EMFF Operational Programme 2014-2020

Blue Growth & Marine Spatial Planning Scheme

Mapping Ireland's Marine Invasive Species

Applying modelling approaches to assess the spread and management of marine non-indigenous species

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An Roinn Talmhaíochta, Bia agus Mara
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Marine Institute
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Executive Summary

The growth in marine activity and global trade has led to rising numbers of marine species being transported around the world. This has led to a global trend of increasing numbers of marine non-indigenous species (NIS) becoming established, including in Irish waters. A small proportion of NIS may go on to spread widely within their recipient habitats and cause environmental or socio-economic impacts. These are known as invasive alien species (IAS), and the potential harm some of these IAS can cause has led to international and national policies that require measures to manage these introductions and their impacts.

To make informed decisions to manage IAS, policymakers need information on the potential of species spread, the potential impact, and the viability of potential control strategies. However, information is often lacking either because an IAS is yet to establish in an area, or because resources are insufficient to survey the entire IAS's range. One source of information comes from combining the results of larval transport modelling (that determines where larvae may spread to from a given source) and species distribution modelling (that determines where larvae may settle, survive and become established). This combination allows a spatial map of an IAS's potential spread to be estimated.

This project aimed to develop larval transport and species distribution models for two focal NIS, in order to demonstrate the applicability of this combined approach for decision-making about NIS management in Ireland. The two focal NIS were: the Pacific oyster (*Magallana gigas*) and the Asian shore crab (*Hemigrapsus sanguineus*). These two NIS were chosen to provide contrasting life history characteristics and scenarios of introduction and spread. The Pacific oyster is cultivated as an aquaculture species and has wild populations already established in Ireland. The Asian shore crab is a horizon species predicted to arrive in Ireland within the next decade.

Environmental and socio-economic impacts of both focal species in their invaded range were reviewed, as well as potential control strategies. Combined results from larval transport and species distribution models provided predictions of the spread of the species within Galway Bay (a region with high quality hydrodynamic data) and to a lesser extent the Bantry Bay area. These results provide species spread scenarios at a very local spatial scale. The results also allow the contributions of individual source populations to the overall spread and impacts to be deduced. These results can be visualised as maps and made available as GIS layers. This modelling evidence was then combined with the species' environmental, social and economic impacts to generate species specific management recommendations that take the local environment into account. For example, predicted spread and connectivity at local scales can be used to identify vulnerable sites and likely major sources of larvae and evaluate control strategies and estimate risk of recolonisation. This information can be used to inform development of species management plans and can facilitate discussion with stakeholders.

The combined modelling approach also provides evidence of how climate change translates into changing risk of species spread. Records of *M. gigas* in Ireland are relatively sparse and their environment and ecosystem-service impacts are limited. This is in contrast to parts of mainland Europe where *M. gigas* form dense reefs that changes the appearance and composition of coastal assemblages and have significant impact. As climate change increases sea temperatures, *M. gigas* recruitment will become more common around Ireland and the risk of substantial impact will increase. The modelling quantifies this by predicting the increase in oyster larval survival as a result of warming seas. Increased survival increases the risk of more extensive reef formation around Ireland's coastline. Targeted intervention at an early stage would be required to prevent such spread while populations remain small.

Decision makers must balance multiple complex factors in setting policy and management, and must ensure compliance with directives such as the Marine Strategy Framework Directive. Our approach provides diverse quantitative evidence to support this decision making and help target specific interventions.

Table of Contents

List of Tables

List of Figures

List of Boxes

1 Introduction

1.1 An introduction to marine invasive species

Human movement around the world has created opportunities for the movement of species across natural geographic barriers. Marine species have been transported across the globe intentionally and unintentionally, associated with the movement of ships or introduced for aquaculture, food, or use in aquaria. The number of these introductions is increasing as globalisation and developments in technology increase connectivity between regions (Galil et al. 2014; Seebens et al. 2016). Introduction and spread of these marine non-indigenous species (NIS) has occurred across all marine regions, including in polar systems. As the marine environment is increasingly looked to as a solution for global problems, it will become more important to manage the potential impact marine activities will have on the environment including via the intentional and accidental spread of NIS.

The introduction of novel species is an important driver of change in marine systems (Simberloff et al. 2013). The environmental impacts of invasive NIS are recognised globally as a major threat to biodiversity (Molnar et al. 2008; Pyšek et al. 2020). Furthermore, invasive species may also cause direct and indirect changes to the economy, the interaction of people with their environment, and even human health. These environmental and socio-economic impacts may be well-known in terrestrial and freshwater landscapes but can be less noticeable and harder to research in marine environments (Chan and Briski 2017).

Clarifying the terminology used when talking about species introductions is important, as there are multiple terms used to refer to non-indigenous species (NIS) and different interpretations of when a NIS can be considered invasive. In this document the following terms will be used, with the definitions provided by Mirimin et al. (2019):

> *"Non-Indigenous Species (NIS) (synonyms: alien, exotic, non-native) are defined as species, subspecies or lower taxa introduced outside of their natural range and occurring beyond the limits of their natural dispersal potential. This includes any part, gamete or propagule of a species that might survive and subsequently reproduce. Their presence outside their natural range arises from intentional or unintentional introduction by humans. Natural shifts in distribution ranges (e.g., due to climate change or dispersal by ocean currents) do not qualify a species as a NIS. However, secondary introductions of NIS from the area(s) of their first arrival can occur without human involvement due to spread by natural means."*

> *"Invasive Alien Species (IAS) are a subset of established NIS which have spread, are spreading or have demonstrated in other locations a potential to spread, and have an adverse effect on biological diversity, ecosystem functioning, socio-economic values and/or human health in invaded regions."*

The ability of a NIS to spread within introduced regions is considered by many researchers to be a key component in its consideration as an invasive species, and in some cases the only component. Using the above definition, if a species has had a documented impact in other areas of its introduced range we consider it an IAS, even if no impact has been recorded in Ireland so far.

Where there is uncertainty over the origin or status of a species, it is referred to as cryptogenic (Carlton 1996). This uncertainty can arise due to the widespread or cosmopolitan distribution of a species, or

due to lack of knowledge about when or where an introduction may have originated. Determining the exact introduction pathway can be difficult, and different species will have different levels of uncertainty attached to them.

In the following sections, a brief overview is given of the known marine IAS in Ireland, the process of NIS establishment in marine environments, their potential impacts and the framework of management measures used to prevent or control spread and impact. In this context the applicability of larval tracking models to support IAS management decisions is discussed.

1.1.1 Marine NIS present in Ireland

In Ireland, NIS have been introduced to all environments, on land, in freshwater rivers and lakes and in marine and coastal areas. Invasive NIS introduced to Ireland have caused changes in community composition, and in some cases cause environmental and economic damage (Stokes et al. 2004).

A review of the marine NIS currently present in Ireland was undertaken by Mirimin et al. (2019). The authors compiled a list of 122 marine and brackish NIS reported in Irish waters which had at least one verified occurrence. The NIS comprise a variety of species from across 24 phyla, with animals (mainly arthropods and molluscs) making up the majority of species (66%), followed by algae (18%) and chromista (13%) (Mirimin et al. 2019). This review followed a checklist of all aquatic NIS compiled by Minchin et al. (2007a) which identified 59 brackish and marine species, approximately half of which were thought to have established reproducing populations in Irish waters. The majority of marine NIS originated from the north Pacific, followed by the north-west Atlantic (Minchin 2007a).

Of the NIS included in the latest review, 20 were identified as high risk in terms of their potential impact. Examples of high-profile NIS already established in Ireland include the macroalgae *Sargassum muticum* and *Undaria pinnatifida.* The establishment of *S. muticum* can negatively impact seagrass communities especially those which may already be damaged, and the species can also dominate intertidal rockpool communities. The ascidians *Corella eumyota, Botrylloides violaceus, Styela clava* and the high profile *Didemnum vexillum* are known to cause fouling problems in other areas of their introduced range and may dominate fouling systems (Minchin 2007b).

New NIS continue to be introduced to Irish waters. In the five years from 2013 – 2018, three new potentially invasive marine species were detected, *Undaria pinnatifida, Schizoporella japonica* and *Perophora japonica*, demonstrating that marine species introductions continue to occur. In 2017 a horizon scanning exercise was conducted for Ireland to identify the most likely IAS to arrive on the island of Ireland within the next ten years (2017-2017) (Lucy and Davis 2020). Of the top 40 horizon species there were seven marine species (Table 1.1), including the estuarine Chinese mitten crab, *Eriocheir sinensis* (Lucy and Davis 2020). These species are considered by expert consensus to be likely to have high environmental impacts and to be introduced to Ireland in the near future. Indeed, two female Chinese mitten crabs were captured in Waterford harbour in winter 2020/2021 including one female with eggs, the newest record for this species since individual crabs were documented in 2006 & 2009 (Cassidy 2021).

Table 1.1 – Top marine horizon species identified as likely to arrive and have a high impact in Ireland. Table an abbreviated version of that presented in Lucy and Davis (2020).

Species	Group	Common name	Native range
Hesperibalanus fallax	Crustacean	Warm water barnacle	Atlantic coast of tropical Africa
Caulacanthus okamurae	Algae	Pom-pom weed	Asia (Pacific)
Ensis leei	Mollusc	American razor clam	NW Atlantic
Hemigrapsus takanoi	Crustacean	Brush-clawed shore crab	Asia (Pacific)
Celtodoryx ciocalyptoides	Sponge		NW Pacific
Hemigrapsus sanguineus	Crustacean	Asian Shore crab	Asia (Pacific)
Eriocheir sinensis	Crustacean	Chinese mitten crab	Asia (Pacific)

1.1.2 The invasion process

The process by which a novel species is introduced into a new system, successfully survives a new environment and goes on to create a reproducing population has been described by many researchers. Blackburn et al. (2011) proposed a unified framework for discussing this process, prompted by differences in framing between researchers studying terrestrial plant and animal invasions. The invasion process is divided into several stages, separated by barriers which must be overcome for a species to become progress to the next stage. These stages are as follows:

1) Transport
2) Introduction
3) Establishment
4) Spread

Transport & Introduction: Vectors of marine NIS

Globally, the primary vectors of marine species introductions are shipping and aquaculture (Minchin et al. 2013). The growth of trans-oceanic trade has resulted in greater numbers and sizes of commercial ships, reduced shipping times, and greater potential for the transport of novel organisms (Hulme 2009). Marine organisms may be carried as larvae or adults entrained within ballast water, trapped within sea chests, or attached to ship hulls (Gollasch 2002). Transport of organisms by smaller vessels is also increasingly being recognised as an important mechanism for introduction, and especially the secondary spread of organisms (Acosta and Forrest 2009; Clarke Murray et al. 2011). Secondary spread may occur via a wider range of vectors, including natural dispersal of larvae or adults (Stokes et al. 2004).

The world's aquaculture industry depends on the culture of NIS for much of its stock. Aquaculture species are moved around the world as stock and in some cases have escaped into the wild and formed self-sustaining populations. However, the movement of stock and equipment has also resulted in the unintentional introduction of species used as packing material or moved as epibionts or parasites of stock (Minchin 2007c). Aquaculture biosecurity practices are becoming increasingly important, but this has been the mechanism by which several historical introductions have occurred.

Other vectors include the transport of organisms for use in the aquarium and pet trade, as live food, or as live bait for fishing and angling which are then intentionally or accidentally released (Katsanevakis et al. 2013). While rarer than introductions via shipping and aquaculture, these vectors

have resulted in the introduction of IAS with significant environmental impacts such as the macroalgae *Caulerpa taxifolia* in the Mediterranean, or the lionfish *Pterois volitans* in the Caribbean.

In Ireland the primary pathways of NIS introduction are vessels, aquaculture and secondary spread from other European countries (Mirimin et al. 2019). Of the existing species reported in Ireland, 61% are estimated to have been introduced from vessel traffic via ballast water or hull fouling, 22% from aquaculture, and 12% from neighbouring countries (Mirimin et al. 2019).

Establishment and spread

The introduction of a species to a novel area does not guarantee its survival or establishment, with only a few species forming self-sustaining populations. Following introduction, a new species population may fail to establish because of mortality or failure to reproduce. Environmental conditions often contribute to the failure of a novel population to survive (e.g., unsuitable climatic conditions prevent survival or reproduction), but may also be specific to the nature of the introduction event (e.g., only one sex of adults introduced etc.) (Blackburn et al. 2011).

A suitable environment invasive range is not the only factor affecting establishment. Characteristics of the receiving environment which may facilitate establishment, include disturbance, degraded habitats, and presence of artificial structures (Britton-Simmons and Abbott 2008; Airoldi and Bulleri 2011; Airoldi et al. 2015).

Even if survival and reproduction do occur, establishment may not occur in the long term if population growth is not high enough to sustain the population. Understanding whether a novel NIS is likely to spread can be hard if there is little information on its biology or habitat preferences, but is easier when the species already has evidence of spread elsewhere.

Larval dispersal is the primary mechanism for spread of many benthic species (Swearer et al. 2019). The characteristics of the pelagic larval phase of marine NIS may influence how quickly these NIS spread within recipient habitats. However, long dispersal phases are not necessarily required as anthropogenic vectors may assist in the secondary spread of a species. Many marine NIS are fouling ascidian species with short planktonic durations such as the high impact *Didemnum vexillum* (Beveridge et al. 2011). For example, marinas and ports have been shown to be highly interconnected with corresponding similarities in biofouling communities between highly connected sites (Ferrario et al. 2017).

Species which spread within an invaded range can cause impact at a range of scales, some may become part of the ecosystem without displacing native species, while others can cause impact for species, entire ecosystems or the people which interact with them.

1.2 Impacts of marine IAS

A small proportion of NIS which establish may spread widely and cause environmental, social, or economic impacts. When considering the impact of an IAS there are several components to help understand the impact. Jehske et al. (2021) identify these as the directionality of impact (i.e. whether it is positive or negative), the system impacted (biodiversity or socio-economic), the ways in which impacts are classified and measured (e.g. taking into account human values or nor), as well as the scales on which impacts are assessed (e.g. spatiotemporal, number of species, population size impacted) (Jeschke et al. 2014, 2021) .

Understanding the impact of an IAS can be complicated as impact may vary dependent on environment and invasion stage (Simberloff et al. 2013; Guy-Haim et al. 2018). Thus, the apparent impact of an IAS may vary depending on when after introduction it is assessed. Impacts may only become evident once IAS are widespread (Pyšek et al. 2020); alternatively, after an initial phase of rapid spread an IAS may become integrated into the receiving community (Kraemer 2019).

1.2.1 Environmental impacts

IAS can cause environmental impacts through direct and indirect mechanisms which can affect individual species, change community structure, alter habitats and affect ecosystem processes (Katsanevakis et al. 2014). The mechanisms by which impact is caused can include competition and predation, hybridisation, spread of parasites and disease, or through poisoning and toxicity (Grosholz 2002; Hawkins et al. 2015). In an assessment of 101 marine and coastal IAS in Europe, Katsanevakis et al. (2014) found that most of the IAS affected multiple native species, of these 65% had some negative impact and 35% some positive impact reported.

The impacts of IAS on native species can be minimal to extensive, affecting only local populations up to leading to whole species extinction. IAS' which alter the chemical or physical characteristics of habitats can be termed ecosystem engineers. The impacts of ecosystem engineers may be complex, potentially changing ecological communities in ways where it is hard to determine if the overall effects are good or bad. For example, habitat forming species may create structures for colonisation by native species, but may also alter community composition or reduce habitat suitability for native species (Bruno 2005; Rodriguez 2006; Gribben et al. 2015). Classification of the impact of these species can thus be semi-subjective, as the effects of these species may be viewed as positive or negative depending on the species valued and biodiversity metrics considered.

Increasing dominance of invasive species within communities can alter the structure of those communities, potentially changing food-web dynamics. For instance, a 30 year study of changes in subtidal seaweed assemblages in the Gulf of Maine demonstrated a shift towards dominance of invasive seaweeds, which reduced canopy height and increased biodiversity of species at the base of the food web (Dijkstra et al. 2017). IAS can also facilitate the establishment of other NIS, further contributing to marine community change (Simberloff et al. 1999).

Understanding the full impact of an IAS can be hard to detangle from other anthropogenic impacts, especially as they may combine to affect ecosystems synergistically (Pyšek et al. 2020). Habitat degradation or pollution may facilitate IAS establishment (Crooks et al. 2010; Airoldi and Bulleri 2011), while the construction of marine artificial structures may create more opportunities for colonisation of fouling species or facilitate spread between habitats (Mineur et al. 2012; Airoldi et al. 2015). Impacts of IAS on species or habitats may exacerbate or be masked by impacts such as eutrophication, habitat

degradation, or resource extraction. For example, large blooms of *Mnemiospsis leidyi* in the Black Sea which impacted commercially important fish species are thought to have been facilitated by extensive eutrophication from agricultural run-off.

1.2.2 Ecosystem service impacts

IAS which cause changes to the environment and native biota may affect the way in which the ecosystem functions. This can lead to changes in the ecosystem services provided, although the ways in which these are affected by IAS remains under researched (Walsh et al. 2016). This is especially true in marine systems, where research assessing and valuing ecosystem services is still relatively novel. If the effects of IAS on ecosystem services are not quantified, this could lead to inaccurate assessments of the costs and benefits of management action (Walsh et al. 2016).

Ecosystem-services are generally classified into regulating, provisioning, and cultural services. The impacts of IAS on provisioning and regulating services have researched in greater detail than impacts on cultural values (Pejchar and Mooney 2009).

In an assessment of the ecosystem services impacted by marine IAS in European seas, the services most frequently negatively impacted are food provision, followed by ocean nourishment, recreation and tourism, and lifecycle maintenance (Katsanevakis et al. 2014). Food provision was also most frequently positively impacted by IAS, with IAS also providing cognitive benefits, water purification and climate regulation (Katsanevakis et al. 2014). Of the marine IAS reviewed, the macroalgae and vascular plants (*Spartina* cordgrasses) impacted the most ecosystem services.

Some of the smallest IAS have caused significant negative impacts to ecosystem services. For instance, dinoflagellate blooms can cause largescale mortality of fish and shellfish including farmed shellfish (Silke et al. 2005; Katsanevakis et al. 2014).[1] In addition to direct impacts on species used for food, IAS can also indirectly affect commercial fish stocks by causing changes to important fish-supporting habitats such as seagrass communities, but can also potentially provide novel habitat.

Those IAS which act as ecosystem-engineers are likely to have the most complex impacts on ecosystem services, as they could result in changes to the chemical and physical structure of ecosystems. However, assessing ecosystem-service impacts of a species may be complicated as the negative impacts of an IAS on processes supported by native species may be compensated for if NIS fulfil functionally similar roles. For example, in an assessment of intertidal rock pool habitats in Spain, the invasive macroalgae *Sargassum muticum* was shown to inhibit the carbon storage capacity of native species but to provide functionally similar levels of carbon storage (Rossi et al. 2019).

1.2.3 Socio-economic costs

NIS impacts on native species, habitats and ecosystem services can affect the ways in which people interact with and use the environment (Jones 2017). This can include impacts on industries which directly rely on the marine environment, whether through changes to stock or target species or impacts on infrastructure. For example, fishing yields have been impacted due to loss or collapse of fisheries stock due to competition or predation by NIS (Oguz et al. 2008), or due to reduced fishing

[1] It should be noted that there is some disagreement regarding the classification of some dinoflagellates as non-indigenous as many now have a cosmopolitan distribution and distinguishing native and non-native ranges can be difficult (Gómez 2008).

efficiency (Galil 2007). Marine environments which are already under stress due to other human activities may be especially vulnerable to the establishment of invasive species (Oguz et al. 2008).

Socio-economic impacts associated with the spread of fouling species can result from increased cleaning costs, reductions in the yield of aquaculture species, loss of aquaculture stock and fewer opportunities for sale (Watson et al. 2009; Fitridge et al. 2012; Ferguson et al. 2017). Fouling species such as the ascidians *Styela clava* and *Didemnum vexillum* may also cause problems when found on artificial structures such as harbour or marina pontoons.

NIS may also impact the way in which people interact with their environment. This could be minor changes to how accessible coastal environments are, or behavioural changes associated with perceived risk of encounters with harmful species such as jellyfish or lionfish. However, they may also cause more serious impacts to public health, as is seen in the Levantine Sea where *Rhopilema nomadica,* a tropical scyphozoan represents a health risk due to the severity of its stings (Galil et al. 2014).

Costs associated with NIS can result from costs associated with attempts at management and control but can also stem from direct impacts to marine industries or users. The cost of IAS across all terrestrial and aquatic environments to the Irish economy were estimated as at more than €200 million in 2013 (Kelly et al. 2013).

Box 1 – Impact Assessment Scoring, comparing impacts across IAS

In recent years, there have been a number of attempts to standardise the ways in which impacts are assessed across species. These include the development of frameworks for impact assessment, both for biodiversity and socio-economic impacts. These tools can be used to support management decisions, by allowing identification of priority species for surveillance, prevention, or control. Comparison of impacts across species is an important component of identifying horizon species likely to imminently arrive and cause detrimental impacts (Roy et al. 2014). Horizon scanning exercises increase awareness of high impact species and can be used to guide research and management.

Examples of impact assessment frameworks include the IUCN Environmental Impact Classification for Alien Taxa (EICAT) scheme for environmental impacts (Hawkins et al. 2015; IUCN Species Survival Commission (SSC) 2020), and the Socio-economic Impact Classification for Alien Taxa (SEICAT) scheme for socio-economic impacts (Bacher et al. 2018). The INSEAT framework evaluates the impact of IAS on ecosystem-services and includes consideration of both the positive and negative consequences of IAS (Martinez-Cillero et al. 2019).

The choice of framework depends on the context which the scores are going to be used for. The EICAT scheme only considers the negative impacts of IAS spread, and has been critiqued for not considering the potential positive consequences an IAS may have on recipient environments (Vimercati et al. 2020). Where EICAT scoring is used to identify horizon species or prioritise prevention measures this is justified, but where management decisions regarding eradication and long-term control are made based only on negative consequences this could lead to conflict between different stakeholder groups which perceive IAS impacts differently (Vimercati et al. 2020).

In assessing species impacts to compare or prioritise species management, expert judgement is often sought which allows for review even if evidence is absent (Martinez-Cillero et al. 2019).

EICAT Impact mechanisms	SEICAT Constituents of human well-being
Competition	Safety
Predation	Material and immaterial assets
Hybridisation	Health
Transmission of disease	Social, spiritual, cultural relations
Parasitism	
Poisoning/toxicity	
Biofouling or other direct physical disturbance	
Grazing/herbivory/browsing	
Chemical impact on ecosystem	
Physical impact on ecosystem	
Structural impact on ecosystem	
Indirect impacts through interactions with other species	

EICAT Impact Scores	
Massive (MV)	Causes local extinction of at least one native taxon which is natural irreversible even if the alien taxon is no longer present the native taxon cannot recolonise the area
Major (MR)	Causes local or sub-population extinction of at least one native taxon which is naturally reversible if the alien taxon is no longer present.
Moderate (MO)	Causes population decline in at least one native taxon but no local population extinction
Minor (MN)	Causes reduction in individual performance (e.g. growth, reproduction, defence, immunocompetence) but no decline in local native population sizes
Minimal Concern (MC)	Negligible level of impact: no reduction in performance (e.g. growth, reproduction, defence, immunocompetence) of individuals of native taxa

INSEAT Ecosystem service impact assessment semi-quantitative scores	
4	The species leads to an **increase** in the provision of the ecosystem service, which is both **intense** and **irreversible**
3	Substantial **increase** in the provision of the ecosystem service; the effect is **reversible** if the species is managed or removed
2	**Noticeable increase** in the provision of the ecosystem service but **reversible** if the species is managed or removed
1	The **increase** in the provision of the ecosystem service is **too small** to be significant
0	**No impacts** detectable/ecosystem services **not applicable** to this species
-1	The **reduction** in the provision of the ecosystem service is **too small** to be significant
-2	**Noticeable reduction** in the provision of the ecosystem service; it is **reversible** if the species is managed or removed
-3	**Substantial** reduction in the provision of the ecosystem service; the effect is **reversible** if the species is managed or removed
-4	The species leads to a reduction in the provision of the ecosystem service, which is both **intense** and **irreversible**
-	I don't know
-	Data deficient

1.3 Management of marine IAS

The potential consequences of IAS invasion have led to the need to manage or control invasive species. This has been recognised in international legislation such as in the Convention on Biological Diversity (CBD) and Sustainable Development Goals (SGDs).

Management activities are usually targeted at different stages of a species invasion. Given the difficulties associated with removing a species once it has established in a new environment, focusing on preventing species arrival has been seen as a priority (Hulme 2006).

A hierarchy for management of IAS has been recognised in policy as well as scientific literature, with the CBD identifying preventative measures as the most cost-effective mechanism to prevent IAS impacts. This is followed by early detection and rapid response, and when these options are unsuccessful or inapplicable long-term management and control. This hierarchy recognises that eradication of an IAS once it has become established and widespread is often unfeasible, especially in the marine environment (Vilà et al. 2010).

This basic hierarchy has been elaborated on and further refined by many studies across the field of invasion research. The need to use a consistent terminology in the management and assessment of invasive species has led to the development of a management framework based on potential actions targeted at different stages of the invasion process (Robertson et al. 2020). Eight types of active management are identified in Table 1.2 below.

The framework proposed by Robertson et al. (2020) distinguishes between active management and other methods which may support management measures, such as public education, awareness raising, monitoring and surveillance and the decision-making processes which can include risk analyses and cost-benefit analysis. However, it is these supporting tools that are an essential element of management success, as inadequate consideration can lead to failed management attempts. Understanding what the aims are and likelihood of success of any management intervention is critical.

Multiple management plans may be needed to address different species, environments, and spatial scales. A national framework with more detailed management at a local level or targeted to a specific species could enable the best use of local knowledge and expertise, and can draw on the expertise of local stakeholders and volunteer groups to help monitor for target species and increase compliance with biosecurity measures.

Table 1.2 – Management terms and objectives proposed by Robertson et al. (2020), with potential management examples in the marine environment given.

	Management term	Management objective	Decision support tools	Example in the marine environment
Prevention	Pre-border pathway management	Reduce the uptake of the species and its transport outside the area of interest	Pathway risk analysis	e.g. Anti-fouling paint upkeep; Ballast-water exchange
	Interception	Intercept the species on first entry into the area of interest	Identification of Critical Control Points (HACCP)	e.g. Screening of vessel fouling on arrival in ports
Captive management	Limits to keeping	Limit the keeping or cultivation of the species within the area of interest	Sector-wide biosecurity plans	e.g. Aquaculture species assessments & import risk assessments
	Secure keeping	Ensure the security of the species help in captivity/cultivation within the area of interest	Site biosecurity plans	e.g. Growing infertile stock species
Eradication	Eradication	Remove the entire population from the area of interest – with no immediate risk of re-introduction	NNRM Scheme – Assess eradication options	e.g. wrapping and treating with chemicals
Long-term management / Control	Complete reproductive removal	Remove the entire population from the area of interest – but with remaining risk of re-invasion or further reproduction if not managed	NNRM Scheme – Assess eradication options	
	Containment	Limit the spread of a reproducing population within the area of interest		
	Suppression	Reduce the distribution or abundance of a population within the area of interest		e.g. Physical removal of IAS in vulnerable habitats e.g. seagrass meadows
Impact adaptation and ecosystem restoration		Actions taken to reduce associated impacts without or in conjunction with actions to reduce spread		e.g. native oyster restoration
No management		When a species is widespread, and no action is taken to reduce its spread		

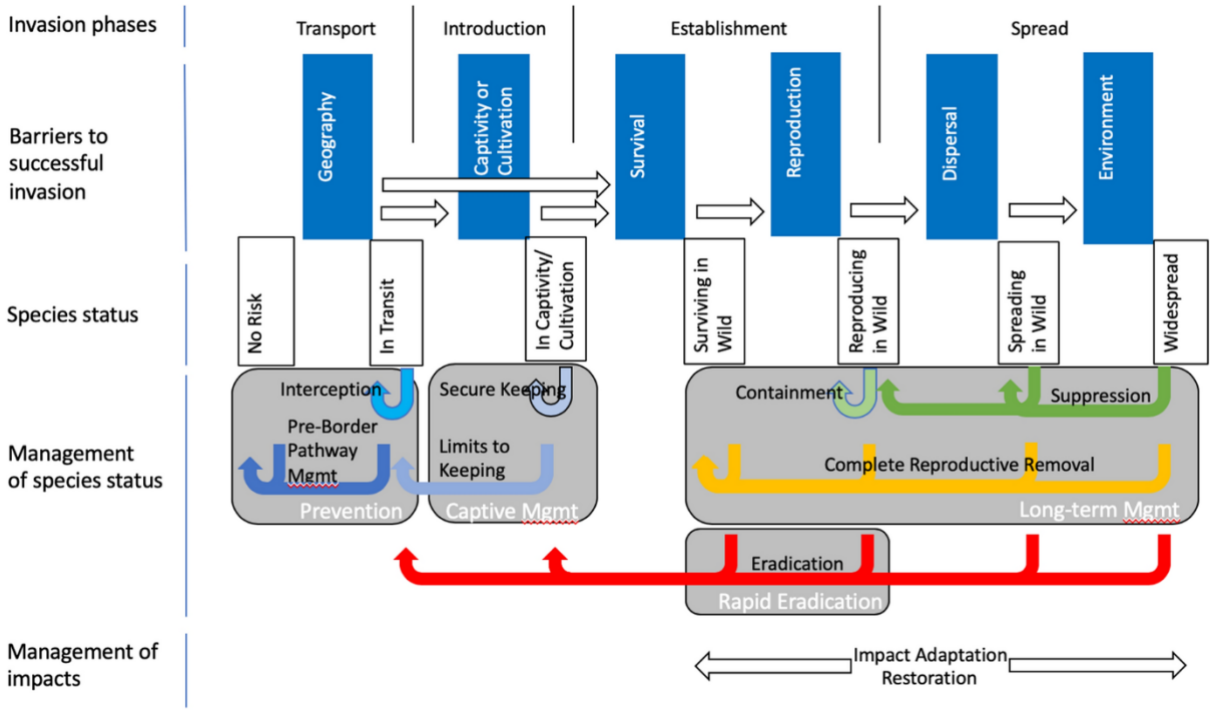

Figure 1.1 – Figure reproduced from Robertson et al. (2020). Possible management actions during a biological invasion with barriers to successful invasion described by Blackburn et al. (2011) represented by blue bars.

1.3.1 Active management measures

Prevention

Preventing species arrival primarily involves addressing the different pathways of species transport. There have been a number of international policies developed to address the movement of species by the shipping industry. The IMO adopted the Ballast Water Management Convention[2] and has set out guidelines on the management of hull fouling. Biosecurity measures for the aquaculture industry also help prevent transfer of unwanted organisms, although this is often more focused on disease than NIS. Development, adoption and compliance with biosecurity measures is still low in many sectors, and there is often little regulatory framework to aid with this. Recreational water users are often excluded from the requirements which commercial stakeholders have, but remain a high risk vector for NIS. Increasing awareness of the risks of NIS transfer, and of the steps which can be taken to reduce spread could lead to fewer introductions of NIS.

The association of species introductions with areas of high maritime activity makes ports and harbours high risk areas. These areas also introduce large volumes of artificial structures into the marine environment, providing hard substrate for the settlement of fouling species. There is a high level of connectivity between Irish ports and those in the UK and Europe, with Dublin Port, Cork Harbour and Belfast Lough the main link with British ports (Mirimin et al. 2019). Furthermore, some species have larval durations long enough that they could be capable of dispersing naturally between the UK and Ireland.

[2] International Convention for the Control and Management of Ship's Ballast Water and Sediment

Rapid response and eradication

Eradicating an IAS once it has established in the marine environment is incredibly difficult and has only been achieved in a few select cases. Eradication has a far higher chance of success if attempted early in the invasion process, before a species has become widespread (Simberloff 2003). However, the low densities of individuals which make eradication possible may also reduce the likelihood of detecting a species' arrival in the early stages of its invasion (Mehta et al. 2007). The establishment of a monitoring and surveillance plan which facilitates early-detection and rapid response to novel arrivals is thus a critical component in creating a window for eradication. This is discussed in more detail in Section 1.3.2.3.

There are examples of successful eradications of marine NIS around the world, such as the removal of *Mytilopsis sp.* from harbours in Australia, and the localised removal of *Caulerpa taxifolia* in California (Anderson 2005). Whether eradication is feasible depends on a number of criteria, including the status of the invasion and the availability of appropriate control tools. In order to prioritise management of species where eradication is more likely to be successful, a framework for assessing eradication feasibility was developed to allow comparison between species, this is described in more detail in Box 2 (Booy et al. 2017, 2020). Using this tool as part of contingency planning for the arrival of horizon species, combined with a process for deciding on and licensing control action, would greatly enhance the likelihood of eradication success.

Box 2 - Assessing eradication feasibility: The Non-native Risk Management (eradication) Scheme (NNRM)

A framework to assess the feasibility of NIS eradication has been developed that can be used to evaluate eradication strategies and prioritise NIS for management action (Booy et al. 2017). The framework can be used to evaluate eradication strategies for established or horizon species. Where a NIS has not yet established, identification of a possible eradication strategy can form part of contingency planning for IAS with a high likelihood of arrival. The assessment framework can be used at different spatial scales and has been used to prioritise management across countries and at national and regional scales (Booy et al. 2020).

In the UK the NNRM tool is used in conjunction with other risk analysis tools to help make management decision for NIS at a national level. Guidance information and a template can be found on the GB Non-native Species Secretariat (GB NNSS) website on the Risk Management page.[1]

To carry out the scoring for published prioritisation exercises, a similar approach to that used in horizon scanning exercises combining expert judgement with review and consensus building has been used (Roy et al. 2014, Booy et al. 2017). This approach aims to reduce subjectivity and bias but is sensitive to the expertise and experience of the assessors.

The table below shows the scoring system for the different criteria an eradication strategy is evaluated against.

Table 1.3 – The assessment criteria for response scores used in the Non-native Risk management (eradication) Scheme (NNRM). This can be used to compare which species should be prioritised for management, but also provides a useful overview of the constraints and advantages of management action.

Criteria	Response Score				
	1	2	3	4	5
Effectiveness	Very ineffective	Ineffective	Moderate effectiveness	Effective	Very effective
Practicality	Very impractical	Impractical	Moderate practicality	Practical	Very practical
Cost	>£10M	£1-10M	£200k-1M	£50-200k	<£50k
Negative impact	Massive	Major	Moderate	Minor	Minimal
Acceptability	Very unacceptable	Unacceptable	Moderate acceptability	Acceptable	Very acceptable
Window of opportunity	< 2 months	2 months - 1 year	1 – 3 years	4-10 years	>10 years
Likelihood of reinvasion	Very likely	Likely	Moderate likelihood	Unlikely	Very unlikely
Conclusion (overall feasibility of eradication)	Very low	Low	Medium	High	Very high

For every response score, a confidence rating should be given based on the level of evidence and uncertainty. This is scored on a 3-point scale: 1 (low), 2 (medium), 3 (high).

The NNRM scheme does not identify whether species are a priority for management based on risk of establishment or likely impact but can be combined with risk scores from risk assessments or horizon scanning exercises. While the criteria and scoring are intended to be used to assess eradication, the same considerations apply to long-term management and the assessment could be used to guide species management plans.

The results from this process can act as a first step in identifying eradication or control strategies, and aid decisions on whether action is needed. Following this a species management plan can be prepared, which requires more thorough consideration of the practicalities of species management and more detailed knowledge of the local area. Input from local stakeholders, IAS experts, local authorities should be considered during this process.

[1] GBNNSS Risk management https://www.nonnativespecies.org/non-native-species/risk-analysis/risk-management/ (Accessed 29th Aug 2022)

Long-term management and control

While eradication may be the primary goal of IAS management strategies, if eradication is not feasible the long-term management of established NIS may still deliver benefits including preventing further spread, limiting impacts on vulnerable habitats, or reducing impact on social or economic concerns.

Long-term management through population control involves suppressing population size or abundance, and Robertson et al. (2020) divide this based on the change in species status. Three management objectives are identified: (1) complete reproductive removal of the entire population from the area of interest, (2) containment to prevent spread to new areas, and (3) suppression to reduce distribution or abundance (Figure 1.1). Monitoring the outcomes of long-term control activities is crucial and often not effectively implemented, but without this knowledge it can be hard to determine if management aims have been achieved (García-Díaz et al. 2021).

The methods used to eradicate a species or contain or supress its expansion may be similar. These can range from physical removal, treatment with chemicals or exposure to conditions beyond its physiological tolerances e.g. air drying or freshwater. Use of control measures needs to consider the collateral consequences for the supporting habitats and species. The public perception of control measures may vary, depending on the species in question which can affect political and financial support for control measures. For example, control of terrestrial mammals can be seen by the public as cruel or inhumane, even if awareness of impacts is high.

A number of IAS policy objectives include the goal to reduce IAS impacts, including the draft of the post-2020 Convention on Biological Diversity (CBD) target on IAS (Essl et al. 2020; García-Díaz et al. 2021). Despite the importance given to reducing environmental impact, if eradication or population control is not considered feasible further management options to mitigate IAS impact are often not explored. García-Díaz et al. (2021) advocate for an impact-based approach to long-term IAS management, which could incorporate population control measures, adaptation, and mitigation measures (see below), and which may also allow for consideration of the potential positive impacts IAS may be providing for some sectors of the community.

Impact adaptation and restoration

Where reducing the spread of a species is not feasible, management actions may be taken to reduce associated impacts (Robertson et al. 2020). For example, in the Netherlands the spread of the invasive Asian oyster drill, *Ocinebrina inornate*, is affecting Pacific oyster cultivation. To reduce impact on aquaculture yield, a transition to growing oysters in bags on trestles as opposed to bottom culture on the seafloor is being recommended to remove access by the oyster drill (Babaran 2017).

In some cases, restoring native populations may reduce impacts and prevent spread of NIS. An Irish example of this from terrestrial environments is the regional decline in grey squirrel populations following recovery of native pine martens (Sheehy and Lawton 2014; Flaherty and Lawton 2019).

1.3.2 Decision making and management tools

There is a need for evidence to support decisions about management to allow management action to be taken. Implementation of active management measures is often hindered by lack of access to funds, no clear layout of the responsibility for management action, or lack of regulatory power to permit management intervention. Tools and planning procedures which increase the likelihood of management action being taken can address one or several of the stages of an IAS invasion.

Biosecurity planning

Biosecurity planning is the process by which individual sectors or regions can identify opportunities to prevent species introductions occurring and set out contingency measures to allow rapid action should a species introduction occur. Biosecurity planning may range from site, operation, species plans (Cook et al. 2014a). The development of a biosecurity plan will usually involve experts working with stakeholders to identify the critical control points in their operations where changes to practice could help reduce the likelihood of species introduction (Cook et al. 2014b). For example, this could involve cleaning gear before it is moved between water bodies. Supporting local authorities, stakeholders and event organisers in the development of biosecurity plans could reduce the number of introduction events.

Surveillance and monitoring

Establishing a monitoring and surveillance plan for IAS is essential to the effective management of invasive species. A monitoring and surveillance plan may fulfil several objectives. The early detection of new arrivals allows for rapid response, and for eradication to be considered before a species is established or widespread (Simberloff 2014). Monitoring of known IAS that have already arrived in the country may enable assessment of range expansion and establishment status (Inglis 2006). This is important in assessing the impact on the recipient environment and determining whether control action is needed.

Surveying large areas of the coast to detect the arrival of novel species is impractical, thus a risk assessment process can be used to establish priority areas to survey on a regular basis (Hulme 2009). In developing a surveillance plan, spatial data on pathways, points of entry, and available habitat can be combined to identify high-risk sites (Hulme 2009). These are frequently based around ports but could include bays with high numbers of aquaculture sites or sites with large amounts of artificial structures. Examples of surveillance schemes which have been developed based on spatial assessments of high risk pathways can be found in the Shetland islands (Shucksmith and Shelmerdine 2015). In New Zealand, habitat preferences and dispersal models were combined to inform survey designs at eleven high-risk ports (Inglis 2006; Inglis et al. 2006).

Reports from marine users and the general public can be crucial in detecting new species across wider areas. Citizen science monitoring schemes such as Seasearch (https://seasearchireland.ie) can also be utilised to help track spread of IAS. The use of citizen science and reports from the public to detect novel invasions takes advantage of significantly larger numbers of people. To be effective, publicly available information on target IAS and established reporting structures need to be in place to support identification of suspicious specimens. This can be difficult for many marine IAS, as some species may only be distinguished through close analysis of identifying features (e.g. bryozoans). However, training events and the publication of ID guides may aid in helping to develop skills of key stakeholders. Data collected through public reporting may also not be suitable to understand population numbers as sighting frequency may be driven by media reporting and outreach as opposed to changes in populations (Koen and Newton 2021).

There are also a number of established long-term surveys which collect marine data which may detect novel species, and a large amount of survey work undertaken by private companies for EIAs and planning purposes. For example, in the UK a baseline survey undertaken by an environment consultancy for planning purposes led to detection of the NIS brush-clawed crab, *Hemigrapsus takanoi*, in Suffolk, England (Ashelby et al. 2017).

Mirimin et al. (2019) present an overview of best practice and different monitoring methods, which are more extensively reviewed in Lehtiniemi et al. (2015). It is likely that in the future molecular methods will play a more significant role in monitoring NIS, especially use of environmental DNA (eDNA) techniques (Mirimin et al. 2019; Fernandez et al. 2021).

Contingency planning

In the event that a novel NIS is detected, a rapid response is the best chance for effective eradication. Contingency plans can improve the speed at which action is taken by setting out a framework for decision making in the event of a novel species introduction.

