impacts

Health and safety: No impacts are known.

Aquaculture Operation: Control of infestations will impose costs on operations through requirements to inspect and remove individuals on seed oysters, and inspection and regulation of oyster transfers and culture equipment. Control of drills in infested areas involves raking the ground, removing debris, and other labor-intensive tasks (Quayle 1969). Impact on operations through clearing costs is assessed as ‘Moderate’ at low confidence.

Aquaculture cultivated species: Oysters-*Ostrea edulis*, *Magallana gigas and mussels*. *U. cinerea* could cause negative impacts on shellfisheries, in particular oyster and mussel cultivation on substrates or off-bottom through predation. The shellfish industry is likely to be impacted from the establishment of *U. cinerea*. The feeding activities of the American oyster drill can decimate commercial oyster populations, 50 % mortalities among oyster spat directly attributable to this predatory snail were commonly reported from Essex oyster beds before *U. cinerea* populations declined during the 1980s (Cole 1942, Carriker, 1955). Impacts on cultivated species are considered to be ‘Moderate’, however, Impacts will likely be density dependent. Higher densities of this shellfish predator, or over time, will see higher mortalities. The impact was assessed as ‘Moderate’ with medium confidence as larger individuals less susceptible to predation coupled with regular control would allow the activity to continue. where infestations are severe and cannot be controlled, activities may be abandoned and the impact would be higher (Major- Massive).

Fisheries Operations: *U. cinerea* is captured in some dredges and its retention in mobile gears will depend on mesh size. *U. cinerea* is strongly associated with prey species and would not be expected to be present in high densities where these are absent. There is no suggestion that *U. cinerea* would have an impact on fisheries operations by preventing gear deployment and it is therefore considered to be of ‘Minimal concern’ (high confidence) for fishery operations (but see target species below).

Targeted species-Finfish, crustaceans and whelk: Minimal concern (high confidence). No interaction was predicted, no commercially targeted species are dependent on bivalves and *U. cinerea* is not considered to modify nursery or feeding habitats.

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Annex 17. American lobster *Homarus americanus*

Common name(s): American lobster; Maine lobster; northern lobster; Atlantic lobster; rueu lobster; Canadian Reds; Canadian lobster (NBNatlas, 2017; FAO, 2019).

Synonyms: *Astacus marinus; Astacus americanus; Homarus mainensis* (FAO, 2019).

Domain: Phyla: Anthropoda, Class: Malacostraca, Order: Decapoda, Family: Nephropidae, Genus/species: *Homarus americanus* (WoRMS, 2019).

Description: This is a clawed lobster that resembles the native European lobster, *Homarus gammarus*. It is larger, up to 64 cm in length and around 2 kg in weight (NBN atlas, 2017), with a green/brown body colour although several colour morphs have been observed (van der Meeren *et al*., 2010; NBN atlas, 2017). Its walking legs have a green tinge to them while the claws, spine tips and underside are red/brown in colour. This is in contrast to the European lobster which has a cream/white colour to its spine tips and coalescing spots on a blueish carapace with a yellowish underside (Hayward & Ryland, 2017; Stebbing et al., 2012a). It has a characteristic small spine on the ventral margin of the rostrum which is often used to distinguish it from the European lobster (*H. gammarus*), however the latter does also occasionally have these spines (Holthuis, 1991; Stebbing et al., 2012a). While molecular testing has been employed in Norway to definitively identify *H. americanus*, Addison & Bannister (1994) suggest there are many fewer UK native lobsters (*H. gammarus*) with the same lower rostrum spines as *H. americanus*.

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Figure 17.1. American lobster. *Homarus americanus*

Homarus americanus: Habitat

Native range: In their native range they are found in a wide variety of habitats. Populations living inshore are found on shallow water on mud, cobble, bedrock, peat reefs, eelgrass beds and occasionally in sandy depressions (Lawton & Lavalli, 1995). Habitats that provide shelter for juveniles, such as cobbles are important and availability may limit recruitment (Wahle and Steneck, 1991). Offshore populations are found on similar substrates including clay but are at a greater depth.

Substratum type: In general, *H. americanus* prefers hard substrates to live on and inhabits rocky and hard mud habitats (FAO, 2019). In their invasive ranges they are found in habitats consisting of rock, sand, pebble and mud which are near the coast at a depth of 10-50 m (van der Meeren *et al*., 2010). Allen and Van der Meeren, (2012) state that *H. americanus* can be found inhabiting coastal, intertidal and littoral areas including littoral mud flats as well as brackish estuaries.

Benthic recruitment habitats and nursery grounds of *H. americanus* were identified in the Gulf of Maine, USA. Settlement size and early benthic stage individuals were primarily restricted to cryptic and shelter providing substratum which tended to be subtidal cobble containing vegetation. They were not found in ledge and sedimentary substrate without vegetation, conversely adult individuals were found in these habitats (Wahle & Steneck, 1991). May (2015) found that juvenile *H.gammarus*, prefer complex structured habitats that allow them to hide, while adults were habitat generalists.

Salinity: Adults: Usually found in > 25 ppt, optimum salinity is 30-35 ppt (Allen & van der Meeren, 2012; Lawton & Lavalli, 1995; van der Meeren; 2010). They can survive limited periods of low salinity (Lawton & Lavalli, 1995) and are reportedly able to tolerate a reduced salinity of >8-14 ppt (Allen & van der Meeren, 2012). Larvae: The optimum salinity for the larval stages is 30-31.5 ppt (Allen and van der Meeren, 2012).

Depth: Littoral fringe to 480 m but more commonly found 4-50 m (Holthuis, 1991; FAO, 2019). In their invasive range they are caught at depths of 10-50 m near the coast (van der Meeren *et al*., 2010). These depths are usual fishing depths around the coast of the UK which could be causing sampling bias. It may be that *H. americanus* occur in similar numbers at other depths which are not being surveyed.

Wave exposure: There is little evidence in the literature regarding the tolerance of *H. americanus* to wave exposure. However, they do need water flow through their burrows for oxygenation (Lawton & Lavalli, 1995). Tolerance for wave exposed biotopes is based on the European lobster (*Homarus gammarus)* (JNCC, 2015) as both species live in similar habitats.

Homarus americanus: Establishment in Wales 2020

There is no definitive evidence to suggest that *H. americanus* has established breeding populations in UK waters, however, individuals have been recorded, with two adult specimens recovered from northern Wales (2016 and 2017). No Welsh records of establishment to date (NBN atlas, 2017; Stebbing et al., 2012a).

There have been incidents where individuals have been released into the sea around the UK. An incident in Brighton involved the release of 361 live American lobsters as part of a Buddhist ceremony. The Buddhists involved were unaware that the species was a non-native species and local fishermen helped to retrieve some of the crustaceans from the Sussex coast (Gov.uk, 2017). Not all the released individuals were re-captured and some female lobsters retrieved were found to be carrying viable eggs (Stebbing pers comms). Released individuals like these pose a threat to the native population, *H. gammarus*.

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Homarus americanus: Impact pathways based on species traits, biology and ecology

Competition: *H. americanus* have the potential to be competitively superior to *H. gammarus* through their larger size, aggressiveness and greater fecundity (Stebbing et al., 2012a). The two lobster species have the same food preferences (Chapman, 1980; Lawton & Lavalli, 1995; Collins, 1998) and it is possible that *H. americanus* could out-compete *H. gammarus* for food and shelter resources *H. americanus* is reportedly more aggressive than *H. gammarus* and appears to be more dominant in shelter conflicts (van der Meeren *et al*., 2010). Juvenile *H. americanus* have been shown to out competed *H. gammarus* for both shelter and food under laboratory conditions, with *H. americanus* predating on *H. gammarus* at times during the trails despite being fed ad libitum (Stebbing pers comms).

It is also possible that *H. americanus* will compete with *Cancer pagurus*, the edible crab, because their niches overlap (Stebbing et al., 2012a). It could also have a competitive impact on other large crustaceans of both commercial and environmental importance (Stebbing et al., 2012a).

