Document Control

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Revision History

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18/05/2017	V Gravestock	1.2	Draft	Small amendments & Reference List	
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This document has been distributed for information and comment to:

Title	Name	Date sent	Comments received
HRA Studland to Portland SCI – Fish traps – v1.1	Natural England	17/05/2017	Yes – 23/08/2017
HRA Studland to Portland SCI – Fish traps – v1.4	S Pengelly, Southern IFCA	14/02/2018	Yes
HRA Studland to Portland SAC – Fish traps – v1.5	Natural England	26/02/2018	Yes – 26/03/2018
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HRA Studland to Portland SAC – Fish traps – v1.8	Natural England	07/06/2018	14/06/2018
HRA Studland to Portland SAC – Fish traps v1.9 FINAL	Natural England	15/06/2018	

Southern Inshore Fisheries and Conservation Authority (IFCA)

Fisheries in EMS Habitats Regulations Assessment for amber and green risk categories

European Marine Site: Studland to Portland SAC

Feature: Reefs

Site Specific Sub-Feature(s): Circalittoral rock, Infralittoral Rock, Subtidal stony reef

Generic Sub-Feature(s): Subtidal bedrock reef; Subtidal boulder and cobble reef; Subtidal mussel bed on rock

Gear type(s) Assessed: Fish traps

Technical Summary

A fishery for live wrasse developed in 2015/2016, a portion of which occurs within the Studland to Portland SAC and as such fell outside the deadline of the revised approach. In 2016 and 2017 the fishery took place seasonally from April to October. In 2017, a total of 10 vessels engaged in the fishery, although not all are believed to fish in the SAC. Wrasse are typically targeted using fish traps close to the shore in waters no deeper than 10 metres, over infralittoral rocky ground typically characterised by heavy kelp and seaweed cover. The dominant target species are Ballan wrasse (*Larus bergylta*), although other wrasse species may be targeted.

The potential pressures likely to be exerted by the wrasse fishery upon the designated features were identified as abrasion, removal of non-target species and removal of target species. Scientific literature shows the physical impacts of potting on temperate rocky habitats are negligible or limited in extent, especially when compared to impacts resulting from periods of adverse weather.

Impacts related from the removal of target species are considered as direct impacts on wrasse populations and indirect impacts on the wider ecosystem. The wrasse fishery is size-selective and as such can remove certain groups from the population leading to variety of implications related to population dynamics, demography and reproduction. Wrasse species however do not appear within any species list within Conservation Packages associated with the site. Any direct impacts on wrasse populations are therefore not considered relevant in the context of this assessment.

When considering the wider indirect ecosystem impacts of wrasse removal, there is a lack of evidence on ecological function of wrasse species and subsequent impacts on temperate reef habitats. As such, best available evidence was used to infer potential impacts, with research highlighting potential concerns around the removal of wrasse as an epibenthic grazer (of small algal grazing invertebrates) and subsequent changes in algal biomass.

When considering the scale of the fishery, the relatively small area subject to fishing (~1.5% of the SAC and ~0.8% of the total reef feature) and the targeting of one predominant wrasse species, it was concluded the potential indirect effects of wrasse removal on algal growth of notable/ representative brown algae will not occur at levels significant enough to have an adverse effect on site integrity and is therefore not considered to hinder the sites conservation objectives.

Wrasse fishery guidance, introduced in June 2017, outlines a wide range of different measures and as such makes the fishery one of the most restricted in the Southern IFCA district. Whilst aimed at ensuring the long-term sustainability of the fishery through preventing over-exploitation of wrasse populations, the fishery guidance will also benefit the wider ecosystem. In particular, safeguarding against potential impacts related to the ecological function and wider ecosystem and thereby reducing potential risks associated with uncertainties surrounding these effects. Additionally, the fishery will be closely monitored and a feedback process established to allow for regular review and adaptive management.