In the UK, national contingency plans have been developed for terrestrial, freshwater and marine environments. The contingency plans set out the responsibilities for overseeing and carrying out management action triggered by detection of a novel NIS. For example, in England the Department for Food, Environment, and Agriculture (DEFRA) has overall policy responsibility, with the responsible authority for delivering action dependent on where the species is detected and which sector (fisheries, aquaculture or the environment) is most likely to be affected. The responsibility authority leads a Response Group, which recommends action for which an Operational Group implements on the ground (GB NNSS 2020).

Priority species are identified via horizon scanning and risk assessments processes. If a non-priority species arrives, a rapid risk-assessment can be used to determine if response is warranted.

Box 3 – The New Zealand Marine High Risk Site Surveillance (MHRSS) Scheme

In New Zealand, a Marine High Risk Site Surveillance (MHRSS) scheme[1] was developed to monitor priority locations within and near eleven international shipping ports and marinas. The primary objective of the MHRSS is to:

- "detect incursions of high-risk non-native organisms that are not known to be present in New Zealand",

but the scheme also has two secondary objectives, to "

- Detect incursions of other non-native or cryptogenic organisms that have not previously been recorded in New Zealand, and
- Detect range extensions by established non-native or cryptogenic organisms that exhibit characteristics of pests."

The specific locations sampled within each harbour area are informed by target IAS habitat preferences and likely hotspots for dispersal. Surveys are conducted bi-annually, conducted by an experienced team, and utilise a variety of methods to survey artificial structures, intertidal, and subtidal habitat (Woods et al. 2020). These methods include benthic sled tows, crab traps, diver, and shore searches.

Sample plans for target IAS were developed using niche models and particle dispersion models (Inglis et al. 2006). Other aspects of the sampling methods such as efficiency, cost-effectiveness, collateral impacts, and feasibility were considered when designing the surveillance strategy.

[1] https://www.marinebiosecurity.org.nz/surveillance/ (Accessed 29th Aug 2022)

1.4 Modelling as a tool in IAS management

Access to the appropriate evidence to support management action or feed into decisions on management action is crucial; however, this can be challenging to acquire if a NIS is newly introduced to a region. Predicting the impact an IAS may have and developing plans to manage an IAS requires an understanding of how an IAS will spread in its introduced environment (Blanco et al. 2021). There are a number of components which influence this including the species characteristics (e.g. reproductive strategy, fecundity, mobility), the physical characteristics of the receiving environment (e.g. current dynamics, suitable salinity temperature range), secondary dispersal vectors, and the invasibility of the receiving biological community (Robins et al. 2017).

Predicting IAS dispersal in new environments therefore requires consideration of a range of different factors. Models can be used to help understand how a species may spread in new environments, but also to test how these factors can influence dispersal (Clarke Murray et al. 2014). This is especially useful in the research and management of IAS as data on the distribution of a species may be limited due to lack of survey effort, or because the aim is to predict the potential spread of a species which has not yet or only recently been introduced.

Different species disperse using various methods which affect the rate and distance at which a NIS will spread into new areas. For many benthic species, the larval stage is the most dispersive and in some cases the only mobile stage for many NIS where the adult phase is attached to the substrate. Ballast water may transport larvae directly or spawning may be precipitated by the arrival of a species into a novel environment. For example, changes in temperature or salinity upon entry of a vessel into port may trigger spawning of organisms within the hull fouling community (Ferrario et al. 2017). In other cases, where culture of adults is occurring, spawning may be the means of escape into the wild.

1.4.1 Species Distribution modelling

Developing effective management plans relies on knowledge of the spread potential of invasive species as this can be used as one indicator of the potential impact of a species (Blanco et al. 2021). Species distribution models have been used to map the probability of species presence by using information on environmental and physical factors which might affect establishment and population persistence (Elith and Leathwick 2009). Thus knowledge of species physiological tolerances can be combined with environmental data to predict the global range of a species or to identify areas within a country which could support a novel species (James et al. 2015).

Sea temperature is often the predominant physio-chemical factor influencing a species' distribution (Sorte et al 2013). Salinity is another important variable, although models may not be detailed enough to account for localised coastal variation in salinity. Species distribution models alone may also fail to account for hydrological regimes which may influence the dispersal of species.

In the case of IAS it has been suggested that it may also be useful to incorporate anthropogenic data as this can correlate with the main vectors of species introduction. A recent study by Blanco et al. (2021) predicting the occurrence of invasive macroalgae on the Iberian Peninsula included anthropogenic variables as well as environmental factors. While physio-chemical factors remained the most important variables, including the size of vessels, distance from ports, and size of population improved the estimates of probability of occurrence for five invasive macroalgae species, excluding *Codium fragile* (Blanco et al. 2021). The species distribution maps produced reliable estimates of

target species presence and absence, enabling detection of high-risk areas for invasive macroalgae introduction in the area (Blanco et al. 2021). A similar approach was used by Gallardo et al. (2013).

Species distribution modelling has a range of uses in the marine environment, many of which can be used to help manage marine systems (Guisan et al. 2013); from identifying areas suitable for restoration of vulnerable habitat or predicting occurrence of rare or priority species that are difficult to survey (Stuart et al. 2021). It can also be used to supplement survey data and create predicted habitat maps for areas with sparse survey data (Blanco et al. 2021). In relation to IAS, species distribution models can be used to predict current or future range limits, and estimate probability of occurrence. This can be used to guide survey efforts or supplement information of species distribution where survey data is sparse, enabling decisions to be made on IAS management.

Management decisions need to be based on the best available information however the window in which to make these decisions can be very short, and waiting for large scale surveys to be carried out might be prohibitive to management action. Using species distribution models to predict the extent and area of habitat which might be suitable for an IAS ahead of its arrival can provide useful information on the level of risk a species might pose for a region. The use of horizon scanning workshops to identify likely novel introductions has resulted in horizon lists for species where SDM could proactively be developed

Habitat suitability can be calculated based on current environmental conditions and for future climate change scenarios (Goldsmit et al. 2018). As climate change affects local environmental conditions, the list of NIS capable of tolerating Irish seas may change, as may the extent of habitat available to them (Stachowicz et al. 2002).

1.4.2 Particle tracking models

Particle tracking models (also known as biophysical models, Lagrangian transport models or larval transport models) simulate larval dispersal by combining predicted water movements from an oceanographic model with a species' larval biology (e.g. swimming ability, buoyancy, daily activity budget). These particle tracking models can predict the transport of an IAS larva away from its site of introduction (Robins et al. 2017). By tracking the movements of many thousands of simulated larvae areas of high larval "propagule pressure" can be identified. This predicted larval propagule pressure can then be combined with habitat suitability estimates from a species distribution model to identify areas at high risk of IAS establishment from specific sources sites of IAS. This modelling approach can produce predictions with a detailed spatial and temporal resolution that can then be further analysed to guide monitoring and marine spatial planning applications.

Accounting for larval behaviour in a particle tracking model is crucial, since it has important consequences on the predicted transport of the simulated larvae by the ocean currents. Studies have shown that differences in larval behaviour can lead to bigger differences in the predicted transport than the variability in current patterns (North et al. 2008; Robins et al. 2017).

It is also important that the oceanographic model has a fine temporal and spatial resolution if the model is being applied close to the coast. A temporal resolution of less than six hours is required in order for tidal currents to be resolved. Coastal waters also require a spatial resolution that can capture important coastline features and important changes in bathymetry.

1.5 Project Development

1.5.1 Project objectives

As detailed in the previous section, established IAS may have a range of impacts on their invaded environment. Understanding the potential impacts of an IAS is an important element in determining the need for management interventions. Applying control and eradication measures can also be very costly, thus determining which interventions are most likely to succeed and where to target them can help identify best value for money. To understand which management interventions are appropriate for a species, knowledge is needed of the species current distribution, the potential extent of its spread, what damage this spread could cause and what potential management interventions are available. Conducting field surveys to gather this information could be time consuming without information on where a species may have spread to and could threaten the rapidity with which management measures can be applied.

Combining species distribution and particle tracking modelling allows estimation of the likely spatial distribution of the spread of an IAS. These tools can then be used to guide and inform decision about IAS management. This project aims to develop these models for two focal IAS to demonstrate the applicability of this approach in decision-making for IAS management in Ireland.

This project chose two focal species to use to understand the potential applicability of modelling in aiding management decisions. The first is the Pacific Oyster, *Magallana gigas*, a species widely cultivated in Ireland, which has already been observed to have recruited in the wild in some locations in Ireland. The second species, the Asian shore crab, *Hemigrapsus sanguineus*, has not yet been found in Ireland but has established extensive populations in areas of mainland Europe, with individuals also found in Great Britain. This species has been identified as a horizon species of concern for Ireland (Lucy and Davis 2020).

This report presents each species as a case-study, with an assessment of potential environmental and socio-economic impacts of spread in Ireland and the management measures which could be applied. The spread maps produced from the modelling work have then been used to assess how management measures could best be applied, as a demonstration of how this approach could be used to guide decisions on invasive species management.

1. Develop Lagrangian transport models for the larvae of focal species
2. Map the IAS spread potential of focal species larvae
3. Characterise the ecological and socio-economic consequences of IAS spread
4. Summarise potential management actions and their efficacy
5. Map the impact of selected management actions on IAS spread

1.5.2 Focal invasive alien species

The Pacific Oyster was introduced in Europe as a commercial aquaculture species in the 1960s, and in some areas quickly became established in the wild (Troost 2010). It has since formed extensive wild reefs in some parts of its invaded range within Europe, and as sea temperatures have warmed, wild populations are being found further north in Scandinavia, the UK and Ireland (Kochmann et al. 2013; Dolmer et al. 2014; Smith et al. 2015). In Ireland, feral individuals have been found in a number of locations with more established populations found in Lough Swilly, Lough Foyle, and in Strangford Lough in Northern Ireland (Guy and Roberts 2010; Kochmann et al. 2013).

In Europe, two *Hemigrapsus* species have been introduced both of which have their origins in East Asia*; H. sanguineus*, the Asian shore crab, and *H. takanoi*, the brush-clawed shore crab. The primary vectors of introduction are thought to be associated with shipping pathways, either in ballast water or associated with hull fouling communities (Epifanio 2013). However, there is also the possibility of transfer with aquaculture material and via natural dispersal within regions. The two species are considered invasive in areas of mainland Europe where they can reach very high densities. Both *Hemigrapsus* spp. have also spread to Great Britain, where there are individual recording of *H. sanguineus* and evidence of a reproducing population of *H. takanoi* (Seeley et al. 2015; Ashelby et al. 2017). Both species have been identified as horizon species in Ireland (Lucy and Davis 2020). The choice of using *H. sanguineus* for this study was guided by the need for detailed information about the larval behaviour of the species in the Lagrangian transport model, as there has been more extensive research on the larval stages of this species than for *H. takanoi*. Nonetheless, in some studies the larval behaviour of *H. sanguineus* has been used as a proxy when investigating *H. takanoi* (Ashelby et al. 2017).

The species were chosen as exemplars as they differ in key aspects of their invasion biology and status in Ireland. As mentioned above, *M. gigas* is already present in the wild in Ireland but has not yet spread to the same extent as seen elsewhere in Europe. *H. sanguineus* is thought likely to arrive in Ireland but so far has not been detected. This has implications for opportunities for management and control. Furthermore, they have different primary vectors and present different problems in terms of management of a mobile species in comparison to a settled species. The larval stages of the species are also characterised by different behaviours and thus allow us to explore how larval behaviour in the pelagic environment affects the estimated spread of the species. While these species allow exploration of some of the variation in marine IAS introduction, they will by no means represent a full exploration of all possible issues in marine IAS management in Ireland. However, they will hopefully provide an exemplar framework of modelling and its application that can be built upon for other species.

1.5.3 Investigating IAS impact and potential management methods

To characterise the ecological and socio-economic impacts of the target IAS, a systematic search of published literature and invasive species databases was conducted. The literature was reviewed to determine the applicability of reported impacts to the Irish context, using the EICAT, INSEAT and SEICAT frameworks to guide assessment of environmental, ecosystem-service, and socio-economic impacts. While these frameworks are intended for use in species comparisons, the scales of impact provide a useful standardised scoring system.

To identify potential management actions, a review of published academic and grey literature was conducted. From this review, experts with experience of management in areas with established populations of the target IAS were contacted. Informal interviews were conducted to determine whether management approaches trialled elsewhere could be applicable in Ireland.

The results from this analysis have been used to interpret the outputs from the spread models for each species, both in terms of identifying areas vulnerable to impact and in assessing the effectiveness of potential management interventions.

1.5.4 Habitat suitability for establishment

Suitable habitat for establishment of the target IAS in Ireland was estimated using species distribution models. These rely on using independent environmental variables and their association with existing

occurrence data for the species of interest. Two machine learning models were used, Maxent and Boosted regression trees with the sdm R package (Friedman 2001; Phillips et al. 2006; Naimi and Araújo 2016).

Species occurrence data from around Europe was sourced from GBIF and complemented from other atlases and affiliated peer-reviewed articles (OBIS, NBN). The environmental variables included in the model were physical (e.g., temperature), chemical (e.g., dissolved oxygen) and biological (e.g., Chlorophyll A concentration) data. The variables were sourced from open-source interfaces (Marine Copernicus, MODIS, EMODnet) and gridded over the European waters. The models did not include information on substrate or marine community type because of discontinuous information with heterogenous levels of precision.[3]

The output from the *M gigas* and *H. sanguineus* species distribution model was incorporated as a post-process condition to the Lagrangian transport model, such that settlement of model-transported larvae would only occur if the habitat was considered suitable for establishment.

1.5.5 Modelling larval spread of target IAS in Ireland

Two regions within Ireland were used to test the application of larval tracking models to understanding NIS spread, both located on the west coast of Ireland. The first was Galway Bay, and the second the area around Bantry Bay.

Galway Bay is located in between Co. Galway and Co. Clare, it is a large embayment characterised by the presence of a number of shallow enclosed sub-basins, and semi-protected from the Atlantic Ocean swell by the Aran islands and Black Head. Bantry Bay is in west Cork, and the region is characterised by long narrow bays.

The choice of these two areas was influenced by the presence of two local scale hydrodynamic models. While the choice of region is restricted by the availability of high-resolution hydrodynamic models, the methodology developed by this study could be adapted for other areas of the coastline as higher resolution data becomes available in the future. Detailed information on the methodology used to run the models is presented in a separate technical report.

Tracking larval dispersal

A hydrodynamic model and Lagrangian transport model were coupled with knowledge of larval spawning and behaviour to simulate the dispersal of larvae following spawning.

The hydrodynamic models used in this project are based on the Regional Ocean Modelling Systems, commonly named ROMS (Shchepetkin and McWilliams 2005).

The Galway Bay area is covered by a local scale model (named CONN) covering an area on the west coast of Ireland from 10.8°W to 8.9°W and from 52.95°N to 53.73°N. The Bantry Bay area is covered by a local scale model (named BANT) which covers an area on the south-west coast of Ireland from 10.8°W to 8.9°W and from 52.95°N to 53.73°N. The horizontal resolution of these models is c. 200 m, a much finer scale than is available in the larger resolution regional hydrodynamic model (NEATL)

[3] For a more detailed description of species distribution model development please refer to the technical report.

which covers much of the northwest European continental shelf, but which has a variable horizontal resolution between 1.1 – 1.6 km in Irish coastal waters.

Lagrangian transport modelling was used to simulate larval transport. The open source LTRANS software (version 2b) was used to carry out transport simulations (Schlag and North, 2012). The transport models were run for three years, 2018, 2019 and 2020 for which there is an hourly time resolution of the hydrodynamic data. Biological information on spawning location and release condition, larval behaviour and pelagic life duration were incorporated into the Lagrangian transport model. This is summarised below for each species.

Spawning and larval behaviour

Magallana gigas

Information on known locations of wild and farmed oysters within Galway Bay was used to inform the locations of larval release sites for *M. gigas* (Figure 1.2). Known farm locations were obtained from aquaculture licenses with information on the use of diploid or triploid spat noted. Data on the location of wild oysters in Ireland was collated from a variety of sources including targeted surveys for Pacific Oysters carried out in 2009 (Kochmann et al., 2013). Data on wild oyster was also sourced from record databases including GBIF, and the National Biodiversity Data Centre. Any duplicate records were removed.

To assess connectivity and allow for investigation of the impact of further *M. gigas* spread, another level of modelling was implemented by assuming hypothetical releases of larvae over all sites with favourable water temperature conditions for spawning (see below). These sites were restricted to intertidal and subtidal locations up to 1 m depth, as there have been no subtidal records of *M. gigas* in Galway Bay.

Spawning of *M. gigas* usually occurs between July and August, and is dependent on water temperatures (Troost 2010). In our model, spawning of larvae was triggered at each site based on a set of temperature conditions. For each year, the site was required to exceed the 592 day degrees required for gamete development.[4] Once the day degrees were reached, spawning was triggered if during a following period of 6-7 consecutive days, the water temperature exceeded 16 ºC. This is the minimum temperature observed for Pacific oyster spawning, although the optimum temperature is 18 ºC (Ruiz et al. 1992; Dankers et al. 2004; Castaños et al. 2009) . The models were run for three years (2018, 2019, and 2020) and differences in temperature patterns between the three years resulted in some variation in the timing of spawning, and which release sites met the temperature requirements (Figure 1.2).

M. gigas larvae can spend three to four weeks in their pelagic phase before settling, with pelagic larval duration influenced by temperature (Troost 2010). During this phase they transition through three stages, trochophore, veliger and pediveliger, before settling on hard substrate such as rock, cobble or shell (Kochmann et al. 2013; Robins et al. 2017). For the simulation of Pacific oyster dispersal, larval tracking lasted for 31 days to simulate the pelagic life-span of Pacific oyster larvae in cold waters (Kheder et al. 2010; Troost 2010).

[4] The day degree is the cumulated difference between the daily water temperature and a 10.55 ºC threshold (Mann, 1979; Mills, 2016). Day degree is an indicator setting the days when the oyster spawning could start.

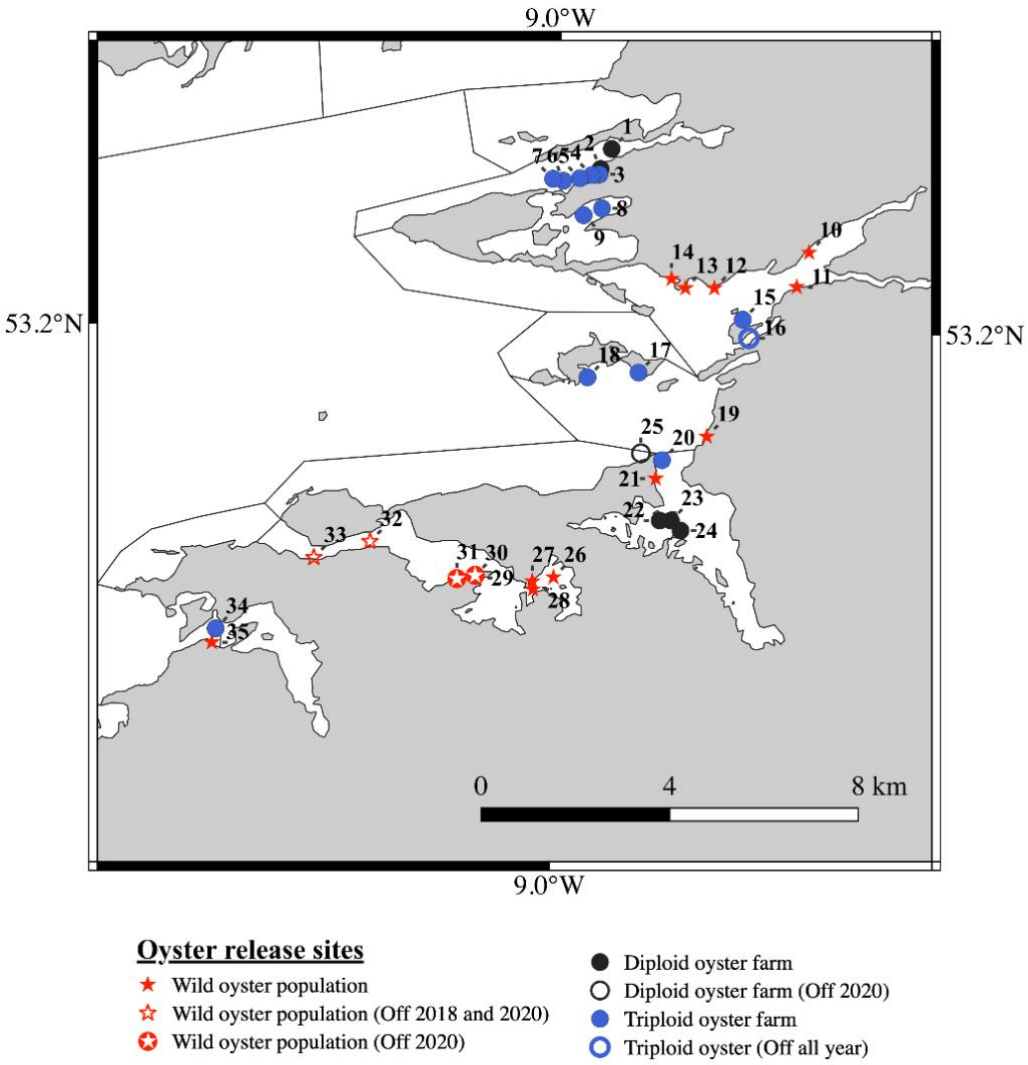

Figure 1.2 – Distribution of *Magallana gigas* release sites in Galway Bay divided by type of site. Oyster farm locations are shown by dots, wild previous locations by stars. Variation in the symbols shows where releases from some sites did not occur in certain years due to variation in environmental conditions.

The behaviour of *M. gigas* larvae during their pelagic phase is not well known. Three larval behaviours were simulated, (passive particle movement, diel migration, and tidal migration) and details are presented in the technical report. However, as a number of studies have observed tidally induced vertical migration in bivalve species (e.g. *Crassostrea virginica*) (Knights et al. 2006), the results from this behaviour have been used in this report. This behaviour occurs during the veliger stage, with larvae swimming nearer the surface during flood tides and lower in the water column during ebb tides which favours larval retention in near shore environments (Arakawa 1990).

Settlement of *M. gigas* larvae was considered to occur between 21 and 31 days if larvae were in a site with suitable habitat.

Hemigrapsus sanguineus

In contrast to the Pacific Oyster, there have been no documented occurrences of *H. sanguineus* in Irish waters. A release site was chosen based on the most likely pathway for *H. sanguineus* introduction, movement in ballast water or on hull fouling. The model thus simulated release from a point within Galway harbour, as this is the area with the highest concentration of vessel activity within the inner Galway Bay area.

Spawning of *H. sanguineus* occurs during the nocturnal high tide (Park et al. 2005), reportedly from late April to early October according to the winter temperature and Galway Bay latitudes (Epifanio et al., 2013), with one female crab capable of spawning several batches of eggs per year (Epifanio et al. 1998; McDermott 1998).

The pelagic larval duration of *H. sanguineus* larvae ranges from 16 to 55 days depending on the temperature (25 ºC to 15 ºC). Giménez et al (2020) estimated a logarithmic relationship between the water temperature and the pelagic larval duration. Using the summer average water temperature of 18ºC at the release location, the pelagic larval duration of the crab in Galway Bay was set to 35 days. During that period, *H. sanguineus* larvae go through five zoeal stages and one megalopa stage, which lasted 5 days and 10 days for a zoeal stage and megalopa stage respectively

The early zoeal stages of *H. sanguineus* exhibit geotaxis and barokinensis behaviours in favour of offshore water dispersal, allowing seaward export of early larval stages by maintaining a surface position (Park et al. 2004). The late zoeal stages are cued by the light and are assumed to have a diel vertical migration (Cohen et al. 2015). Then, the megalopae return to intertidal areas and crawl near the bottom to settle (Park et al. 2004). Settlement was allowed if the larvae reached the intertidal zone and suitable habitats between 25 and 35 days. Settlement is cued by the exudates of the adults and the biofilm of the substrate (Epifanio et al., 2013). This specific information is not given in the Galway Bay and therefore was not accounted for.

1.5.6 Using spread models to guide management actions

The outputs from the models of IAS spread were analysed using QGIS software and R to understand the connectivity between sites, and to investigate relationships between the spread data and existing spatial information on marine communities, the locations of artificial structures, and release sites. Additional spatial data was obtained from the Marine Atlas, NPWS, and county records. The details of the GIS files used, and any analysis is given in the accompanying Technical Report.

2 Case study: Pacific oyster, *Magallana gigas* (Thunberg, 1973)

Figure 2.1 – A photograph of a feral Pacific oyster *Magallana gigas* in the Solway Firth. Photograph by Christine Beveridge, SAMS.

2.1 Species profile

2.1.1 General life history

The Pacific oyster, *Magallana gigas* (Thunberg, 1793) (previously *Crassostrea gigas*) is an oyster species which can reach 20-30 cm and reach an age of up to thirty years (Nehring 2011). The shells are whitish with purple streaks, irregularly shaped with large radial folds in mature adults and unequally sized left and right valves (Nehring 2011). Pacific oyster shells can vary in appearance depending on the substrate. The Pacific oyster can be distinguished from the native European oyster, *Ostrea edulis*, by a wavier shell margin and a more elongated shape; *O. edulis* tends to be more circular with more numerous but weaker ridges (Bishop 2020). The Pacific oyster grows attached to the substrate either individually, in clumps or in large reefs. The native range of *M. gigas* is the northwest Pacific and around the waters of Japan where it is found from the lower intertidal to approximately 40m depth (Herbert et al. 2016).

M. gigas has been widely introduced around the world as an aquaculture species, with most introductions in Europe an attempt to replace collapsed native European oyster fisheries. The initial introduction of Pacific oysters for aquaculture in Europe took place in the late 1960s and 1970s (Drinkwaard 1998). Pacific oysters may be cultured on trestle tables or longlines, or directly on the substrate with cultivation practices varying depending on location.

2.1.2 Habitat and biology

In the Solway Firth, *M. gigas* is found on macroalage-free exposed bedrock and large boulders, and dense macroalgal coverage is thought to be unsuitable for settlement and establishment (Kochmann et al. 2013; Cook et al. 2014a). Kochmann et al. (2013) assessed oyster preference with different habitat types, finding that *M. gigas* presence in Ireland was positively associated with hard reef habitat (bedrock, boulders, cobbles, pebbles or biogenic reef). Oysters were not found attached to macroalgae.

2.1.3 Physiological tolerances (range and preferences)

Pacific oysters have a broad environmental tolerance which has contributed to their global distribution. As an intertidal species, adult *M gigas* have a broad thermal tolerance ranging from -1.8 to 35 C, and tolerate salinities from 10-50. Prolonged exposure to extreme cold conditions may affect oyster survival, with high mortality in Scandinavian wild populations seen during a very cold winter in 2009-2010 (Laugen et al. 2015). Temperature thresholds for gametogenesis, spawning and larval development may play an important role in the population expansion of this species (Herbert2016, Dutertre et al. 2010). Prolonged periods of high sea temperatures have been observed to correspond to higher recruitment of *M gigas* elsewhere in Europe (Wehrmann et al. 2000; Guy and Roberts 2010).

Settlement typically occurs in the lower intertidal or subtidal down to 15 m, although *M. gigas* has been observed at depths up to 40 m (Nehring 2011; Robins et al. 2017). Establishment in subtidal areas seems to lag behind intertidal establishment, although some of the delay could be attributed to sampling bias. In the Wadden sea, the Pacific oyster population expanded from intertidal populations into the subtidal (Diederich et al. 2005). In Sweden, density increases since initial establishment in 2008 have been seen extending into deeper sub-tidal areas in recent years (Laugen et al. 2015).

2.1.4 Timing of reproduction and recruitment

Pacific oysters are highly fecund, with an individual oyster capable of producing between 50 and 200 million eggs per spawning event (Quayle 1988, McAfee & Connell 2021). Pacific oysters are oviparous, with spawning occurring when temperatures are highest, this usually occurs in July and August in the Northern Hemisphere (Troost 2010). The pelagic larval stage usually lasts three to four weeks depending on temperature, transitioning through trochophore, veliger and pediveliger stages before settlement occurs. The planktonic larval duration in shorter in warmer temperatures. Spawning and larval survival are temperature dependent, with temperatures of over 17 °C thought to be required for spawning and larval development although survival is possible at lower temperatures (Child and Laing 1998; Ruesink et al. 2005; Castaños et al. 2009).

Pacific oysters settle on hard substrates, with rocks, pebbles, and bivalve shells providing opportunities for settlement in muddy or sandy areas (Kochmann et al. 2013). Pacific oysters become attached to the substrate on which they settle by excreting a cement which fixes the lower valve to

the substrate (Arakawa 1990; Troost 2010). Settlement may occur on other oyster shells, with the potential for large oyster reefs to develop as clumps of oysters join.

2.1.5 Route of introduction and invaded range distribution

The widespread culture of *M. gigas* in the aquaculture industry has led to the establishment of wild populations in North America and Europe, as well as S. America, Africa, and Australia and New Zealand (Ruesink et al. 2005; Dolmer et al. 2014).

The history of *M. gigas* aquaculture introductions into Europe is relatively complex, with initial importation events into different countries from British Columbia followed by trade of spat between European countries (Lallias et al. 2015). *M. gigas* settled in the wild relatively quickly in some areas following their introduction for aquaculture. By the end of the 1960s, wild settled oysters were found along the Atlantic coast of France (Lallias et al. 2015). In the UK, warm summers in the early 1990s led to oyster spat settling in the wild in the south of England and in the Menai straits (Spencer et al. 1994).

Established populations of wild *M. gigas* are now found across many countries in Europe, with extensive reef-forming populations in the Wadden Sea in the Netherlands and Germany and on the coast of France. More recently *M. gigas* reefs have developed in England. The distribution of the invaded range of *M. gigas* is extensive, spreading into Scandinavia (Dolmer et al. 2014), with records of wild individuals reported from Scotland as far north as the Shetland Isles (Shelmerdine et al. 2017).

In areas with extensive tidal flats such as the Wadden Sea, *M. gigas* has been shown to form large reefs. For instance, in the Oosterschelde estuary, Netherlands, large feral oyster reefs formed in intertidal and subtidal areas with densities estimated to increase from 0.25 km in 1980 to 8.1 km in 2003 (Troost 2010). In regions with fewer large shallow intertidal areas, oysters are more commonly observed in sites with high water flow such as small shallow bays or along narrow sounds and beaches (Laugen et al. 2015). This is seen in Sweden where the tidal range is typically small. Wild *M. gigas* populations have become common along the norther part of the west coast of Sweden since its first detection in 2006, but are usually found at low to medium densities as opposed to in closely packed reefs (Hollander et al. 2015). In the North Adriatic, *M gigas* is found in the medio-littoral zone on natural rocky shore or artificial structures, where it can form dense colonies, and is uncommon below 1 m depth (Ezgeta-Balic et al. 2019; Stagličić et al. 2020).

M. gigas is expanding its range in Europe with warming sea temperatures. Thomas et al. (2016) demonstrated a large northward shift in the spawning temperature limits of *M gigas* over a 30 year period to 2003. *M. gigas* is found at the edge of its temperature range and as sea temperatures warm, is accompanying the shift in appropriate temperatures poleward (Faust et al. 2017, Rinde et al. 2017). In most regions there has been a delay between initial escape of the species and population expansion. This delay has been between 15-20 years depending on the area (Diederich et al. 2005; Cardoso et al. 2007). Years with warmer summer temperatures may support for greater recruitment and allow for population expansion.

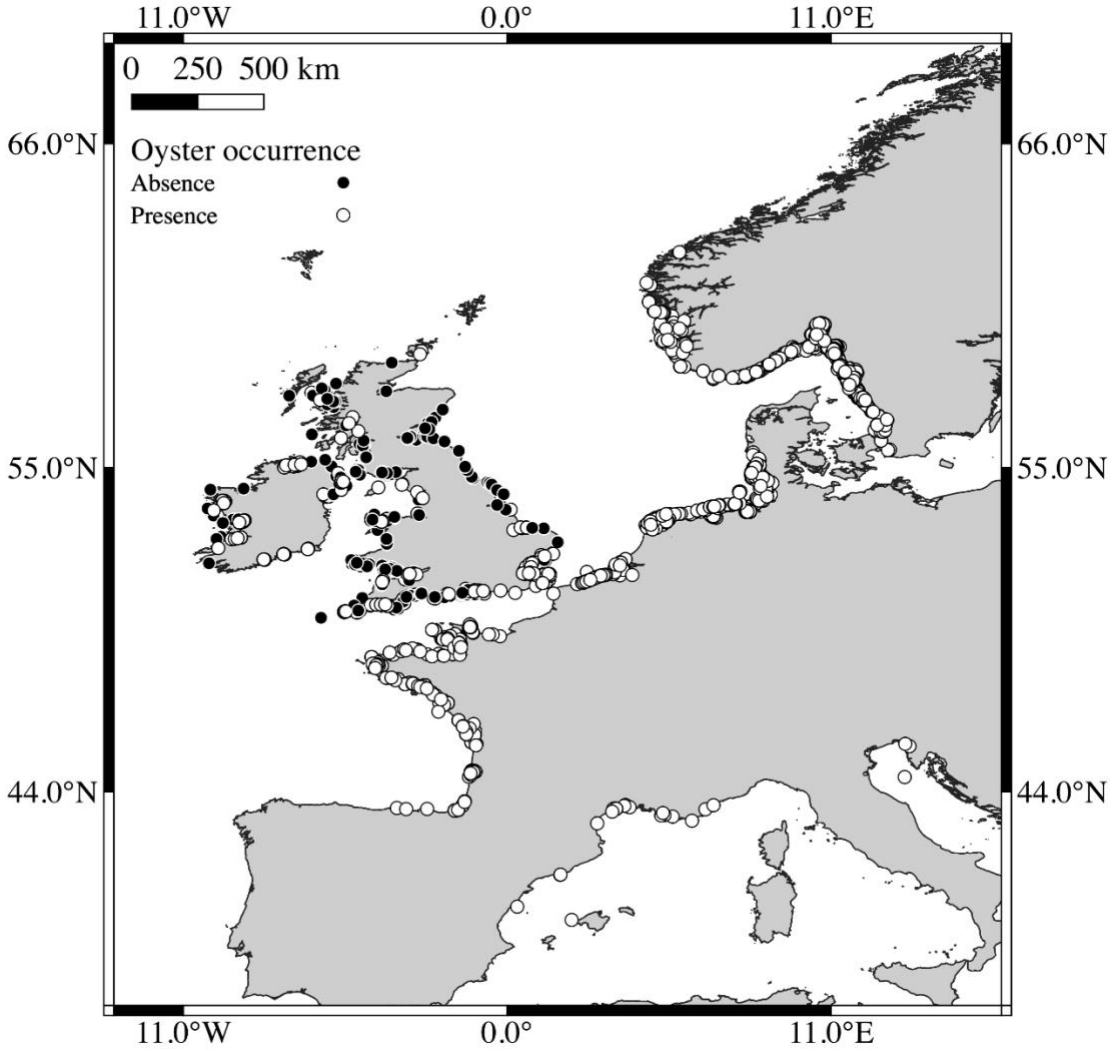

Figure 2.2 The distribution of *Magallana gigas* in European waters. Presence is indicated by white circles, absence by black circles, with records taken from GBIF, NBN Atlas, OBIS and Kochmann et al. (2013).

2.1.6 Methods of sampling

Surveys for *M. gigas* are commonly conducted using transects over areas of the low intertidal zone, conducted at low spring tides. Collection of individuals allows an assessment to be made of the age structure of the population based on shell morphology (Richardson et al. 1993; Guy and Roberts 2010). Methods previously used for sampling *M. gigas* in Ireland involve 45 minute timed searches for oysters on intertidal shores, using the SACFOR scale to estimate abundance (Kochmann et al. 2013; Zwerschke, Kochmann, et al. 2018).

Subtidal populations may be identified using side-scan sonar techniques (Nehls and Büttger 2007). Remote sensing imaging has been used to map the spatial distribution of wild oyster reefs, with some success shown in using visible-near infrared (VNIR) hyperspectral imaging to detect the horizontal extent of oysters in rocky areas (Bris et al. 2016).

2.2 Pacific oyster in Ireland

2.2.1 Magallana as an aquaculture species in Ireland

Pacific oysters were first introduced as an aquaculture species in 1965 (Browne et al. 2007) and are now widely cultivated across Ireland. Oyster farms operate mainly in the intertidal bays located in the west, southeast and northeast coasts of Ireland, and takes place on approximately 2000 ha of the intertidal. Almost all of the culture occurs on trestle tables in the lower intertidal, in areas exposed only on low spring tides. Oysters are cultured in mesh bags on the tables, grown from spat supplied by hatcheries and harvested when grown to a suitable size.

Farmed shellfish accounted for 65% of the overall aquaculture production volume in Ireland in 2018, with production predicted to increase in the future. The sector turnover was estimated at €176 million (SEMRU 2019). The contribution of Pacific oysters to the sector is significant. In 2019, the total volume of farmed pacific oysters in Ireland was 10,460 tonnes, 21% of the total aquaculture output volume, and was valued at €45.2 million (BIM 2020). Over 80% of this was from triploid stock, and 20% from diploid stock (BIM 2020).

There are 144 businesses farming Pacific oyster in Ireland, the majority of which are small businesses employing less than 6 people. Over 1350 people are employed by the Pacific oyster industry, and the sector provides the most employment in aquaculture (BIM 2020).

2.2.2 Wild population distribution in Ireland

Wild Pacific oyster populations in Ireland and typically found near areas where *M. gigas* is cultured, with the first records of wild establishment from the 1980s. The majority of wild *M. gigas* records are found on the west and northern Irish coasts, with records from the east coast found in the northeast in Strangford Lough and Carlingford Lough (Figure 2.3) (Zwerschke, Kochmann, et al. 2018). Densities in Ireland in most places remain low with limited spread away from introduction points (Zwerschke, Kochmann, et al. 2018). The most abundant intertidal populations (1 – 9 oysters/m^2) are found in Lough Swilly and the Shannon estuary, but these self-sustaining populations are at much lower densities than those found elsewhere in Europe (Zwerschke, Kochmann, et al. 2018).

Targeted surveys of wild *M. gigas* populations were last conducted in Ireland in 2012 -2013 by Zwerschke et al. (2018) who sampled 14 new sites and re-visited 18 sites in Ireland previously surveyed by Kochmann et al. (2013), and five sites in Northern Ireland previously surveyed by Guy and Roberts (2010). Kochmann et al. (2013) sampled 69 sites in 2009 and identified Pacific oyster presence at 18. These surveys were all in the lower intertidal area, with only one subtidal area surveyed in Lough Swilly. Low-density subtidal populations of *M. gigas* have also been documented in Lough Foyle (Tully and Clarke 2012; Zwerschke, Kochmann, et al. 2018).

Pacific oysters are most widespread and abundant in Lough Swilly, with a subtidal mussel bed recorded in 2009 with densities estimated at 12.5 individuals per m^2 (Kochmann et al. 2013). This is one of the most northerly established populations of *M. gigas* in Europe (Robins et al. 2017). Intertidal Pacific oyster densities were recorded during native oyster surveys as ranging from 0-8 m^2 in 2011, with the total number of oysters in the surveyed area in 2011 was estimated at 5.64 million oysters (Tully and Clarke 2012). The range of Pacific and native oysters overlapped in the survey area, but Pacific oysters were more dominant in the intertidal with native oysters more common in shallow sub-tidal area (Tully and Clarke 2012). The increase in Pacific oyster populations in Lough Swilly is thought

to have occurred since 2001, and is suggested to be impacting native oyster recovery as large areas of previous native oyster beds now have high densities of Pacific oyster (Tully and Clarke 2012).

In Northern Ireland, there is an established population of Pacific Oysters in Strangford Lough on the east coast (Guy and Roberts 2010). Reports of wild oysters in Strangford Lough have been documented since the 1990s, but an assessment of the age of oysters collected in 2008 indicated that successful recruitment has occurred annually since 2002 (Guy and Roberts 2010).

The low densities seen in self-sustaining wild *M. gigas* populations in Ireland may be due to environmental conditions infrequently supporting spawning and larval development. Modelling of ocean bottom temperatures showed that thresholds for recruitment of *M. gigas* are only exceeded infrequently around much of the Irish coast (less than 3 out of 10 years) under current climate conditions (King et al. 2020). In the parts of Europe where high-density *M. gigas* reefs have established, recruitment thresholds are exceeded regularly (King et al. 2020). This could change in the future as anthropogenic climate change leads to changes in ocean temperatures. Using a regional climate model projection to identify changes in bottom seawater temperatures which could allow recruitment of *M. gigas* under future climate conditions, King et al. (2020) showed that by 2040 recruitment thresholds for *M. gigas* could be exceeded in Irish waters in the majority of years (more than 7 out of 10 years).