The larvae and postlarvae of *H. americanus* feed on zooplankton (NOAA, 2019) and may compete with filter feeders.

The impact of competition on MPA seabed habitat features is assessed as 'Minimal concern' as the species with which *H. americanus* competes are not characterizing species of any MPA feature (for species assessments see the fisheries section, below). Confidence is low due to the lack of evidence.

Predation: *H. americanus* is omnivorous (Allen & Van der Meeren, 2012) eating a variety of prey items. They are opportunistic hunters and feeders resulting in regional diets. Adult *H. americanus* diet can include starfish, molluscs, crabs, fish, sea urchins, congeners and macroalgae. Their larvae and postlarvae are carnivorous and eat zooplankton during their first year (NOAA, 2019). No evidence, due to a lack of relevant studies, have been found to suggest that their diet has any negative impacts on any native taxa that characterise the MPA features and this impact pathway is classed as 'Minimal concern', confidence in the assessment is low.

Hybridisation: Hybridisation with the native lobster, *Homarus gammarus*, has been recorded in Norway (A.-L. Agnalt pers. comm, cited in Stebbing et al., 2012a) and experimentally induced in aquaria. Hybrid eggs have also been recovered from female *H. americanus* found in UK waters (south coast of England) (Stebbing pers comms). Hybrids when backcrossed with European lobsters within the laboratory produced infertile males and fertile females which suggest genetic alteration of native lobsters is potentially possible if the two species interbreed (van der Meeren *et al*., 2010; Stebbing et al., 2012a and references therein). Talbot *et al*. (1984) found lobsters have a preference to mate with conspecifics when given the choice. Similarly, van der Meeren *et al*., (2008), found that female European lobsters chose male European lobsters over American lobsters to court and mate with, even if the latter was dominant. It should be noted that *H. gammarus* is not a characterizing species of any of the MPA features so would not result in loss or reclassification of these, the impact is assessed as 'Minimal concern' with high confidence based on seabed habitat features only. An impact assessment for *H*. *gammarus* populations is presented in the fisheries section below.

Transmission of disease: There is a significant risk of spread of disease to the native lobster. *H. americanus* is susceptible to WSSV (white spot syndrome virus) and is a carrier of epizootic shell disease, Gaffkaemia (red tail disease) and there is potential for it to be a carrier of other unknown diseases (Cawthorne, 2011). A spread of these diseases in European waters could have a serious impact on local fisheries and a worst case scenario could close them completely (van der Meeren *et al*., 2010).

<u>Gaffkaemia</u> (red tail disease) is a lethal bacterial disease that originates in North America and can be carried by some American lobsters (Stewart *et al*., 1996. Imported *H. americanus* in holding pens in Europe have been found to be infected (Wiik *et al*., 1987; Mortensen, 2002) and high mortalities may result due to crowding (Stebbing et al., 2012a). The disease has subsequently spread to wild populations and has been reported in Norway and the Orkneys (Stebbing *et al*., 2012a and references therein). Although it is 100% lethal in European lobsters, in wild purebred European lobster populations, the disease has little chance to spread, as the lobsters die within days after contamination (van der Meeren *et al*., 2010). Gaffkaemia (*Aerococcus viridans var. homari*) has been found in several native populations of *H. gammarus* in the UK, and it was determined had been introduced by *H. amaericanus*,

but the prevalence was low (Stebbing *et al*., 2012b). The presence of *H. americanus* and/or hybrids may however increase the risk of spread and mortality.

Epizootic Shell Disease (ESD) is a disease found on American lobsters and is suggested to be caused by the bacterium *Aquamarina* (Kircun, 2019). The shell becomes infected, if the shell can be moulted in time then the disease is lost with the moulted shell. If the disease penetrates though the shell and further through the epidermal skin layer below then the bacterium will get into the circulatory system and kills the lobster (Kircun, 2019). Adults moult less frequently than juveniles leaving them more vulnerable. Egg carrying females delay their moult by a further six months leaving them even more susceptible to the disease. As reproductively successful females are the individuals needed to sustain a population it is worrying that these are the adult individuals that are most susceptible to this disease (Kircun, 2019). This disease has been spotted in *H. americanus* in Norwegian waters (van der Meeren *et al*., 2010).

White spot syndrome virus (WSSV), so called by the white spots that appear on the carapace of infected individuals of penaeid shrimp, is known to be the most economically devastating viral pathogen to shrimp aquaculture production globally (Clark *et al*., 2013). Not all infected individuals will show signs of disease and diagnosis requires laboratory testing (Anon, 2013). It is thought that this virus could infect all decapod crustaceans including the European lobster, *H. gammarus* (Clark *et al*., 2013). The disease is spread between crustaceans by cannibalism and contaminated water while humans can spread the disease by moving infected live or dead crustaceans, contaminated aquaculture gear or water (Clark *et al*., 2013).

For MPA seabed features characterised by burrowing decapod crustaceans such as *Nephrops norvegicus* and other burrowing decapods, the impact of disease was assessed as ‘Moderate’ as populations of these species may decline. Impacts to *C. pagurus* and *H. gammarus* are discussed in the fisheries section (below). Assessing risk is subject to inherent uncertainty regarding prevalence, spread and mortality rate and confidence is low.

Parasitism: While *H. americanus* can be a carrier of parasites (van der Meeren *et al*., 2010) it does not parasitize other taxa. This impact pathway is considered ‘Not applicable’.

Poisoning/toxicity: There is nothing to suggest that this species poses a threat to native taxa due to toxicity or poison which is why this pathway has been considered as ‘Not applicable’.

Bio-fouling: It is possible that *H. americanus* could be a vector for biofouling species however, it is not a bio-fouling organism so this pathway is considered ‘Not applicable’.

Grazing/herbivory/browsing: *H. americanus* is a predator so this pathway is classed as 'Not applicable'.

Indirect impacts through interactions with other species: *H. americanus* are often carriers of encrusting species such as barnacles and polychaete worms. There is a significant possibility that these species would also be non-native species or carry disease and could cause their own impacts when introduced (van der Meeren *et al*., 2008). However, there is little in the literature on this issue and it was not possible to make an assessment and 'Data deficient' has been recorded in the EICAT table.

Homarus americanus: Impact pathways – Habitats

Chemical impact on ecosystem: There is no evidence in the literature to suggest that *H. americanus* has any chemical impact on the ecosystem. Therefore, this pathway is considered to be 'Not applicable'.

Physical impact on ecosystem: 'Not applicable'. There are no known physical effects aside from burrowing and sediment movement and these are assessed through 'structural impact' below.

Structural impact on ecosystem: Large decapods alter their environment and sometimes play a key role by supressing grazers and space competitors (Boudreau and Worm, 2012). *H. americanus* digs into and re-engineers the substrate leading to disturbance and alteration (Pottle and Elner, 2011; van der Meeren *et al*., 2010). Since they are long-lived it is possible that this behaviour could have a structural impact on the environment (van der Meeren *et al*., 2010). Little is known about any deleterious effects that *H. americanus* may have on the MPA features. Cooper and Uzman (1980) state that the type of shelter and substrate chosen are similar between *H. americanus* and *H. gammarus* and that the dimensions of shelters excavated or occupies are similar. However, *H. americanus* tend to be more sociable with multiple occupancy of shelters recorded (Cooper and Uzmann, 1980).

As the species appear functionally similar this pathway is recorded as 'Minimal concern' based on functional replacement. Confidence is low due to a lack of empirical evidence for rates of burrowing.

Homarus americanus: Interactions with MPA Features

MPA features that provide suitable habitat

Atlantic and Mediterranean high energy infralittoral rock (A3.1); Atlantic and Mediterranean moderate energy infralittoral rock (A3.2); Atlantic and Mediterranean moderate energy circalittoral rock (A4.3). Both adults and juveniles are found in these rocky habitats which provide ledges and crevices for shelter as well as vegetation which juveniles need as refugia from predators (Allen and Van der Meeren, 2012; Wahle and Steneck, 1991).