1. Introduction

1.1 Need for an HRA assessment

Southern IFCA has duties under Regulation 9(3) of the Conservation of Habitats and Species Regulations 2010 as a competent authority, with functions relevant to marine conservation to exercise those functions so as to secure compliance with the Habitats Directive. Article 6.2 of the Habitats Directive requires appropriate steps to be taken to avoid, in Natura 2000 sites, the deterioration of natural habitats and habitats of species as well as significant disturbance of the species for which the area has been classified.

Management of European Marine Sites is the responsibility of all competent authorities which have powers or functions which have, or could have, an impact on the marine area within or adjacent to a European Marine Site (EMS). Under section 36 of the Species and Habitats Regulations (2010):

"The relevant authorities, or any of them, may establish for a European marine site a management scheme under which their functions (including any power to make byelaws) are to be exercised so as to secure in relation to that site compliance with the requirements of the Habitats Directive."

In 2012, the Department for Environment, Food and Rural Affairs (Defra) announced a revised approach to the management of commercial fisheries in European Marine Sites (EMS). The objective of this revised approach is to ensure that all existing and potential commercial fishing activities in European Marine Sites are managed in accordance with Article 6 of the Habitats Directive. Articles 4.1 and 4.2 of the Birds Directive also require that the Member States ensure the species mentioned in Annex I and regularly occurring migratory bird species are subject to special conservation measures concerning their habitat in order to ensure survival and reproduction in their area of distribution. This affords Special Protection Areas (SPAs) a similar protection regime to that of Special Areas of Conservation (SACs).

This approach was implemented using an evidence based, risk-prioritised, and phased approach. Risk prioritisation was informed by using a matrix of the generic sensitivities of the sub-features of the EMS to a suite of fishing activities as a decision-making tool. These sub-feature-activity combinations were categorised according to specific definitions, as red[1], amber[2], green[3] or blue[4].

Activity/feature interactions identified within the matrix as red risk had the highest priority for implementation of management measures by the end of 2013 in order to avoid the deterioration of Annex I features in line with obligations under Article 6(2) of the Habitats Directive.

[1] Where it is clear that the conservation objectives for a feature (of sub-feature) will not be achieved because of its sensitivity to a type of fishing, - irrespective of feature condition, level of pressure, or background environmental conditions in all EMSs where that feature occurs – suitable management measures will be identified and introduced as a priority to protect those features from that fishing activity or activities.

[2] Where there is doubt as to whether conservation objectives for a feature (or sub-feature) will be achieved because of its sensitivity to a type of fishing, in all EMSs where that feature occurs, the effect of that activity or activities on such features will need to be assessed in detail at a site specific level. Appropriate management action should then be taken based on that assessment.

[3] Where it is clear that the achievement of that conservation objectives for a feature is highly unlikely to be affected by a type of fishing activity or activities, in all EMSs where that feature occurs, further action is not likely to be required, unless there is the potential for in combination effects.

[4] For gear types where there can be no feasible interaction between the gear types and habitat features, a fourth categorisation of blue is used, and no management action should be necessary.

Activity/feature interactions identified within the matrix as amber risk required a site-level assessment to determine whether management of an activity was required to conserve site features. Activity/feature interactions identified within the matrix as green also required a site level assessment if there were “in combination effects” with other plans or projects.

Site level assessments were carried out in a manner consistent with the provisions of Article 6(3) of the Habitats Directive, but were also required to meet the 6(2) responsibilities of Southern IFCA as a competent authority. The aim of the assessments was to consider if any activity could significantly disturb the species or deteriorate natural habitats or the habitats of the protected species. From this, a judgement was made as to whether or not the conservation measures in place were appropriate to maintain and restore the habitats and species for which the site has been designated to a favourable conservation status (Article 6(2)). If assessments identified that additional conservation measures were required, these had to be implemented or be in the process of implementation by the end of 2016. Southern IFCA completed this process by the 2016 deadline. Following the end of 2016, the need for assessment i.e. if a change in the status of an existing fishery or a new fishery arose, will be reviewed by Southern IFCA on an as and when basis.