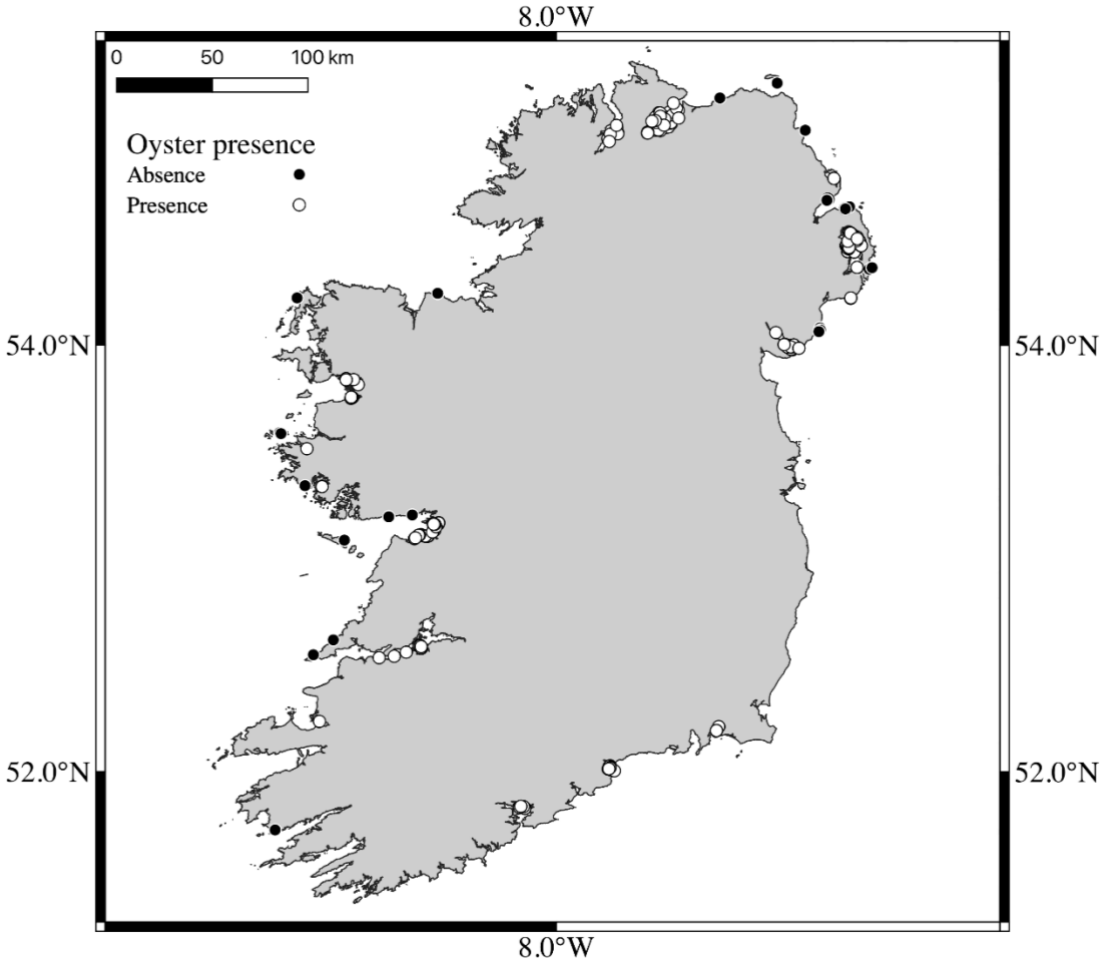

Figure 2.3 - The distribution of *Magallana gigas* in Ireland. Presence is indicated by white circles, absence by black circles, with records taken from GBIF, NBN Atlas, OBIS and Kochmann et al. (2013).

2.3 Review of the Impact of *Magallana gigas*

2.3.1 Ecological impacts:

There have been a number of reviews of the impacts of *M gigas* (Herbert 2012; Herbert et al. 2016; Martínez-García et al. 2021). The most recent review of the global introduction and impact of *M gigas* by Martinez-Garcia et al. (2021), found that studies on the impact of wild oysters have been conducted in 11 counties, primarily those with a higher GDP. These include the USA, Canada and Argentina, six European countries and Australia and New Zealand (Martínez-García et al. 2021). The impacts of *M. gigas* have also been reviewed extensively in Herbert et al. (2012,2016).

The impact of *M. gigas* on marine ecosystems has been demonstrated to vary depending on oyster density and the local habitat (Green and Crowe 2014; Zwerschke, Hollyman, et al. 2018). The major impacts and many of the studies of *M. gigas* impacts come from locations where extensive reefs have formed. In Ireland, reef formation on an extensive scale has yet to occur and thus the impacts of individual or small clumps of oysters on the environment are also not likely to be extensive. The system into which species are introduced will also affect what impact an IAS has, the majority of the research into *M. gigas* as an IAS has come from the Wadden Sea where large changes in habitat have been caused by *M. gigas* establishment. The large tidal flats present in the Wadden Sea are not found in Ireland but reef formation has also occurred in rocky shore habitats on the coasts of England and France. The population dynamics of IAS are not static, and changes in population size and density could lead to some of the impacts seen elsewhere occurring in Ireland.

Intertidal rock

Rocky shore habitats occupy much of the Irish coastline. Pacific oysters are capable of colonising large areas of the intertidal zone and can be found from Mean High Water to mean Low Water. In France, Brittany, low to moderate energy rocky shores support reef formations of *M. gigas* (Lejart and Hily 2011; Herbert 2012). In Kent, areas of protected chalk reefs have also been colonised by *M. gigas* reefs (McKnight and Chudleigh 2015). The highest densities of oysters appear to be in the mid- and mid-low intertidal zone on rocky shores, with oysters capable of colonising a variety of hard substrata (Castaños et al. 2009).

Studies from France suggest that presence of *M. gigas* on rocky shores may alter community structure by increasing habitat heterogeneity, with higher biomass and species richness on oyster reefs compared to adjacent rock habitat (Lejart and Hily 2011; Herbert 2012). Barnacle abundance has been shown to increase with oyster cover in rocky environments in Ireland (Zwerschke, Hollyman, et al. 2018), as did the abundance of some limpet species on oyster reefs in France (Lejart and Hily 2011). However, in the Irish study rocky shore habitats were the only environment in which limpet and *Fucus vesiculous* abundance decreased with increasing *M. gigas* density (Zwerschke, Hollyman, et al. 2018), demonstrating the complexities of determining impact in an invaded range even when examples can be studies in other regions. Green and Crowe (2013) also found complex relationships between oyster density and impacts on rocky boulder habitats, with highest diversity of other biota occurring at the lowest density of oysters. They also demonstrated inhibition of the honeycomb worm *Sabellaria alveolata*, a species which generates reefs protected under the EU Habitats Directive (see below).

Soft-sediment habitats

It is in intertidal soft-sediment habitats such as intertidal mud and sand that the most extensive changes in habitat have been observed as a result of the introduction of *M. gigas*. The transformation of large areas of the Wadden Sea to oyster reefs has changed these soft-sediment habitats to hard substrate, which leads to large changes in the communities associated with these habitats (Reise 1998; Nehls and Büttger 2007; Markert et al. 2013). Oyster reefs provide habitat heterogeneity and hard substrate for the attachment of a large number of epibiotic species. The abundance of both species within the reef and in the sediment under reefs has been shown to be higher than in adjacent mudflat areas (Lejart and Hily 2011, Green and Crowe 2014). In the Wadden Sea *M. gigas* reefs have typically formed in the low intertidal, leaving a refuge zone for soft-sediment species that might have been negatively affected by the formation of hard habitats (Herbert et al. 2012).

As *M. gigas* larvae require hard substrate on which to settle, reef development has occurred through settlement on shells and former mussel beds, before further settlement on other oysters allowed for expansion of the reef. The impact of *M. gigas* on mussel beds is discussed below.

Biogenic habitats

Habitat forming species are frequently a priority for conservation, and there are several species in Ireland which can form biogenic reefs. These habitats frequently support diverse marine communities but may also provide a hard substrate for colonisation by *M. gigas*.

Sabellaria reefs

Shallow intertidal reefs can be formed by the polychaete worm *Sabellaria alveolata* in sandy areas of the lower intertidal (Dubois et al. 2006). These reefs are protected under Annex I of the Habitats Directive as they are relatively rare, but the species can also be found encrusting boulders in intertidal rocky shore habitats in Ireland. Data on the distribution of this species in Ireland is poor, but a recent review of records shows that there is a discontinuous distribution of intertidal populations around the coastline (Firth et al. 2021)

M. gigas has been found attached to *S. alveolata* reefs, although this was rare in surveyed locations (Zwerschke, Kochmann, et al. 2018). In a study of an intertidal boulder field in Lough Swilly, Co Donegal, Green and Crowe (2013) found that coverage of *S. alveolata* was reduced on boulders with *M. gigas* attached, and that *S. alveolata* settlement was reduced on boulders with live or dead oysters attached. The exact mechanisms for reduced settlement were not clear, but direct competition for space was deemed to be unlikely as in the experiment *M. gigas* was attached to the topside of boulder, while *S. alveolata* is generally found on the underside (Green and Crowe 2013). The authors suggest that alteration of flow regimes or changes in community structure brought about by the presence of *M. gigas* could indirectly affect *S. alveolata* settlement or post-settlement success.

In France, colonisation of *S. alveolata* reefs by *M. gigas* has been observed in Mont Saint-Michel, where the largest European examples of these reefs are found (Cognie et al. 2006; Dubois et al. 2006). Dubois et al. (2006) found that *S. alveolata* reefs colonised by oysters had significantly higher species richness and diversity than those without, and that oyster presence also altered community composition. Oyster presence on *S. alveolata* reefs could lead to competition for food and space, but the exact influence of oyster presence on *S. alveolata* populations are not yet clear (Dubois et al. 2006). Areas of the reef subject to disturbance by humans appear to be more prone to oyster colonisation, and oyster presence appears to have led to further damage by those collecting wild oyster from the reef (Herbert 2012).

Subtidal biogenic reefs may be formed by *Sabellaria spinulosa* polychaete worms, which create reefs by joining together sand grains. In north-east Kent, *M. gigas* is displacing an area of *S. spinulosa* reef (McKnight 2012, Herbert 2012).

Blue mussels *(Mytilus edulis)*

In much of the introduced range of *M. gigas* in Northern Europe, intrusion of *M. gigas* into mussel beds is one of the most evident changes *M. gigas* has had on native biodiversity. In the shallow Wadden Sea, large mussel beds are found across large areas of the intertidal and shallow subtidal zone. The shells provide a hard substrate for the settlement of *M. gigas* larvae, which has led to many mussel beds turning into mixed mussel/oyster reefs, with oysters the dominant species in some reefs. In Ireland, mussels are primarily found on tide-swept rocky shores without the extensive shallow reefs seen in the Wadden Sea.

Body condition of mussels in mixed mussel/oyster beds was shown to decrease with increasing oyster dominance (Waser et al. 2016). Experimental evidence from field experiments shows that competition in mixed mussel-oyster groups led to lower body condition of both species, but did not affect growth rate or mortality (Joyce et al. 2021). Competitive interactions may be mediated by differences in water flow between sites, and mussel condition may be affected by external factors unrelated to oyster presence. Differences in tolerance to air exposure also led to mussels having higher dominance in the upper intertidal zone (Waser et al. 2016).

Research has suggested that the additional habitat structure created by *M. gigas* may offer a refuge from predators for *M. edulis* growing within the reef, and reduce overgrowth by epibionts such as barnacles (Buschbaum et al. 2016). However, Green and Crowe (2014) found that impacts varied with density, such that diversity in mussel beds was enhanced by the addition of oysters at low densities, but reduced as oyster densities increased.

European oyster *(Ostrea edulis)*

Pacific oysters have been shown to compete with native oyster populations in the USA and Australia. In Australia, *M. gigas* was shown to outcompete the native oyster *Saccostrea glomerata* in areas of the lower intertidal where the species overlap (Krassoi et al. 2008). At higher tidal elevations *S. glomerata* was not affected by *M. gigas* presence as mortality of *M. gigas* as a result of abiotic stress was significantly higher (Krassoi et al. 2008).

Impacts of *M. gigas* on the native European oyster, *Ostrea edulis*, may vary depending on the habitat and density of both species. Native populations of *O. edulis* in Europe suffered large declines as a result of overfishing and disease and are now present at much lower abundances. This is also true of *O. edulis* populations in Ireland, which have declined since an estimated peak in the mid-late 19th century when these beds were important fishery sites (Tully and Clarke 2012). Many of the remaining native oyster sites in Ireland are found in estuarine areas, many of which are designated as Special Areas of Conservation (SACs).

In much of the invaded range of *M. gigas, O. edulis* and *M. gigas* are thought to occupy different depth zones, with *M. gigas* occupying more of the intertidal and *O. edulis* found at lower depths (Reise 1998). This has been observed in areas of the Mediterranean, where Pacific oyster are currently only found in intertidal areas down to 1 m depth and *O. edulis* is restricted to the subtidal (Stagličić et al. 2020). However, in Ireland *O. edulis* occupies both intertidal and subtidal distributions, and there is evidence for an overlap in the depth distribution of both oyster species (Tully and Clarke 2012; Zwerschke et al. 2016). In Northern Ireland in Strangford Lough, both native and pacific oysters occupy similar zones in the lower intertidal (Zwerschke et al. 2016), likely as a result of the high water retention in the

northern part of the Lough (Zwerschke et al. 2016). In Lough Swilly, Pacific oysters have been observed in areas previously occupied by *O. edulis* although some segregation based on depth was seen (Tully and Clarke 2012).

Direct competition between *O. edulis* and *M. gigas* has been observed in manipulative field experiments conducted in Strangford Lough with *M. gigas* negatively impacting *O. edulis* in subtidal habitats on horizontally oriented substrata, but positively impacting native oysters on vertical substrata (Zwerschke, van Rein, et al. 2018). In intertidal habitats with higher abiotic stress species interactions were weaker, and growth rate and biomass of *O. edulis* was unaffected by *M. gigas* presence (Zwerschke, van Rein, et al. 2018). Zwerschke et al. (2018) concluded that the presence of *M. gigas* is unlikely to negatively affect *O. edulis* recovery apart from in specific habitats (subtidal vertical substrates).

Native and Pacific oysters were shown to support different epibiotic communities in Strangford Lough, with lower species richness of epibiotic communities supported by *M gigas* (Guy et al. 2018). However, in an experimental investigation of the benthic assemblages associated with *O. edulis* and *M. gigas*, Zwerschke et al. (2016) found that species diversity and benthic assemblage structure were similar in hard substrate communities. The authors suggest that native and Pacific oysters may play functionally similar species facilitation roles due to similarities in the physical structures created (Zwerschke et al. 2016).

Seagrass beds

There are three species of seagrass in Ireland, *Zostera marina, Zostera noltii,* and *Ruppia maritima* which is only present in a few locations. Shallow subtidal meadows are formed by *Z. marina* which can also be found in intertidal pools, whereas *Z. noltii* is found on intertidal mudflats forming short lawns (Dubsky and Campos 2019). These habitats provide habitat for a variety of marine organisms, but also provide food for important wading birds such as Brent Geese (Dubsky and Campos 2019). Loss of seagrass beds is driven by pressures including eutrophication, disturbance and presence of other invasive species. Seagrass beds provide a variety of ecosystem services, including carbon sequestration and coastal protection by stabilising sediments and reducing wave action (Kelly and Volpe 2007).

In British Columbia, Canada, seagrass beds (*Zostera marina*) were observed to be absent directly below intertidal oyster beds, although the two species were found in similar habitats at the regional scale (Kelly and Volpe 2007). Field experiments showed that seagrass in low-intertidal zones below high-intertidal oyster beds had reduced growth rates compared with control sites (Kelly and Volpe 2007). The exact mechanisms for this inhibition are not clear, although it has been suggested that deposition of large volumes of organic matter through the filtering activity of *M. gigas* may create conditions that result in high levels of sulphide in the sediment. High sulphide levels have been shown to reduce growth and photosynthetic activity of seagrass (Kelly and Volpe 2007).

The impact of Pacific oysters on seagrass beds has also been investigated in the context of aquaculture impacts. *Zostera marina* beds in the USA were shown to be negatively impact by oyster cultivation as a result of both biological interactions between oysters and eelgrass, as well as the disturbance from aquaculture operations (Tallis et al. 2009). Seagrass beds may thus also be impacted by control efforts or by Pacific oyster harvesting and could cause long-term disturbance of eelgrass *(Zostera marina)* beds (Wisehart et al. 2007; Dubsky and Campos 2019). This is covered in more detail in the management section below (Section 2.4.1).

Invasive species facilitation

There are many cases where the transport of Pacific oysters for aquaculture has resulted in the unintentional introduction of other invasive species. High profile examples include the introduction of wireweed, *Sargassum muticum*, to Europe. In Ireland, two exotic copepod species (*Mytilicola orientalis* and *Mytilicola* intestinalis) were introduced accompanying Pacific oyster stock movements (Holmes and Minchin 1995). Other species which have been introduced include the invasive wireweed, *Sargassum muticum,* and the seagrass *Zostera japonica* (Holmes and Minchin 1995; Miossec 2009; Martínez-García et al. 2021).

The introduction of other IAS with Pacific oyster aquaculture imports may have negative environmental and socio-economic effects but can be considered more a consequence of aquaculture as a vector than an impact of the wild spread of *M gigas*. However, the interaction of *M gigas* with other IAS may affect the impact of one or both species.

The hard substrate provided by Pacific oyster reefs may facilitate the establishment of other invasive alien species. For example, the hard substrate and more heterogenous habitat of the oyster reefs may provide habitat for invasive barnacles *Austrominius modestus* (Green and Crowe 2014) or a spatial refuge against predators for small species such as the Brush-clawed shore crab, *Hemigrapsus takanoi* (van den Brink et al. 2012; van den Brink and Hutting 2017). As mentioned above, in the Wadden Sea high densities of *Sargassum muticum* on Pacific oyster reefs were shown to reduce settlement of native sessile organisms and macro-algae on oyster shells (Lang and Buschbaum 2010).

Fish communities

There are fewer studies into the impact of *M. gigas* establishment on fish communities than on more directly associated species (Herbert et al. 2012). In a comparison of oyster reef, seagrass and unstructured mudflat habitats in the northeast Pacific, fish and decapod communities were shown to be influenced more by location within an estuary than by habitat (Hosack et al. 2006). Oyster reefs could be utilised by fish as foraging and nursery areas, but there has been little investigation as to whether there have been any changes in fish habitat use as a result of *M. gigas* reef formation.

Wading birds

The impact of Pacific oyster establishment on wading birds has been assessed in the Netherlands (Markert et al. 2013; Waser et al. 2016), the UK (Herbert et al. 2018) and Argentina (Escapa et al. 2004).

In Europe, intertidal mudflats and sandflats provide important feeding areas for many overwintering migratory birds on the East Atlantic flyway, and the transformation of these habitats into hard substrate by the formation of oyster reefs could impact the abundance and feeding success of wading birds (Herbert et al. 2018). However, higher macro-invertebrate density and biomass within oyster reefs may provide additional prey species for some foraging bird species. Higher densities and biomass of large invertebrate prey species such as annelids and shore crabs were observed on reefs in the UK (Herbert et al. 2018), and in Argentina greater numbers of epifaunal species on reefs were found to correlate with higher abundance of American oystercatcher (*Haematopus palliates*) and lesser yellowlegs (*Tringa flavipes)* (Escapa et al. 2004).

The only birds which have been observed eating Pacific oysters are the Eurasian oystercatcher (*Haematopus ostralegus)* and Herring Gull (*Larus argentus*) (Markert et al. 2013). In south-east

England, higher numbers of these species were observed on oyster reefs compared with uncolonized mudflats, although the higher feeding success of oystercatchers was due to the greater amount of benthic prey available on the oyster reef (Herbert et al. 2018). Eurasian curlew (*Numenius arquata)* also had high feeding success and biomass intake rates on oyster reef than on uncolonized mudflats;(Herbert et al. 2018) however, black-tailed and bar-tailed godwits were seen in lower abundances on oyster reef (Herbert et al. 2018).

The adaptation of Eurasian oystercatchers and curlew to foraging on oyster reefs confirms previous observations in the Wadden Sea where bird densities at two Pacific oyster reefs with different densities were compared with adjacent sandflats (Markert et al. 2013). However, when looking at intertidal mixed mussel/oyster beds in the Dutch Wadden Sea, increasing oyster dominance of mixed reefs led to a decrease in abundance of oystercatchers (Waser et al. 2016). Three other species also had lower abundances in beds with higher densities of oysters (Common Gull (*Larus canus*), red knot (*Calidris canutus*) and dunlin (*Calidris alpina*)), but most other bird species which foraged on the bivalve beds had no association between abundance and increasing oyster density (Waser et al. 2016). Sanderling (*Calidris alba*) and Common Ringed Plover (*Charadrius hiaticula*) showed an aversion for all bivalve beds, preferring to forage on other intertidal habitat.

The consequences of controlling Pacific oysters may cause greater disturbance to wading birds than taking no management action. Waser et al. (2016) speculated that as birds avoid oyster fishing areas, the disturbance caused by removing oysters from mixed beds would likely do more harm than good. The consequences of Pacific oyster reef removal on wintering shorebirds were assessed by Herbert et al. (2018).

2.3.2 Impacts on ecosystem services

As a habitat forming species, *M. gigas* supports many ecosystem processes which also provide ecosystem services. Individual oysters may contribute to some ecosystem services, but some will also derive from the structural formation of oyster reefs (Lemasson et al. 2017). The complex hard structures formed by oyster reefs create a heterogenous habitat and support a wide variety of epibiotic species and can provide shelter and food for more mobile species. The reef structure can also stabilise shorelines, improve water quality and sequester carbon (Fletcher et al. 2012). Oyster reefs generally provide ecosystem-services valued at up to US$99,000 per hectare per year (Grabowski et al. 2012, McAffe & Connell 2021).

Native oyster in Europe were once a dominant feature of much of the coastline, occupying significant areas in mainland Europe and in Ireland. Globally, 85% of native oyster reef habitat has been lost due to overfishing following the industrialisation of oyster dredging, disease, and other impacts (Beck et al. 2009). The greatest declines in native oysters were seen in a 50 year period between 1840-1890 (McAfee and Connell 2021).

M. gigas reefs can provide many of the same ecosystem services and may replace some of the ecosystem services originally provided by native oyster or mussel reefs by providing similar functional services (Zwerschke et al. 2020). However, there are differences in the habitat distribution of native and Pacific oyster reefs with the former being primarily subtidal, while Pacific oyster forms large intertidal reefs with some subtidal colonisation (McAfee and Connell 2021). The extent to which *M. gigas* is valued as a replacement to *O. edulis* will vary depending on the context, and the likelihood of native oyster regeneration. In the UK and Ireland, there has been a recent resurgence in native oyster restoration projects but many of these are still in their infancy. Much of this is driven by the potential for native oyster reefs to provide ecosystem services.

Table 2.1 – Summary of the ecosystem services potentially affected by establishment of *M. gigas*. Ecosystem services listed from CICES, with CICES code given in parentheses. Impacts mostly derive from places where *M. gigas* have formed extensive reefs, but may also be true of less dense established populations. Impacts are colour coded where the effect increases provision of a service (green) or reduces or negatively impacts a service (red).

<table>
<tr><th colspan="3">Supporting services</th></tr>
<tr><td colspan="3">Biodiversity
- Reef formation can provide habitat for a variety of epibenthic species. Increases habitat heterogeneity and species richness in muddy sediment areas of Wadden Sea.
Sediment stabilisation</td></tr>
<tr><td colspan="3">Water filtration</td></tr>
<tr><td colspan="3">Biodiversity
- Potential negative impact on habitat forming species (e.g. mussels, seagrass, Sabellaria reefs)
Competition with other filter feeding species (e.g. mussels)</td></tr>
<tr><th>Provisioning services</th><th>Regulating services</th><th>Cultural services</th></tr>
<tr><td>Animals reared by in-situ aquaculture for nutritional purposes (1.1.4.1)
- Economically important aquaculture species in Ireland</td><td>Control of erosion rates (2.2.1.1)
- Sediment stabilisation can reduce erosion</td><td>Characteristics of living systems that that enable activities promoting health, recuperation, or enjoyment through active or immersive interactions (3.1.1.1)
- Negatively impact recreational water users by causing cuts and injuries to beach users.
- Reduce beach access
- Injure dogs walked on beach</td></tr>
<tr><td>Wild animals (terrestrial and aquatic) used for nutritional purposes (1.1.4.3)
- Hand collection & fishing of feral oysters</td><td>Filtration/sequestration/storage/accumulation by micro-organisms, algae, plants, and animals (2.2.1.2)
- Carbon sequestration</td><td>Characteristics or features of living systems that have an existence value (3.2.2.1)
Potential negative impact on vulnerable species valued for their rarity (e.g. maerl, seagrass, Sabellaria reefs)</td></tr>
<tr><td>Fibres and other materials from wild animals for direct use or processing (excluding genetic materials) (1.1.4.4)
- Shells used for cultch material (potential for restoration of native oysters)</td><td>Pest control (including invasive species) (2.2.3.1)
- Facilitate introduction & establishment of IAS
Provide habitat for settlement of other IAS</td><td></td></tr>
</table>

- Shells used for ornamental purposes - Shells used for construction material	
Animal material collected for the purposes of maintaining or establishing a population (1.1.4.5) - Feral oysters harvested to supplement aquaculture stock	
Animals reared by in-situ aquaculture for nutritional purposes (1.1.4.1) - Potential negative impact on cultured mussels by competing for food (only proposed in certain systems with low carrying capacity)	

Provisioning services

The global culture of *M. gigas* as an aquaculture species demonstrates its value as a food source but will not be covered in extensive detail in this report. Wild Pacific oysters may also be harvested or fished as a food source, although this is likely to only occur where population density is high enough to make fishing economically viable and not so high as to distort the shells and substantially diminish their market value. Where commercial fishing is not viable, hand-picking of oysters is some areas also takes place (Mortensen 2017).

Pacific oyster shells can be used as substrate for the settlement of other shellfish species, including for the native *O. edulis*. This already occurs in some parts of Ireland (Herbert 2012), although a study investigating differences in cultch type for settlement of *O. edulis* spat found that Pacific oyster shells had the lowest settlement compared to mussel shells or mixed clam shells (Tully, O'Halloran, et al. 2018). This could represent a way of using Pacific oysters fished or collected via control activities, as long as only the shells are returned to the environment.

Substantial feral populations of *M. gigas* can also have significant negative effects on provisioning services as they can compete for phytoplankton with commercially cultured oysters or other filter feeders such as mussels and reduce the overall carrying capacity of semi-enclosed systems (Herbert et al. 2016, and see below).

Regulating and Maintenance services

Oysters influence a range of ecosystem processes that underpin regulating and maintenance services, including water column filtration, sediment stabilisation, nutrient cycling and benthic pelagic coupling (Herbert et al. 2016).

In field experiments in Lough Swilly, *M. gigas* has been shown to affect nutrient cycling both directly and via impacts on microbial communities (Green et al. 2012). Effects depend on the cover of oysters

and the habitat affected, for example with sediment–water fluxes of NH_4+ and Si(OH)4 and benthic turnover rates increasing with increasing cover of oysters in mudflats but decreasing in mussel-beds with the greatest cover of oysters (Green and Crowe 2013). High densities of oysters increased total microbial activity, chlorophyll content and CO_2 and CH_4 emission from sediments (Green et al. 2012), but even at only 10% cover, *M. gigas* increased the concentration of total oxidised nitrogen and pore-water NH_4^+ and altered microbial assemblages involved in nutrient cycling. Zwerschke et al. (2020) found similar nutrient cycling rates for *M. gigas* and O. edulis. Functional similarities between the two species suggest that further establishment of *M. gigas* in areas of Ireland previously occupied by *O. edulis* would not affect nutrient cycling rates in the ecosystem (Zwerschke et al. 2020).

The structures formed by Pacific oyster reefs can modify tidal flow and wave action (Walles et al. 2011). Reef formation by *M. gigas* may protect the seafloor under the reef from erosion, with the layer of shells providing protection from waves and currents and stabilising the substrate they're embedded in (Wijsman et al. 2008). The filtration and deposition of fine sediment on the seafloor may also contribute to reducing erosion, as sediment under oyster reefs was shown to have a higher mud-content (Wijsman et al. 2008). The sediment stabilisation capabilities of oyster reefs may extend beyond the immediate area of the reef, helping to prevent erosion of soft-substrate habitat (Walles et al. 2015).

In the tidal flats in the Oosterschelde estuary, construction of artificial dams and storm surge barriers led to changes in tidal dynamics causing loss of tidal sand and mud-flats. Following the wild establishment of oyster reefs in the area, artificial reefs using *M. gigas* shells to promote further settlement were constructed in areas most vulnerable to erosion (Walles et al. 2011)

As covered in the section above on environmental impacts, *M. gigas* may also facilitate the introduction and establishment of other IAS. The establishment of *M. gigas* could thus negatively contribute to the biological resistance of a system, and reduce the ability of an ecosystem to prevent establishment of other marine NNS.

Cultural services

The impacts of IAS on cultural services provided by marine systems are best understood in the context of impacts on recreation and tourism (Pejchar and Mooney 2009). Understanding how colonisation by IAS may impact how people perceive the environment is harder, with the cultural value of marine systems still under assessment in many areas.

In the Netherlands, areas with large Pacific oyster reefs are marked with warnings for beach goers to take care of the shells, as they may cause deep cuts (Wijsman 2021, *pers. comm.*). In Lake Grevelingen, a saltwater lake popular with recreational water users, many holidaymakers are injured every year by the sharp shells. The Pacific oyster population in the lake spread away from culture sites, colonising breakwaters, recreational beaches and surfing areas where swimmers and surfers may be injured by encounters with the oysters in the shallows (Wijsman et al. 2010).

In some areas, large oyster reefs have transformed soft-sediment beaches into inaccessible hard substrate. This can affect how people use coastal areas, as reefs may be less easy to walk on and can cause injury to people and pets, with anecdotal evidence of injuries to dogs caused by sharp shells (McKnight 2021, *pers. comm.*).

2.3.3 Direct impacts to industry

Aquaculture industry

The presence of extensive wild populations of Pacific oysters may benefit oyster farmers by providing a source of seed for growing on, as well as supporting fishing of mature adults. Shells collected from fishing of wild oysters can also be used as substrate for culture of farmed stock. Fishing of wild *M. gigas* populations has been documented in the Netherlands, the UK and sporadically in Lough Swilly in Ireland (Wijsman et al. 2008; Herbert 2012; Tully and Clarke 2012).

In Australia, culture operations of native Australian oysters, the Sydney rock oyster Saccostrea glomerata, are impacted by overgrowing of Pacific oysters which reduces yield and increases cleaning costs (Martínez-García et al. 2021). Settlement of oysters on cultured stock species could reduce growth and compete for food.

The high filtration capacity of Pacific oysters may reduce the quantity of planktonic food available for other filter feeding species. In open water systems this is unlikely to be a problem; however, in systems with limited water exchange such as the Oosterschelde in the Netherlands this could lead to problems. The quantity of oysters present in the Oosterschelde was thought by mussel farmers to be competing for food with mussels, reducing yield from mussel farming operations (Wijsman et al. 2008). Based on the biomass of Pacific oysters in the Oosterschelde, Wijsman et al. (2008) showed that Pacific oysters could impact the carrying capacity of the Oosterschelde for shellfish culture. The high volume of Pacific oysters was estimated to be responsible for 2/3 of the total filtration capacity by shellfish in the Oosterschelde.

Recreation and tourism

Pacific oysters have frequently been observed growing on artificial structures. In the Netherlands they have been found on sluices and dykes (Wijsman 2021, *pers. comm.*). Fouling of artificial structures can lead to increased cleaning costs and potentially injuries if access is impeded by oyster growth.

The injuries caused by encounters with sharp shells by swimmers and surfers may translate into costs for recreational area managers. Removal of *M. gigas* in Lake Grevelingen, the Netherlands, was prompted by the number of injuries and was paid for by the site management (Wijsman et al. 2010). Increased likelihood of injury also requires water users to be adequately informed of the risks through visible signage and represents costs in terms of medical care needed to treat injuries.

In the UK, concerns over injuries from a dense *M gigas* reef found near a slip-way led to removal trials (Herbert et al. 2018).

2.4 Managing *Magallana gigas*

Attempts to remove established *M gigas* populations have occurred in areas of the Netherlands, Norway, Sweden, UK, Australia, and South Africa (Martínez-García et al. 2021). A summary of management opportunities is given in Table 2.2

Table 2.2 – Active management opportunities for *Magallana gigas* in Ireland at each stage of the management hierarchy.

	Management term	Management objective	*M. gigas* active management methods	Active management examples
Prevention	Pre-border pathway management	Reduce the uptake of the species and its transport outside the area of interest	n/a	n/a
	Interception	Intercept the species on first entry into the area of interest	n/a	n/a
Captive management	Limits to keeping	Limit the keeping or cultivation of the species within the area of interest	Prevent use of diploid spat in new regions with no existing aquaculture operations	In operation in Ireland?
	Secure keeping	Ensure the security of the species held in captivity/cultivation within the area of interest	Aquaculture farm practices: - Use triploid stock - Culture oysters in trestles and remove old mature adults if not being sold	Aquaculture plan
Eradication	Eradication	Remove the entire population from the area of interest – with no immediate risk of re-introduction	Not possible if aquaculture operational in region	?
Long-term management / Control	Complete reproductive removal	Remove the entire population from the area of interest – but with remaining risk of re-invasion or further reproduction if not managed	Only possible if only a few feral oysters present.	No example of complete removal.
	Containment	Limit the spread of a reproducing population within the area of interest (e.g. apply control measures at range edge)	Long-term control options: - Mechanical removal - Hand removal - Fishing	
	Suppression	Reduce the distribution or abundance of a population within the area of interest (e.g. vulnerable habitat or with socio-economic damage)		NE Kent, UK – hand removal by volunteers in most vulnerable habitat UK – dredging where impacting people accessing beach Lake Grevelingen, NL – dredging where causing harm to recreational users

			Oosterschelde, NL – dredging trial driven by concern over impacts to mussel farming yields
Impact adaptation and ecosystem restoration	Actions taken to reduce associated impacts without or in conjunction with actions to reduce spread	Strengthening diversity of native predators/competitors Native oyster restoration Signage to reduce injuries	 Lake Grevelingen, Netherlands
No management	When a species is widespread, and no action is taken to reduce its spread	No action taken with recognition that the future state of wild population may make control impossible.	

2.4.1 Captive management – Aquaculture industry measures

Triploidy

Reproductive sterility of cultured stock has been suggested as a method to reduce the spread of wild oysters through spawning from farms. In Pacific oysters, induced triploidy (i.e. where cells contain three sets of chromosomes instead of two) can result in oysters being unable to produce viable gametes and thus being sterile (Herbert 2012). Triploidy can be induced in two ways, either through crossing a tertraploid male and diploid female to create a triploid offspring (mated triploids).

This prevents spread from cultured oysters, and has led to the majority of farms in Ireland (80%) culturing triploid stock (BIM 2020). However, the triploid condition may not always be stable, and triploidy does not always render oysters completely sterile.

Farm practices and licensing conditions

There are a number of measures which oyster farmers could take to improve biosecurity and prevent spread from farms. Herbert et al. (2012) describe a management plan proposed by an oyster farm in Wales which includes measures to reduce the likelihood of escapes and to check for spawning. More detail on the proposed measures can be found within the review, but in brief the measures included:

- "Use of triploid stock
- Calculation of degree days and checks for evidence of spawning
- Surveys for oyster spat if spawning is observed
- Regular removal of mature oysters that are too large for sale
- Removal of old (>5 years) oysters that have not reached market size
- Containment of oysters to trestles
- Removal of oysters and associated infrastructure should operations cease
- Limit the number of farms within a region or water body
- Limit the area of the seabed licensed for production
- Does off-bottom bags and trestles or suspended culture change risk of gametogenesis
- Limit the density of adult stock grown on-bottom
- Control the number of bags or trays
- Manage the size at harvest
- Consider harvest prior to spawning
- Monitor and prevent farm escapes or remove farm escapes

In working with aquaculture farmers, it will be essential to maintain trust. The BIM have already started working on ways in which aquaculture farmers can increase biosecurity on farms, and report sightings of target individuals. Any work on oysters will need to incorporate the perspectives on aquaculture farmers and should be mediated via established relationships with BIM.

2.4.2 Eradication, long-term management, and control

Mechanical destruction: dredging and grabs

Dredging has been trialled in the Oosterschelde, Netherlands in an area where high densities of Pacific oysters were thought to be competing with cultured mussels. In 2006, mussel farmers worked with

scientists to remove 50 ha of Pacific oysters (12,500 tons) from natural littoral and sub-littoral beds using mussel dredges (Wijsman et al. 2008). At the time, Pacific oysters were estimated to cover 1475 ha of the 35,100 ha Oosterschelde estuary, with roughly half the area covered in the littoral zone, and half in the sub-littoral. Complete removal of *M. gigas* from the region was suggested to be impossible; however, control of dense reefs through dredging was considered to have potential but was very labour intensive requiring 20 boat hours per hectare and costing around €300,000 (Wijsman et al. 2008). It was predicted that if fishing efforts were not maintained, the oyster reef in the intertidal zone would be restored within three to six years, whereas those in the sublittoral may take longer. Complete removal of all shells was recommended to slow recolonisation (Wijsman 2021, *pers. comm.*). The 50 ha cleared were not thought to have a large impact on the filtration capacity of the system (Wijsman et al. 2008), and concerns were raised over whether the disposal methods for oysters would result in their being properly contained. Since the trial, oyster densities have been reduced through impacts related to the introduction of invasive oyster drills and the oyster Herpesvirus, and there has been no additional dredging for management purposes (Wijsman 2021, *pers. comm.*). However, the wild oysters in the eastern Oosterschelde have also been fishing by oyster farmers (see Fishing and Harvesting section below).

In the Colne estuary in south-east England, an oyster-dredge was used to remove a dense *M gigas* reef that had developed near a slip-way due to concerns over injuries from the sharp shells (Herbert et al. 2018). Over a five-year period, the reef had attained densities of 200 oyster per m^2. Dredging was carried out as part of regular fishing activity in early spring 2013, and was supplemented by hand removal of remaining clumps of oysters at low spring tides (Herbert et al. 2018).

Options for managing Pacific oysters in areas where proliferation has led to injury by recreational water users have also been explored in the Netherlands. Lake Grevelingen is a saltwater lake used for recreational water sports and swimming, but injuries from oysters in shallow estuaries led to the management investigating potential removal options for these areas (Wijsman et al. 2010). Grabs and were recommended to fish oysters from shallow (<2 m) areas causing problems for water users, with a layer of sand used to cover shells left behind following fishing efforts (Wijsman et al. 2010).

Dredging of oysters may also impact other vulnerable species. In the Wadden Sea, the volume of dredging that would be required to reduce *M. gigas* populations is considered to cause too much habitat damage to be a viable management option (Herbert 2012). In Strangford Lough the decline of horse mussel (*Modiolus modiolus*) reefs had led to a ban being implemented on the use of mobile fishing gear, making this an unviable option for control of subtidal *M. gigas* populations (Guy and Roberts 2010).

Concerns have been raised by an Irish group working on seagrass protection and restoration about potential impacts of dredging with heavy equipment for Pacific Oysters in Lough Swilly (Dubsky and Campos 2019). In the US, the periodic disturbance caused by oyster farming is a concern as it has been shown to cause long-term disruption of eelgrass *(Zostera marina)* beds (Tallis et al. 2009). However, Wisehart et al. (2007) showed that recovery of eelgrass beds following dredging was possible as reduced eelgrass densities potentially provided greater opportunities for seed germination and seedling growth. When carrying out IAS control work it will be important to assess potential impact on protected or vulnerable species or habitats as well as their recovery potential.

Hand-held hammering

At the early stage of *M gigas* establishment, hand removal may represent an effective means of reducing population size. This relies on small teams trained in manual removal, with a small-scale cull

trialled in Northern Ireland and a decade long control effort in Kent, UK. In both cases, hammers are used to break open shells. The advantages of the methodology are that it is low in cost and equipment and has minimal collateral damage on ecosystem. The number of people needed are low but training of participants in the techniques is required, and multiple repeat control efforts are likely to be needed if more than a few individuals are present. The techniques can only be carried out in the intertidal zone, with efforts limited by access at low spring tides.

Strangford Lough, Northern Ireland

A pilot cull using hand removal was trialled in Strangford Lough in August 2008 (Guy and Roberts 2010). A baseline survey identified sites with the highest densities of *M gigas*, which were selected to test the efficiency of manual control. Densities in Strangford Lough at the time were relatively low, only in some locations did densities exceed 1 individual per m^2 (Guy and Roberts 2010).

A small team of three people used a hammer to break the valves on *M gigas* individuals encountered along set transects along the lower intertidal zone. The pilot cull resulted in lower densities of oysters in the following year in culled sites, whereas control sites generally showed an increase in oyster density. Average population declines of 89% were achieved at culled sites. The recommendation was to repeat the cull annually to take account of likely continued recruitment, with density surveys repeated every 5 years to evaluate the influence of management and recruitment on *M gigas* populations in the area (Guy and Roberts 2010).

Northeast Kent Marine Protected Area (NEKMPA)

The extent and ecological impact of *M gigas* has been studied in the NEKMPA since 2007. A small volunteer group was set up to attempt to manage *M gigas* numbers in the NEKMPA, with efforts targeted on a particularly vulnerable area of chalk reef. Given the potential to damage the chalk reef, an initial pilot study was set up to determine the effectiveness of control methods (McKnight and Chudleigh 2015).

The volunteer group has continued to work in this area since 2012, with a small, trained team (~15 volunteers in total) conducting removal of oysters from the reef using the methodology set out in McKnight and Chudleigh (2015). In 2020, the cumulative effort over the year resulted in 322,000 oysters removed and over 1543 hours spent on site. In the area where removal has been ongoing, there has been no spread of the oysters into the mudflats and oysters are limited to scattered single specimens or small clusters. This can be contrasted with the formation of reefs and spread into the mudflats that can be seen on the north coast of the NEKMPA.