Sublittoral coarse sediment (A5.1) Areas of cobble and gravel provide suitable habitat (SwAM, 2016).

Sublittoral mud (A5.3) and Peat and clay exposures are considered suitable habitat for *H. americanus* based on a preference for hard mud substratum to dig and burrow into (Allen and Van der Meeren, 2012; van der Meeren *et al*., 2010; Lawton and Lavalli, 1995). These biotopes would suit adult lobsters more than juveniles since the latter need a more complex habitat structure where they can hide from predators (May, 2015; Wahle and Steneck, 1991).

Subtidal seagrass beds (except A5.545 and A5.5343) and *Zostera* beds (except A5.545) are classed as suitable habitats based on the habitat preferences of *H. americanus*. They have been recorded living in eelgrass habitats (Lawton and Lavalli, 1995). Similarly, sublittoral macrophyte-dominated sediment (A5.5) A5.53; A5.533; A5.5331 and A5.545 are suitable biotopes for *H. americanus* based on their liking of eelgrass as a habitat (Lawton and Lavalli, 1995) which provides a suitable burrowing substratum as well as some shelter camouflage.

MPA habitats considered potentially suitable for *Homarus americanus*

Atlantic and Mediterranean high energy circalittoral rock (A4.1) Rocky habitats provide suitable shelter for both adults and juveniles lobsters (Allen and Van der Meeren, 2012; Yunnie, 2015).

Subtidal estuarine rocky habitat is potentially suitable habitat for adult *H. americanus*. These habitats are reduced salinity (18-30 ppt) and adult *H. americanus* can tolerate lower salinities than this (>8-14 ppt) but with an optimum salinity preference of 30-35 ppt (Allen and van der Meeren, 2012). Larval stages cannot survive these reduced salinities (Allen and van der Meeren, 2012) so it is unlikely that recruiting larvae would settle in these biotopes. Since it is recorded in the literature that adult *H. americanus* are found in these habitats they have been scored with a high confidence.

Tide-swept channels; Sublittoral mixed sediments; Subtidal mixed muddy sediments; Sheltered muddy gravels; Sea-pen and burrowing megafauna communities; Mud habitats in deep water (except A5.371 and A5.372). These biotopes provide suitable burrowing substrates for *H. americanus* (Factor, 1995; Lawton and Lavalli, 1995).

Boulder communities are potentially suitable habitat for postlarvae *H. americanus* due to their size and vulnerability. These habitats are unlikely to be suitable for adults but since small crab species and juvenile crabs are found in these habitats (JNCC, 2015) it is likely that postlarvae and juvenile lobsters could also be found in these two biotopes. Sine there is little in the literature regarding *A. americanus* and these habitats it has been scored with a low confidence.

Sublittoral sand (A5.2) A5.241; A5.242; A5.243; A5.244 Based on habitat preferences (Allen and Van der Meeren, 2012).

Seagrass beds A5.545 and A5.5343 and *Zostera* beds A5.545 are potentially suitable habitats. While *H. americanus* likes eelgrass habitats (Lawton and Lavalli, 1995) these biotopes have reduced salinity which is not optimal for adult *H. americanus* which prefer 30-35 ppt (Allen and van der Meeren, 2012). They can tolerate the reduced salinity which is why it has a medium confidence score.

Sublittoral macrophyte-dominated sediment (A5.5) (except A5.53; A5.533; A5.5331; A5.545 which are suitable biotopes) are considered potentially suitable based on substrate preferences to shelter in/under and to bury in to (NOAA, 2019; van der Meeren *et al.,* 2010). A5.54; A5.541; A5.542 are maerl biotopes. These complex habitats could make potentially suitable habitats for juvenile lobsters (May, 2015). The latter have been scored as low confidence since there is no evidence in the literature for an association between maerl and *H. americanus*.

Sabellaria alveolata reefs; Blue mussel beds; *Modiolus modiolus* beds; Maerl beds; *Musculus discors* beds; *Ostrea edulis* beds are all unlikely habitats for adult *H. americanus* due to lack of suitable substrate to burrow or crawl into. It is possible that post-larvae and juvenille *H. americanus* will find these habitats more suitable as refugia than the adults since they prefer complex structural habitats (May, 2015) as well as using burrows (Factor, 1995). There may be space within these structures for small burrows. Since the literature is scarce for any association between these habitats and *H. americanus* the assessments have been assigned a low confidence.

MPA habitats considered unlikely to be suitable for *Homarus americanus*

Fragile sponge and anthozoan communities on subtidal rocky habitats A4.211; A4.2111; A4.2112. These biotopes are mostly found in Scotland and Ireland which is why it is unlikely they will be found in Wales. Since there is still a likelihood of these biotopes being discovered it has a medium confidence.

Sublittoral sand (A5.2) A5.21; A5.22;A5.221; A5.222; A5.223; A5.23; A5.231; A5.232; A5.233; A5.234 The mobile nature of the sediment in these biotopes are unlikely to be suitable to build burrows in. Since there is little evidence for this in the literature it has been scored with a low confidence.

Sublittoral mud (A5.3) A5.323; A5.324; A5.343; A5.372 based on the soft nature of substratum which is likely to be unsuitable for *H. americanus* since this species prefers hard mud to dig and bury into (FAO, 2019). The assessment has been assigned a low confidence due to the lack of specific evidence and wide range of suitable habitats *H. americanus* inhabits.

MPA features unsuitable for *Homarus americanus*

Littoral (intertidal habitats) are unsuitable (NOAA, 2019; Allen and van der Meeren, 2012), although juveniles may shelter in pools. These habitats may be used for foraging when inundated (Jones and Schulman 2008), but are considered unlikely to be suitable since they are often exposed to the air and are only inundated at high tides. Coastal saltmarshes and saline reedbeds (A2.5); High energy littoral rock (A1.1); Moderate energy littoral rock (A1.2); Low energy littoral rock (A1.3); Littoral coarse sediment (A2.1); Littoral mud (A2.2); Littoral mixed sediments (A2.4); Intertidal *Mytilus edulis* beds on mixed and sandy sediments; Littoral biogenic reefs (A2.7); Intertidal Mytilus edulis beds on mixed and sandy sediments;

Atlantic and Mediterranean high energy circalittoral rock (A4.1) A4.133; Fragile sponge and anthozoan communities on subtidal rocky habitats. The biotope A4.133. is only found in Scottish sea lochs.

Summary of the impacts on the MPA features

Hard muddy habitats provide favourable habitats for adult *H. americanus*, providing stable substrates for burrowing (FAO, 2019). Unlike juveniles, adults are more likely to inhabit deeper, offshore areas that lack crevices and vegetation to hide in (Wahle and Steneck, 1991). Adults are known to travel large distances while juveniles are much more restricted to nursery grounds and travel far less (Wahle and Steneck, 1991). (Relevant habitats: Mud habitats in deep water, Sublittoral mud and Peat and clay exposures).

Rocky areas (Relevant habitats: High energy littoral rock (A1.1); Moderate energy littoral rock (A1.2); Low energy littoral rock (A1.3); Atlantic and Mediterranean high energy infralittoral rock (A3.1); Atlantic and Mediterranean moderate energy infralittoral rock (A3.2); Atlantic and Mediterranean moderate energy circalittoral rock (A4.3)) are favoured by *H. americanus* because they provide suitable habitat. This species requires crevices to hide in and prey items to feed on. *H. americanus* niche preference overlaps with the European lobster (*Homarus gammarus*) and the edible crab (*Cancer pagurus*) (Yunnie, 2015). Both of these species can be found within these habitats also.

(Relevant habitats: Littoral sediments dominated by aquatic angiosperms and Sublittoral macrophyte-dominated sediment (A5.53; A5.533; A5.5331; A5.545).

Homarus americanus-Socio-Economic Impacts

Health and safety: There is a potential for minor injuries caused by claws if handled. Impact was assessed as ‘Minor’ at high confidence.

Aquaculture: Infrastructure: No evidence for impacts, assessed as ‘Minimal concern’ with high confidence.

Aquaculture cultivated species: No evidence for impacts. Note that it is illegal to hold American lobsters in open waters in the UK under the Lobster (Control of

Deposit) Order 1981[12] (Stebbing et al., 2012a). Assessed as ‘Minimal concern’ with high confidence.

Fisheries Operations: No impacts on fisheries operations are anticipated. Pots and traps, fixed nets and passive nets are the main fishing activities used in Wales to catch crustaceans. Assessed as ‘Minimal concern’ with high confidence.

Fisheries Target species:

Native lobster - *Homarus gammarus:* *H. americanus* are imported to the UK where they are held in aquaria for live consumption (Rowley, 2012). There is the threat from potential hybridizing and direct competition, in addition to as vectors for diseases (van der Meeren *et al*., 2010). *H. americanus* are known to undertake long migrations, as much as 322 km along the coast of Maine, USA (Campbell and Stasko, 1985; Campbell, 1986) so released individuals could potentially travel far from their release area. Reports of individuals being caught by fishermen are reported from time to time and in 2002 an individual was found in the English Channel (Liang, 2002). It has been reported that this individual may have been thrown overboard from a passing cruise liner (Liang, 2002) although it could have been released by other means and/or travelled to this area. Escapees in Norway have been found to travel 30 km distance from their holding cages (Campbell and Stasko, 1985; Campbell, 1986). It has been estimated that the loss of the GB native lobster fishery would cost GB PLC £26.5 million (Yunnie, 2015).

Increases in the population of *H. americanus* could potentially impact *H. gammarus*, through hybridisation, competition, predation or the introduction and/or spread of disease. The impact is assessed as ‘Moderate’ as the population of *H. gammarus* is considered likely to decline. Impacts may be more significant and confidence in the level of impact is low. SwAM (2016) report that expert opinion is divided on whether *H. americanus* establishment will result in total losses of Homarid lobsters or replacement of one species by another without a change to the total stock.

Edible/brown crab - *Cancer pagurus:* There is also the potential for economic impacts on the edible crab fisheries. In invaded areas *H. americanus* have mostly been caught in shallow water by lobster or crab pots although there is a report of one being captured in a trawl from deeper waters (van der Meeren *et al*., 2010). The American lobster could compete with the edible crab for food and shelter (Stebbing et al., 2012a). No evidence was found to assess competition or disease impacts, the likely impact was assessed as ‘Moderate’ for a potential population decline at low confidence.

Crustaceans: Edible/brown crab (*Cancer pagurus*); Lobster (*Hommarus gammarus)*; Norway lobster (*Nephrops norvegicus*); Spider crabs (*Maja squinado*) and the

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[12] https://www.legislation.gov.uk/uksi/1981/994/contents/made

common prawn (*Palaemon serratus*): All crustaceans are potentially at risk from white spot syndrome virus (WSSV) which can be transmitted between crustaceans through cannibalism (Clark *et al*., 2013). Assessing risk is subject to inherent uncertainty regarding prevalence, spread and mortality rate and confidence is low.

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Annex 18. Comb jelly: *Mnemiopsis leidyi*

Name: *Mnemiopsis leidyi* Agassiz, 1865 (Mills, 1998).

Common name(s): American Comb jelly (GISD, 2020); Sea walnut (Didžiulis, 2013; Tennessen, 2011); Warty comb jelly (Le Page, 2019).

Synonyms: *Mnemiopsis mccradyi* Mayer, 1900 (Mills, 1998); *Mnemiopsis gardeni* L. Agassiz, 1980 (Didžiulis, 2013 and references therein).

Domain: Phylum: Ctenophora, Class: Tentaculata, Order: Lobata, Family: Bolinopsidae, Genus/species: *Mnemiopsis leidyi* (Mills, 1998).

Description: A small marine comb-jelly (c. 10cm x 2cm). Key features are the oral lobes spanning nearly the entire body length (Faasse & Bayha, 2006; Fuentes *et al*., 2010).

Image by © Lars Johan Hansson

Figure 18.1. Comb jelly**.** *Mnemiopsis leidyi*

Mnemiopsis leidyi: Habitat

Native range: *Mnemiopsis leidyi* is a planktivorous ctenophore. It is native to temperate and sub-tropical estuaries and coastal waters along the East Coast of the Americas, from New England to Argentina (GESAMP, 1997; Purcell et al., 2001, cited from GBNNSS. 2019).

Substratum type: ‘Not applicable’. *Mnemiopsis leidyi* lives in the water column.

Salinity: Marine and brackish: < 2 to 39 PSU (Purcell & Arai, 2001; Mills, 1998). It is thought to reproduce only when salinity exceeds ~10 PSU (Lehtiniemi *et al*. 2012; Jaspers *et al*. 2011a).

Depth: Can be found at which ever depth the current takes it (Telnes, 2017).

Wave exposure: Likely to be found across a range of wave exposures as it is dispersed by water currents (Jaspers *et al.,* 2018).

Mnemiopsis leidyi: Establishment in Wales as of 2020

Not recorded (NBN Atlas, 2017).

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Mnemiopsis leidyi: Impact pathways based on species traits, biology and ecology

Competition: There is possible competition between fish and *M. leidyi* for food in the water column. *M. leidyi* feeds on zooplankton (Jaspers, *et al.,* 2011b; Mills, 1998) as do many pelagic fish species and fish larvae (Abo-Taleb, 2019; Turner *et al*, 1985). This competition occurs in the water column and does not impact the benthic marine features (see 'Socio-economic impacts' for more information) and is assessed as 'Not applicable' for benthic habitats and species.

Predation: 'Data deficient'. *M. leidyi* is a voracious predator of zooplankton (Mills, 1998) and effective at consuming mesozooplankton (Shiganova, 1998). Hamer *et al*.(2011) found *M. leidyi* to primarily feed on plankton with fish eggs being of minor importance. Such predation could have an impact on benthic species with a pelagic larval stage (i.e. meroplankton) by altering dispersal and recruitment. In the Caspian Sea benthic crustacean numbers dramatically decreased after the invasion of *M. leidyi* while annelid numbers increased (Roohi *et al.,* 2010).

Hybridisation: 'Not applicable'. *M. leidyi* is a self-fertilising, simultaneous hermaphrodite (Ivanov *et al.,* 2000).

Transmission of disease: 'Not applicable'. While *M. leidyi* is known to host parasites, *Neopechona pyriformis* and *Edwardsiella* sp., but there is no evidence to suggest this invasive ctenophore transmits disease.

Parasitism: 'Not applicable'. It is a host of *Neopechona pyriformis* (a Platyhelminthes-trematode or fluke) (Mills, 1998). Parasitic sea anemone larvae (*Edwardsiella* sp.) have also been found infecting *Mnemiopsis leidyi* in the North East Atlantic (Selander *et al*., 2010). Coelenterates often serve as intermediate hosts for fish eggs (Purcell & Arai, 2001) but there is no evidence to suggest that *M. leidyi* is a parasite.

Poisoning/toxicity: ‘Not applicable’**.** Ctenophore comb jellies do not have stinging cells and are non-toxic (Gershwin *et al*., 2014). Instead their tentacles have sticky, adhesive cells called colloblasts to help catch their prey. The ctenophores within the class Nuda have hooks and poison secreting gland within their gullet for paralysing their prey. Since *M. leidyi* is not in this class it is assumed not to be poisonous or toxic (Ramel, no date).

Bio-fouling: ‘Not applicable’. *M. leidyi* does not attach to other animals, algae or substratum.

Grazing/herbivory/browsing: ‘Not applicable’. *M. leidyi* is a carnivore.