The success of the operation in this small area is hampered by recolonisation from other nearby areas, including the local port and marina which are more difficult to access to remove source populations from (McKnight 2021, pers. comm.). This highlights the difficulties of management if the potential for recolonisation is not adequately assessed. However, the efforts of the group show that containment of impact for particularly vulnerable sites is possible even when oyster numbers are high.

Fishing and harvesting

Fishing of wild oyster populations has taken place in areas of Europe where *M. gigas* has established in high densities. This is only likely to be viable if densities of *M. gigas* are high enough to sustain fishing effort, and is unlikely to lead to eradication of the population. However, collaboration with oyster farmers is likely to be an important aspect of management and may offer opportunities for monitoring spread of wild *M. gigas*.

In the Blackwater estuary in southeast England, early introductions of *M. gigas* as an aquaculture species led to spawning as early as 1965 with wild oysters recorded in the 1970s (Herbert 2012). However, since the 1990s more extensive spatfall has led to larger wild populations which are self-sustaining and these wild populations are now commercially fished (Herbert 2012). The cultivation of oysters on trestles in other parts of the estuary also uses seed obtained from the wild spawning population, which reduces operating costs (Herbert 2012). The oyster farmers contribution to reducing Pacific oyster populations and aiding in restoration of native oysters is thought to be important in managing *M. gigas* population numbers in the estuary (Herbert 2012).

In the Oosterschelde area of the Netherlands, wild oysters are fished by oyster farmers for spat and to use the shells as substrate for cultured stock (Wijsman 2021, *pers. comm.*). This has occurred in the eastern part of the Oosterschelde since 2006. Although the volumes removed are unlikely to lead to eradication or control, along with other pressures this may contribute to reducing population numbers to levels where fewer impacts are seen.

In Lough Swilly, Co. Donegal an intensive commercial fishery operated in 2010 and 2011 based on fishing wild populations of Pacific oysters, showing the high biomass of the species in this area. High catches and high market prices made this viable (Tully and Clarke 2012).

2.4.3 Impact adaptation and ecosystem restoration

Control actions for IAS have been prompted by injury or inconvenience to people in local communities, where access issues or harm caused by the development of oyster reefs have led to trial removals (Wijsman et al. 2010; Herbert et al. 2018). In areas where injuries may be caused to recreational users, signage advising caution and appropriate footwear may be required to prevent harm to people and pets.

There is currently a lot of interest in the restoration of marine habitats including native oyster reefs, driven partly by the carbon sequestration potential of oyster reefs. It is possible that restoration of marine habitats could increase the biotic resistance of these systems to invasion by IAS. Efforts to reduce other anthropogenic stressors could also decrease disturbance and reduce opportunities for IAS establishment.

2.4.4 No management

In much of the invaded area where *M. gigas* has established no management action has been taken. In the Wadden Sea, the decision to take no management action to remove *M. gigas* reefs was influenced by the extensive distribution of the reefs and the possibility of collateral damage to the associated habitats. However, the choice to take no management action needs to be made in full awareness of the potential for future impacts should the window for action be missed.

2.5 Managing *Magallana gigas* in Galway Bay

The example of *M gigas* in Galway Bay illustrates how species distribution and larval tracking models can be incorporated into the decision-making framework for IAS. This is intended as a demonstration of the applicability of these models and the management measures discussed should be interpreted as examples not firm recommendations.

2.5.1 Site description – Galway Bay

Galway Bay is located on the west coast of Ireland, with most of the coastline located in Co. Galway but the southern part of the bay located in Co. Clare. It features several smaller shallow bays and inlets and numerous small islands in the inner eastern part of the bay, as well as the larger Aran Islands at the mouth of the bay (NPWS 2015). The Aran islands and Black Head partially protect the site from exposure to Atlantic swells (NPWS 2015).

Large areas of intertidal sand and mudflats are found on the sheltered eastern coast between Oranmore Bay and Kinvarra Bay. Moderately exposed intertidal communities are supported by the intertidal sediments found in Galway Bay. For many centuries, European native oysters were commercially fished in Galway Bay before stocks became depleted due to factors such as poor fisheries management and disease (Tully, Clarke, et al. 2018). However, some native oyster stocks remain and there is interest in restoring stocks to former levels.

The inner part of Galway Bay is a protected area, designated as the Galway Bay Complex Special Area of Conservation (SAC) (000268) and the Inner Galway Bay Special Protected Area (SPA) (004031).

Galway Bay is designated as an SPA due to the large variety of wintering waterbirds which it supports, providing feeding and roost sites for internationally important wintering populations of Great Northern Diver and Brent Goose, and nationally important populations of sixteen other wetland species[5] (NPWS 2013a). As a wintering site the bay is also important for a number of gull species, and has nationally important breeding colonies of Cormorant and two species of tern, the Sandwich Tern and Common Tern (NPWS 2013a).

The Galway Bay Complex SAC is designated for three marine Annex I habitats (Large shallow inlets and bays, mudflats and sandflats not covered by seawater at low tide, and reefs), and the Annex II species *Phoca vitulina* (harbour or common seal). The Large Shallow inlets and bays habitat may incorporate other Annex I habitats within its area. Within the Galway Bay Complex SAC there are 12 community types which have been described and which are found in one or more of each of the Annex I habitats.

- Mudflats and sandflats not covered by seawater at low tide [1140]
    - Intertidal sandy mud community complex
    - Intertidal sand community complex
- Reefs [1170]
    - *Mytilus*-dominated reef community
    - *Fucoid*-dominated reef community
    - *Laminaria*-dominated reef community
    - Shallow sponge-dominated reef community complex

[5] Black-throated Diver, Cormorant, Mute Swan, Wigeon, Teal, Shoveler, Red-breasted Merganser, Ringed Plover, Golden Plover, Lapwing, Dunlin, Bar-tailed Godwit, Curlew, Redshank, Greenshank, Turnstone.

- Large shallow inlets and bays [1160]
    - Intertidal sandy mud community complex
    - Intertidal sand community complex
    - Maerl-dominated community
    - *Zostera*-dominated community complex
    - Fine to medium sand with bivalves community complex
    - Mixed sediment dominated by Mytilidae community complex
    - Shingle
    - *Fucoid*-dominated reef community
    - *Laminaria*-dominated reef community
    - Shallow sponge-dominated reef community complex

Intertidal and subtidal surveys were carried out at this site in 2006, 2009 and 2010 to map the habitats and determine the physical and biological nature of the site (NPWS 2013b). More detail on the habitats and surveys used to determine their area and location can be found in the site conservation objectives and supporting documents on the NPWS website (NPWS 2013b).

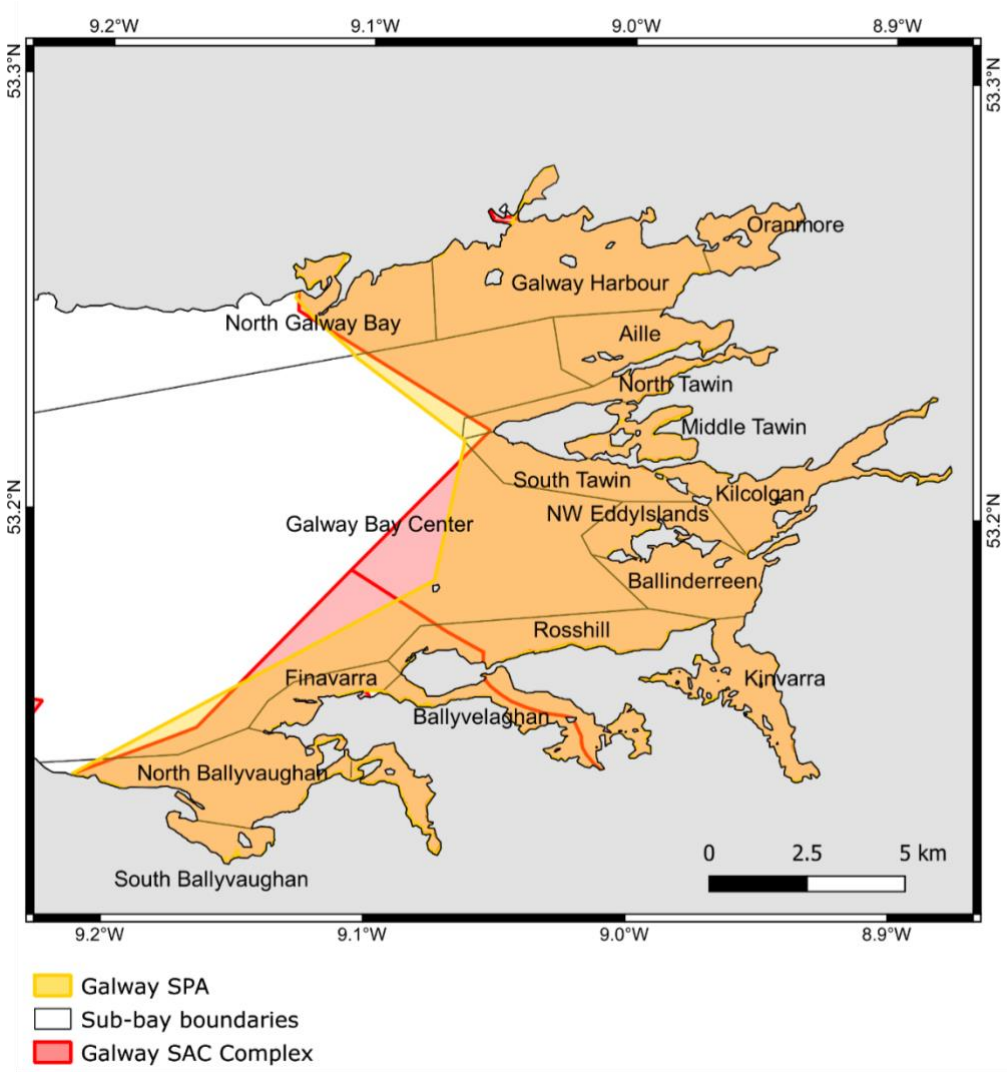

Figure 2.4 Map of Galway Bay showing the limits of the Galway Bay SAC Complex and SPA area. The larger bay has been divided into smaller regions which are used within the modelling

part of this study. The geographic area of these smaller regions is shown as well as the name given.

2.5.2 Known distribution of *M gigas* in Galway Bay

Pacific oysters are cultivated in Galway Bay, on several sites in the north and east parts of the bay. The majority of the oyster cultivation in Galway Bay is on trestle tables, although there may be some fishing of Pacific oysters on the seabed in the Clarin river (Kilcolgan bay) part of the Bay (Tully and Clarke 2012). The classification of Galway Bay as free of the oyster herpesvirus led to farms only sourcing stock material locally, thus having higher numbers of diploid stock cultivated than in other parts of the country. This was ended in 2021, opening opportunities for import of better-quality triploid stock from France.

Zwerschke et al. (2018) carried out five surveys within Galway bay in 2012/2013, repeating previous surveys carried out in 2009 (Kochmann et al. 2013). In 2012/2013 Pacific oyster abundances were recorded as rare (<0.009/m^2) in Parkmore (Kinvarra) and Ballvelaghan, occasional (0.01-0.09/m^2) in Dunbulcaun and Finvarra, and frequent (0.1-0.9/m^2) in Ballinacourty (Kilcolgan bay) (Kochmann et al. 2013; Zwerschke, Kochmann, et al. 2018).

In 2018, an intertidal survey of inner Galway Bay was carried out to investigate the distribution of native oysters (*Ostrea edulis)* but also documented occurrences of Pacific oyster, mussels and periwinkles (Tully, Clarke, et al. 2018). The quadrat surveys found that Pacific oysters mainly occurred in the Clarin River estuary (5% prevalence) with lower prevalence in Kinvara Bay and Tawin (1%). However, no extensive populations were found with numbers much lower to those observed in Lough Swilly, with only individual oysters were recorded in Eddy Island, Oranmore Bay, and St Georges (Tully, Clarke, et al. 2018).

2.5.3 Predicting the spread of *Magallana gigas* in Galway and Bantry Bay

The results of the habitat suitability model predict that habitat suitable for the establishment of Pacific oysters can be found in much of the eastern and southern part of Galway Bay (Figure 2.5).

The availability of suitable substrata may not correlate directly with oyster presence, as hydrodynamic conditions may affect the dispersal of spat. Observations of wild oysters in Strangford Lough are thought to be influenced by the hydrological regime, with the cooler temperatures and higher exchange rates in the southern part of the Lough suspected to be a barrier to settlement despite availability of suitable substrate (Guy and Roberts 2010).

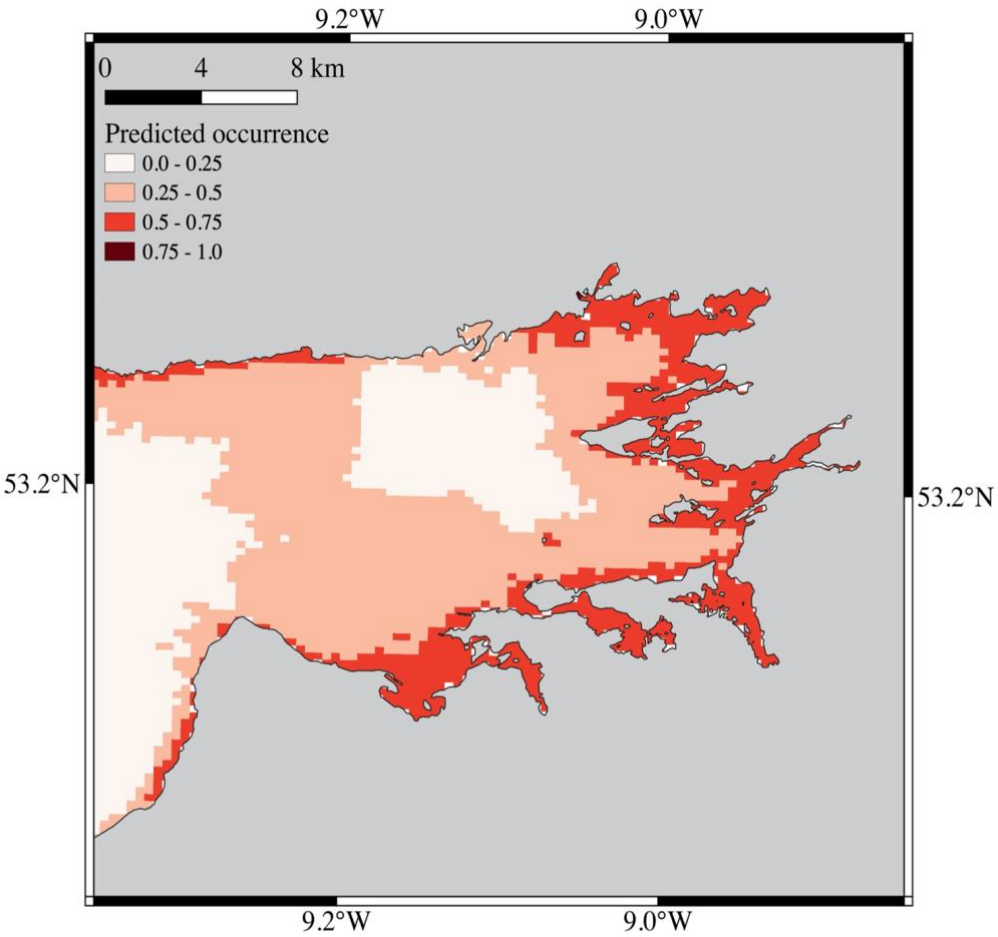

Figure 2.5 – Habitat distribution map for *M. gigas* in Galway Bay based on species distribution modelling.

The results from the models show the likely distribution of oyster spread in Galway Bay from farms and wild populations averaged over three years from 2018 – 2019 (Figure 2.6). The areas with the highest density of larval settlement are in the eastern and southern part of the bay, often located within the more restricted sub-bays. In analysing larval density from different sources (see Figure 2.10, Figure 6.3, and Figure 6.4) it is possible to see that spread from wild populations is largely restricted to the southern part of the Bay, whereas spread from farm sources is spread across the eastern part of the bay (Figure 2.10).

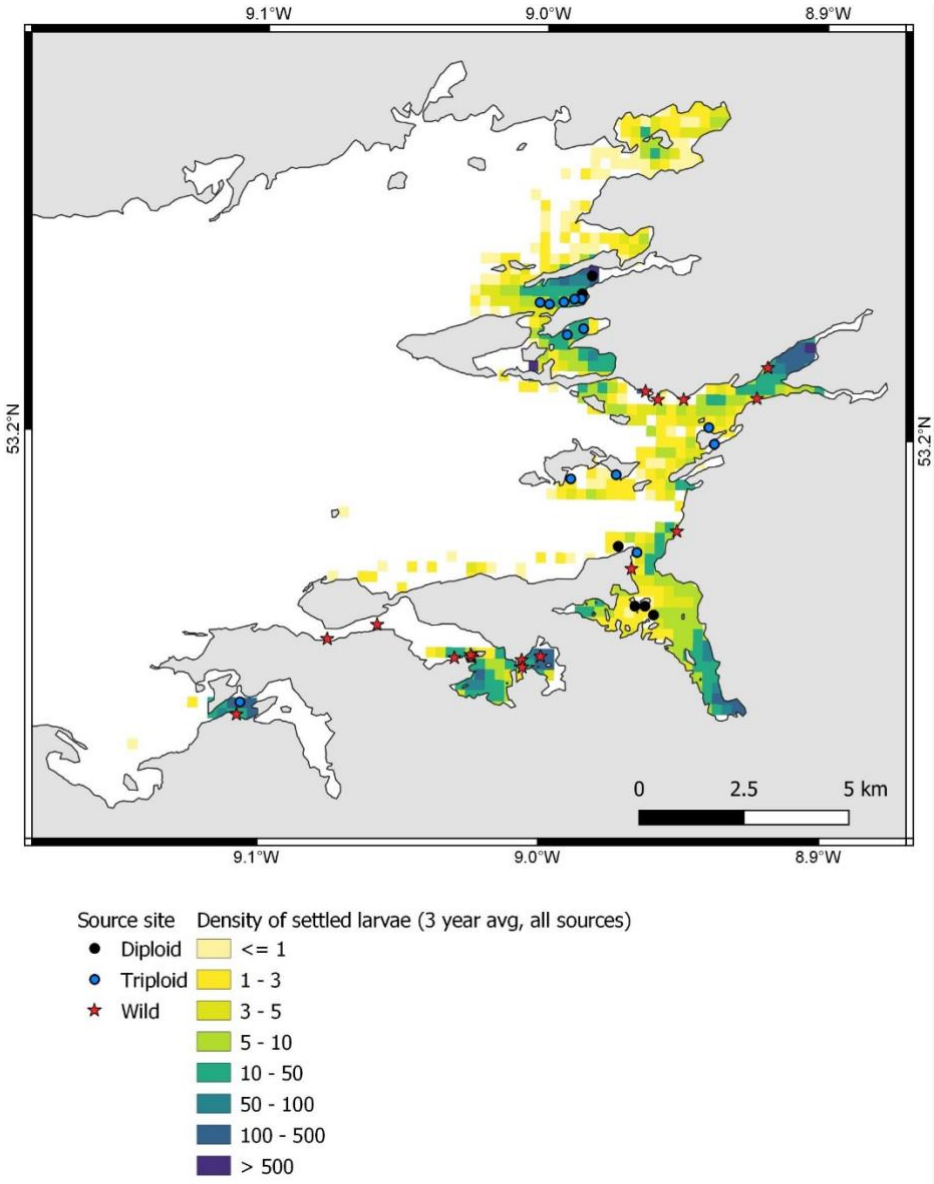

Figure 2.6 – The distribution and density of modelled larvae settlement showing a three year average for models run in the years 2017, 2018 and 2019. Density includes larvae from all sources, both wild and farm sites.

Patterns of larval dispersal in Galway Bay

An alternative approach to estimating risk of *M gigas* establishment was taken by simulating larval release from multiple random points within inner Galway Bay. This was intended to help understand the influence of bay geography and ecosystem hydrodynamics on the export and retention rates of *M gigas* larvae within smaller sub-bays (Figure 2.7). The larval behaviour of *M gigas* should favour retention (Hills et al., 1991; Knight et al., 2006).

Bays which have high levels of retention may be more likely to have established populations of wild oyster – results from previous surveys shown that bays with high retention rates are more likely to have wild populations. These bays also more likely to have a higher temperature and reach the required number of day degrees earlier, making it more likely that larvae will experience the temperatures needed for settlement.

Sea temperature may also vary depending on local geography, with shallow enclosed bays reaching higher temperatures that are more likely to favour recruitment even in cooler years. This has been suggested as a reason for why recruitment has been observed in sub-optimal cooler years in Strangford Lough (Guy and Roberts 2010). This can be observed in the variation in the time at which the necessary degree-days for gametogenesis and spawning are attained across the bay, with earlier spawning triggered in the inner shallow sub-bays in the east (Figure 2.8).

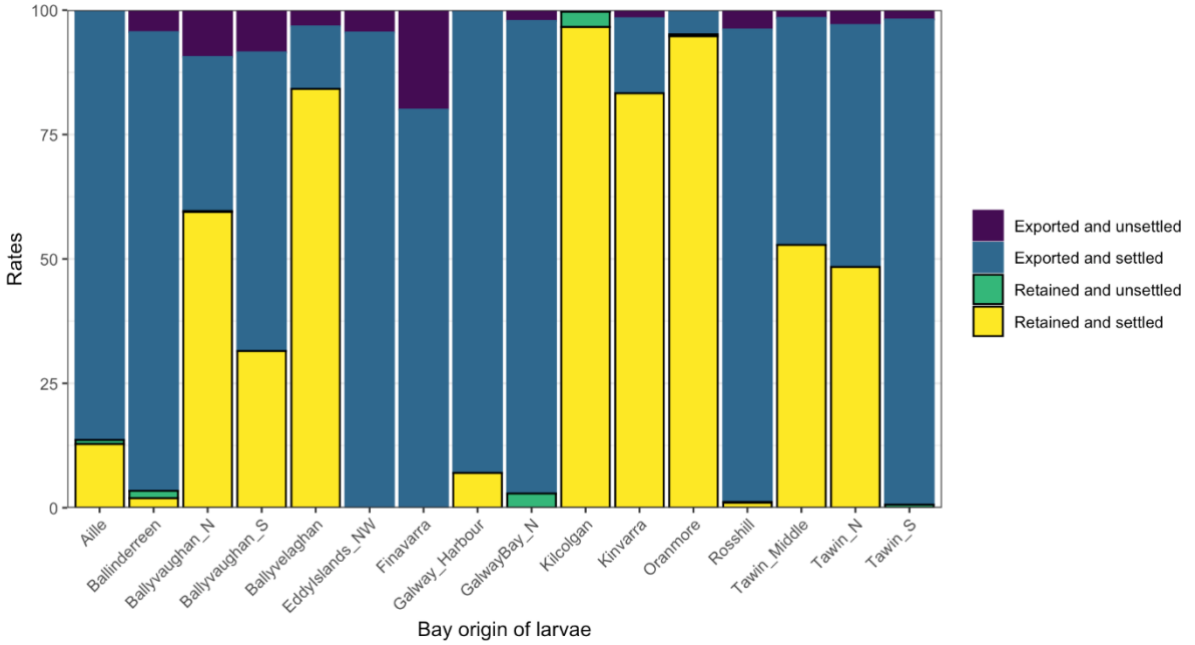

Figure 2.7 Bar chart showing the proportion of larvae which are either retained or exported from sub-bays within Galway Bay, combined with their settlement status. Sub-bay boundaries are displayed in Figure 2.4.

In terms of transported distance, the rate of exported larvae with retentive-oriented behaviour were similar in both regions (Table 1). Less than a third (~28%) of the exported larvae with a retentive-oriented behaviour travelled as far as 5 km away from their source, raising the possibility that larvae dispersed to a nearby area, and around 25% of them were transported between 5 to 10 km away. However, the rates of larvae exported further than 10 km away from the source sites were two times higher in the Bantry region (10.4%) than in the Galway region (5.4%). Furthermore, these rates had the highest variations from passive larval simulation (decreasing by 33% and 50%). The settlement rates for these larvae were limited and no higher than 9% in both regions.

Exportation of larvae was increased along more exposed straight coastlines, whereas retention increased if source sites were located in semi-enclosed bays. The characteristics of the seascape may thus influence retention rates and propagule pressure.

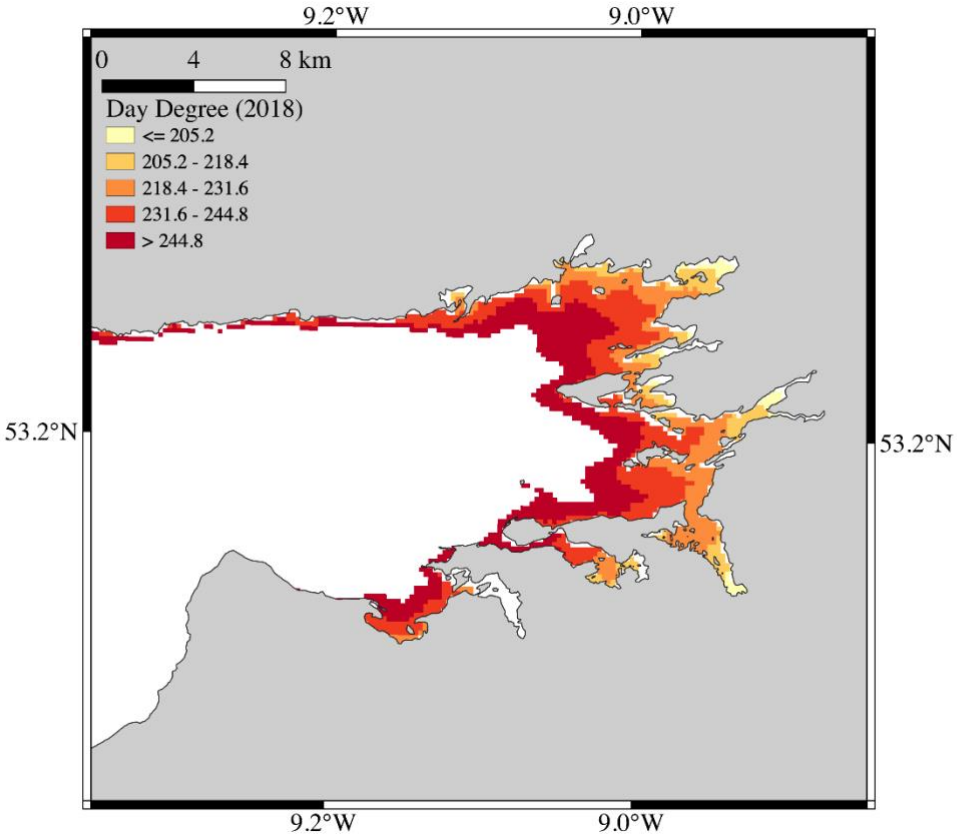

Figure 2.8 – Time at which the number of day degrees required for spawning was reached across Galway Bay.

2.5.4 Identifying vulnerable habitats using spread maps

The impact of *M gigas* on the habitats in the Galway Bay SAC is likely to change depending on the density of *M. gigas* populations. Isolated individuals or clumps are unlikely to have large impacts on species or habitats. However, should more extensive reef formation occur this could lead to changes in community composition and habitat use with potential impacts for the environment and people.

Significant alteration of the type of community present in the SAC could result in changes to the integrity of the habitats for which the site has been designated (Herbert 2012). It is unlikely that at current levels *M. gigas* populations will damage the integrity of the whole designated site. However, the maps of predicted larval settlement and habitat suitability indicate that large amounts of the eastern and southern part of the bay could support recruitment of *M. gigas*.

The status of Galway Bay as an SAC means there is detailed information available on the marine community types present within the area which might be affected by *M. gigas* establishment. Overlaying the larval density maps over the spatial data on marine community type found within the Galway SAC allows identification of habitats which might be vulnerable to oyster settlement. This information can be used to recognize habitats at risk, target surveillance and monitoring efforts within sites that are more vulnerable, and identify where control efforts might be restricted due to the presence of sensitive habitat.

The average number of larvae predicted to settle within each habitat type over the three years tested is shown in Figure 2.9, although it should be noted that this is not standardised by the area covered by each habitat. Most settlement within the intertidal zone falls within the fucoid dominated

community complex. There is no settlement predicted to occur in *Zostera sp.* seagrass beds. Settlement is also low in the intertidal sand community complex, the shingle habitat, the laminaria-dominated community complex, and the shallow sponge-dominated community complex.

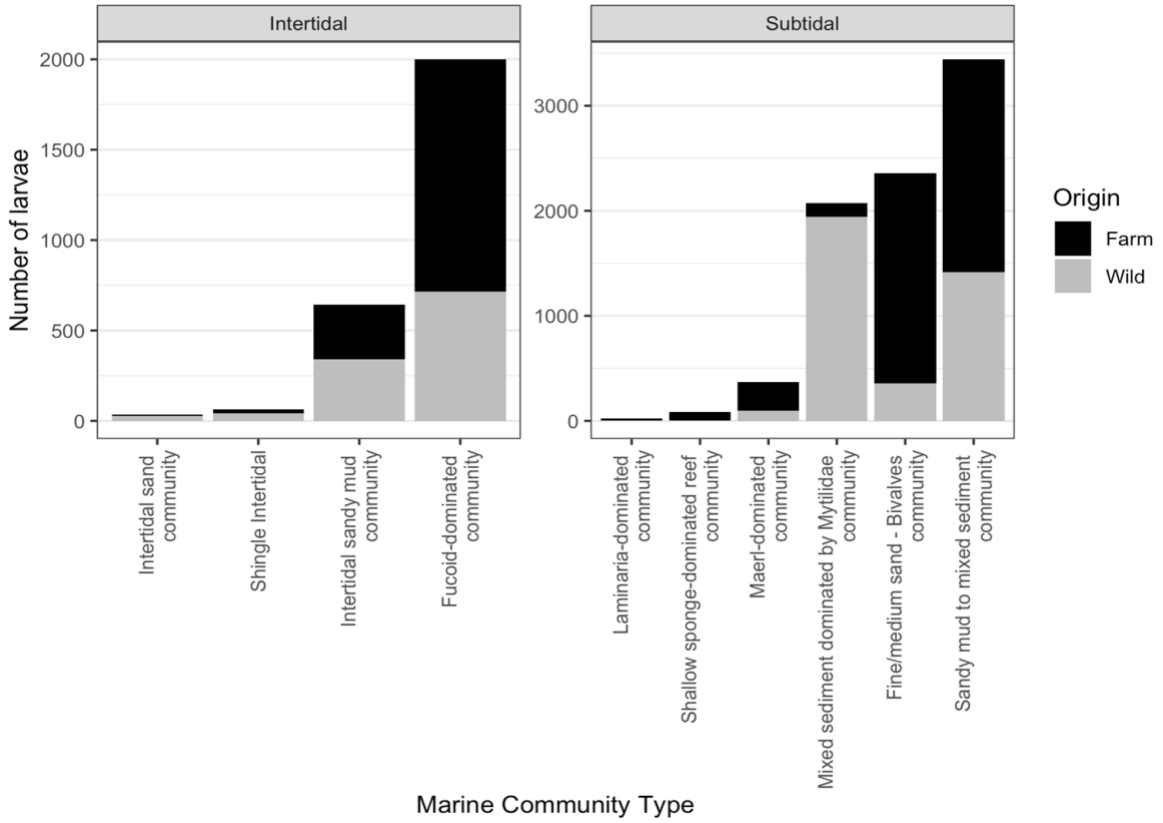

Figure 2.9 - The average number of larvae across three years of simulated larval tracking settling within different marine community types within the Galway SAC. Larvae are divided by source location as either originating from wild release sites (grey) or aquaculture farms (black).

Intertidal habitats

A1.1 Littoral rock and other hard substrata (*Fucoid* dominated community complex)

In the intertidal zone, the highest densities of larval settlement occur within the *Fucoid*-dominated reef community complex which is found as a narrow band across most of the coastline of eastern and southern inner Galway Bay.

The density of fucoids and other macro-algae within this community type may affect the likelihood of Pacific oyster establishment, as dense macroalgal coverage may prevent establishment (Kochmann et al. 2013; Cook et al. 2014a). It is hard to determine without more extensive surveys the level of macro-algal coverage in this habitat. Surveys of intertidal bivalves have shown that colonisation by native oysters and Pacific oyster do occur in this habitat type (Tully, Clarke, et al. 2018), but perhaps the density of macroalgae have prevented population expansion .

Intertidal sandy mud/sand community complex

In some areas of Europe such as in the Wadden Sea, large areas of intertidal sediments have been transformed into oyster reef (Reise 1998; Herbert 2012). Oysters settle on the remains of bivalve shells or scattered hard substrate, and once small clumps of individuals are established can settle on other

oysters to form more extensive reefs. In the Wadden Sea these reefs are mostly found from around MLW to the shallow subtidal (Herbert 2012).

Formation of these reefs in Galway Bay is a possibility and could have consequences for the type of benthic community found, the feeding and behaviour of bird species found, and the use of the habitat by people visiting and in the local communities.

Subtidal habitats

Determining the impact of *M. gigas* on subtidal soft and mixed sediment communities in Galway Bay is more complex, as it is hard to predict if there will be enough hard substrate for settlement of *M. gigas* larvae. It is likely at low densities establishment and spread into subtidal habitats will be limited, but this could increase if the number of reproducing wild oysters in the bay increases. There is some suggestion that there are subtidal oysters occurring in the eastern part of Galway Bay in the Clarin estuary, but it is hard to know how much of this is supplemented by farm spawning or escapes or historic fishing activities (Tully, Clarke, et al. 2018).

Biogenic habitats

Native mussels (Mytilus edulis)

Shoreline surveys in 2018 documented low prevalence of *Mytilus edulis* in Inner Galway Bay despite evidence of spatfall, with higher distributions found only in a couple of locations in the Clarin estuary and Oranmore Bay (Tully, Clarke, et al. 2018). There are no extensive mussel reefs documented within the bay, so it is unlikely *M. gigas* will have any significant impact on *M. edulis* in Galway Bay.

Native Oysters (*Ostrea edulis)*

There has been extensive fishing of native oyster populations for many centuries in Galway Bay, although populations have been negatively impacted by fishing pressure, disease and loss of habitat. There is a native oyster restoration project in Galway Bay run by a local community group with support from the Marine Institute.

Sabellaria alveolata

Records of intertidal *Sabellara alveolata* populations have recently been collated and show that densities are much higher on the northern coast of Galway bay, with fewer records on the eastern and southern shores (Firth et al. 2021). The larval density spread maps show little spread of *M. gigas* on the northern shore, so it is likely that there will be low impact on S. alveolata populations in the Bay based on the current distribution of *M. gigas* in the bay.

Seagrass beds (Zostera spp.)

There is no overlap between predicted spread of oyster larvae and the small areas of seagrass (*Zostera* sp.) habitat within Galway Bay. However, oyster beds in close proximity to seagrass beds have been shown to reduce growth, likely through changes to sediment composition (Kelly and Volpe 2007). Monitoring of the extent of oyster establishment in areas nearby may thus be warranted. Furthermore, control efforts may damage sensitive seagrass beds so any management action should take care to avoid disturbance of this habitat.

Maerl

There are three main sites where maerl beds are found in the southern and eastern parts of Galway Bay, these are mostly *Lithothamnion coralloides*. An extensive bed of *Phymatolithon calcareum* is found in Mucknish Bay supported by strong tidal currents.

There is some settlement within the maerl-dominated community complex, the most notable site is in Ballyvaughan. In this bay, most of the settlement is from retention of larvae from wild oyster source sites which have been documented here. There are very few studies investigating the impact of *M. gigas* on maerl populations, thus monitoring whether settlement occurs within this habitat could be useful to determine whether further study of impacts is warranted. Most subtidal methods of *M. gigas* control have the potential to negatively impact maerl reefs, and are thus unlikely to be permitted.

Wading birds

Impacts to wading birds are species specific, with some species able to adapt to feed on *M. gigas* and others negatively impacted by presence of oyster reefs. In Galway Bay, more significant pressures to the birds may stem from other anthropogenic sources, including sewage effluent and aquaculture industry detritus and disturbance. Shoreline habitats are also at risk from urban expansion of Galway City. If management activities or surveys undertaken, the risk of disturbance to feeding and roosting wading birds should be assessed.

2.5.5 Monitoring and surveillance

Baseline survey

A full baseline survey to understand the current distribution and state of wild populations in Galway Bay is essential in determining whether the window for management remains open. A recent intertidal survey carried was out by Tully et al. (2018), but there are a few gaps in the southern part of the bay where there are older records of wild *M. gigas*. Repetition of surveys in locations previously surveyed by Kochmann et al. (2013) and Zwerschke et al. (2018) using similar methodologies will allow for changes in population abundance to be estimated.

Surveys for *M. gigas* can be carried out by surveying the low intertidal at low spring tides. In most places in Northern Europe populations have established in the intertidal before spreading into the subtidal.

Finally, although the majority of predicted spread is from triploid farm sources with only low densities of settlement from diploid farms, the far northern part of the bay should be included in a survey plan to check that additional establishment has not occurred in this area. Data from Tully et al. (2018) shows that there are records of *M. gigas* individuals in this area, suggesting that spread from diploid farms continues to occur.

Monitoring and surveillance strategy

Following completion of a baseline survey, routine monitoring to keep track of new wild populations or changes in population status will enable potential changes in impact, or the window of opportunity for management to be identified. More extensive and systematic routine monitoring of the area covered by wild *M. gigas* can help show changes in spread and impact over time. For example, in the Netherlands the extent and range of oyster and mussel beds is surveyed on a yearly basis with the data hosted on a publicly available website.[6] Given the lower densities of Pacific oysters in Galway Bay, monitoring may not need to be as frequent as this, especially if the quantity of diploid stock cultured in the bay is reduced.

[6] https://shiny.wur.nl/Schelpdiermonitor_Banken/

In developing a surveillance plan to monitor future spread of *M gigas* from existing populations, the spread models can be used to target efforts in regions with the highest likelihood of settlement and establishment (Figure 2.11). The spread models identify the north-eastern part of the bay as vulnerable to spread from farm sites, while higher retention within sub-bays in the southern part of the bay could enable expansion of the wild population within these sites. The larger Kinvarra and Kilcolgan areas are predicted to have high densities of settlement from both wild and farm sources, so depending on the current extent of wild populations may need more frequent monitoring.

Artificial structures may provide a point of access and easily identifiable reference point for intertidal surveys (Figure 6.7). Deteriorating piers and walls may also provide hard substrate for the attachment of Pacific oysters, and thus could support establishment of new populations.

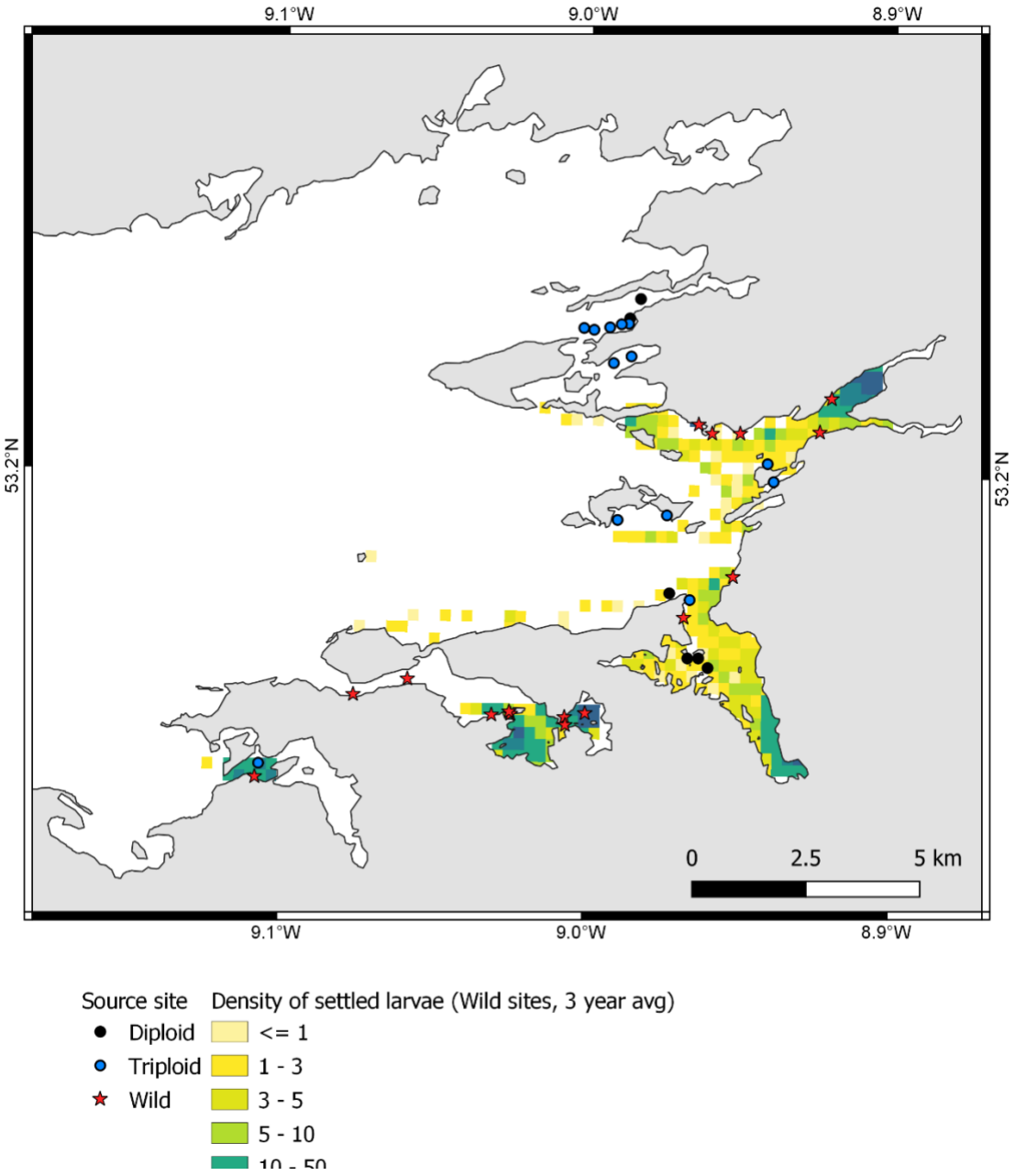

Figure 2.10 – Density of settled *M. gigas* larvae in Galway Bay averaged over three years from 2018 - 2020 with larvae released from wild source sites with suitable environmental conditions.