Indirect impacts through interactions with other species: *M. leidyi* has been linked to the collapse of anchovy (*Engraulis encrasicholus*) fish stocks in the Black Sea in the late 1980s. Recent studies suggest that overfishing of top predators, causing a trophic cascade, led to an increase in planktivorous fish (Daskalov, 2002). This in turn led to a rise in the consumption of zooplankton which led to a decline in zooplankton biomass. The consequences of this was a reduction of grazing pressure on phytoplankton which, coupled with the increasing euthrophication of the Black Sea at the time, allowed for an increase in phytoplankton biomass (Daskalov, 2002). The fishing industry changed their catch preference to match the now abundant planktivorous fish stocks which were also subsequently overfished (Daskalov, 2002).

M. leidyi was introduced into the Black Sea in the 1980s probably through boat ballast tanks. Planktovorous fish are its main predator so when they became overfished it helped to allow *M. leidyi* numbers to increase (Shiganova, 1998). The high abundance of these gelatinous invaders led to a sharp decline in zooplankton, ichtyoplankton and mesozooplankton, abundance and diversity (Shiganova, 1998). *M. leidyi* had a competitive advantage of food consumption over the anchovy population thanks to the enhanced carrying capacity through eutrophication, high spring –summer temperatures promoting its production, and the high fishing pressure which further induced the anchovy stock collapse (Oguz, 2008).

Blooms of *M. leidyi* have been recorded in the coastal waters of Israel (Galil *et al*., 2009), Italy (Boero *et al*., 2009), and Spain (Fuentes *et al*., 2010). Similar blooms have been recorded in Northern Europe; e.g. Netherlands, 2006 (Faasse & Bayha, 2006); western Baltic Sea, 2006 (Javidpour *et al*., 2006); in Danish waters, 2007 (Tendal *et al*., 2007; Riisgård *et al,* 2007) and the Baltic Sea, 2009 (Jaspers *et al*., 2013). If *M. leidyi* does enter Welsh waters there is the possibility that blooms could occur. The alteration of food webs that occurred in the Black Sea with the eventual collapse of the anchovy stocks should serve as a cautionary tale.

Mnemiopsis leidyi: Impact pathways – Habitats

Chemical impact on ecosystem: 'Not applicable'. None reported.

Physical impact on ecosystem: 'Not applicable'. *M. leidyi* lives in the water column.

Structural impact on ecosystem: 'Not applicable'. *M. leidyi* lives in the water column.

Mnemiopsis leidyi: Summary of impacts on MPA features

M. leidyi occurs in the water column and does not have a direct impact on the benthic MPA features.

Mnemiopsis leidyi: Socio-Economic Impacts

Health and safety: No impacts are known

Aquaculture Operations: Unknown, large blooms may impede operations by clogging gear but no evidence was found describing this. 'Data deficient'.

Aquaculture cultivated species- Mussels, oysters & scallops: *M. leidyi* is a predator of zooplankton (Mills, 1998; Jaspers, *et al.,* 2011b) which may have a preference for motile prey such as larvae (Jaspers *et al.,* 2011b). Mussels, oysters and scallops have a pelagic larval stage which could be impacted through predation should *M. leidyi* arrive in Welsh waters and reach abundant levels. What impact this would have is unknown due to a lack of data. Based on its notorious history with fish stocks collapse in the Black Sea (Shiganova, 1998) its introduction to other areas is of major concern (Van Ginderdeuren *et al.,* 2012). This pathway has been assessed as 'Major' but with low confidence since there were so many variables that led to the eventual anchovy stock collapse in the Black Sea.

Fisheries operations: Unknown, large blooms may impede operations by clogging gear but no evidence was found describing this. 'Data deficient'.

Fisheries target species- Finfish: As mentioned above in the 'Indirect impacts through interactions with other species' section *M. leidyi* has been linked to the collapse of the anchovy fishery (*Engraulis encrasicholus*) in the Black Sea in the late 1980's / early 1990s (Daskalov, 2002; Kideys, 2002; Shiganova *et al.*, 2001). It is unlikely that *M. leidyi* was directly responsible for this collapse, it seems more likely that a combination of events were to blame (Bilo & Niermann, 2004). Bilo & Niermann (2004) suggest multiple factors including overfishing and food competition from *M. leidyi* played a prominent role as well as the ctenophores predation upon the early life stages of the anchovy which likely enhanced pressures on anchovy stocks. Additionally, changes in atmospheric and oceanic patterns in the northern hemisphere in the late 1980s may well have altered plankton communities which would alter the food supply for small pelagic fish species like the anchovy (Bilo & Niermann, 2004).

While landings of anchovy in the Black Sea dropped to one-third of their previous levels, leading many fishermen to abandon fishing, other plantivorous fish species were also affected (Ivanov *et al.,* 2000). Declines in the Mediterranean horse mackerel (*Trachurus mediterraneus ponticus*) and sprat (*Sprattus sprattus phalericus*) amongst others occurred alongside the loss of the anchovy (*Engraulis encrasicolus ponticus)* (Ivanov *et al.,* 2000). *M. leidyi* arrived in the nearby Sea of Azov in the late 1980s where it now reaches high densities in the warmer months (Shiganova & Bilgakova, 2000). Here it also caused fisheries decline, that of the Azov anchovy (*Engraulis encrasicolus maeticus*) and Azov kilka (*Clupeonella cultriventris*) (Ivanov *et al.,* 2000).

The Caspian Sea has also seen declines and near fisheries collapse which have been linked to the introduction of *M. leidyi* (Daskalov & Mamedov, 2007). Again, it is unlikely that the ctenophore is directly to blame with multiple factors being involved. The Caspian Sea has seen anthropogenic impacts including overfishing, pollution from domestic and industrial waste and oil and gas field development (Ivanov, 2000). *M. leidyi* arrived in the Caspian Sea in the late 1990s and reached its highest abundance between 2001 and 2002 (Roohi *et al.,* 2010). Monitoring of the sea between 2001 and 2006 showed the ctenophore explosion coincided with a decline in abundance and diversity of mesozooplankton. “While this decline appeared to have reduced the nourishment of sprat (also known as kilka), it seemed to have affected phytoplankton favourably mainly due to the decrease in grazing pressure” (Roohi *et al.,* 2010). The disappearance of edible zooplankton was one of the first observations after the arrival of *M. leidyi.* Benthic deposit feeders biomass increased, species of oligochaetes and polychaetes, while benthic crustaceans decreased in number during 2001-2003 and completely disappeared during 2001-2003 (Roohi *et al.,* 2010). The kilka (*Clupeonella* spp.) fishery of Iran, which had been the most abundant and widespread zooplanktivorous fish before the arrival of *M. leidyi* decreased significantly in 1999. The kilka landings had decreased by ~70% from 1995-2000 to 2001-2006 (Roohi *et al.,* 2010). This resulted in a loss to the economy of at least 125 million US dollars (Roohi *et al.,* 2010).

A loss in kilka predators like mullet (*Liza* spp., Mugilidae) and kutum or white fish (*Rutillus frisii kutum*) were also recorded between 1991 and 2006 (Roohi *et al.,* 2010). Anchovy kilka (*Clupeonella engrauliformis*) in the Caspian Sea is the main diet for piscivores in the Caspian Sea which include the sturgeon (*Acipenser spp.* and *Huso huso*) and seals (*Phoca caspica*) (Daskalov & Mamedov, 2007). Ivanov *et al.* (2000) speculate that with all the pressures that the Caspian seal (*Phoca caspica*) is already under: “The entry of Mnemiopsis into this complex equation of population stress and demise could prove to be the final straw for the Caspian seal”. The UKs population of grey seals (*Halichoerus grypus*) accounts for 45 % of the total population worldwide (SCOS, 2008 in Leeney *et al*., 2010) with 4 % of the UKs population breeding around Pembrokshire, Wales (Stringell *et al.,* 2015). Considering the negative impact that *M. leidyi* has had on the Caspian seal should it arrive in Wales there could be cause for concern.

M. leidyi’s economic impacts (via fisheries losses) are potentially huge, as demonstrated in the Black Sea and the Caspian Sea in the late 1980s early 1990s. Faase and Bayha (2006) caution that patterns seen in the Black Sea might be replicated in northern Europe, with impacts on zooplankton species and their predators such as plaice (*Pleuronectes platessa*) and sole (*Solea solea*). These two fisheries have an annual combined catch of around EUR 300 million (European Commission, 2006). *M. leidyi* may impact fish stocks by preying on eggs and larvae and by competing with larvae for food (Fasse & Bayha, 2006). Hamer *et al*. (2011) found in experiments that *M. leidyi* preferred copepods over fish eggs. They concluded that *M. leidyi* was not a direct predator of fish eggs but it may compete for food with larval fish in the North Sea. Fasse & Bayha (2006) suggest that it is unclear if there are any species that will feed heavily enough on *M. leidyi* in Dutch waters to control their numbers.

It is possible that *M. leidyi* populations may be controlled by native predators and other food competitors. Examples of these are: the compass jellyfish *Chrysaora* sp. which is known to control *M. leidyi* population in the Chesapeake Bay, USA (Purcell & Decker, 2005); the jellyfish, *Cynea* sp., are also known predators in Danish waters (Riisgård, 2007), and predation by another ctenophore (*Beroe ovata*) is a well-known predator of *M. leidyi* (Galil *et al.* 2009; Shiganova *et al.,* 2003; Van Walraven *et al.,* 2013). However, the establishment of *M. leidyi* in Welsh waters with its important fishing and spawning grounds is cause for concern due to its high feeding, growth and reproduction rates (Jaspers *et al.,* 2015). While *M. leidyi* has not yet reached Welsh waters a precautionary assessment of ‘Major’ with low confidence was made for finfish populations likely to be impacted.

Mnemiopsis leidyi: References

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Annex 19: Asian/Japanese oyster drill: *Ocinebrellus inornatus* (formerly *Ocenebra inornata*)

Common name(s): Asian oyster drill, Japanese oyster drill, Japanese oyster borer

Synonyms: *Ocinebrellus inoratus; Murex inornatus, Murex endermonis, Murex japonicas, Murex talienwhanensis, Tritonium, Tritonium (Fusus) submuricatum, Trophon incompta* (Worms, 2019). *Ceratostoma inornatum, Ocenebra japona* (ISSG, 2019).

Domain: Phyla: Mollusca, Class: Gastapoda, Order: Neogastropoda, Family: Muricidae, Genus/species: *Ocenebra inornata* (Worms, 2019).

Description: The shell is up to 6 cm in length. It is conical with 6-7 whorls and a pointed spire. The shell colour is variable from beige or brown to orange and striped (Duckwall, 2009). Shells in this family are identified via a labral tooth however *O. inornata* can be found with or without this tooth in the same population, both in its native and invasive ranges (Amano & Vermeiji, 1998). This has caused confusion and has left to misidentification. It can be confused with the similar but smaller (up to 4 cm in length) invasive drill, *Urosalpinx cinerea* (Duckwall, 2009). There is also the possibility of confusing it with, morphologically similar, *Ocenebra erinacea.*

(Image by G & Ph. Pope).

Figure 19.1 Japanese oyster drill: *Ocinebrellus inornatus*

Ocinebrellus inornatus: Habitat

Native range: The native range is from Northern China through Korea and all seas around Japan to Sakhalin and the Kurile Islands (Choe and Park 1997;

GarciaMeunier *et al.* 2003) and from 33º to 51ºN (Radwin and D'Attilio 1976). The distribution of *O. inornta* partially overlaps that of the Pacific oyster, *Magallana gigas*, which forms the snails' principal diet and substrate for deposition of its egg capsules (Lützen *et al*., 2012) and was a vector for introduction. It is therefore likely that both these species occur in similar habitats.

Substratum type: Rocky, intertidal and shallow subtidal habitats. It is also found on marine man-made structures, biogenic reefs and bivalve beds. It has been found on both natural and cultured oyster beds in high numbers. However, it seems to prefer natural oyster reefs, and is found on these at higher densities than in aquaculture areas (Buhle and Ruesink, 2009). Natural reefs are structurally more complex and provide better shelter (Buhle and Ruesink, 2009). Gravel, mud and sand sediments provide suitable habitat (Fofonoff, 2018 and references therein; Duane Sept, 2019) although abundances are lower on bare tidal flats without hard substrate (Buhle and Ruesink 2009). They have been found surviving in winter in estuaries in the Netherlands where temperature drop to -1°C. They have also been found surviving for several months in the Limfjord, Denmark where salinity is 23 PSU (Fofonoff, 2018 and references therein). However, in Willapa Bay, Washington (USA) they were found in the more saline areas of the bay when compared with the Atlantic oyster drill (*Urosalpinx cinerea*) which was found in the areas closer to the rivers (Buhle & Ruesink, 2009).

Salinity: Duckwall (2009) records *O. inornatus* as having a limited tolerance to very low salinity, but can survive for several months at 23 PSU (Fofonoff, 2018 and references therein). In Willapa Bay, Washington (WA), *O. inornatus* was most abundant in the more saline regions of the bay, while the Atlantic Oyster Drill (*Urosalpinx cinerea)* predominated in areas closer to rivers (Buhle *et al*., 2009).

Depth: *O. inornatus* is found on the mid-shore on littoral sediments and rock (Pigeot *et al*., 2000 cited from Lützen *et al*., 2012). No evidence was found to indicate *O. inornatus* is found above mid shore height, probably due to increased emersion times and lack of favoured prey. In the Limfjord (Denmark), *O. inornatus* were caught in dredges on a sandy bottom at depths of 5–6 m (Lützen *et al*., 2012). No records were found for occurrence at greater depths.

Wave exposure: No evidence was found for wave exposure tolerances.

Other information: *Ocinebrellus inornatus* have survived in estuaries in the Netherlands where temperatures fall to -1°C in winter (Faasse and Ligthart 2009).

Ocinebrellus inornatus: Establishment in Wales as of 2020

None. *O. inornatus* does not have a free swimming stage or planktonic larval phase rather its eggs are attached to substrate (Martel, *et al*., 2004). Its natural means of transport within an area are limited to crawling or through attachment to drifting algae or other material. Oyster imports has been the usual vector for introductions (Cohen, 2004). *O. inornatus* arrived in the USA from Japan via the transport of oysters at the beginning of the 20th Century. It then spread around the USA (Duckwall, 2009). In 1995, it arrived on the Atlantic coastline France via oyster shipments from the USA (Martel *et al*., 2004).

For more up-to-date information on distribution please visit the National Biodiversity Network https://nbnatlas.org/ or contact the Natural Resources Wales Marine and Coastal Ecosystems team at Marine.Coastal.Ecosystems.Team@cyfoethnaturiolcymru.gov.uk.

Ocinebrellus inornatus: Impact pathways based on species traits, biology and ecology

Competition: *O. inornatus* competes with native carnivorous snails and drills (e.g. in France with *O. erinacea* (Martel *et al*., 2004a). Its feeding rate is higher than the native European Oyster Drill (*Ocenebra erinacea*) (up to 2.4 times more oysters consumed in experiments) (Martel *et al*., 2004b). When the invasive *O. inornatus* is documented in high numbers the reverse is true of the native *O. erinacea* (Pigeot *et al*., 2000). It seems that *O. inornatus* competes with, and possibly outcompetes, *O. erinacea.*

No native oyster drills or other carnivorous snails characterize MPA features, therefore the replacement of these with the functionally similar *O. inornatus* is not considered likely to lead to the loss or reclassification of MPA features through competition. This pathway is assessed as ‘Minimal concern’ for all MPA features. Note: predation on native fauna and aquaculture species is assessed separately below.