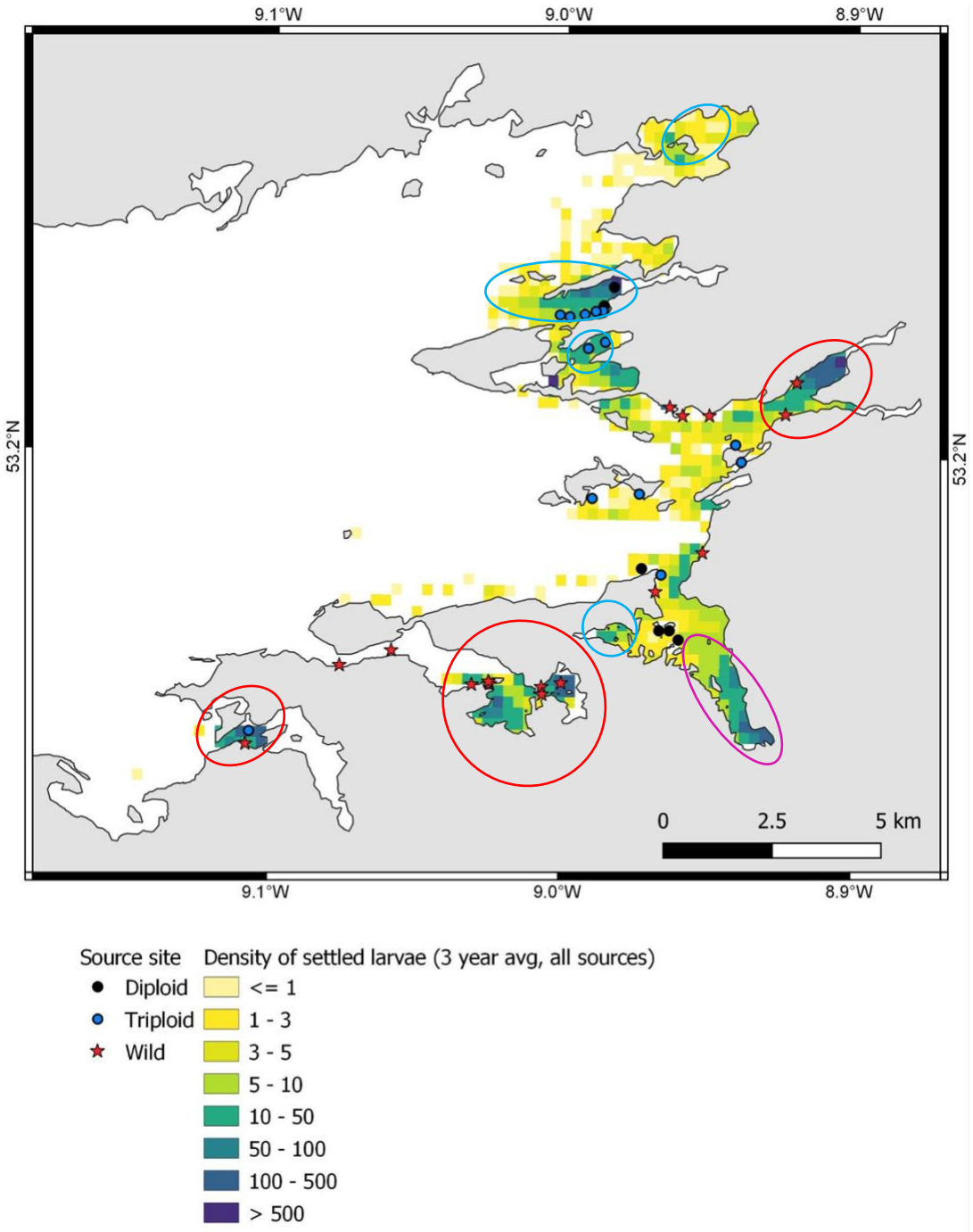

Figure 2.11 – Density of settled *M. gigas* larvae in Galway Bay averaged over three years from 2018 - 2020 with larvae released from all sites with suitable environmental conditions, including diploid and triploid farms and wild individuals.

2.5.6 Captive management of *M. gigas* in Galway Bay

The maps of predicted larval settlement from farm sources show the potential for continued introduction of wild oysters into large areas of the eastern part of Galway Bay (Figure 6.3, Figure 6.4). In North Tawin, where the majority of diploid farm sites are located, predicted settlement is most dense near to source sites within the Tawin area, although there is potential for exported larvae to reach the northern part of Oranmore Bay (Figure 2.12). Spread from diploid farm sites located in Kinvarra is further reaching with most exported larvae transported to Kilcolgan bay, although the highest densities of larvae are still retained in Kinvarra bay (Figure 6.3, Figure 2.12Figure 6.14).

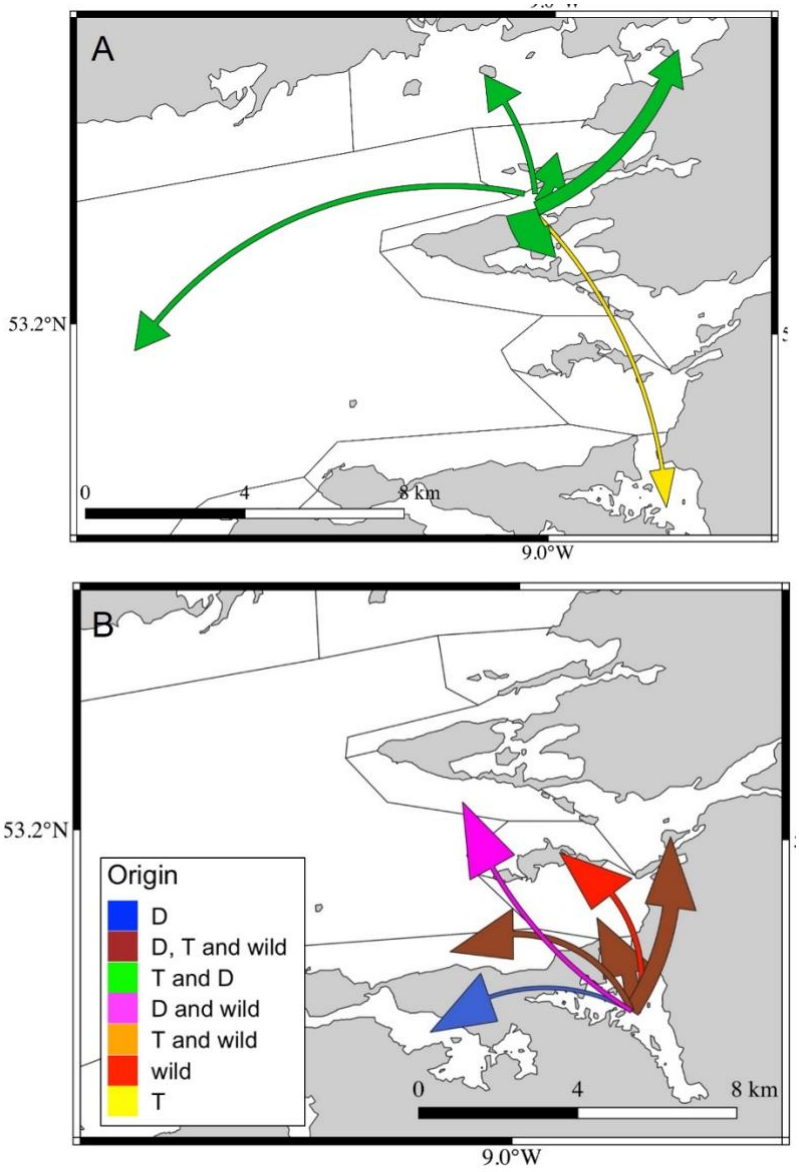

Figure 2.12 – Connectivity within sub-bays of Galway Bay, showing export to other sub-bays from release sites in (a) North Tawin, and (b) Kinvarra, where the majority of diploid farm release sites are located. Arrows indicate relative quantity of larvae exported and are colour coded by release source. For additional information on average numbers of larvae exported and retained from each site see (a) Figure 6.17, and (b) Figure 6.14 in the appendices.

Given the potential spread from diploid sources in Tawin North, promoting the culture of triploid stock in this area could help limit further range extension of wild populations into the north of the bay where they will also be closer to the city of Galway. Records from 2018 show that wild individuals have already been recorded in Oranmore bay (Figure 2.13). Connectivity maps show that dispersal into this area from diploid and triploid farm sites but not from existing wild source sites was shown to be possible by the model outputs, although of course the models are for years after this dispersal would have occurred (Figure 6.19)

The change in the disease-free aquaculture status of Galway Bay will likely open opportunities for a shift from diploid to triploid stock that can be imported from France. This would help reduce continued re-settlement from farm sources, and as wild population numbers are relatively low could help reduce the likelihood of self-sustaining wild populations forming.

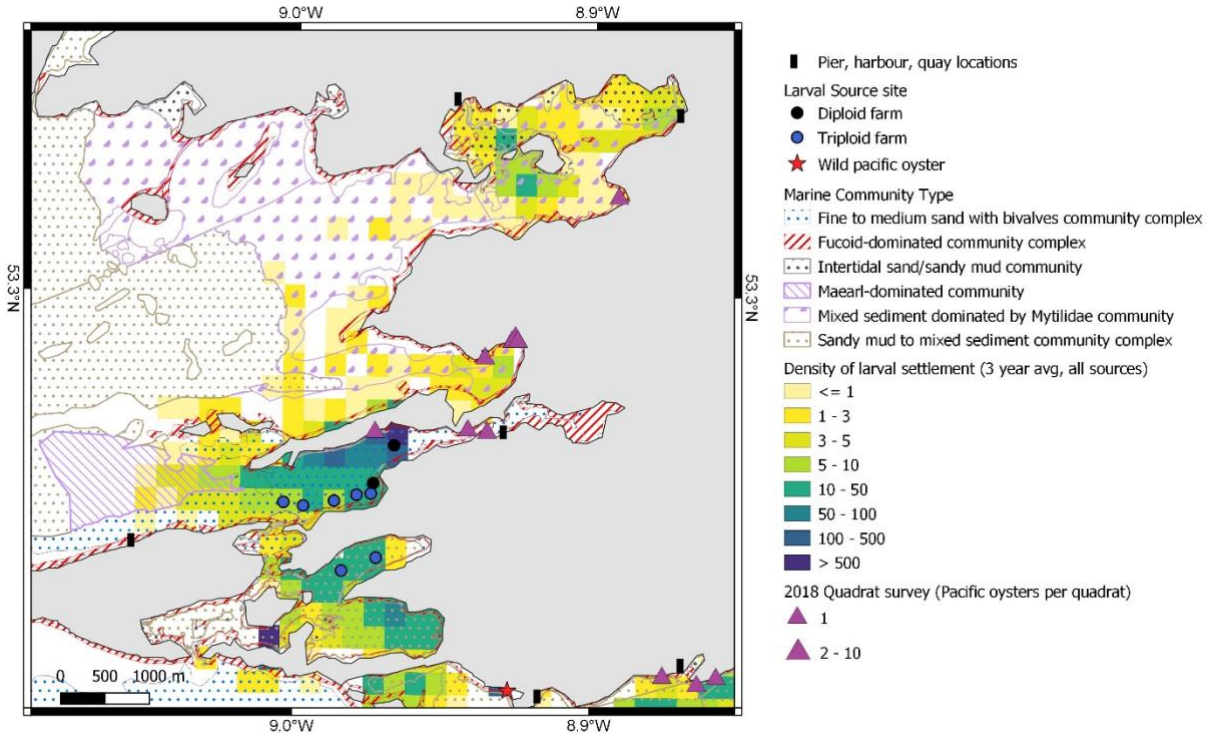

Figure 2.13 – A detailed map showing the predicted density of larval settlement in Oranmore bay, in the north-eastern part of Galway Bay.

2.5.7 Long-term management: Containment & Suppression

Long-term management of *M. gigas* in Galway Bay could aim to suppress population numbers and prevent establishment of high-density reefs. The presence of oyster aquaculture operations in Galway Bay means that complete eradication of wild oysters from Galway Bay in unlikely, as reintroduction will always remain a possibility. However, even if eradication is not possible, long-term management can still deliver benefits to the local environment and communities.

Long-term management may become more likely to be effective if there is a shift towards cultivating only triploid stock, as suppression of wild oyster populations could reduce population sizes to low enough levels that the likelihood of reef establishment is reduced. Carrying out control activities across the whole of Galway Bay may be difficult, but the abundances of Pacific oysters in most areas seem to be low enough that control could be a possibility. Long-term management may become more

viable if all aquaculture operations in the bay change to using triploid stock, as this could reduce re-settlement of cleared areas from farm sources.

While diploid stocks are still cultured, trial control studies could be carried out in the southern part of the bay where there is limited settlement of larvae from farm sources and thus lower chances of recolonisation. The majority of larvae produced by wild populations are retained within the bays of Ballyvelaghan and Ballyvaughan, with no to little import of larvae from other parts of Galway Bay seen over the three years simulated (Figure 6.10, Figure 6.11).

Control methods

Intertidal populations:

As identified in the review above, hand removal or collection of Pacific oysters represents a cost-effective method of keeping *M. gigas* numbers low if extensive reefs have not yet formed. Where reefs have formed it is possible to suppress population numbers in limited areas to reduce impacts on vulnerable or protected habitats.

Subtidal populations:

The feasibility of controlling subtidal populations in Galway Bay is likely to be low. Given the status of the bay as an SAC, dredging using mobile fishing gear could have a negative impact on protected features, especially on seagrass or maerl beds or other bivalve species (see Section 2.4.2). Any subtidal management would need to assess impacts on native species and habitats and take accordance of existing restrictions.

Any fishing or harvesting activity of *M. gigas* in Galway Bay should be permitted and monitored, as it is possible in the future that establishment of *M. gigas* in habitats sensitive to disturbance could occur (e.g. where accessing oysters disturbs seagrass beds). Furthermore, fishing and harvesting activities could also disturb wading birds and the potential adverse impacts of disturbance should be considered.

Managing *M. gigas* in Galway Bay – taking account of local features

The recommendations in this section are for illustrative purposes only, intended to demonstrate the type of management actions that could be prompted by model outputs. Management action would need to take better consideration of recent survey data, the experience of local stakeholders and those with more detailed knowledge of the sites in question.

Ballvaughan

The presence of a Maerl bed in Ballvaughan warrants closer examination of the status of *M. gigas* within the bay as there is some overlap with larval settlement predicted based on spread from the existing record of feral pacific oysters.

The models showed that over the three years simulated, no larvae arrived from elsewhere in Galway Bay, suggesting if control actions within this site were taken then there would be minimal recolonisation from elsewhere (Figure 6.9). If there are only few isolated *M. gigas* individuals, management action could be taken at the same time as a survey of status is conducted. The use of triploid stock at the aquaculture site in Ballyvaughan should be confirmed, and if management action is taken the farm should continue to use triploid stock to ensure recolonisation from the site does not occur.

Ballvellaghan

The number of wild records within the small shallow Ballyvelaghan bay indicates that at least in the past conditions are suitable for *M. gigas* establishment, although spawning from these sites was not consistent across all 3 years.

The high retention of larvae within the bay could facilitate recruitment of *M. gigas* from the wild population. However, as arrival of spat from outside the bay is limited (Figure 6.11), control of the existing wild populations would have higher chances of eradicating *M. gigas* from this small area as there would be few chances for recolonisation. The habitats found within the bay could support *M. gigas* settlement, especially if there are dead shells providing hard substrate within the mixed sediment dominated by Mytilidae community complex.

Kilcolgan

Kilcolgan Bay has reports of a number of wild oysters potentially extending into the subtidal (Tully, Clarke, et al. 2018). Analysing the import and retention of larvae into and within Kilcolgan Bay shows that it receives larvae from a number of nearby sites as well as retaining the majority of its larvae (Figure 6.12). Larvae arriving into Kilcolgan Bay are from both wild and from triploid farms (although it is hoped that settlement from triploid sites would be minimal). Thus while the potential higher numbers of wild M. gigas in the bay would suggest that management action might be necessary, it might be better to first address surrounding sites exporting larvae into Kilcolgan before attempting control action within the Bay. The first step would be to survey the areas to give a more accurate assessment of the status of wild oysters within this part of Galway Bay.

2.6 Conclusions regarding management of *M. gigas* in Ireland

The Pacific oyster is established in the wild in a number of locations mostly on the west and north coast but is currently mostly at low densities which are unlikely to have major impacts on the environment or ecosystem-services. However, as climate change drives increases in sea temperatures, environmental conditions suitable for *M. gigas* recruitment will become a more frequent occurrence. This could increase recruitment and density of wild populations. Where *M. gigas* forms dense reefs, it causes changes in community assemblages, and can affect ecosystem processes and impact people who use the environment in which it is established.

Management of wild populations of *M. gigas* is possible at low densities or in restricted areas of vulnerable habitat. Larval tracking models that show predicted spread and connectivity at local scales can be used to identify vulnerable habitats, evaluate control strategies and estimate risk of recolonisation. This information can be used to inform future development of species management plans and can be used as a tool to facilitate discussion with stakeholders.

Understanding the dispersal potential of larvae from wild populations is reliant on up-to-date information on the extent and status of wild Pacific oysters in Galway Bay. However, a survey for wild *M. gigas* could also offer opportunities for verifying predictions from the larval dispersal models. The models can be updated following this or any other changes to farm status, to remove populations that no longer represent a spawning risk.

A new survey of the current distribution of *M. gigas* across Ireland would help understand where *M. gigas* is along the invasion pathway in Ireland. It has been nearly ten years since the last targeted survey for wild *M. gigas* in Ireland (Zwerschke, Kochmann, et al. 2018). Regular monitoring of *M. gigas*

populations is especially important as impacts and management opportunities are density dependent. The potential for applying existing management measures relies on early intervention when recruitment to the wild adult population remains at the stage of isolated individuals and small clumps of oysters. Given the current low densities of *M. gigas* around much of Ireland the potential for control of wild populations using manual hand-removal is possible. Given the need for repeated efforts if spawning populations remain in the region, identifying appropriate areas to target control is needed.

The experience of management in the UK and the Wadden Sea suggest attempting to control expansion once more extensive reefs have been formed is not feasible. The climate model projections run by King et al. (2020) suggest that by the period 2040-2059, recruitment thresholds for *M. gigas* will be regularly exceeded around Ireland even though ocean bottom temperature increases will be less pronounced in Irish waters than in other parts of northwest Europe. In this case, it may be that management interventions could be targeted in areas where vulnerable habitat exist

The results from the assessment of management options for *M. gigas* in Galway Bay could act as a first step in identifying eradication or control strategies and could be used to aid decisions on whether action is needed. This project was intended as an exemplar of the uses of localised particle tracking modelling in aiding management decisions, and before further management action is taken for *M. gigas* a species management plan should be prepared, which requires more thorough consideration of the practicalities of species management and more detailed knowledge of the local area. Input from local stakeholders, IAS experts, local authorities should be considered during this process.

3 Case study: Asian shore crab, *Hemigrapsus sanguineus*

3.1 Species profiles

Hemigrapsus sanguineus De Haan, 1835 [in De Haan, 1833-1850] - Asian shore crab

Figure 3.1 – A photo of *Hemigrapsus sanguineus*, by Ansel Oommen, Buglife.org.

Hemigrapsus takanoi Asakura & Watanabe, 2005 – Brush-clawed shore crab

Figure 3.2 – A photo of *Hemigrapsus takanoi*, by Hans Hillewaert, Flikr (use under attribution, non-commercial license)

The Asian shore crab (*Hemigrapsus sanguineus)* and the Brush-clawed shore crab (*Hemigrapsus takanoi)* are small brachyuran crabs native to east Asia. The two species have been introduced to the eastern USA and the northern coast of Europe where they rapidly spread, and were identified as high-risk horizon species likely to arrive in Ireland within the decade from 2017-2027 (Lucy et al. 2020).

H. sanguineus has a squarish carapace with 3 spines found behind the eyes, with distinctive light and dark bands on the walking legs. The carapace is generally dark in colour with orange, green or purple hues. Crabs can reach a carapace width of 35-40 mm, and male crabs have a fleshy lobe at the base of the largest claw (Bishop 2020). *H. sanguineus* has a large native range along the coast of the northwest Pacific, extending from Hong Kong (20 ºN) to Sakhalin Island (50 ºN) in Russia (Epifanio 2013).

H. takanoi adult crabs are very small, also featuring a square carapace with 3 spines, and reaching a carapace width of 25 mm. A small patch of 'fur' is found at the base of the largest claw in *H. takanoi* (Bishop 2020). *H. takanoi* was described as a separate species from the closely related *H. penicillatus* in 2005 by Asakura and Watanabe (2005). The native range of *H. takanoi* is the northwest Pacific, around the waters of Japan and Taiwan, and along the coast from Russia, Korea and northern China (Asakura and Watanabe 2005).

3.1.1 Habitat and biology

H. sanguineus can be found in intertidal and shallow sub-tidal zones in coastal marine and estuarine habitats. In its native range, *H. sanguineus* is typically found in the mid- to upper-intertidal zone but is also known to occupy the lower intertidal (Epifanio 2013). The species is commonly found on exposed rocky shores with high water flow where it shelters under rocks and boulders. The density of *H. sanguineus* in the USA was shown to increase with the rock coverage in intertidal areas, as this increases the availability of shelter (Lohrer et al. 2000; Jouett and Child 2014).

In Europe, *H. sanguineus* may migrate between littoral zones in different seasons, with one population in the Wadden sea found in the upper mid-littoral during spring-summer, but in shallow subtidal zones during winter months (Landschoff et al. 2013). In France, adult *H. sanguineus* are found in the mid-shore zone under boulders, but the absence of very small individuals led to the proposal that larvae of *H. sanguineus* may settle within mussel beds (Pezy and Dauvin 2015). Newly settled and small crabs were found within mussel beds which may also provide a predator refuge (Pezy and Dauvin 2015).

H. takanoi occupies intertidal and shallow subtidal muddy and rock shores in its native habitat, mostly in sheltered locations where it is found under boulders (Asakura and Watanabe 2005). It is also found in sheltered estuaries and ports (Wood et al. 2015). In the Dutch Oosterscheldt, *H. takanoi* is found within shellfish beds and under rocks in the intertidal zone, as well as within artificial oyster reefs constructed for coastal protection (van den Brink et al. 2012). On the French coast, *H. takanoi* shows a preference for settling in fine sediment under boulders, but may also display burrowing behaviours in soft-sediment habitats (Gothland et al. 2014). In the UK and Dutch populations, larger males were observed occupying subtidal habitats (Ashelby et al. 2017).

The two species display differences in their habitat preferences, with *H. sanguineus* more commonly found on moderately-exposed shores with higher water flow and coarse-sediment substrate, and *H. takanoi* on sheltered low-energy shores with fine sediment habitats, within harbours or on oyster/mussel reef on mudflats (Landschoff et al. 2013; Epifanio 2013). This habitat partitioning is seen in the species' native range as well as in invaded ranges in the USA.

H. sanguineus and *H. takanoi* can also be found associated with fouling communities on anthropogenic structures, including piers and breakwaters and on anthropogenic debris (Geburzi et al. 2015). In the

Wadden Sea, both *Hemigrapsus* species occupy artificial-boulder shoreline habitat, (Landschoff et al. 2013).

3.1.2 Physiological tolerances

H. sanguineus can tolerate a wide range of conditions in temperate regions spanning a wide range of salinities and temperatures. Populations of *H. sanguineus* survived a three-year period of very cold winters from 2010-2012 where sea water temperatures fell below freezing and ice was present on the shoreline (Landschoff et al. 2013). While *H. sanguineus* has been recorded in estuarine habitats, salinities higher than 19 are required (Gittenberger et al. 2010). *H. sanguineus* larvae also tolerate a wide temperature range from 15 to 30 ºC (Epifanio et al. 1998). *H. takanoi* larvae tolerate a wide salinity range from 15 – 35 (Gothland et al. 2014).

3.1.3 Timing of reproduction and recruitment

Female *H. sanguineus* crabs are capable of spawning several batches of eggs during one summer season (Epifanio et al. 1998; McDermott 1998). In the USA, the spawning period of *H. sanguineus* is from April to early October (Epifanio et al. 1998). Larvae undergo five zoeal stages and one megalopa stage, with pelagic larval duration dependent on water temperature (Giménez et al. 2020).

The larvae display behaviours at an early stage which favour export from estuarine habitats to coastal waters, from which they return at later developmental stages (Cohen et al. 2015). The early zoeal stages respond to gravity and pressure cues to maintain a surface position which favours offshore dispersal (Park et al. 2004). This behaviour and the long larval duration favours long-distance dispersal increasing the likelihood of spread to novel areas (Park et al. 2004).

The metamorphosis of *H. sanguineus* megalopae is cued by exudates from adult individuals and by marine biofilms associated with adult habitats (Epifanio 2013; Mygas et al. 2020). Molting can be stimulated by the presence of available hard substrates alone, but was shown to be significantly quicker in the presence of biofilms including those from non-adult habitats (Mygas et al. 2020).

The reproductive biology of *H. takanoi* has not been researched as extensively as that of *H. sanguineus*, but is assumed to be similar to that of *H. sanguineus* and *H. penicillatus* (van den Brink and Hutting 2017). Female crabs can produce several broods each summer. Reproductive maturity is reached when the crabs are still small (carapace width 6-7 mm), which can be reached within 1 year. The early reproductive maturity of this species is thought to aid in its ability to rapidly establish and spread in new areas (Gothland et al. 2014).

3.1.4 Invaded range distribution

Established populations of both *Hemigrapsus* spp. are found in European waters. The first record of *H. sanguineus* in Europe is from Le Havre harbour in Normandy in 1999 (Dauvin 2009), while that of *H. takanoi* is from 1994 in La Rochelle, France (Noél et al. 1997; Karlsson et al. 2019).

Hemigrapsus sanguineus

In the USA, *H. sanguineus* is found extensively along the eastern coast from Maine to North Carolina. It is thought to have been first introduced via ballast water to the Delaware estuary in New Jersey in 1998 where populations became established (Williams and McDermott 1990; McDermott 1991). Following its initial introduction it spread rapidly (estimated at a rate of 12 km per year) (Leppäkoski

and Olenin 2000), and is now the dominant crab species in rocky intertidal habitat in much of its invaded range in the USA (Epifanio 2013). In a study of *H. sanguineus* populations in Long Island Sound, densities of over 350 individuals per m^2 were recorded in 2000-2001 (Kraemer et al. 2007). Since then average intertidal density has decreased by 5% per year and the abundance of the largest females has fallen (Kraemer 2019).

H. sanguineus populations are well established on the French coast of the English Channel and their distribution stretches eastward over 1,500 km of coastline to Denmark (Dauvin and Dufossé 2011). Individuals have been reported from Guernsey and Jersey in the Channel Islands, with individual adult *H. sanguineus* males recorded from a number of sites in Great Britain, including from south Wales and Kent (Seeley et al. 2015). Where studies of population abundance have been conducted in France, regular increases in population abundance and density have been observed with the highest densities recorded as reaching over 50 individuals per m^2 (Dauvin and Dufossé 2011).

Hemigrapsus takanoi

H. takanoi has a more extended range than *H. sanguineus* in Europe with a patchy distribution stretching from the Atlantic coast of northern Spain to the Baltic Sea (Gothland et al. 2014; Geburzi et al. 2015). Populations have been studied on the Atlantic coast of France in the Bay of Biscay and in select locations on the French coast of the English Channel, with two large populations in Dunkirk harbour and Honfleur harbour (Dauvin and Dufossé 2011; Gothland et al. 2014). The species has been introduced to the Dutch Oosterschelde (van den Brink et al. 2012), the Wadden Sea, the Swedish west coast (Karlsson et al. 2019), and was first recorded in the Baltic Sea in the inner Kiel Fjord in July 2014, where it was thought to have already established a reproducing population (Geburzi et al. 2015). *H. takanoi* populations are frequently found within or near harbours, possibly both due to the suitability of the low hydrodynamic environments created by harbours, and potentially the high frequency of introduction events due to vessel movement (Gothland et al. 2014).

Two specimens of *H. takanoi* were found in Great Britain in an estuary in Essex in 2013 and in North Kent in 2014 (Wood et al. 2015), both of which are soft sediment estuarine environments. Further to this, additional sightings were recorded in North Kent, and retrospectively the first account of this species in the UK was discovered to have been from a citizen science Shoresearch survey in 2011 in Essex (Ashelby et al. 2017). An established large reproducing population was detected in the Orwell estuary in Suffolk during a development environmental baseline survey in (Ashelby et al. 2017).

The presence of a large population in the Orwell estuary in Suffolk demonstrates that the initial arrival of these crabs in an area may go undetected for some time. Ashelby et al. (2017) estimate that the population was likely introduced in 2014, two years before the reproducing population was detected in 2016.

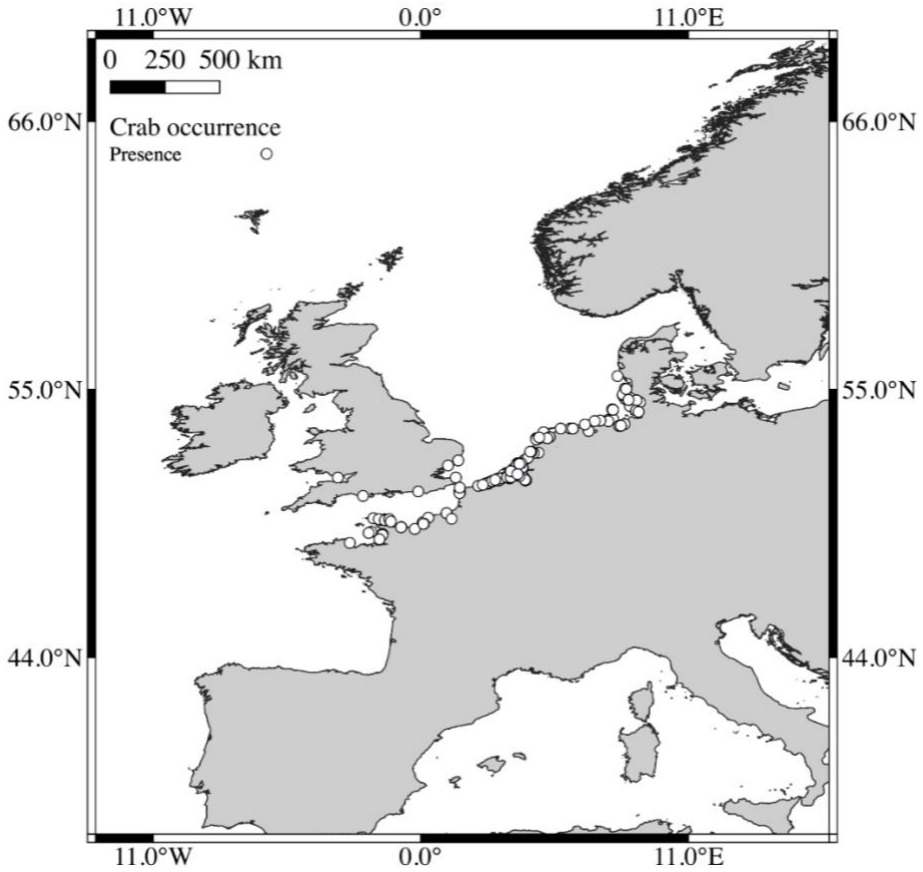

Figure 3.3 - The distribution of *Hemigrapsus sanguineus* in European waters. Presence is indicated by white circles, with records taken from GBIF.

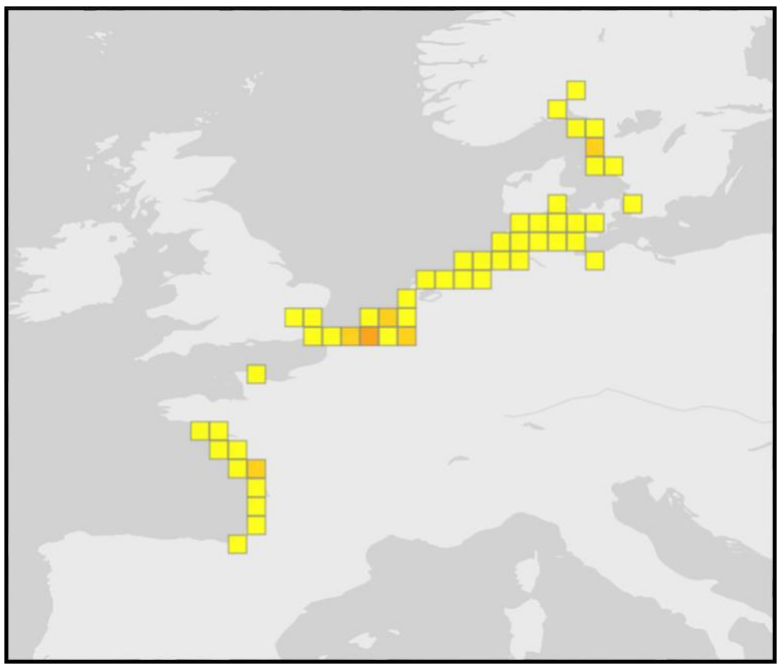

Figure 3.4 – The distribution of *H. takanoi* in European waters. Map generated by GBIF using records from GBIF database. Grid cells with greater densities of records are darker coloured. Checklist dataset https://doi.org/10.15468/39omei (Accessed via GBIF.org on 11th Aug 2021).

3.1.5 Route of introduction

Shipping is thought to be the primary introduction vector of *H. sanguineus* and *H. takanoi* to the USA and Europe, likely within ballast water (McDermott 1998). Genetic analysis suggests that invasive populations of *H. sanguineus* in the USA are likely to have originated in Japan, with multiple introductions via shipping likely (Blakeslee et al. 2017). However, transport of crabs within hull fouling communities is also possible as crabs are capable of clinging tightly to surfaces (Landschoff et al. 2013). For example, *H. takanoi* individuals were found in samples of hull fouling communities on a commercial ship in a German port in 1993 (Wood et al. 2015).

Transport of crabs with the movement of aquaculture stock is also thought to be a likely mechanism of secondary dispersal (Noél et al. 1997). Transport with the importation of Pacific oysters may be responsible for secondary introductions of *H. takanoi* within Europe, with this suggested as the mechanism for the introduction of *H. takanoi* to the Netherlands in the 1990s (van den Brink et al. 2012).

Following their introduction, *H. sanguineus* spread rapidly within the USA and in Europe. This can occur through secondary introductions, but also the natural dispersal of both adults and larvae. The capacity of *H. sanguineus* to produce several broods within an extended breeding season, and rapidly reach reproductive maturity increases the potential successful spread via larval dispersal (Park et al. 2005; Brousseau and McSweeney 2016). Adult *H. sanguineus* have also been show to be highly mobile and movement of adults may also contribute to population spread (Brousseau and McSweeney 2016).

The rapid spread of *H. sanguineus* along the coastline of Europe is attributed to dispersal of larvae by prevailing longshore currents, as well as secondary spread from other vectors (Landschoff et al. 2013). In Europe, genetic studies of *H. takanoi* populations show that genetic diversity across Europe is of similar levels with no evidence for genetic bottlenecks resulting from introduction events (Geburzi et al. 2020).

3.1.6 Method of sampling

Intertidal surveys for *Hemigrapsus* species typically involve using quadrats or total counts of specimens within an area of the intertidal or under a certain number of boulders to estimate abundance (Dauvin and Dufossé 2011). *H. sanguineus* are typically more inactive and shelter under rocks at low tide and are therefore easily sampled by capturing crabs underneath rocks at low tides (Rodriguez 2018). Crabs can be sexed, and the carapace width measured to gain information on population structure. Baited crab traps can also be used to collect individuals (Karlsson et al. 2019), and *H. takanoi* has been collected using subtidal survey methods including grabs, trawls, fyke nets and seine nets (Ashelby et al. 2017).

The tendency for *H. sanguineus* megalopae to cling to filamentous structures such as seagrass or fibrous material has led to researchers exploiting this to collect juveniles in the field (Mygas et al. 2020).

3.2 Impacts of *Hemigrapsus*

Globally, invasive crab species are some of the highest profile marine invasive species due to their impacts on recipient ecosystems. There have been numerous examples of introductions of brachyuran crabs, including the European shore crab *Carcinus maenas*, the blue crab *Callinectes sapidus*, and the Chinese mitten crab *Eriocheir sinensis* (Breen and Metaxas 2012). However, there has been less research into the impacts of the small *Hemigrapsus* species in their invaded range compared to the larger crab species (Epifanio 2013). Furthermore, most studies on the impacts of *H. sanguineus* have taken place on the east coast of the USA, with fewer studies in Europe. This distinction is important, as many of the reported impacts are on the European shore crab, which is also considered invasive in the USA.

On the east coast of the USA, *H. sanguineus* has become the dominant shore crab in a number of locations. Densities of *H. sanguineus* have been reported reaching over 150 crabs per m^2, and in one case reaching greater than 300 crabs per m2 (REF).

3.2.1 Ecological impacts of *Hemigrapsus* spp.

Predation on native species

H. sanguineus is an opportunistic omnivore, and in Europe, has been reported feeding on mussels, algae, detritus and a wide range of other invertebrate species (Jungblut et al. 2018). Invertebrate prey of *H. sanguineus* include amphipods, barnacles, bivalves, gastropods and polychaetes (Lohrer and Whitlatch 1997; Brousseau and Goldberg 2007).

Higher foraging activity has been observed at high tide when crabs most actively search for algae and invertebrate prey (Griffen et al. 2012, 2015). There may be differences in food preference of *H. sanguineus* at different developmental stages, with juvenile *H. sanguineus* observed feeding on algae (Griffen 2011, 2012, Tyrell and Harris 2000). Conclusions from laboratory tests may also not accurately reflect feeding behaviour in the wild, with algae making a more significant contribution to diet in the field (Griffen 2011, Karlsson 2019).

High densities of crabs could lead to declines in the abundance of prey species. Much of the research into the potential impacts of *H. sanguineus* is from areas in the USA where it became the dominant intertidal crab species.

A large proportion of the diet of *H. sanguineus* in invaded areas of the USA is comprised of small common mussels, *Mytilus edulis* (Lohrer and Whitlatch 2002). In caging experiments in Long Island Sound, crab predation affected mussel survival and was responsible for up to 25% of mussel mortality in the intertidal zone in the study area (Brousseau et al. 2014). In the same area, predation on barnacles by *H. sanguineus* was shown to not have a significant impact on the recruitment success of *Semibalanus balanoides* (Brousseau and Goldberg 2007). In fouling communities, caging experiments in Long Island, USA showed that *H. sanguineus* led to changes in community composition and favoured dominance of non-native ascidians such as *Botrylloides violaceus* and *Diplosoma listerianum* (Freeman et al. 2016).

Prey species may adapt to the presence of novel predators over time. Populations of *Mytilus edilus* showed an inducible defensive response to the presence of *H. sanguineus* in a region of the USA where the crab had been present for 15 years, whereas naïve populations did not (Freeman and Byers 2006).

Competition with other crabs (Carcinus maenas)

Competition between native and non-native Brachyuran crabs is thought to be a major component of the environmental impact of these species as IAS. The distribution of *H. sanguineus* and *H. takanoi* in Europe overlaps with the distribution of native European shore crab, Carcinus maenas, and both species are thought to be to be displacing *C. maenas* in some parts of their invaded range (Dauvin 2009).

H. sanguineus

The impact of *H. sanguineus* predation affects not only prey species but also native competitors. Evidence from the USA which shows that the prey species and prey size preferences of *Hemigrapsus* spp. overlap with *C. maenas* (Griffen et al. 2008, 2012) is supported by recent experimental studies from Europe (Bouwmeester et al. 2020). *Hemigrapsus* spp. were shown to have a prey-size preference range which overlapped with that of the native *C. maenas*, with all three crabs showing a similar positive relationship between size of the individual crab and preferred prey class (Bouwmeester et al. 2020). High densities of invasive crabs may exert pressure on prey populations of M. edulis.

In Long Island Sound, where *H. sanguineus* populations reached extremely high densities, low numbers of other intertidal crab species. Even following a reduction in the density of *H. sanguineus* over a 16 year period from 2001-2017, populations of other crab species remained very low (Kraemer 2019). Declines in native intertidal crabs with increasing H. sanguineus density have been reported from other locations on the east coast of the USA in long-term monitoring studies (Kraemer et al. 2007; Kraemer 2019).

In France, Dauvin et al. 2009 suggest there has been a change in the intertidal crab community in the areas with high densities of *H. sanguineus* (Dauvin 2009). Low *C. maenas* population numbers are observed in areas where *H. sanguineus* is most populous, while there has been no such decline in areas with low *H. sanguineus* abundances (Dauvin and Dufossé 2011). The breeding season of *H. sanguineus* is also significantly longer than that of *C. maenas*.