Predation: Oyster drills are limited to prey with a shell thickness less than the length of the radula, so they are limited to small prey, oyster spat, and juveniles of other bivalves (Duckwall, 2009). They feed on a variety of shelled invertebrates including mussels, clams, barnacles and other gastropods (Chew and Eisler, 1958; Duckwall, 2009; Faasse and Ligthart, 2009; Lützen *et al*., 2012).

Predation impacts were considered to be of ‘Minimal concern’ in MPA features where bivalves were not characterising and/or establishment was considered unlikely. Confidence was assessed as moderate for these biotopes as little evidence was available to assess establishment.

Predation impacts were considered to be ‘Minor’ where some loss of native species was possible but these are not preferred prey (for example barnacles, burrowing bivalves) and densities of *O. inornatus* are likely to be low. Confidence is low due to lack of specific evidence.

Predation impacts on *Mytilus edulis* beds and *Ostrea edulis* beds were assessed as ‘Moderate’ at low confidence as many adults would be expected to have reached a size where predation is limited (see below: impacts on key MPA features and socio-economic impacts for further information). However, at high densities and over longer time periods *O. inornatus* may reduce recruitment and impacts could be more severe. The confidence assessment of low for these MPA features reflects uncertainty in the severity of the long-term impact.

Hybridisation: No evidence was found for hybridisation with native species and this impact pathway is assessed as ‘Not applicable’.

Transmission of disease: No records of disease transmission by *O. inornatus* were found in the literature and this impact pathway is considered to be ‘not applicable’.

Parasitism *O. inornatus* is not a parasite and this impact pathway is assessed as ‘Not applicable’.

Poisoning/toxicity: *O. inornatus* predates on organisms by drilling with a radula and secreting acids to aid drilling through shells. It is not poisonous or toxic and this impact pathway is assessed as ‘Not applicable’.

Bio-fouling: *O. inornatus* is a mobile epifaunal species and is not a bio-fouler, this impact pathway is assessed as ‘Not applicable’.

Grazing/herbivory/browsing: *O. inornatus* is a predator not a herbivore and this impact pathway is assessed as ‘Not applicable’.

Indirect impacts through interactions with other species: Through its consumption of reef forming bivalves *O. inornatus* may indirectly alter the biogenic structure of habitats if it becomes established in mussel and oyster habitats. It is known to feed on mussels and oysters and in its invasive range has been documented in high concentrations, this impact is assessed through ‘Structural impact on ecosystem’ pathway below.

Ocinebrellus inornatus: Impact pathways – Habitats

Chemical impact on ecosystem: changes in filter feeder density through predation by *O. inornatus* may result in indirect impacts on biogeochemical cycling within the ecosystem. Such impacts would be density dependent and site-specific. No direct impacts on the ecosystem were found in the literature and this impact pathway is considered ‘Not applicable’.

Physical impact on ecosystem: As a small, mobile, epifaunal species *O. inornatus* is not anticipated to cause direct physical impacts on the ecosystem and this pathway is ‘Not applicable’.

Structural impact on ecosystem: As a small, mobile, epifaunal species *O. inornatus* is not anticipated to cause direct structural impacts. Indirectly this species may alter habitat structure through the removal of biogenic reef forming species (mussels and oysters). Predation was not considered to result in the loss of the entire oyster reef or mussel bed and impacts were assessed as ‘Moderate’ at low confidence.

Ocinebrellus inornatus: Interactions with MPA Features

MPA features that provide suitable habitat for *Ocinebrellus inornatus*

O. inornatus is found in higher abundances among its preferred prey of oysters although it may also feed on other species. Habitats considered to provide suitable habitat are:

***Ostrea edulis* beds**:

- A5.43 Infralittoral mixed sediments; A5.435 *Ostrea edulis* beds on shallow sublittoral muddy mixed sediment;
- Sheltered muddy gravels; A5.435 *Ostrea edulis* beds on shallow sublittoral muddy mixed sediment;

***Mytilus edulis* beds:**

- A5.6 Sublittoral biogenic reefs: biotopes within A5.62 Sublittoral mussel beds on sediment;
- Intertidal *Mytilus edulis* beds on mixed and sandy sediments;
- Blue mussel beds; and
- A2.7 Littoral biogenic reefs: A2.72 Littoral mussel beds on sediment.

MPA habitats considered potentially suitable for *Ocinebrellus inornatus*

Potentially suitable habitats include intertidal and shallow sublittoral rock, and intertidal and shallow subtidal sediments. Confidence is low, due to the lack of detailed habitat information relating to wave exposure, tidal currents and shore height and subtidal depth tolerance:

Rock

- A1.1 High energy littoral rock;
- A1.2 Moderate energy littoral rock;
- A3.3 Atlantic and Mediterranean low energy infralittoral rock:
    - A3.361 *Mytilus edulis* beds on reduced salinity infralittoral rock;
- A4.2 Atlantic and Mediterranean moderate energy;
    - A4.24 Mussel beds on circalittoral rock.

Sediment

- A2.2 Littoral sand and muddy sand:

    - A2.24 Polychaete/bivalve-dominated muddy sand shores, may occur here as some prey items are present;
    - A2.3 Littoral mud;
- A2.3 Littoral mud:
    - A2.31 Polychaete/bivalve-dominated mid estuarine mud shores;
- A2.4 Littoral mixed sediments (all);
- A5.1 Sublittoral coarse sediment: A5.13 Infralittoral coarse sediment;
- A5.2 Sublittoral sand: A5.24 Infralittoral muddy sand; A5.26 Circalittoral muddy sand based on sediment and depth (Buhle *et al*., 2009) and presence of bivalves;
- A5.3 Sublittoral mud:
    - A5.33 Infralittoral sandy mud; A5.34 Infralittoral fine mud; A5.35 Circalittoral sandy mud; based on sediment and depth (Buhle *et al*., 2009) and presence of bivalves;
- A5.4 Sublittoral mixed sediments;
- A5.43 Infralittoral mixed sediments;
- Sheltered muddy gravels; and
- Peat and clay exposures.

MPA habitats considered unlikely to be suitable for *Ocinebrellus inornatus*

The following MPA features were considered unlikely to be suitable due to a lack of prey items and in many rock habitats, dense macroalgae that were considered likely to reduce suitability. Confidence is low due to lack of specific habitat evidence for absence in these features:

Rock and other vegetated habitats

- A1.3 Low energy littoral rock, based on lack of suitable prey and macroalgae that may limit suitability;
- A3.1 Atlantic and Mediterranean high energy infralittoral rock;
- A3.3 Atlantic and Mediterranean low energy infralittoral rock (A3.31; A3.32; A3.34);
- A3.2 Atlantic and Mediterranean moderate energy infralittoral rock;
- A4.1 Atlantic and Mediterranean high energy circalittoral rock;
- A4.1 Atlantic and Mediterranean high energy circalittoral rock; (A4.12; A4.13);
- A4.11 Very tide-swept faunal communities on circalittoral rock: lack of prey and extremely tide-swept;
- A4.2 Atlantic and Mediterranean moderate energy (A4.21; A4.22; A4.23; A4.25);
- A4.3 Atlantic and Mediterranean low energy circalittoral rock;
- Intertidal boulder communities;
- Tide-swept channels;

- Fragile sponge and anthozoan communities on subtidal rocky habitats (most habitats outside of recorded depth range); and
- Estuarine rocky habitat.

Biogenic habitats

- A5.6 Sublittoral biogenic reefs.
    - A5.61 Sublittoral polychaete worm reefs on sediment.

Sediment habitats

Sediment habitats considered unsuitable due to sediment mobility and/or lack of suitable prey items:

- A2.1 littoral coarse sediments; biotopes with sparse fauna and infauna;
- A5.2 Sublittoral sand:
    - A5.23 Infralittoral fine sand.