Research has suggested that *H.* sanguineus may exclude C. maenas from predator refuges under rocks, which could account for some of the decline (Jensen et al. 2002; Kraemer et al. 2007). More direct impacts may come from predation of *H. sanguineus* on juvenile *C. maenas* (Lohrer and Whitlatch 2002).

Competition experiments with *H. sanguineus* and *C. maenas* undertaken in Sweden showed that male *H. sanguineus* were effective competitors for mussel prey even when present at low densities, while females were passive and did not show high feeding rates (Karlsson et al. 2019). *H. sanguineus* dislodged *C. maenas* from prey, then used their legs and large chelae to push away other crabs and defend their position while feeding (Jensen et al. 2002; Karlsson et al. 2019)

H. takanoi

In the Netherlands, *H. takanoi* was found to outnumber *C. maenas* on hard, intertidal structures in the Eastern Sheldt (van den Brink et al. 2012). Adult *C. maenas* can grow significantly larger than introduced *H. takanoi*, competition may be occurring when juvenile *C. maenas* are the same size as adult *H. takanoi* (van den Brink and Hutting 2017). Competition may occur only during a specific period of a species' lifecycle as juvenile *C. maenas* crabs were shown to be worse competitors than similar sized *H. takanoi*.

Community change

H. sanguineus may serve as prey similar to other native crabs but there have been few studies which have investigated this. Dauvin & Dafusse (2011) suggest the colonisation of the high eulittoral zone by *H. sanguineus* may provide additional prey for bird and fish species, as the invertebrate biomass in this zone is typically low.

3.2.2 Socio-economic impacts of *Hemigrapsus* spp.

The socio-economic impacts of *H. sanguineus* are likely to be of minor or minimal concern, as there is no evidence for direct economic or socio-cultural impacts. The dominance of *H. sanguineus* in intertidal communities in some areas of the invaded range in the USA has not been shown to have significant economic impacts.

Predation of *H. sanguineus* on a variety on invertebrate species could cause problems if these species are commercially cultured or fished. The prey of *H. sanguineus* include mussels and oysters, generally the smaller juvenile shellfish and predation could reduce recruitment. Dauvin et al. (2009) suggest that predation by *H. sanguineus* on mussels and oysters in the intertidal zone could pose a risk to shellfish farm yields. Whether these species will impact shellfish production could be influenced by the intertidal distribution of the two crab species and the shellfish culture methods in use (Dauvin 2009).

High densities of *H. sanguineus* may affect the abundance of the common periwinkle *Littorina littorea*, which could affect periwinkle harvesting activities (Kraemer et al. 2007). However, the abundances of *H. sanguineus* observed in Long Island where this was studied (averaging 120 crabs/m^2) have not yet been observed in Europe.

The potential negative impacts of *H. sanguineus* on intertidal crab populations could affect local fishing of this species. In Ireland, *C. maenas* is used as bait in shore-based sea angling, with crabs in the process of moulting preferred.[7] It is unlikely declines in *C. maenas* will have a negative economic impact, but increased difficulties in finding appropriate bait could mean that anglers change to other bait sources, potentially including *H. sanguineus*. In the USA, fisherman have been reported occasionally using larger *H. sanguineus* individuals as bait (Kraemer 2019).

3.3 *Hemigrapsus sanguineus* in Ireland

The presence of *Hemigrapsus* spp. in Europe and in Britain means that the arrival of these species into Ireland is highly likely. The environmental conditions in Ireland appear to be suitable to support populations of both crab species, with similar habitats and conditions found in the introduced range in mainland Europe (Wood et al. 2015). While latitudinally, Ireland is further north than the native range of *H. sanguineus* in eastern Asia and the invaded range distribution in the USA, the milder environmental conditions in Europe support more northerly populations of *H. sanguineus* and *H. takanoi* (O'Donnell 2021). For example, *H. sanguineus* has already been recorded in Denmark and *H. takanoi* has been recorded in Sweden at nearly 60 °N.

Ecological niche modelling work carried out by Karlsson et al. (2019) indicated that suitable environment for both *Hemigrapsus* spp. is found on much of the eastern coast of Ireland, with smaller areas within bays and estuaries found on the western coast. This is supported by the results of our own species distribution modelling, which shows that large areas of the Irish coastline have a high

[7] https://fishinginireland.info/sea/sbaits/

probability of providing suitable habitat for *H. sanguineus* should the species arrive in the region (Figure 3.5). Similarly, the distribution of suitable habitat is more continuous on the east coast than on the west, where suitable habitats are found within bays.

While information on the ecology of both *H. sanguineus* and *H. takanoi* has been presented above due to similarities in their environmental impacts and management options, it should be remembered that the species differ slightly in their habitat requirements (see section 3.1.1 above) and the *H. sanguineus* species distribution model produced in this project should not be used to identify habitat which could support *H. takanoi*. The potential distribution maps produced by Karlsson et al. (2019) showed differences between the two species, with only *H. takanoi* predicted to spread into the Baltic Sea. Nevertheless, suitable environmental conditions for both species are shown on Irish coasts, with the broad scale maps showing similar patterns of habitat suitability for *H. sanguineus* and *H. takanoi* (Karlsson et al. 2019).

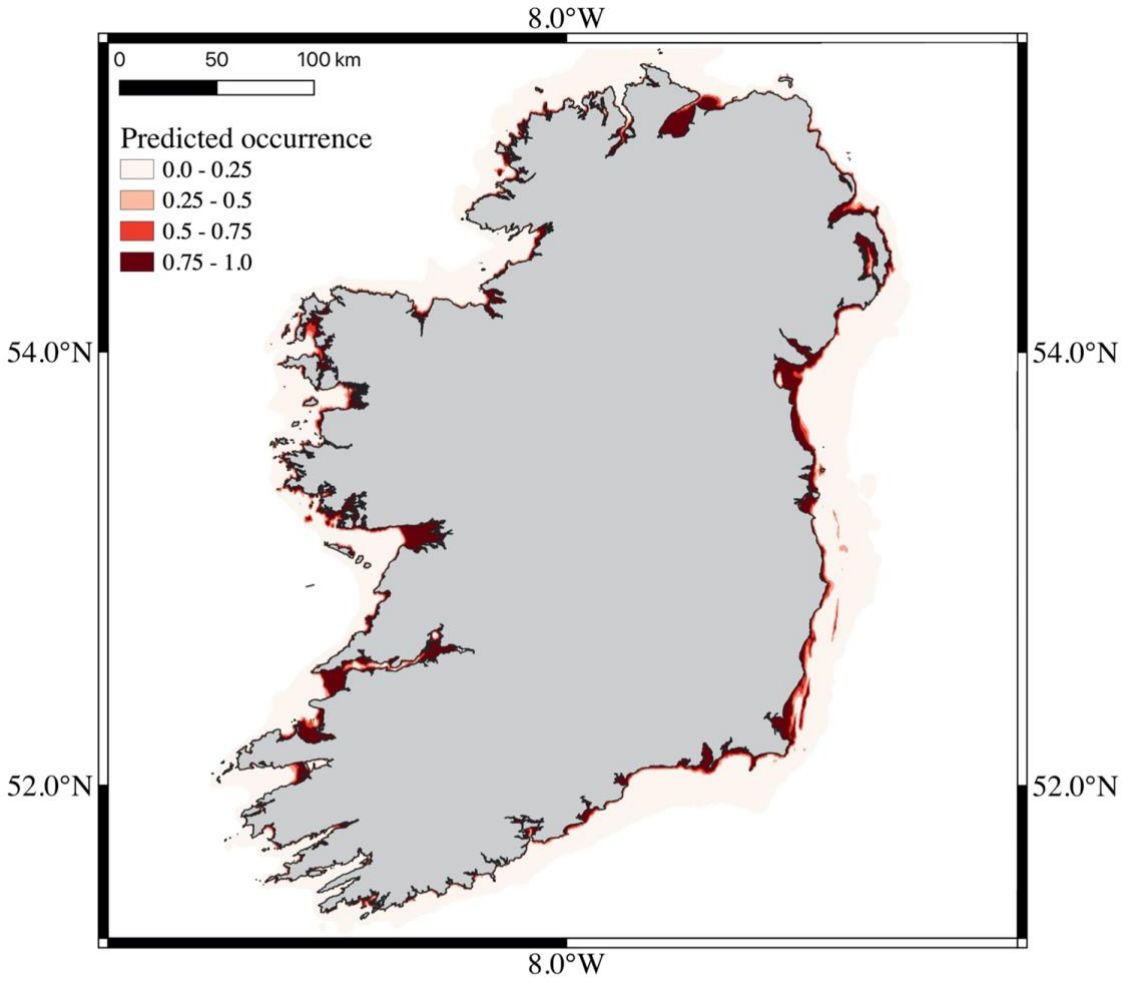

Figure 3.5 - Predicted occurrence of *Hemigrapsus sanguineus* in Irish waters. Predicted occurrence generated by species distribution modelling.

3.3.1 *H. sanguineus* spread

The behaviour of *H. sanguineus* larvae favour long-distance dispersal, increasing the potential ability of this species to spread and colonise novel regions. This larval behaviour could lead to rapid establishment and spread of *H. sanguineus* populations along the coast of Ireland following an introduction event. The maps of predicted settlement of *H. sanguineus* following release of larvae

from Galway harbour demonstrate the large area that could potentially be colonised within one spawning season (Figure 3.6). From the release site in Galway harbour, larvae were transported around inner Galway Bay, as well as along the northern coast of Galway Bay as far as the peninsulas of Connemara (Figure 3.6A).

The influence of larval behaviour is evident by contrasting larval tracking models using passive dispersal with those including *H. sanguineus* larval behaviour. Passive dispersal models predicted spread density much more localised to the release site, whereas when larval behaviour was included in the model there were two areas with the highest densities of larvae, and these were not in the immediate area of the release site. This shows the importance of including species specific behaviour in larval tracking models where it is available.

The spread of *Hemigrapsus* species may be facilitated by the connectivity between Irish, European and UK ports. Natural spread on prevailing currents may also allow spread from UK or French populations. The habitat suitability map for *H. sanguineus* demonstrates that much of the coastline could be susceptible to establishment (Figure 3.5). Combining this map with pathway hotspot analyses could identify priority regions to monitor for *H. sanguineus* arrival. The highest risk entry points associated with shipping and vessel activity are Dublin Port and Belfast Lough on the east coast of Ireland, and Cork Harbour on the south coast (Mirimin et al. 2019).

Surveillance and monitoring schemes should target high-risk areas for introduction. Larval tracking models can be used to identify areas near ports or harbours which should be included in regular surveillance surveys. If routine monitoring is not possible, these maps could be used to direct requests for reports of new species within local areas. Given the use of shore crabs as bait in sea angling, forums and newsletters which address this stakeholder group could be targeted to disseminate requests for *Hemigrapsus* sightings.

3.3.2 Detailed habitat information

H. sanguineus is usually found associated with hard substrates habitats, and is typically found under inter-tidal boulders (Landschoff et al. 2013). The species distribution model did not include information on substrate or recipient habitat as this was not available at suitable scales for the whole area. However, within the Galway SAC Complex more detailed spatial information on substrate and marine community type is available. Large areas of the intertidal zone are characterised by boulders supporting fucoid-dominated communities which give way to Laminaria-dominated communities in the subtidal zone.

The habitat most likely to be susceptible to establishment of *H. sanguineus* within Galway Bay is the fucoid-dominated community complex. This comprises intertidal boulder fields covered in fucoid seaweeds which could provide shelter for *H. sanguineus* individuals. This habitat is present along much of the intertidal coastline in Galway Bay and is predicted to receive the majority of *H. sanguineus* larval settlement in the intertidal zone (Figure 6.22). The model also did not include consideration of the response of crab larvae to settlement cues from adult *H. sanguineus* or biofilms. *H. sanguineus* may be more likely to settle in suitable hard substrate habitats as a result of cues from marine biofilms.

A mask can be applied to the larval density spread maps to only show areas where crab larvae are predicted to settle within suitable habitats, which may help improve the accuracy of model outputs. In Galway Bay, a hard-substrate mask was applied to the larval tracking model outputs (Figure 3.6). Where substrate and habitat data are available, this can be used to better predict the amount of habitat vulnerable to *H. sanguineus* establishment and to guide surveillance and monitoring plans.

There are two regions within Galway Bay predicted to have higher densities of larval settlement, Oranmore bay in the northeast, and a section of the northern coastline west of Galway harbour (Figure 3.6). If a surveillance plan to detect novel IAS in Galway Bay was developed, it should target these areas for repeat monitoring efforts for *H. sanguineus*. Locations in Connemara and the southern part of the bay should be noted as survey locations to include to identify range limits should *H. sanguineus* individuals be found.

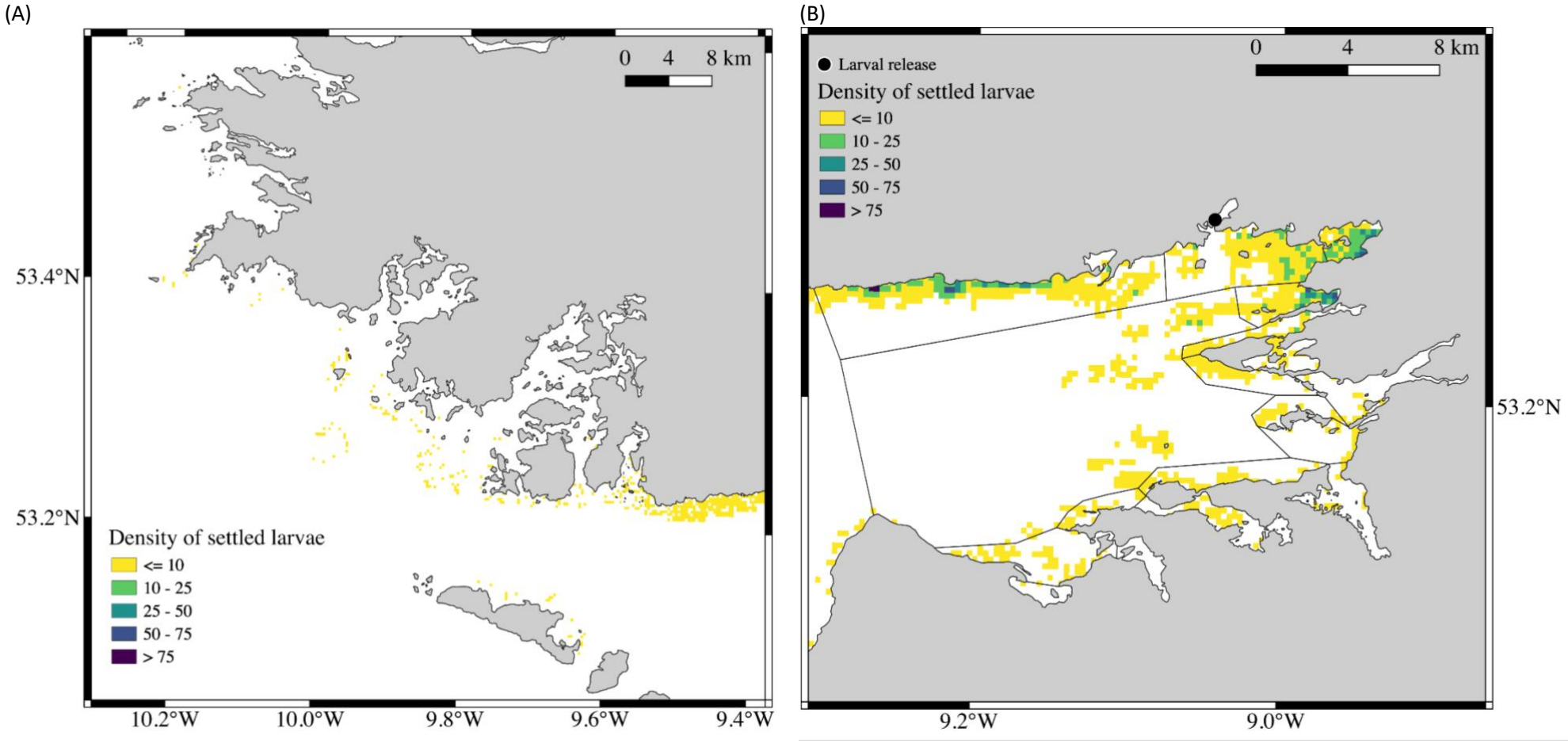

Figure 3.6 – Density of settled *H. sanguineus* larvae along range edges in Connemara in the west of Galway Bay (A) and within inner Galway Bay (B), showing predicted settlement within areas with hard substrate habitat only.

3.4 Management of *Hemigrapsus* spp.

The rapid dispersal ability of *H. sanguineus* and *H. takanoi* makes the likelihood of eradicating this species once populations are established extremely low. There are no species-specific control measures that have been investigated for *H. sanguineus* or *H. takanoi* (Ashelby et al. 2017), partly as they do not receive the same level of attention as more high profile invasive species. In a prioritisation exercise assessing eradication feasibility for horizon species in the UK, both *H. sanguineus* and *H. takanoi* scored 1, the lowest overall score possible (Booy et al. 2017).

As mobile crab species, strategies for eradicating *H. sanguineus* or *H. takanoi* following introduction are limited. If individuals are found and removed at the point of detection it may be possible to prevent population establishment, but it is unlikely that crabs will be detected when only solitary individuals exist (Geburzi et al. 2015; Ashelby et al. 2017).

In the USA, the invasive *C. maenas* was introduced decades prior to the establishment of *H. sanguineus*, and removal attempts were investigated for this species. Perhaps because of the lack of success with these eradication trials, despite the dominance of *H. sanguineus* in some intertidal areas of the eastern USA, there have been no removal experiments attempted in this area.

Given the low feasibility of eradication of *H. sanguineus*, preventing arrival is a priority. Ballast water management may reduce the risk of introduction from other regions; however, given the increasing spread of these species in mainland Europe there are a number of other vectors which are less regulated which could result in the introduction of *H. sanguineus* to Irish coasts. Hull fouling has been shown to be an important mechanism of secondary spread for many marine NIS and has been suggested as a vector for *Hemigrapsus* spp. translocations within Europe (Landschoff et al. 2013). Preventing movement of NIS via hull fouling on recreational vessels can be difficult as there is less regulation of this sector. It is also possible that natural spread from France and the UK may occur, with the timing of larval development allowing transport on westerly currents from the English Channel.

Declines in *C. maenas* in the Netherlands were shown to have predated the establishment of *H. takanoi*. It may be that reducing other pressures on coastal systems may increase biotic resistance to establishment.

Table 3.1 – Summary of manangement actions for *Hemigrapsus* spp. in Ireland

	Management term	Management objective	*Hemigrapsus* active management methods
Prevention	Pre-border pathway management	Reduce the uptake of the species and its transport outside the area of interest	Ballast water management Anti-fouling and hull cleaning Aquaculture import restrictions
	Interception	Intercept the species on first entry into the area of interest	Port/marina biosecurity
Captive management	Limits to keeping	Limit the keeping or cultivation of the species within the area of interest	n/a
	Secure keeping	Ensure the security of the species help in captivity/cultivation within the area of interest	n/a
Eradication	Eradication	Remove the entire population from the area of interest – with no immediate risk of re-introduction	Unlikely unless detected on arrival.
Long-term management / Control	Complete reproductive removal	Remove the entire population from the area of interest – but with remaining risk of re-invasion or further reproduction if not managed	Unlikely
	Containment	Limit the spread of a reproducing population within the area of interest (e.g. apply control measures at range edge)	Risk assessment for movement of vessels and aquaculture material outside invaded region.
	Suppression	Reduce the distribution or abundance of a population within the area of interest (e.g. vulnerable habitat or with socio-economic damage)	
Impact adaptation and ecosystem restoration		Actions taken to reduce associated impacts without or in conjunction with actions to reduce spread	Reduce pressure on native intertidal crabs
No management		When a species is widespread, and no action is taken to reduce its spread	

4 Managing marine IAS in Ireland

4.1 Use of models in supporting IAS management

Understanding how a newly detected NIS may spread following initial introduction will be crucial in determining what range of control measures may be needed. The approach taken to combining species distribution and particle tracking models with information of species behaviour has a number of applications to IAS management. The advantages of tracking dispersal within a relatively small-scales system is that it allows outputs to be matched to the scale at which management action and management decisions can be applied.

Model outputs are localised enough to be used to guide decision making on IAS management within the Galway Bay area, but the lessons learned from this modelling project could be used as a guide for other IAS and other systems.

4.1.1 Guiding risk-based monitoring (allowing for early detection)

Detecting new arrivals of IAS relies on dedicated surveillance and monitoring schemes, or on stakeholders and public who are aware enough of local fauna and flora to report new arrivals to the correct agency. Designing monitoring schemes to take advantage of the best use of financial and human resources is key, especially as there is generally little funding available to deal with IAS management. Monitoring for NIS is also difficult in that there is often a large area to be kept under surveillance, but NIS need to be detected early before their population densities are too high.

Model output can be used to determine which areas are most at risk of settlement, or where settlement overlaps with key features, allowing for resources to be targeted to areas with a high likelihood of species arrival. Combining particle tracking models with species distribution models allows areas to be identified where environmental conditions increase the likelihood of species establishment.

In New Zealand, in drawing up a surveillance plan to detect novel marine IAS in high-risk locations, Inglis et al. (2006) used hydrodynamic and particle dispersion models to simulate larval release from entry points, combined with Habitat Suitability Index models to identify where priority sites for sampling to detect a list of target IAS. Similarly, the model outputs generated for *M. gigas* and *H. sanguineus* can be used to predict where spread of larvae is most likely. Monitoring efforts can then be prioritised for these areas either to help track spread (in the case of *M. gigas*) or identify areas to monitoring for the arrival of a horizon species (*H. sanguineus*). It is important that monitoring also takes place in other areas, but the modelling outputs provide a basis for varying the distribution of effort.

4.1.2 Quantifying spread potential of target IAS

More targeted prediction of species spread could be used to direct rapid assessment surveys following suspected introduction of a novel species, allow for quick identification of suspected range limits and the most likely locations to check for further individuals. For instance, if spawning was observed from a particular wild population of *M. gigas*, the spread of larvae from a particular site could be used to direct targeted surveys.

The habitat distribution model element of the models can be used to identify the total potential area which could be at risk of invasion by a novel NIS. However, at a localised level this may not accurately match the observed distribution of a NIS. The range limits of IAS could be influenced by a number of biotic and biotic variables (Robins et al. 2017), and identifying which components of the receiving habitat influence establishment of a particular species would aid prediction of the extent of a species' spread (Kochmann et al. 2013). However, incorporating these factors into a model can be challenging as data may not always be available. For instance, native species presence may affect settlement of a NIS, as is thought to be the case for dense macroalgal cover on the settlement of *M. gigas* (Kochmann et al. 2013), but finding species data on a fine enough scale is challenging. Indeed, the data shown for Galway Bay was only available due to its designation as an SAC. Furthermore, the estimated spread of a species may increase or change as climate warming brings changes to marine environments, and incorporating this into future predictions adds another layer of complexity.

4.1.3 Guiding and aiding in the development of management action plans

As demonstrated by the *M. gigas* case-study, maps of larval density could be used to help plan and assess the need and effectiveness of management action. Model output could be used to answer management action risk assessment questions, such as how effective action might be in reducing spread, or how likely recolonisation from other sources would be. These models thus offer a way of analysing the effectiveness of management measures and allow for a management plan to be designed based on local information and evidence-based predictions.

The outputs also represent a tool to facilitate discussion with local experts and stakeholders, to demonstrate the need for management measures and allow for communication about specific geographic areas. Manipulating source sites allows for the impact of management action on spread to be analysed. In the case of *M. gigas*, this could involve predicting what would happen if management action was taken in one bay, or if a farm site that was previously using diploid stock changed to triploid.

The identification of sites where high densities of settlement overlap with vulnerable habitats or species could be used to guide impact-based management interventions or identify where other conservation impacts could address degradation to increase resistance to biological invasion.

4.1.4 Distinguishing features of this modelling approach

The fine scale resolution of the hydrodynamic models in Galway and Bantry Bay allow for spread to be assessed at a localised level. There are a number of ways which the outputs from this fine scale modelling can benefit management in comparison to other efforts at modelling *M. gigas* spread which have focused on larger areas at lower resolutions (Robins et al. 2017). Broad scale predictions are useful in determining range extents but the fine scale of the spread predictions from the model in this report allow for control and management measures at a scale which can be much more tangibly interpreted by local stakeholders, managers, and decision-makers. While national level information can be used to inform top-down decisions on the needs of different areas, localised data can allow for cost-effective decisions to be made by managers on the ground.

The detail of the hydrodynamic models used allows for important biological behaviour to be incorporated into the model. The tidal state has been shown to affect the behaviour of larvae, which affects how larvae are dispersed through a system. To incorporate this information requires a model which shows the hourly differences in tide and current, allowing for the influence of these changes on larval spread to be explored. Incorporating this biologically relevant detail can alter the conclusions

drawn from the models, as can be seen by contrasting the export and retention rates of larvae in the simulations run with passive vs. tidally forced larvae.

4.1.5 Model limitations and uncertainties

Model users and those interpreting the outputs as part of a decision-making process need to be aware of the model limitations and uncertainties. A general goal of dispersal modelling is to be able to include species-specific behaviour and characteristics to accurately track larvae within the spatiotemporal hydrodynamic limits of the model. Gaps in the understanding of larval ecology can lead to uncertainties in spread estimates. In this project, the main uncertainties arise from the larval release conditions and the expected larval behaviour.

Larval release conditions matter because the water circulation at the time and place of release can be highly variable, potentially creating various larval trajectories leading to a wide span of larval settlement. In this project, larvae were released from known or estimated sites of population presence. There were knowledge limitations about when to do the releases and which criteria would trigger the releases. If our modelling decisions were to cope with the general reproductive information of the species and the environmental conditions from the hydrodynamic modelling (i.e., estimates of water temperature, tidal cycle), it did not guarantee that modelled releases were made exactly when they would have occurred in nature or over the whole spawning period of the species. The uncertainties linked to the larval behaviour are consequences of the differences in the water current fields along the depth axis experienced by larvae due to vertical repositioning during the transport simulation. In the IAS project, the general behaviours of the species were relatively well known and mimicked in the spread model. However small details such as when a larva is going to modify its behaviour and what is the vertical extent of that change are less well known. These details can change the aggregation rate of larvae dispersing in the pelagic waters and knowledge may be considerably more limited for some other species.

Overall, modelling the larval dispersal of the focal marine IAS was carried out using the available species information, which gives the best dispersal estimations until more knowledge can be described and added. A management mindset that would acknowledge the limitation should first consider the areas with high larval density as they are likely to have the highest risks of invasion, depending on habitat suitability. Then, for other areas, a mindset should either be to wait for additional modelling efforts in order to reduce the modelling uncertainties or impute some caveats in the suggestions.

An additional relevant limitation for the interpretation of the simulations was the absence of larval mortality rates in our simulations. Mortality can seriously compromise the rates of larvae settling in a place and depends on various conditions such as water properties, larval conditions (e.g., feeding, disease), and predation. Mortality rates are not well known for different larval stages of individual species, but with regards to numerical transport the parameter has little to no influence on patterns of larval spread. If known, mortality can be applied on top of the raw modelled dispersal outputs while processing them. This approach has the advantages of being flexible and enabling tests of various scenarios of mortality. In the IAS project, we approached the issue of mortality by considering the water properties at the time of oyster larval settlement (i.e., the competence timeframe for settlement of 10 days for the oyster). The observed minimum temperature of 18 ºC is a threshold above which spat can survive settlement (Figure 6.6). Such information would inform us that among the larvae competent to settle, only 23.5% of them will survive through recruitment in the current conditions of Galway Bay. When projecting the change of sea water temperature due to climate

change, this rate is bound to change, increasing the pressure from the invasive species on the ecosystem (Figure 4.1).

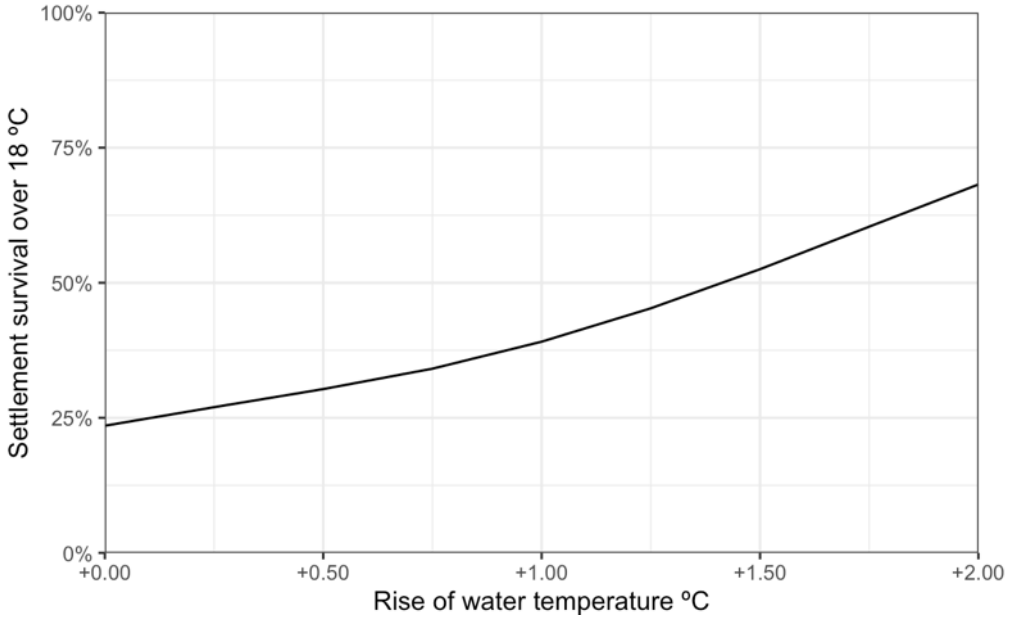

Figure 4.1 - Variation of the oyster settlement survival above 18°C with sea water temperature rising.

The hydrodynamic data also places limitations on the modelling. Most of the larval dispersal estimations (i.e., release day, trajectory, and intrinsic movement of the larvae) depend on the water currents, water pressure, and water properties extracted from the hydrodynamic model. The key limitations of the hydrodynamic data are of spatial and temporal extent. In the IAS project, we were limited to two regions for modelling the larval spread of the IAS because only these two regions had hydrodynamic data with spatial and time resolutions (~200 m and hourly) suitable for applying to the focal species ecology in the coastal water dynamics. Having limited geographical extent of the hydrodynamic data limits the detailed management advice to these two areas. Some general management advice for the whole coastline of Ireland can be inferred from these two regions, but the vast majority of the advice is specific to the region being modelled. In the IAS project, the interannual dispersal simulations were limited to three years because only data from these years had hydrodynamic outputs with a sufficiently fine temporal resolution (1 hour). Having more interannual simulations would have enabled a better estimate of the inter-annual variability in our predictions of larval spread. It would also have allowed the simulation of larval spread in extreme years, marked by strong weather changes. Yet, the three years that were simulated represented a good sample of the average larval spread because, based on the North Atlantic Oscillation (representing atmospheric temperature and precipitation patterns) and the Ocean Niño (representing extreme weather occurrences) indices, these years were having moderate and mild changes: 2018 being between the warmer 2019 and the colder 2020.

Beyond these limitations, modelling the larval spread facilitates the understanding of marine invasion patterns and is a method that can forecast the effect from management decisions with a few modifications of the model parametrisation. Models can be part of the decision tool component for IAS management and be set up to understand, predict, and prevent the different invasion phases. It is not advised to rely only on the modelling to take decisions, or to consider modelled predictions to

be infallible, but the use of models is practical for evaluating the risks under different case scenarios and good models generally represent the best available basis for those evaluations.

4.1.6 Future forecasting and modelling

The modelling approach used in this project has shown the value of localised models in guiding management decision on IAS. However, the availability of the fine-scale hydrodynamic models supporting this work is currently limited. There is a wide range of applications for these models, not limited to IAS management, but the development of models for areas at high risk of IAS introduction in Ireland would allow for more cost-effective monitoring and preventative action to be taken. This is explored further in the following sections.

The models used in this study considered spread over three years between 2018 – 2020. In predicting the impact of IAS in the future, further work could consider the impact of changes in climate under warming climate scenarios. Global warming is leading to increasing sea temperatures and a global shift in the distribution of marine organisms (Burrows et al. 2011). Changes in marine environmental conditions may affect the establishment rate of IAS, as temperature is an important driver in species distributions across a variety of scales.

This is certainly the case for *M. gigas*, with warmer summer temperatures in invaded areas linked to high recruitment of wild Pacific oysters. As sea temperatures increase, impacts of *M. gigas* may begin to be seen in more northern areas as oyster density increases. King et al. (2020) used a regional climate model projection to identify changes in bottom seawater temperatures which could allow recruitment of *M. gigas* under future climate conditions. Projected increases in ocean bottom temperatures varied along the Northwest European Shelf, with greater increases predicted in southern North Sea but less pronounced increases along the coastline of southwest England, Wales and Ireland (max 1.1C 2040-2059).

The study also showed that areas with self-recruiting *M. gigas* reefs along the coast of Europe and in Kent match areas where temperatures regularly exceed *M. gigas* settlement thresholds (King et al. 2020). In Ireland, the thresholds for extensive settlement are infrequently exceeded (less than 3 out of every 10 years), which may explain why the density of wild *M. gigas* remains low in most areas. However, by the period 2040-2059 these thresholds will be exceeded in the majority of years (more than 7 in 10), with some sites predicted to exceed thresholds in the majority of years within the next two decades. This could result in more extensive reef formation in Ireland along with the associated impacts presented earlier in this study.

4.2 Recommendations for managing invasive species in Ireland

There are multiple assessment frameworks and decision support tools which can be used to guide policy and management decisions in Ireland.

The potential ecological and economic damage caused by NIS has led to the development of international, regional and national policy aimed at reducing spread and managing the impacts of biological invasions. However, the implementation of this policy has been slow, especially for marine systems. A thorough overview of the international legislative framework for IAS policy and its relevance to the management of NIS in Ireland can be found in Mirimin et al. (2019).

Lucy et al. (2021) identify the need for cross-jurisdictional cooperation between Ireland and Northern Ireland in managing IAS. The joint Invasive Species Ireland project between the Northern Ireland Environment Agency and the National Parks and Wildlife Service ran until 2013 and the website served as a source of information on NIS across Ireland; however, the website is undergoing updates and will only refer to work in Northern Ireland from 2021.[8] In Ireland, the National Biodiversity Data Centre is now hosting a new dedicated website on invasive species.

IAS are directly identified within the Marine Strategy Framework Directive, which sets out a number of descriptors which are used to assess whether Good Environmental Status (GES) has been achieved in marine environments.

> *Descriptor 2: "Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystems".*

Ireland has adopted the following targets for NIS under the MSFD:

> *Target 1: Effect a reduction in the risk of introduction and spread of NNS through the prioritisation of species and improved management of high-risk pathways and vectors*

> *Target 2: The development of action plans for key high-risk marine non-indigenous species by 2020.*

Ireland's programme of measures for the MSFD include M077 "Maintain and expand as necessary a watch and alert system for non-indigenous species". The potential benefits of creating a national monitoring strategy for IAS are explored below.

4.2.1 Surveillance and monitoring

If a national survey and monitoring strategy for IAS was developed, a risk-based approach could be used modelled on the New Zealand Marine High Risk Site Surveillance (MHRSS) scheme (Box 3). The first step would be identifying hotspots of pathway activity, and much of this work has already been done for Ireland (Tidbury et al. 2016; Mirimin et al. 2019).

Currently, the largest ports in Ireland are Dublin Port, Cork Harbour and Belfast Lough, which are highly connected with UK ports (Mirimin et al. 2019). These sites represent the highest risk areas for NIS

[8] https://invasivespeciesni.co.uk (Accessed 29th Aug 2022)

introduction via ballast water and hull fouling pathways. There are also more marinas on the east coast, which pose a risk for NIS introduction via the movement of recreational vessels (Minchin 2007b). Areas most at risk from introduction via aquaculture activities were identified by Mirimin et al. (2019) as Mulroy Bay and Lough Swilly in the northwest, Waterford Harbour in the southeast, and Carlingford Lough on the east coast. Hotspots for the arrival of NIS via secondary dispersal by natural mechanisms can be harder to pinpoint. A broad-scale assessment of pathway density in the UK and Ireland by Tidbury et al. (2016), showed the south coast of Ireland as most at risk of natural spread from the UK.

This information can be used to prioritise future modelling work, as more detailed hydrodynamic models in these areas would allow the methodology developed for Galway Bay to be carried out in other areas identified as high risk for NIS arrival. Species specific larval tracking models in these areas for target horizon species (such as *H. sanguineus)* could identify sites for repeat monitoring surveys to enable rapid detection of new arrivals. A surveillance plan which combines information on pathways, species ecology and predicted spread and which targets priority species would enable survey effort to be most cost-effectively applied. Guidance on incorporating model results into monitoring can be taken from similar approaches taken in New Zealand (Inglis et al. 2006). Plans can be adapted as new techniques, information and evidence becomes available, or as priorities shift.

In carrying out surveillance and monitoring work, there are a number of established methodologies suitable for surveys in different habitats (Mirimin et al. 2019). The appropriate survey may vary depending on whether there are specific NIS being targeted or routine surveys of different environments. In the future, surveys using eDNA analysis are likely to play a larger role in addition to more traditional survey and taxonomic approaches. Citizen science and public sightings can also generate large volumes of data. Currently, records of public sightings of NIS can be submitted to the National Biodiversity Data Centre.[9]

4.2.2 Preventing arrival

Increasing activity in marine environments raises the risk of NIS introduction with more opportunities for species transport on commercial and recreational vessels. The increasing urbanisation of coastal habitats also provides hard substrate for colonisation by fouling NIS, which often outcompete native species in these environments. Risk of NIS introduction and establishment in areas of high maritime activity should be considered in the risk assessments for coastal and marine construction and development.

The inclusion of NIS in the planning policies in the National Marine Planning Framework could help with this, as the policies specify that risk of NIS introduction should be considered by new proposals:

> *"Reducing the risk of the introduction and/or spread of non-indigenous species is a requirement of all proposals. Proposals must demonstrate a risk management approach to prevent the introduction of and/or spread of non-indigenous species, particularly when:*
>
> - *Moving equipment, boats or livestock (for example fish or shellfish) from one water body to another,*

[9] https://records.biodiversityireland.ie/record/invasives#7/53.455/-8.016 (Accessed 29th Aug 2022)

- *Introducing structures suitable for settlement of non-indigenous species, or the spread of non-indigenous species known to exist in the area of the proposal."*

The model outputs from this report could provide useful evidence in helping assess the risk of introducing novel structures, ports, or aquaculture farms. Modelling spread from a proposed location could show if spread from a site would allow focal NIS to reach vulnerable habitats, or if the site could act as a stepping stone and connect uninvaded habitats with invaded areas (Floerl et al. 2009; Adams et al. 2014; Airoldi et al. 2015). If detailed modelling for all locations around the coastline is not possible, predictions on likely export and retention rates based on features of the coastline could be used as a proxy for broader risk assessment processes (Clavel-Henry et al, 2022).

The outputs from the NMPF process could also be used to help identify areas at risk from NIS establishment, for example sites with high numbers of recreational water users. Areas with high levels of marine activity have been identified as most at risk for NIS introduction, both due to the concentration of vectors and the frequent occurrence of high volumes of artificial structures in these areas which provide substrate for fouling NIS (Shucksmith and Shelmerdine 2015).

Biosecurity

Biosecurity measures need to be implemented by stakeholders. As such, IAS management relies on building trust and good working relationships with stakeholders. An EPA project reviewed some of the major components affecting the efficacy of IAS management in Ireland, including a review of current knowledge and attitudes of stakeholders regarding IAS (Lucy et al. 2021). The project highlighted that across environments few stakeholders are actively implementing or aware of biosecurity procedures (Lucy et al. 2021). Model outputs could be used to demonstrate the potential spread risks or identify high risk areas within ports that site managers should be aware of.

Those working in the marine environment are also best placed to notice novel species arrivals and understand the feasibility of management interventions, so enabling information flow between stakeholders, researchers and responsible authorities is critical. Bord Iascaigh Mara (BIM) has taken action to build relationships with aquaculture farmers to help detect the arrival and spread of novel invasive species.[10] While some of the work has been delayed by the pandemic, the trust between farmers and BIM will be an important feature of IAS management whether aquaculture operations are the vector of spread or the impacted sector (Davis et al. 2018).

4.2.3 Contingency planning

The importance of rapidly responding to novel NIS introductions requires a clear process for evaluating whether a response is required and who should be involved in authorising and carrying out management measures. A contingency plan, which identifies clear steps to be taken, the responsible authority for evaluating and deciding on a course of action, and funding channels, allows for management decisions to be made rapidly and with appropriate consultation of the relevant actors. The development of an overarching contingency plan for marine NIS rapid response for the whole of Ireland would allow for those within relevant departments and agencies to know to whom

[10] https://bim.ie/news-and-events/news/trial-begins-of-new-app-to-help-aquaculture-sector-in-ireland-respond-to-invasive-alien-species/ (Accessed 29th Aug 2022)

responsibility falls, while allowing for nuance such as different actors taking the lead depending on the environment or industry involved.

While an overarching contingency plan would outline a process for deciding on and carrying out rapid measures in response to new arrivals, in order to respond to novel species detailed knowledge of the likely impacts, spread, and control measures available for a specific species is needed. In order that this evidence is available in a timely manner, collating information for identified horizon IAS before their arrival is an important aspect of the rapid response contingency planning process.

The case-study provided in this report for *Hemigrapsus* species provides an example of what a species-specific appendix to a contingency report could include. However, an overarching framework supporting the consideration of this evidence will be crucial in allowing the best decisions on IAS management to be made.

4.2.4 Long-term management of established IAS

The results from the assessment of management options for *M. gigas* in Galway Bay could act as a first step in identifying eradication or control strategies and could be used to aid decisions on whether action is needed. This project was intended as an exemplar of the uses of localised particle tracking modelling in aiding management decisions, and before further management action is taken for *M. gigas* a species management plan should be prepared, which requires more thorough consideration of the practicalities of species management and more detailed knowledge of the local area. Input from local stakeholders, IAS experts, local authorities should be considered during this process.

Citizen science projects can be used to achieve long-term control of small populations of IAS within targeted areas but could run into problems with consistency of effort and monitoring of results. Nevertheless, the involvement of dedicated volunteers could allow for repeated action year on year without extensive funding. There are some existing citizen science initiatives in Ireland that are being used to tackle IAS. For example, in Kilmore Quay removal of *Sargassum muticum* by volunteers as part of a Coastwatch campaign has been carried out in the last two years (Dubsky and O'Doherty 2021). Removal efforts have been hindered by unclear policy on marine IAS removal, especially regarding responsibility and permitting of control actions (Dubsky and O'Doherty 2021). It is currently unclear whether removal of IAS on the foreshore by volunteer groups is legal, with responsibility lying with the Department of Housing, Local Government and Heritage.[11]

[11] https://www.irishtimes.com/news/environment/vital-seagrass-beds-being-overwhelmed-by-invasive-alien-seaweed-coastwatch-warns-1.4608204 (Accessed 29th Aug 2022)

5 Summary

The two case-studies in this report are intended as a demonstration of the applicability of combining larval transport and species distribution models with information on ecological and socio-economic impact to inform the management of IAS in Ireland. Larval transport and species distribution modelling allow for localised patterns of dispersal and establishment of IAS to be predicted based on current knowledge. The outputs from these models combined with information on the consequences of spread can be used to help guide management decisions in a variety of ways at all stages of the invasion process (prior to invasion, early in the invasion, during continued spread and looking to the future and the likely effect of climate change upon spread risk).

The Pacific oyster case-study demonstrates the usefulness of localised spread information in informing monitoring plans, impact assessments, and decisions on control measures. Eradication of wild Pacific oyster is unlikely given the chance of continued reintroduction from aquaculture, but the management of populations to reduce long-term spread and impacts should be considered further, especially where populations are established in or near protected areas or near popular recreational areas, particularly given the predicted increase in risk of spread due to climate change.

The outputs from the Pacific oyster modelling could be used to inform monitoring and surveillance, allowing for the creation of a more cost-effective risk-based strategy. Areas predicted to have a higher density of larvae or to be at the frontiers of spread could be selected for more frequent or intensive surveying depending on the aims of the monitoring strategy. The model output can also be used to show where areas of high Pacific oyster settlement may overlap with vulnerable habitats or artificial structures which might provide hard substrate for settlement. Identifying which source populations contribute to settlement in these areas allows for management areas to be more effectively targeted to areas which might cause environmental or economic impacts.

The recommendations for management measures drawn from the use of these models are contingent on users understanding how data uncertainty and model choices affect the results and outputs. Understanding the dispersal potential of larvae from wild populations is reliant on up-to-date information on the extent and status of wild Pacific oysters in Galway Bay. However, a survey for wild *M. gigas* could also offer opportunities for verifying predictions from the models.

The use of larval transport models to inform monitoring and surveillance also applies to horizon species that are yet to establish populations in Ireland. Furthermore, species distribution models can be used to show whether a species is likely to establish and to map the extent of suitable habitat at risk of colonisation. This can allow for evidence-based decisions to be made about risk to vulnerable ecosystems or industries before a species has established. As demonstrated by the *Hemigrapsus* maps, extensive suitable habitat may be present for this species, with large areas of intertidal habitat vulnerable to establishment.

The combined modelling approach allows climate change predictions (e.g. changes to the marine, terrestrial and atmospheric environment) to be translated into likely changes in species spread risk. This is possible because larval transport models predict the relevant parts of the marine environment experienced by dispersing larvae, whilst species distribution models predict the relevant environmental variables that are associated with a species' population establishment. This information can be used to confer some future resilience to policy and management decisions.

This report's modelling approach is effective in providing evidence for decision making in part because of the high quality hydrodynamic information available in Galway Bay and Bantry Bay. Targeted development (e.g. around major ports in Ireland) of high quality hydrodynamic information will enable a broader evidence-base to be developed that can support IAS policy and management decisions across Ireland. The approach in this report has shown that combining larval transport modelling and species distribution modelling provides effective tools to support a wide range of policy and management decisions at multiple stages of a species' spread. The software and GIS tools accompanying this report can be used as a foundation to develop this approach for other species and in other geographical locations.

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A Appendix – Case-study supporting material

A.1 *Magallana gigas* case-study

A.1.1 Model outputs for *M. gigas*

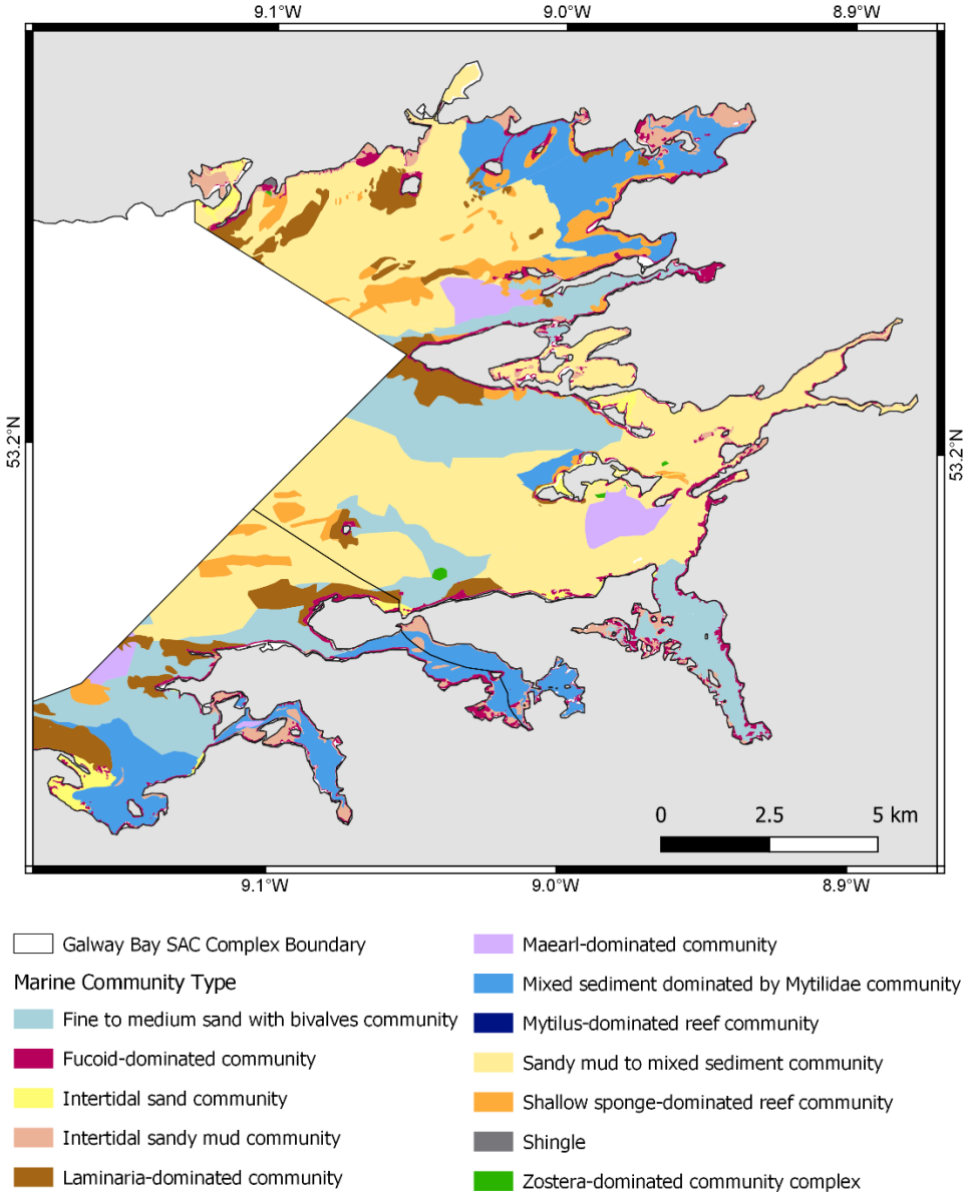


Figure 6.1 Galway Bay SAC Complex showing the boundary of the SAC area and the distribution of marine community types within the SAC. Data on Marine Community Types is from NPWS and available on the Marine Atlas.

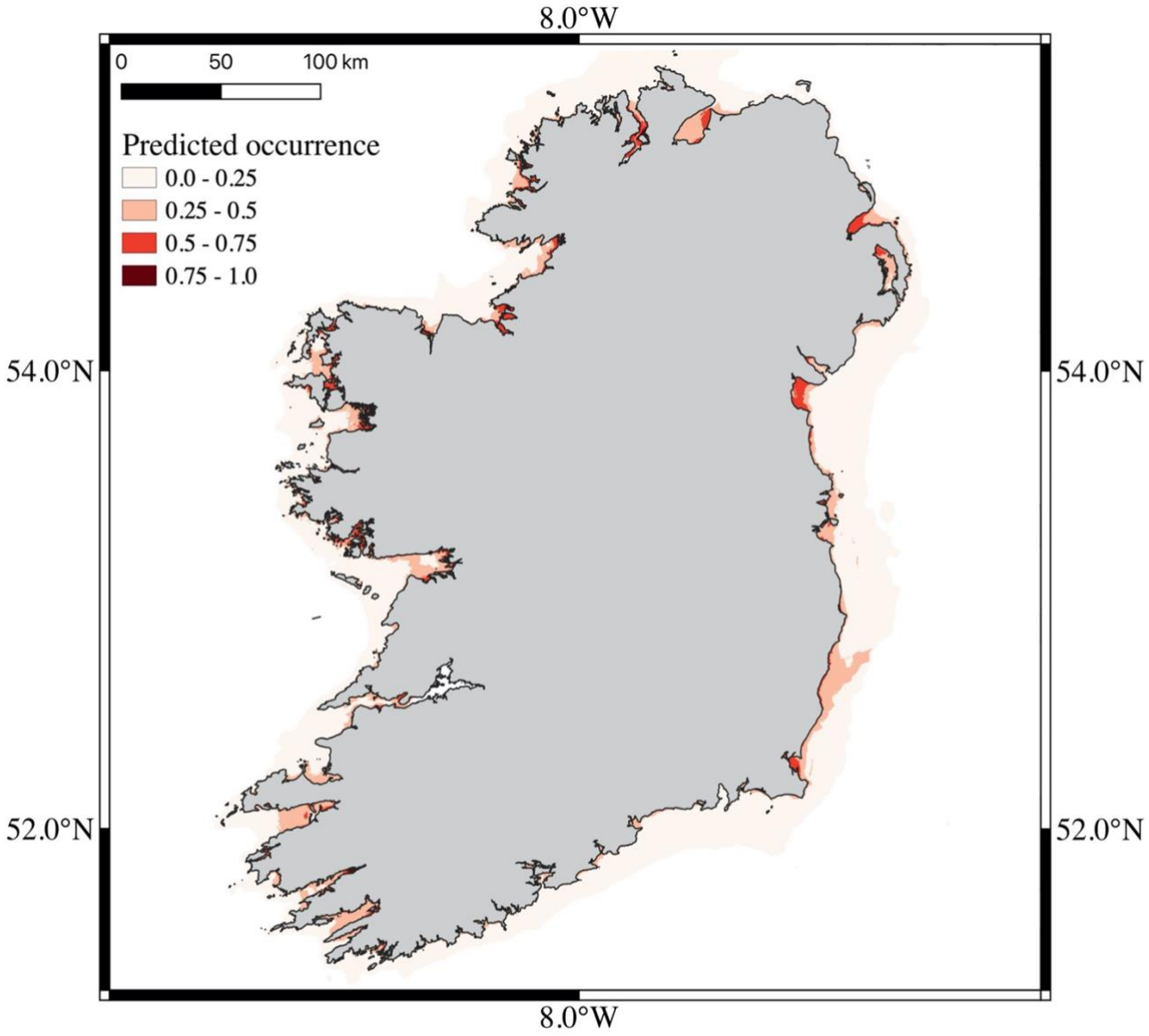


Figure 6.2 – Predicted occurrence of *Magallana gigas* in Irish waters from species distribution modelling results.

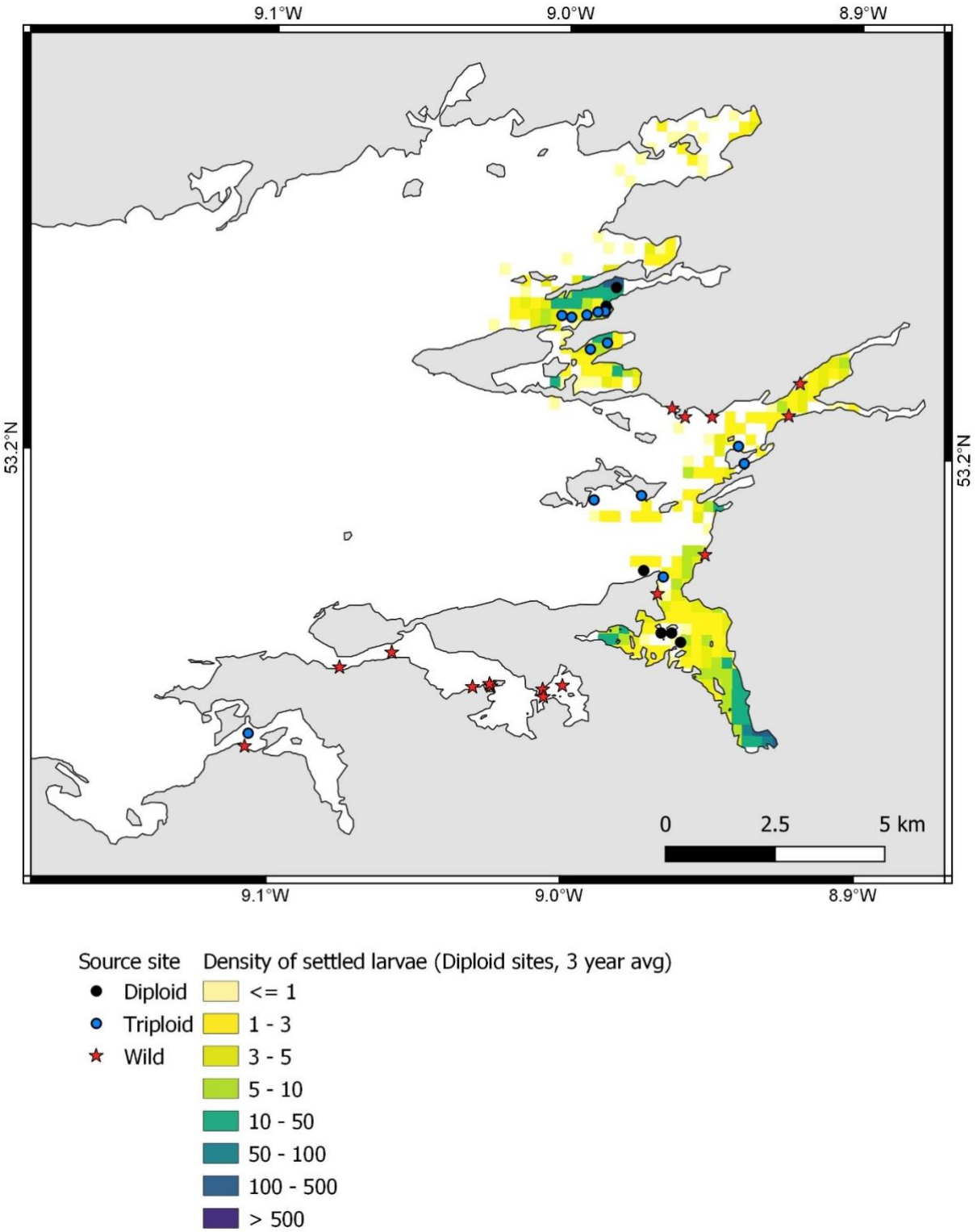


Figure 6.3 - Predicted density of *M. gigas* settled larvae in Galway Bay from diploid farm sources.

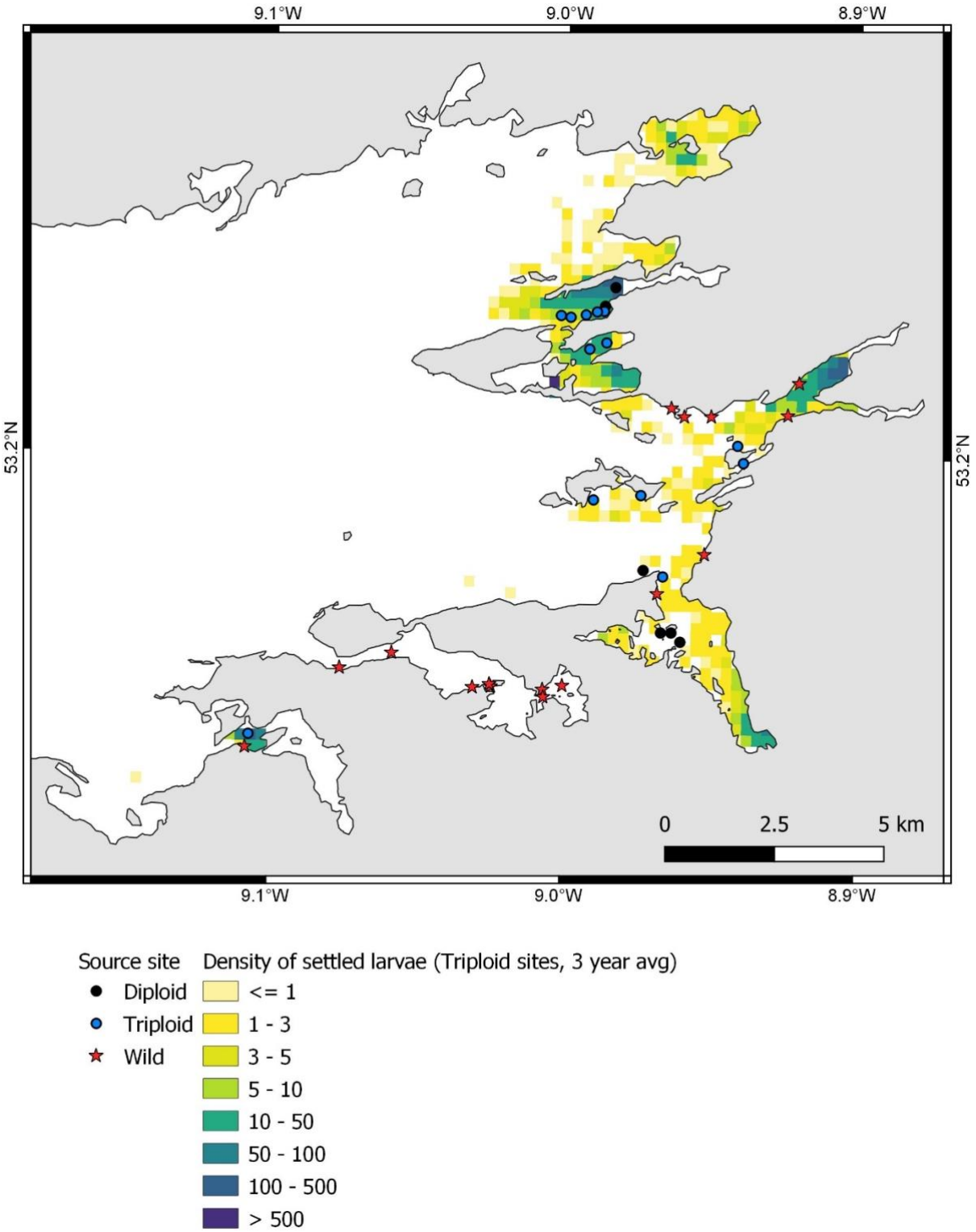


Figure 6.4 – Predicted density of *M. gigas* settled larvae in Galway Bay from triploid farm sources.

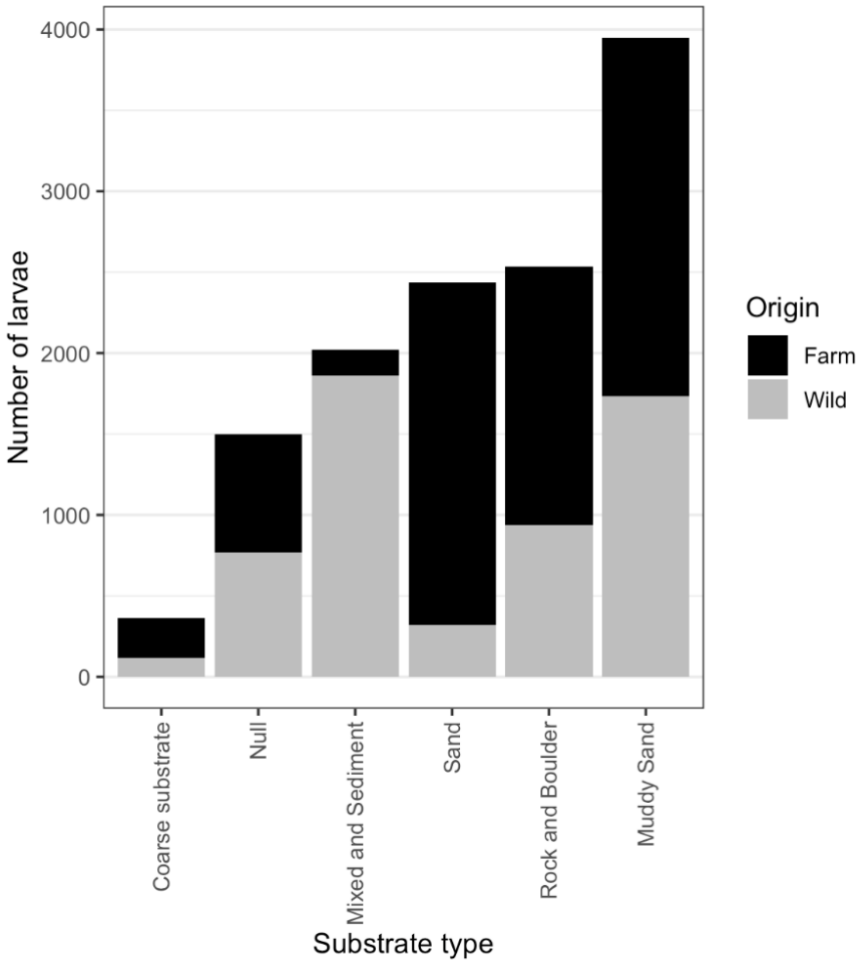


Figure 6.5 - Yearly average numbers of settled larvae over substrate type. "Null" corresponds to areas with no substrate information and areas where substrate type is unclear due to overlaps.

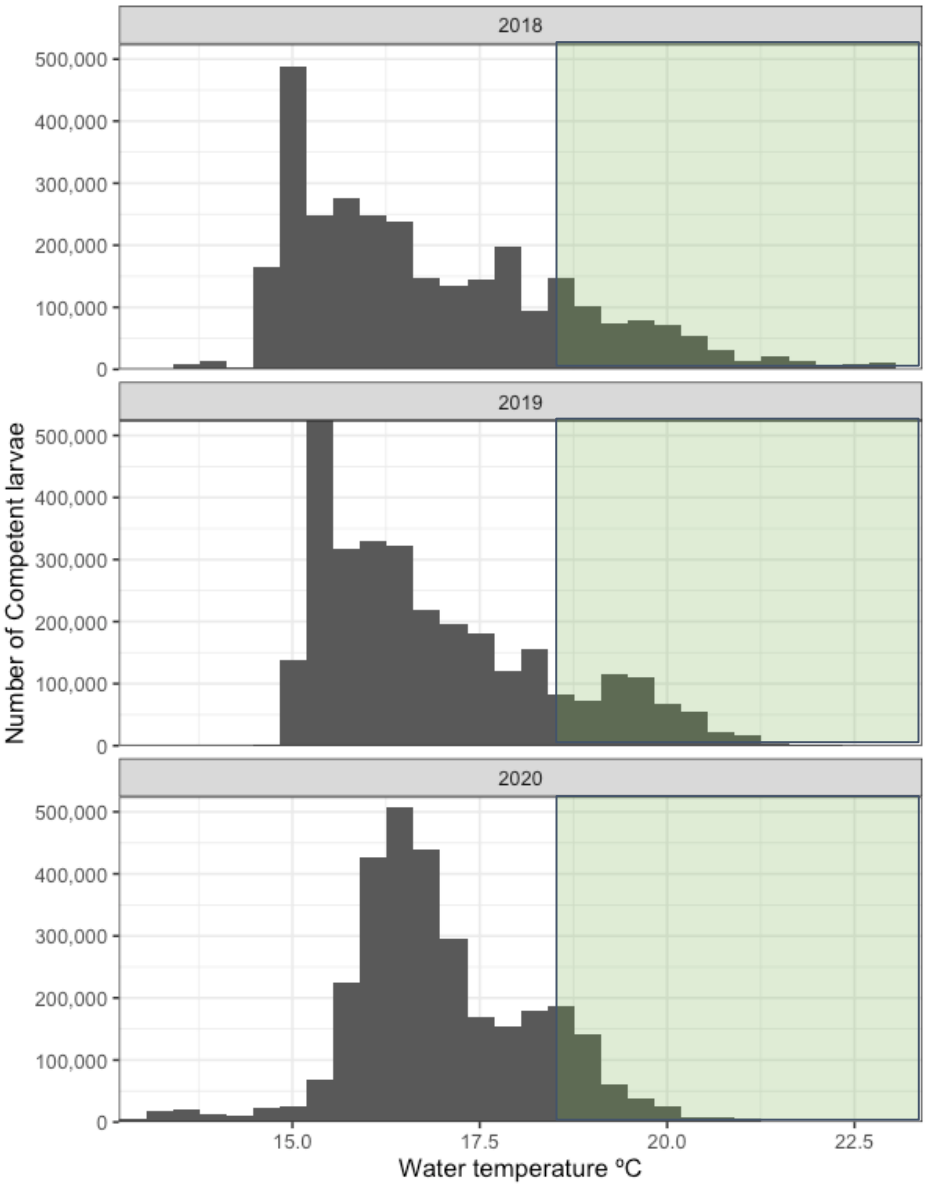


Figure 6.6 - Density of competent larvae according to the bottom water temperature. In green, the water temperature range suitable for spat survival.

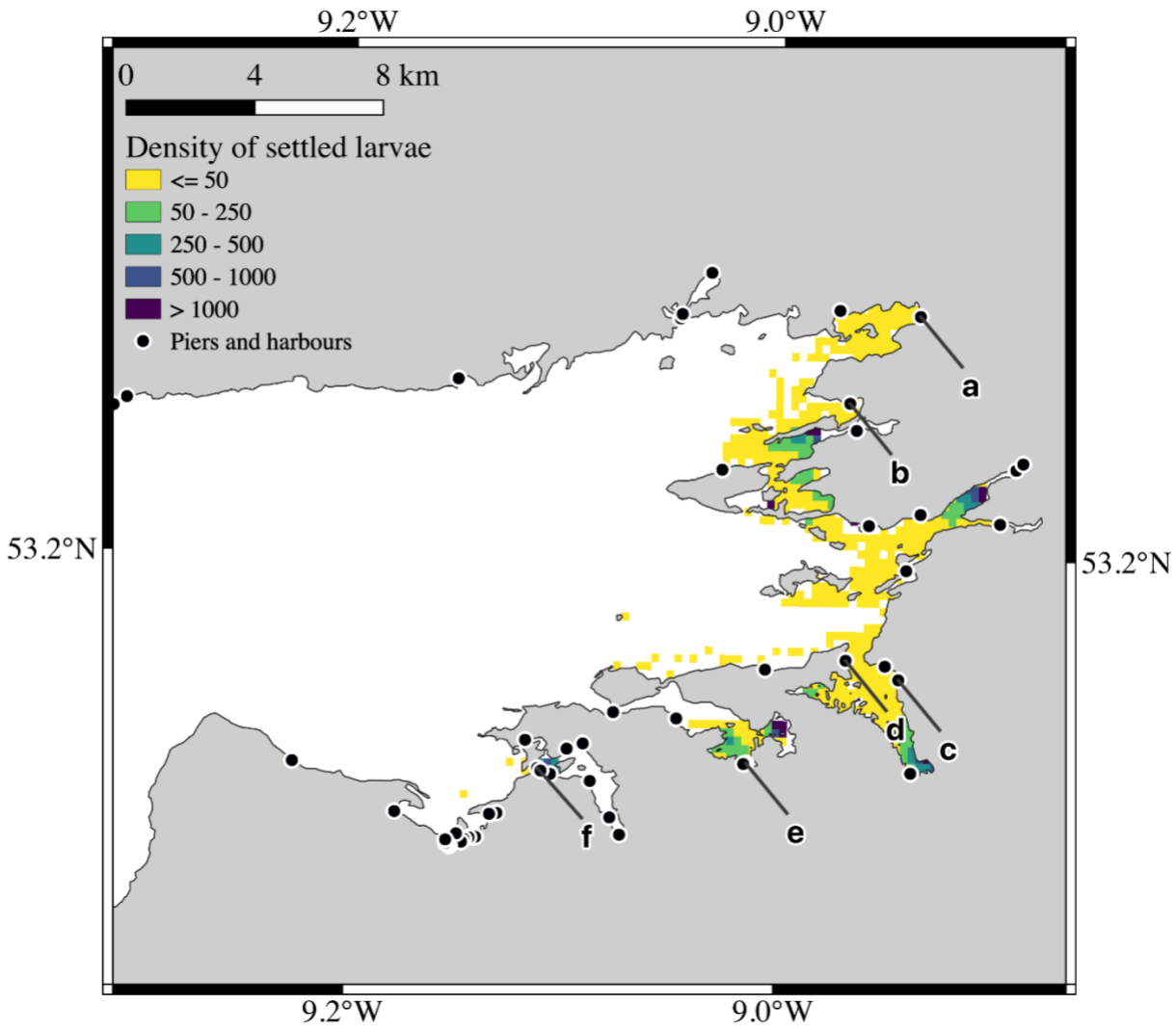


Figure 6.7 - Piers and harbour location in Galway Bay relatively to the density of settled larvae. Sites which overlap with predicted larval settlement are identified by letters, although there are a number of structures in close proximity to high density areas, but the GPS coordinates place them above chart datum. Labelled sites are a) Oranmore, b) Renville, c) Tarrea, d) Parkmore, e) Corranroo, f) Muckinish West (Landing place) and g) Muckinish East (Piers).

Ballyvaughan

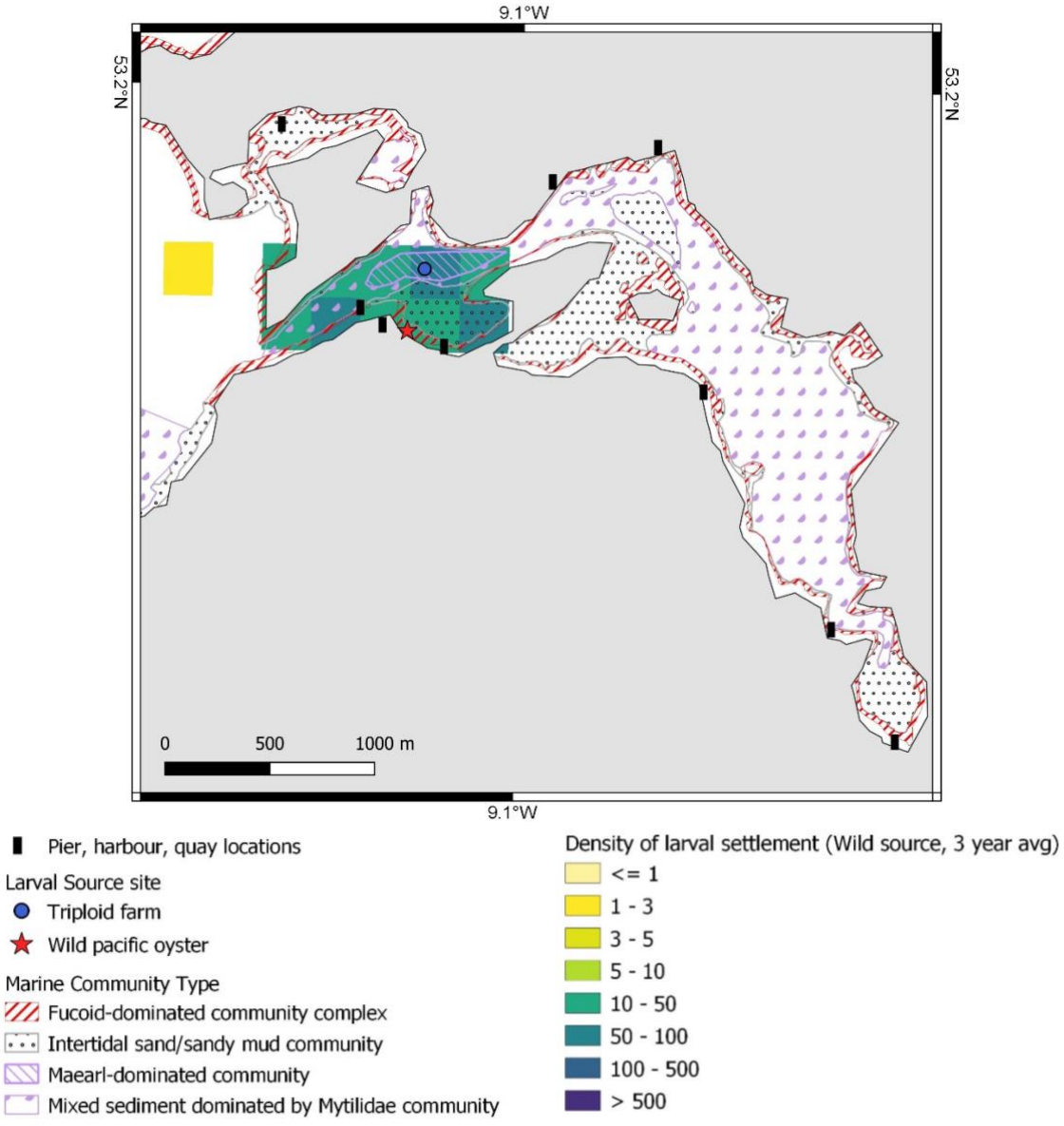
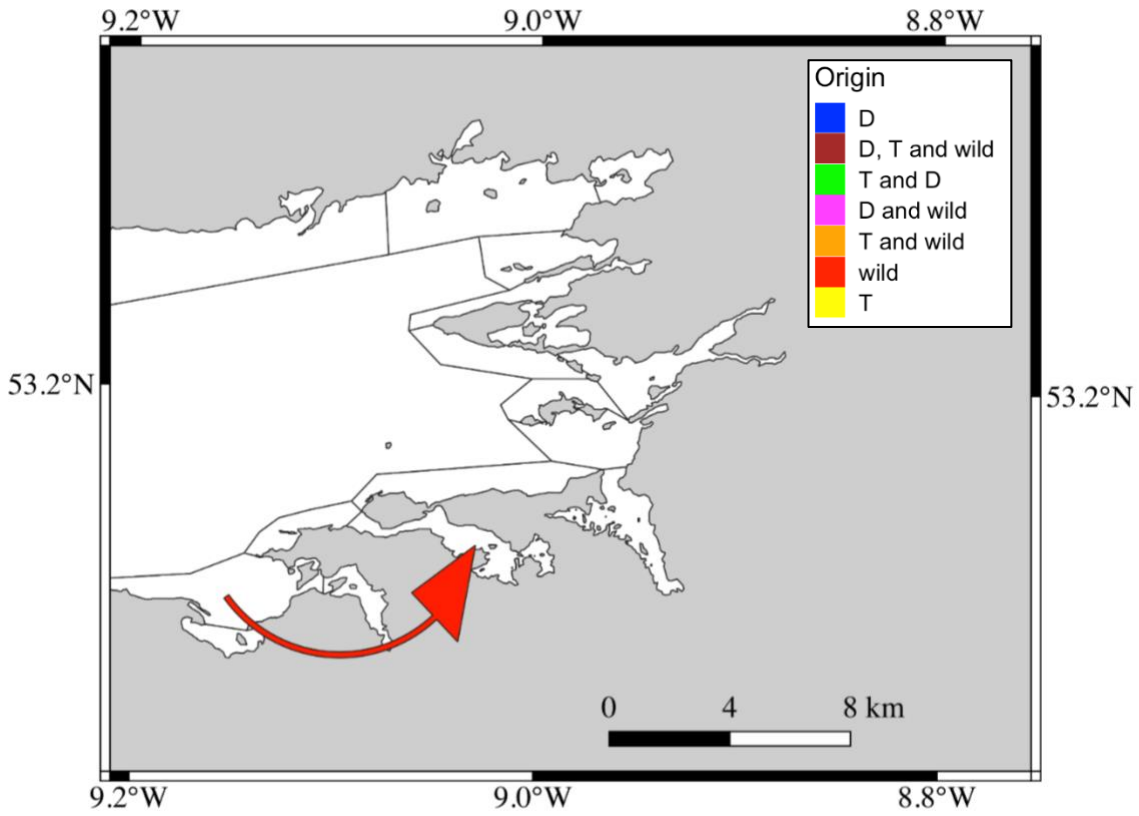


Figure 6.8 - The density of larval spread in Ballyvaughan bay, only from wild source sites. Marine community type is shown based on data from NPWS.



Sink site	Source site type			
	Diploid (%)	Triploid (%)	Wild (%)	Total (N)
Ballyvelaghan			100%	2
Ballyvaughan		50%	50%	783

Figure 6.9 - Source sink connections for *M. gigas* larvae in Ballyvaughan bay. There are no external sources of larvae into Ballvaughan, with high retention of larvae from the documented wild individuals.

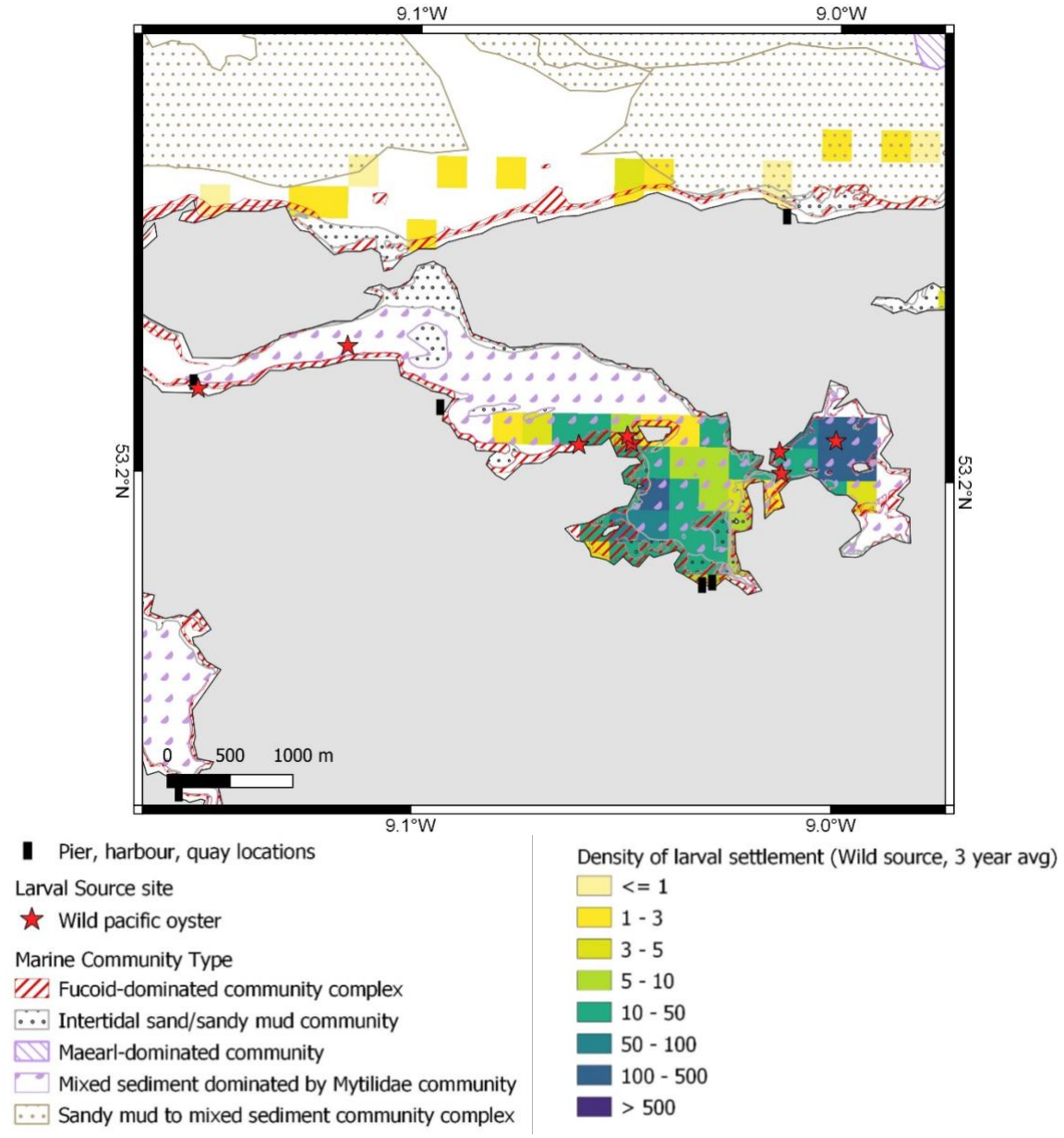
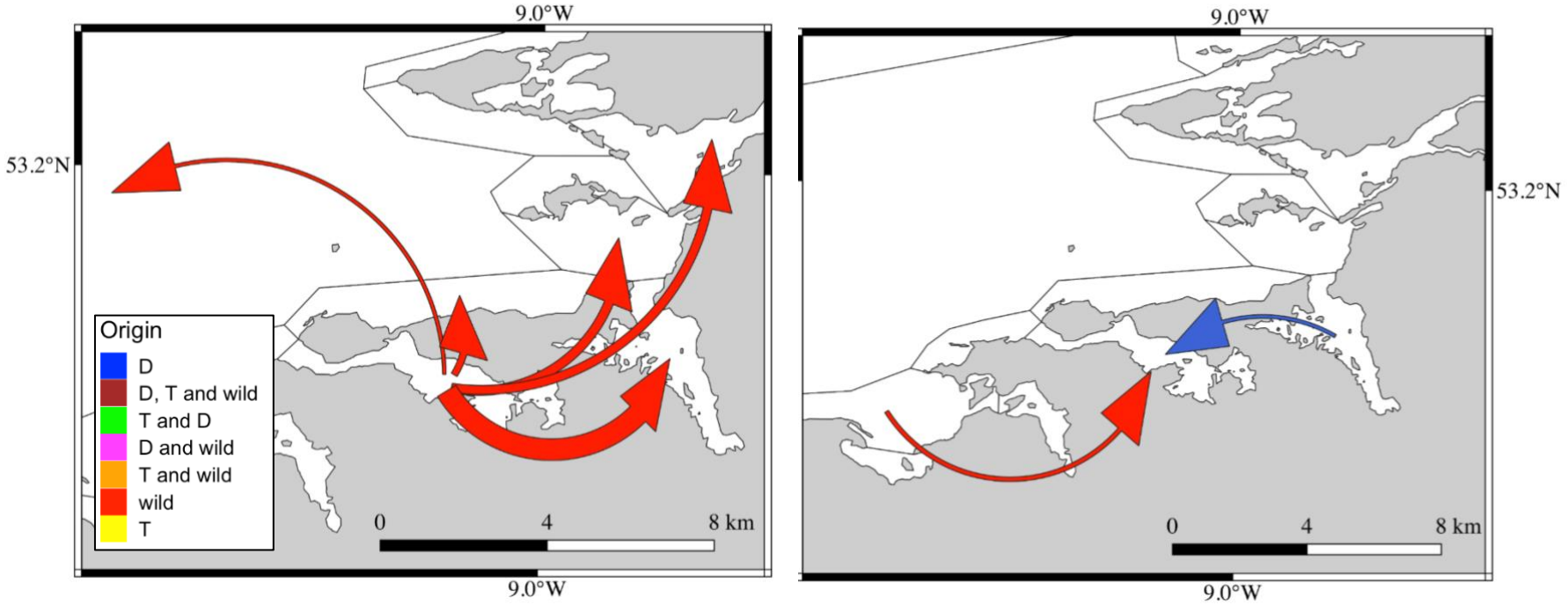


Figure 6.10 – Localised map of the Ballyvelaghan sub-bay area, showing the density of larval settlement predicted from wild sources and the locations of presence records for wild *M. gigas*.