All *O. inornatus* habitat records in the literature were based on intertidal and shallow subtidal habitats. It is unclear if *O. inornatus* occurs below 10m depth if suitable prey items were available and confidence is low for the following habitats, that were considered unlikely to be suitable based on depth range. Confidence is low due to the lack of habitat records:

- Mud habitats in deep water;
- Subtidal mixed muddy sediments;
- Sea-pen and burrowing megafauna communities;
- *Musculus discors* beds;
- *Modiolus modiolus beds*;
- A5.1 Sublittoral coarse sediment: A5.14 Circalittoral coarse sediment; A5.15 Deep circalittoral coarse sediment;
- A5.2 Sublittoral sand:
    - A5.25 Circalittoral fine sand;
    - A5.27 Deep circalittoral sand;
- A5.3 Sublittoral mud:
    - A5.36 Circalittoral fine mud (and note lack of prey);
    - A5.37 Deep circalittoral mud;
- A5.4 Sublittoral mixed sediments:
    - A5.44 Circalittoral mixed sediments;
    - A5.45 Deep circalittoral mixed sediments; and
- A5.6 Sublittoral biogenic reefs:
    - A5.63 Circalittoral coral reefs.

MPA features considered unlikely to be suitable due to reduced or low salinity based on records by Buhle *et al*., 2009 (confidence is moderate):

- A5.1 Sublittoral coarse sediment:
  - A5.12 Sublittoral coarse sediment in variable salinity (estuaries);
- A5.21 Sublittoral sand:
  - A5.21; A5.22 Based on low salinity;
- A5.3 Sublittoral mud:
    - A5.31 Sublittoral mud in low or reduced salinity; A5.32 Sublittoral mud in variable salinity (estuaries) (also lack suitable prey);
- A5.4 Sublittoral mixed sediments:
- A5.41 Sublittoral mixed sediment in low or reduced salinity; A5.42 Sublittoral mixed sediment in variable salinity (estuaries); and
- *Sabellaria alveolata* reefs:
  - A2.711 Unlikely due to sand abrasion; A5.612 variable salinity (Buhle et al., 2009).

MPA features unsuitable for *Ocinebrellus inornatus*

No evidence was found for tolerances to emersion and shore height. Based on Pigeot *et al*., (2000, cited from Lützen *et al*., 2012), that this species is found in the mid littoral zone, habitats occurring above this zone were considered unsuitable due to emersion times and lack of prey. Confidence is low due to the lack of evidence. MPA features considered unsuitable are:

- Coastal saltmarshes and saline reedbeds;
- A2.2 Littoral sand and muddy sand;
- A2.21 Strandline- height on shore;
  - A2.22 Barren or amphipod-dominated mobile sand shores, sediment mobility and lack of prey;
  - A2.23 Polychaete/amphipod-dominated fine sand shores, sediment mobility and lack of prey.

MPA features with no evidence for *Ocinebrellus inornatus*

No evidence was found for presence in the following habitats and no suitable proxy evidence was identified to support an assessment:

- Littoral sediments dominated by aquatic angiosperms
- Seagrass beds
- Zostera beds
- Carbonate reefs
- Maerl beds
- Sublittoral macrophyte-dominated sediment;
  - A5.51 Maerl beds
  - A5.52 Kelp and seaweed communities on sublittoral sediment
  - A5.53 Sublittoral seagrass beds.

Summary of the impacts on key MPA features.

The main ecological and economic impacts are due to predation of oysters, and particularly young 'seed oysters' (Buhle *et al*., 2009; Lützen *et al*., 2012). Buhle and Ruesink (2009) found that per capita effects of invasive drills on native oysters (*O. lurida*) varied nonlinearly with native oyster density and with the density of nonnative alternative prey. *Mytilus edulis* are an alternative prey (Lützen *et al*., 2012). Predation on bivalve species including consuming commercially and environmentally important biotopes and reef building species will impact all habitats characterised by these species with particular effects on bivalve reefs. Where mussels and oysters are not predated or are predated at low rates due to the size of mature individuals, consumption of juveniles preventing natural recruitment may over time result in a loss of reefs. Impacts are assessed as 'Moderate' at low confidence with the potential for more severe impacts in the longer-term.

Ocinebrellus inornatus: Socio-Economic Impacts

Health and Safety: No evidence, interaction considered benign.

Aquaculture operations: *O. inornatus* was regarded as major pest of oyster-culture in its native waters and 'very destructive to native oysters in Puget Sound' within 8 years of its discovery (Galtsoff 1932). Similarly, it was discovered in Atlantic France in 1995, and regarded as a pest species on the introduced Pacific Oyster (*M. gigas*) in culture operations (Buhle *et al*., 2009). Its effects on cultured oysters are greatest on newly planted 'seed oysters', because of the snail's preference for small prey (Carl and Giguet 1972; Goulletquer *et al*., 2002; Buhle *et al*., 2009).

Control of infestations will impose costs on operations through requirements to inspect and remove individuals on seed oysters, and inspection and regulation of oyster transfers and culture equipment. Control of drills in infested areas involves raking the ground, removing debris, and other labor-intensive tasks (Quayle 1969).

The impact on aquaculture operations was assessed as 'Moderate' due to the costs imposed, based on aquaculture operations continuing. If costs of removal or control prove prohibitive in heavily infested areas, impacts could be greater (Major).

Aquaculture target species: Any aquaculture operation that is culturing/growing bivalves such as mussels (*M. edulis* and *M. galloprovincialis)* and oysters (*O. edulis* and *M. gigas*) could be adversely affected by the presence of *O. inorata*. The species is known to cause up to 25% mortality in stocked populations of *M. gigas* (see Elston 1997, cited from Lützen *et al*., 2012) and predation reduces the survival of the Pacific and Olympia oysters in field enclosures and transplant experiments (Buhle and Ruesink, 2009).

O. inornatus is known as a pest species on commercial mussel and oyster beds in France, the Netherlands and Denmark (and Belgium) where it has established (Lützen *et al*., 2012). They are known to feed upon bivalves and in particular mussels and oysters. *O. inornatus* is considered to be a serious threat to French oyster farming. Significant damage has already been reported where it occurs as an invasive species and in smaller areas may cause mortalities of up to 50% (Pigeot *et al*., 2000, cited from Lutzen *et al*., 2012).

Impacts on the cultivated species are considered to be 'Moderate', however, where infestations are severe and cannot be controlled, activities may be abandoned and the impact would be higher (Major- Massive).

Fisheries operations: *O. inornatus* is captured in some dredges and its retention in mobile gears will depend on mesh size. *O. inornatus* is strongly associated with prey species and would not be expected to be present in high densities where these are absent. There is no suggestion that *O. inornatus* would have an impact on fisheries operations by preventing gear deployment and it is therefore considered to be of 'Minimal concern' for fishery operations at high confidence (but see target species below).

Fisheries target species: Finfish, crustaceans and whelk: Minimal concern (high confidence). No interaction was predicted, no commercially targeted species are dependent on bivalves and *O. inornatus* is not considered to modify nursery or feeding habitats.

Shellfish: No evidence was found for predation on scallops, these are mobile and could escape predation. Shellfisheries which harvest naturally occurring bivalves (*M. edulis* and *O. edulis*) could be impacted by this invasive species. High densities of *O. inornatus* could cause significant reductions to *O. edulis* and *M. edulis* beds, through loss of adults and predation on juveniles preventing recruitment. Impacts will depend on the size of the stock and the density of *O. inornatus* coupled with other environmental factors that influence habitat suitability and population maintenance (Buhle and Ruesink, 2009). Impact on *M. edulis* and *O. edulis* fisheries was assessed as 'Major' with medium confidence.

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Annex 20 - Data Archive

Data outputs associated with this project are archived in NRWs Electronic Records Management System on server–based storage at Natural Resources Wales.

The data archive contains:

[A] Environmental and Socio Economic Impact Classification spreadsheets

Metadata for this project is publicly accessible through Natural Resources Wales' Library Catalogue https://libcat.naturalresources.wales (English Version) and https://catllyfr.cyfoethnaturiol.cymru (Welsh Version) by searching 'Dataset Titles'. The metadata is held as record no 124805.

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