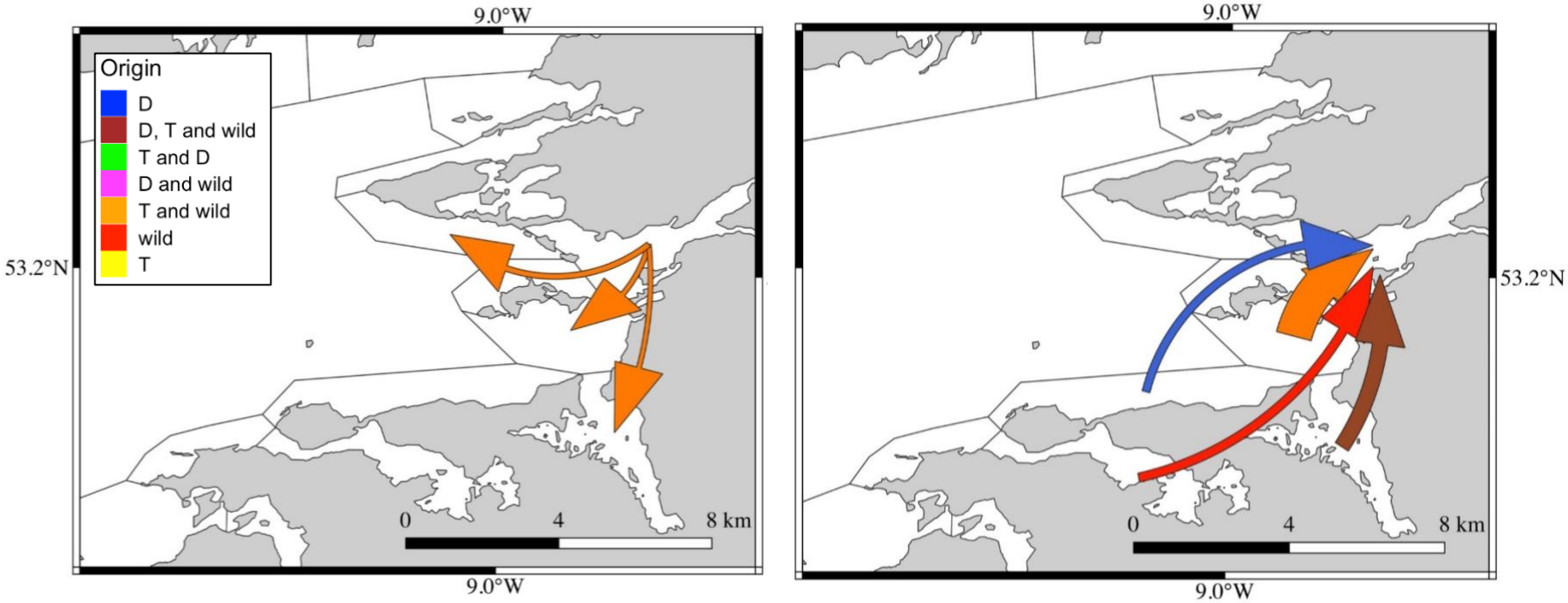
Ballyvelaghan



Sink site	Source site type				Source site	Source site type			
	Diploid (%)	Triploid (%)	Wild (%)	Total (N)		Diploid (%)	Triploid (%)	Wild (%)	Total (N)
Rosshill			100	12	Ballyvaughan			100%	2
Kinvarra			100	231	Kinvarra	100			1
Ballyvelaghan			100	2320	Ballyvelaghan			100	2320
Ballinderreen			100	32					
Galway Center			100	1					
Kilcolgan			100	28.3					

Figure 6.11 - Source sink connections for *M. gigas* larvae in Ballyvelaghan bay.

Kilcolgan



Sink site	Source site type				Source site	Source site type			
	Diploid (%)	Triploid (%)	Wild (%)	Total (N)		Diploid (%)	Triploid (%)	Wild (%)	Total (N)
Tawin S		13%	83%	7	Kinvarra	50%	18%	32%	113
Kinvarra		34%	66%	3	Ballindereen		81%	19%	756.3
Kilcolgan		50%	50%	2344	Ballyvelaghan			100%	28.3
Ballinderreen		50%	50%	4	Kilcolgan		50%	50%	2344
					Rosshill	100%			20

Figure 6.12 - Source sink connections for *M. gigas* larvae in Kilcolgan bay.

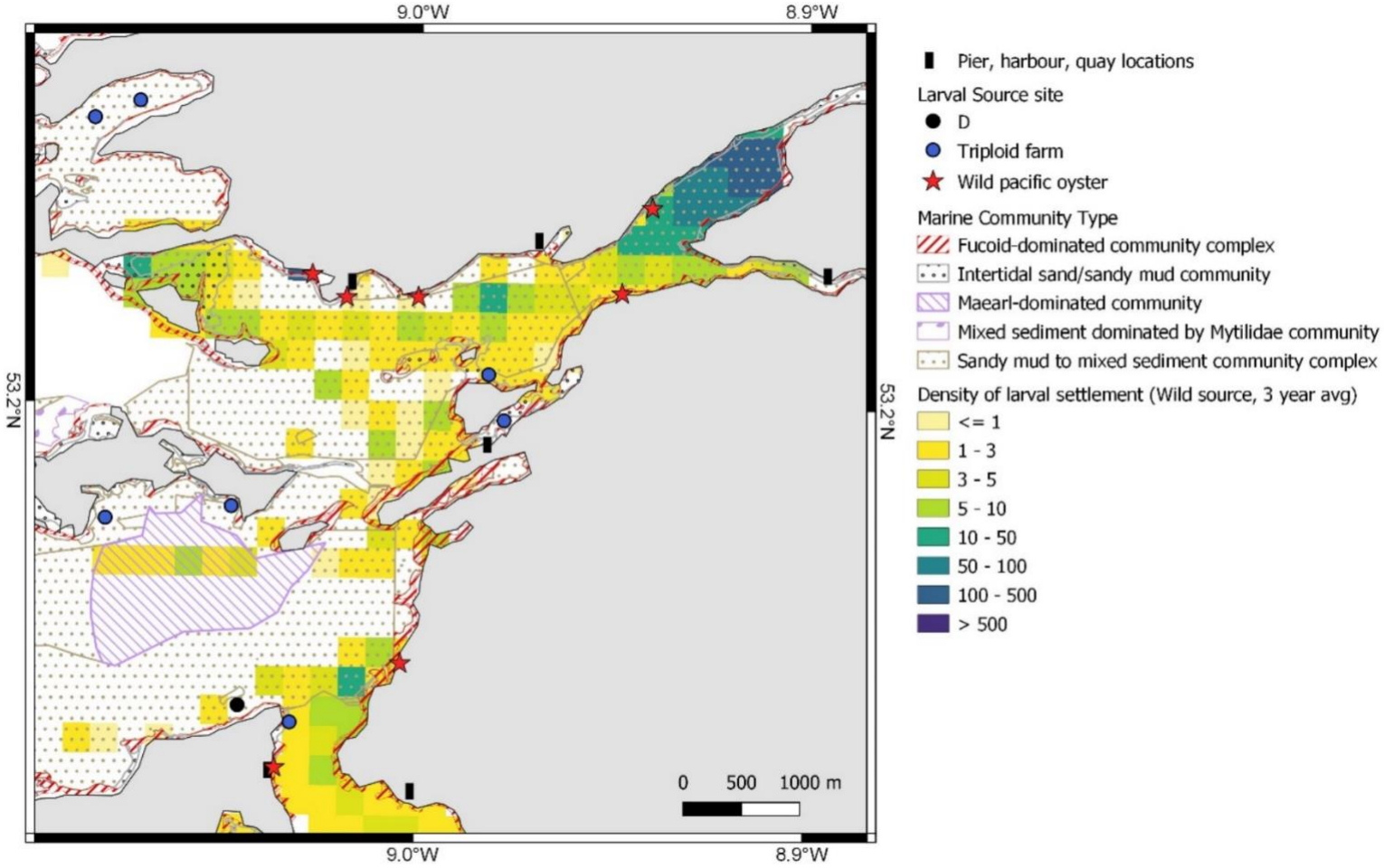
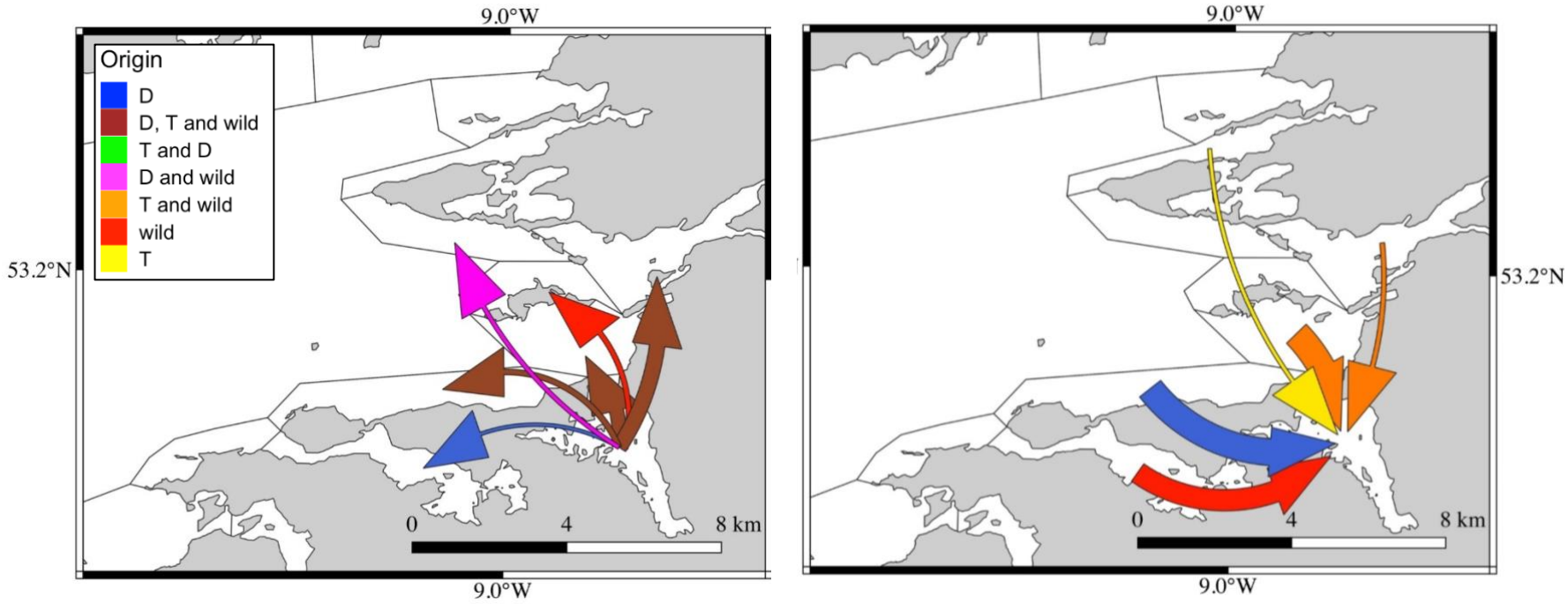


Figure 6.13 – Detailed view of predicted density of settled larvae in Kilcolgan Bay from wild sources averaged over the three modeled years.

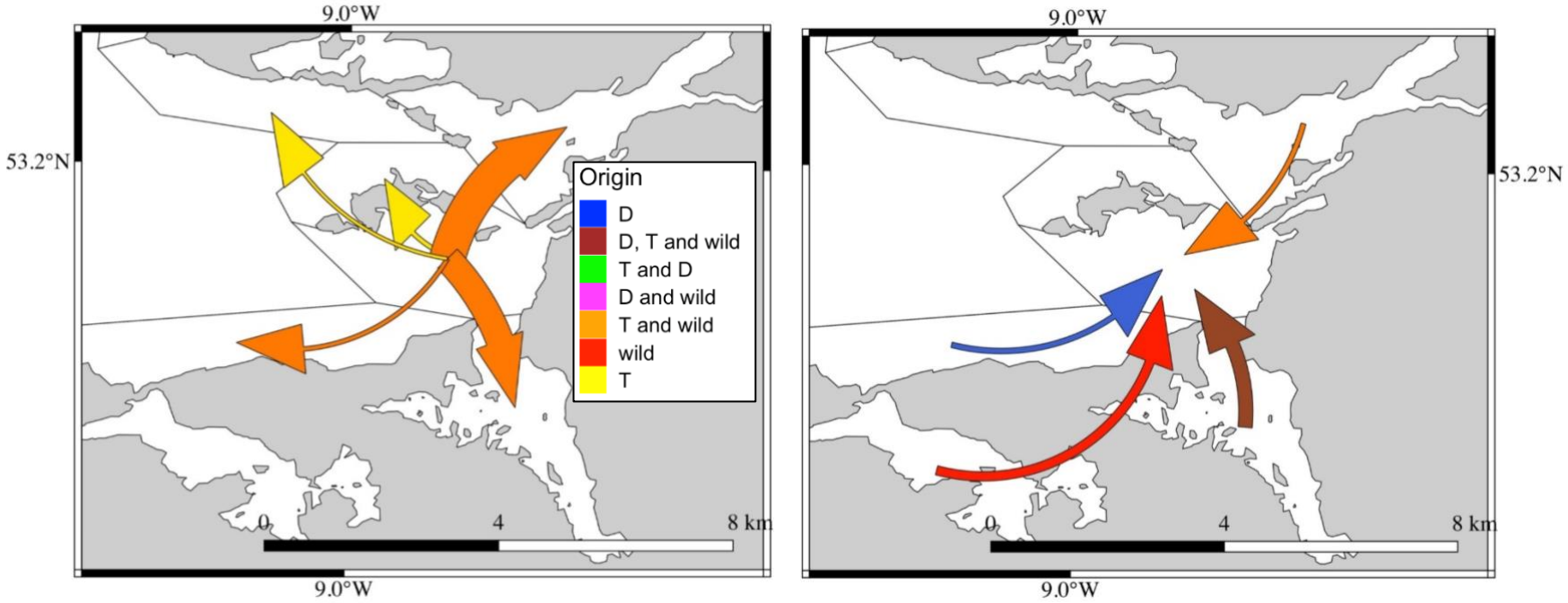
Kinvarra



Sink site	Source site type				Source site	Source site type			
	Diploid (%)	Triploid (%)	Wild (%)	Total (N)		Diploid (%)	Triploid (%)	Wild (%)	Total (N)
Kinvarra	50%	16%	38%	113	Kinvarra	50%	16%	38%	113
Tawin S	50%		50%	2	Tawin N		100%		1.5
Eddy Island N			100%	4	Ballindereen		38%	62%	381.6
Rosshill	50%	25%	25%	8	Ballyvelaghan			100%	231
Kilcogan	61%	20%	19%	1776.3	Kilcolgan		34%	66%	3
Ballinderreen	60%	13%	27%	93.5	Rosshill	100%			367
Ballyvelaghan	100%			1	Tawin N		100%		1.5

Figure 6.14 - Source sink connections for *M. gigas* larvae in Kinvarra bay.

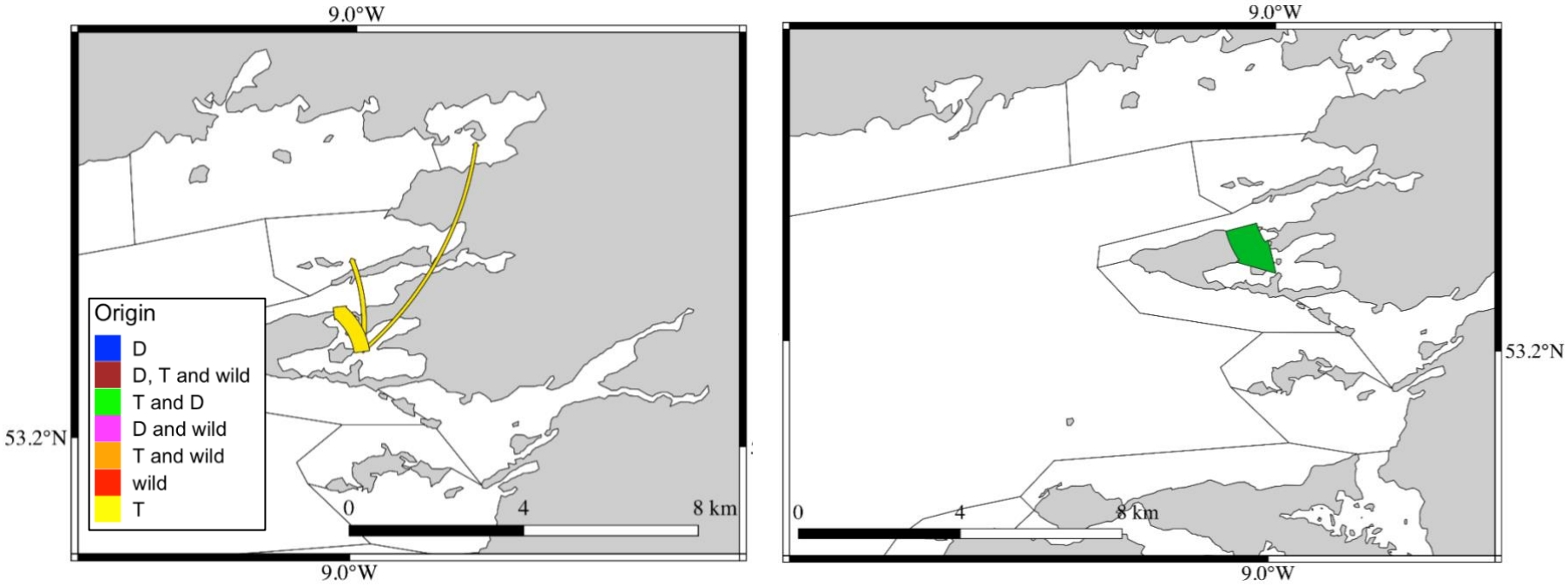
Ballinderreen



Sink site	Source site type				Source site	Source site type			
	Diploid (%)	Triploid (%)	Wild (%)	Total (N)		Diploid (%)	Triploid (%)	Wild (%)	Total (N)
Ballinderreen		67%	33%	33.6	Ballinderreen		67%	33%	33.6
Tawin S		100%		1	Kilcolgan		50%	50%	4
Eddy Island S		100%		1	Ballyvelaghan			100%	362
Kinvarra		38%	62%	381.6	Kinvarra	60%	12%	28%	93.5
Rosshill		60%	40%	2.5	Rosshill	100%			5.5
Kilcolgan		81%	19%	756.2					

Figure 6.15 - Source sink connections for *M. gigas* larvae in Ballinderreen bay.

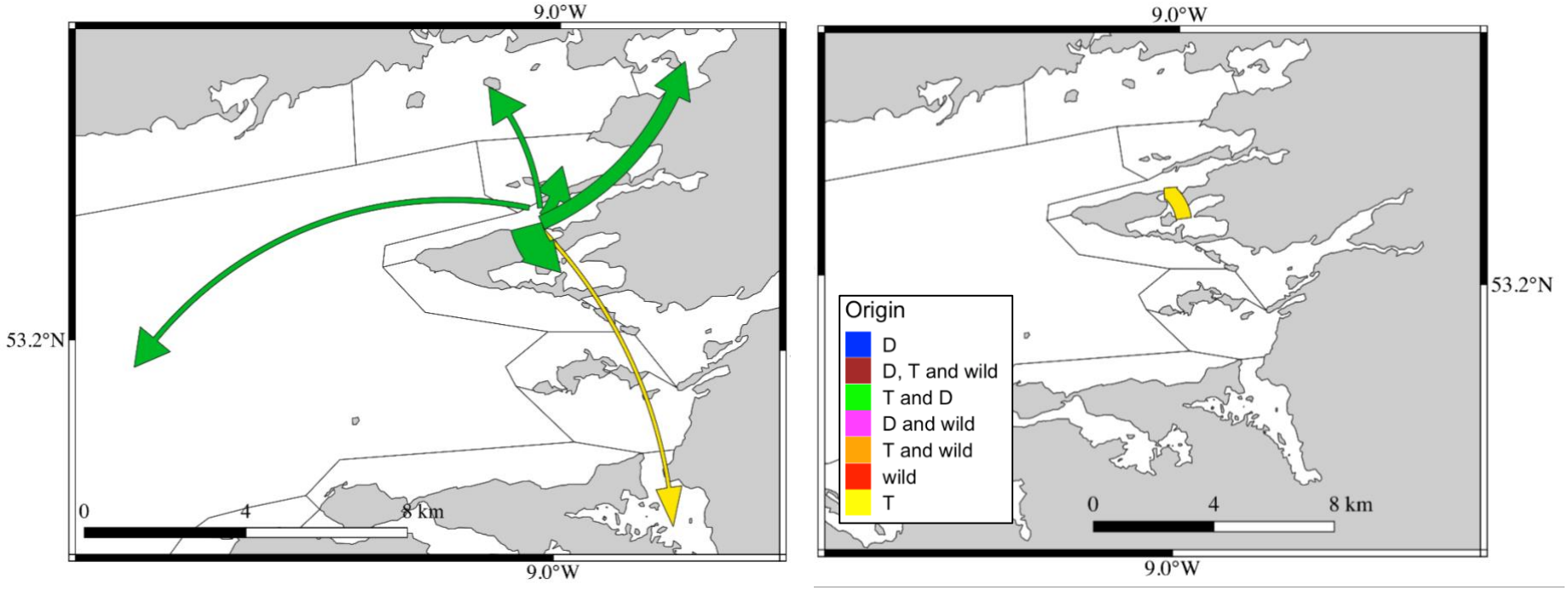
Tawin Inner



Sink site	Source site type				Source site	Source site type			
	Diploid (%)	Triploid (%)	Wild (%)	Total (N)		Diploid (%)	Triploid (%)	Wild (%)	Total (N)
Tawin Inner		100%		718.2	Tawin Inner		100%		718.2
Aille		100%		3.5	Tawin North	16%	84%		607
Oranmore		100%		2.5					
Tawin North		100%		62.3					
Galway Harbour		100%		2					

Figure 6.16 - Source sink connections for *M. gigas* larvae in Tawin Inner bay.

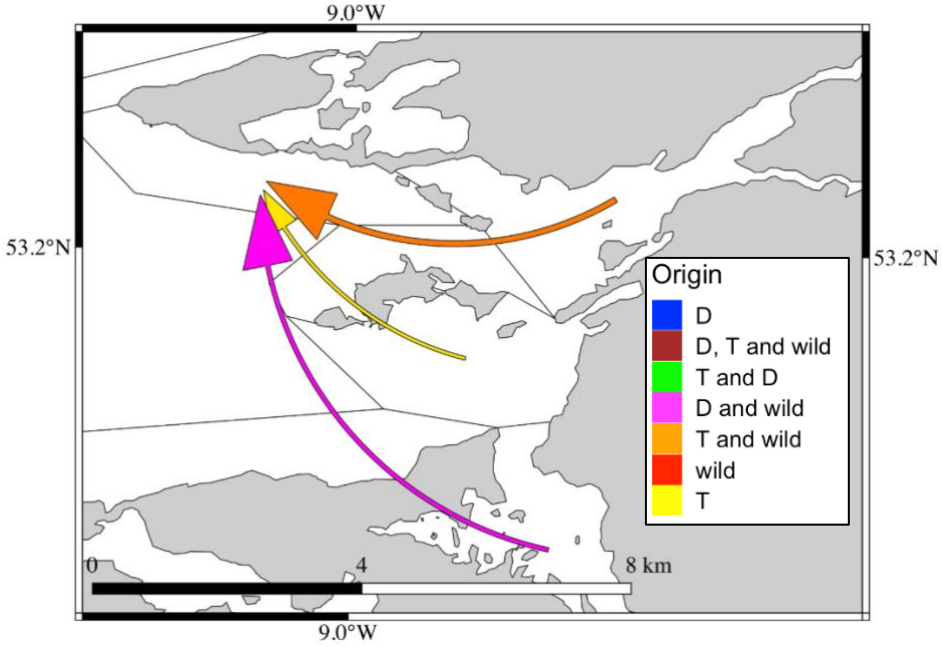
Tawin North



Sink site	Source site type				Source site	Source site type			
	Diploid (%)	Triploid (%)	Wild (%)	Total (N)		Diploid (%)	Triploid (%)	Wild (%)	Total (N)
Tawin North	33.9	66.1		1945,3	Tawin North	33.9	66.1		1945,3
Tawin Inner	16	84		607	Tawin Inner		100%		62.3
Oranmore	10.6	89.4		115					
Kinvarra		1.5		1.5					
Galway Center	20.9	79.1		7.1					
Galway Harbour	20.9	79.1		7.1					
Aille	21.7	88.3		64.6					

Figure 6.17 - Source sink connections for *M. gigas* larvae in Tawin North Bay.

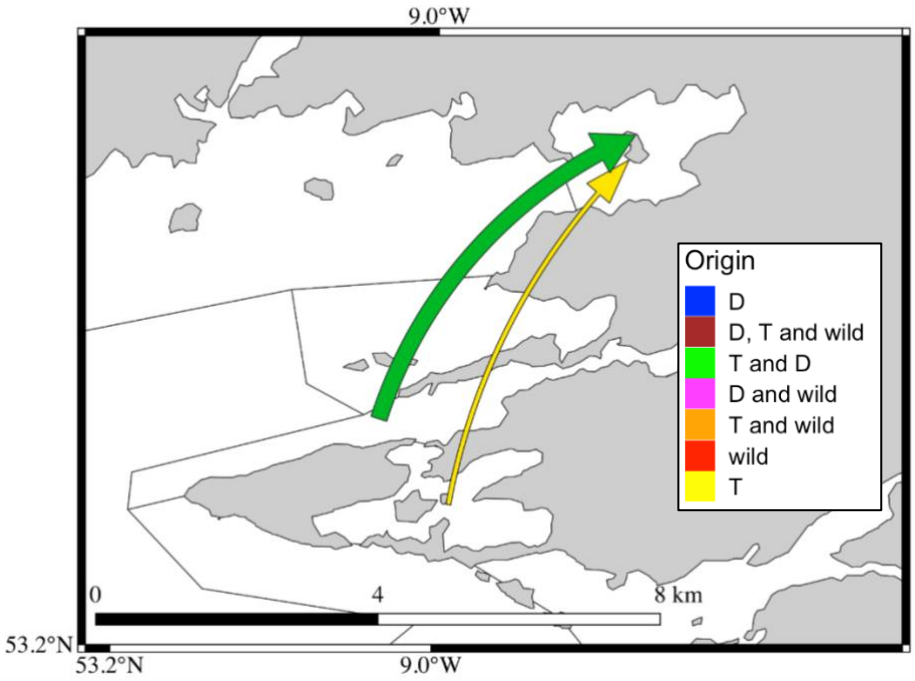
Tawin South



Sink site	Source site type			
	Diploid (%)	Triploid (%)	Wild (%)	Total (N)
Kinvarra	50%		50%	2
Kilcolgan		14%	86%	7
Ballinderreen		100%		1

Figure 6.18 - Source sink connections for *M. gigas* larvae in Tawin South Bay.

Oranmore



Sink site	Source site type			
	Diploid (%)	Triploid (%)	Wild (%)	Total (N)
Kinvarra	10.7%	89.3%		115.3
Kilcolgan		100%		2.5

Figure 6.19 - Source sink connections for *M. gigas* larvae in Oranmore Bay.

A.1.1 Non-native risk management (eradication) assessment for *M. gigas* (Example)

Title	Response	Confidence	Justification
1. Define the scenario	Wild *M. gigas* individuals have been recorded in intertidal areas in Galway Bay, although densities remain low. Currently records are from the east and southern part of the bay.		
2. Define the eradication strategy	The strategy to eradicate this species would be hand removal through hammering of individuals by a small, trained team. An initial baseline survey of the area and pilot cull in the southern part of the bay could be used to test the scheme efficacy. Repeated monitoring and control efforts likely needed, potentially every 5 years depending on success in reducing wild population numbers and presence of cultured diploid stock		
3a. How effective is the strategy?	3 – MODERATE EFFECTIVENESS	2 - MED	Hammering has been shown to be effective at reducing population densities in two case-study areas. The control method is only likely to be effective if densities of oysters are relatively low, and if populations are restricted to the intertidal.
3b. How practical is the strategy?	2 - IMPRACTICAL	2 - MED	Currently there is no permitting system in place for IAS control activities, would need to involve NPWS and would require an impact assessment due to SAC status.
3c. How expensive is the strategy?	5 - <£50k	3 – HIGH	Cost estimate is based experience in the NE Kent MPA where trained volunteers have carried out the majority of the work. Equipment costs are minimal.
3d. How much negative impact would the strategy have?	5 – MINIMAL	3 – HIGH	The impacts on underlying habitat are likely to be minimal, as the method is relatively non-invasive especially in the rocky intertidal.
3e. How acceptable is the strategy?	4 – ACCEPTABLE	2 – HIGH	There are unlikely to be many objections to control of wild Pacific oysters from an ethical perspective, and in Kent interest in the issue has been driven by engagement with the public by volunteers. If fishing of wild populations occurs this may need to be considered.
4. What is the window of opportunity for implementing the strategy?	4 (4 - 10 YRS)	3 – LOW	There has not been much change in Pacific oyster densities in the area for a number of decades. However, in other areas of Europe, rapid increases in population size have been observed following very warm years.

5. What is the likelihood of reinvasion?	1 – Very Likely	2 – MED	Risk of re-establishment high while aquaculture operations use diploid stock. Re-establishment likelihood varies among the sub-bays depending on connectivity with other areas.
6. Conclusion (overall feasibility of eradication)	3 - Medium	2 - MED	Suppression of population feasible while densities remain low, but re-establishment likely.

A.2 Supporting material for *Hemigrapsus* case-study

A.2.1 Spread maps

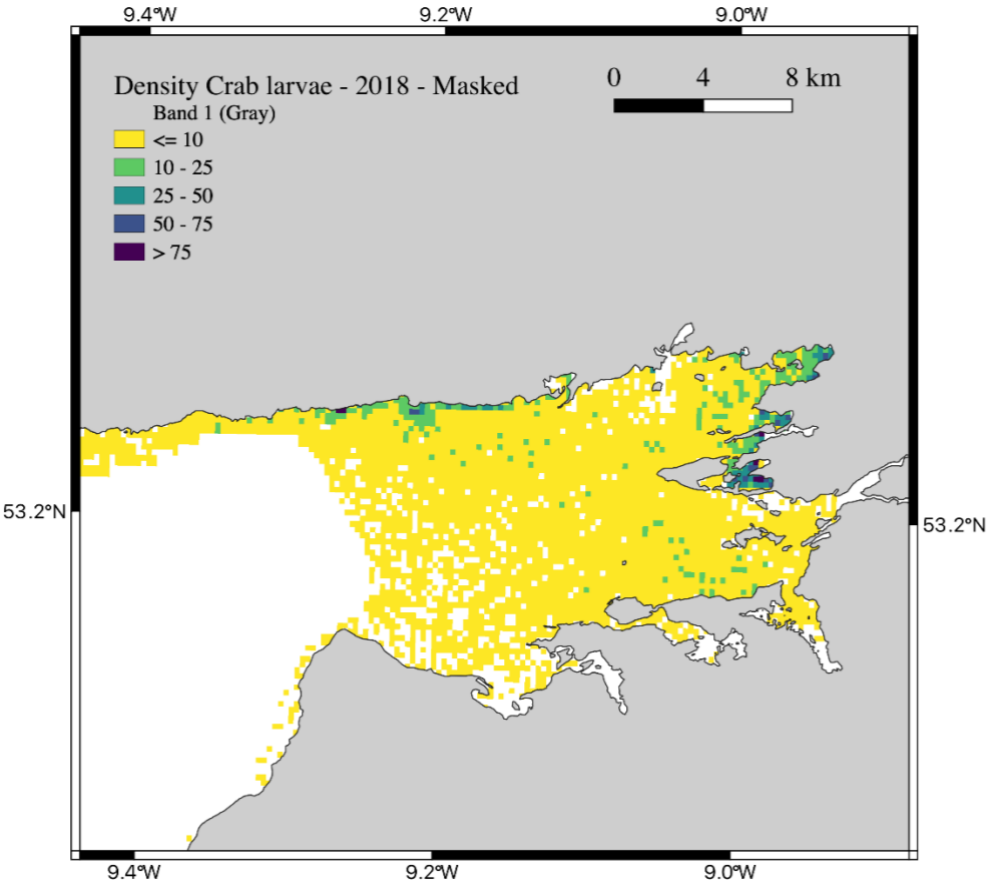


Figure 6.20 – The predicted larval density of *Hemigrapsus sanguineus* in Galway Bay, based on larval tracking models incorporating known *H. sanguineus* larval behaviour with no substrate filter applied.

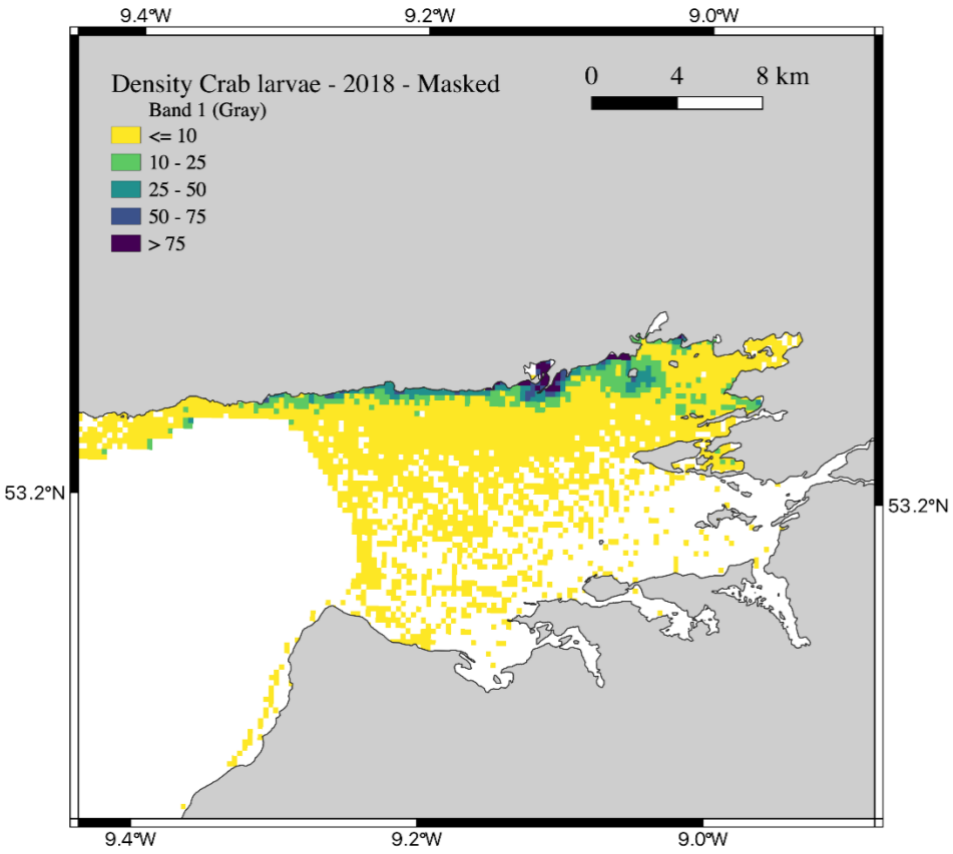


Figure 6.21 – The predicted larval density of *Hemigrapsus sanguineus* in Galway Bay, based on larval tracking models using passive dispersal with no substrate filter applied.

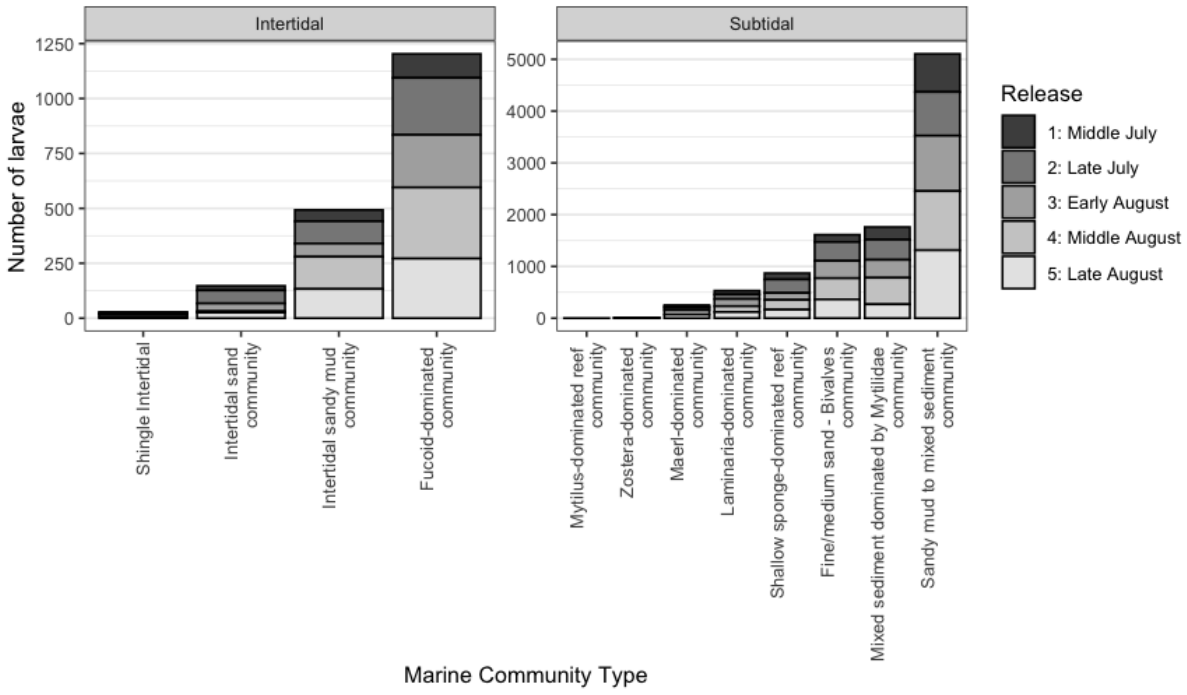


Figure 6.22 – Number of settled crab larvae within each Marine Community Type within the Galway Bay SAC Complex. The number of larvae is split by the date of releases for the year 2018, as *H. sanguineus* may produce multiple broods per summer season.

Further details available on www.emff.marine.ie

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