A new fishery for live wrasse, caught using fish traps, emerged during 2015-16. A portion of the fishery falls within the Studland to Portland SAC. As stated above, Southern IFCA is carrying out a site level assessment to determine whether or not the fishing activity will have a likely significant effect on Reefs of the Studland to Portland SAC, and on the basis of this assessment whether or not it can be concluded that the fishery will not have an adverse effect on the integrity of this EMS.

1.2 Documents reviewed to inform this assessment

- Natural England’s risk assessment Matrix of fishing activities and European habitat features and protected species[5]
- Reference list[6] (Annex 1)
- Natural England’s Regulation 35 Conservation Advice (March 2013)[7]
- Natural England’s Conservation Advice (September 2017)[8]
- Site map(s) – sub-feature/feature location and extent (Annex 2)
- Fishing activity map (Annex 3)
- Fisheries Impact Evidence Database (FIED)
- Natural England Advice on the management of the emerging wrasse fishery (Annex 4)

2. Information about the EMS

- Studland to Portland Special Area of Conservation (UK0030382)

[5] See Fisheries in EMS matrix: http://www.marinemanagement.org.uk/protecting/conservation/documents/ems_fisheries/populated_matrix3.xls

[6] Reference list will include literature cited in the assessment (peer, grey and site specific evidence e.g. research, data on natural disturbance/energy levels etc.)

[7] http://publications.naturalengland.org.uk/publication/3282207

[8] https://designatedsites.naturalengland.org.uk/Marine/MarineSiteDetail.aspx?SiteCode=UK0030382&SiteName=studland&countyCode=&responsiblePerson=&SeaArea=&IFCAArea=

2.1 Overview and qualifying features

- **Reefs.**
    - Circalittoral rock
    - Infralittoral rock
    - Subtidal stony reef

Please refer to Annex 2 for a site feature map.

Studland to Portland SAC lies off the south coast of Dorset and contains numerous areas of reef in many forms, which exhibit a large amount of geological variety and biological diversity. Features of particular interest within the Studland Bay to Ringstead Bay area include a series of limestone ledges (up to 15m across) protruding from shelly gravel at Worbarrow Bay, which support a rich sponge and sea fan community; dense brittlestar beds (*Ophiothrix fragilis*) on shale reefs extending from Kimmeridge; a unique reef feature, known as St Albans ledge, extending out over 10km offshore and subject to strong tidal action; and an area of large limestone blocks known as the “seabed caves”. The Portland Reefs are characterised by flat bedrock, limestone ledges (Portland stone), large boulders and cobbles. On the western side of Portland Bill, rugged limestone boulders provide deep gullies and overhangs. Mussel beds (*Mytilus edulis*) are found to occur in very high densities on bedrock associated with strong currents to the southeast of Portland Bill.

2.2 Conservation Objectives

The site’s conservation objectives apply to the site and the individual species and/or assemblage of species for which the site has been classified (Qualifying features: Reefs).

The objectives are to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the Favourable Conservation Status of its qualifying features, by maintaining or restoring:

- The extent and distribution of qualifying natural habitats and habitats of the qualifying species
- The structure and function (including typical species) of qualifying natural habitats
- The structure and function of the habitats of the qualifying species
- The supporting processes on which qualifying natural habitats and the habitats of qualifying species rely
- The populations of qualifying species
- The distribution of qualifying species within the site

3. Interest feature(s) of the EMS categorised as ‘Red’ risk and overview of management measure(s)

- Reef

A red risk interaction between bottom towed gears and reef features were identified and subsequently addressed through the creation of the ‘Bottom Towed Fishing Gear’ byelaw[9]. The

[9] Bottom Towed Fishing Gear Byelaw:
https://secure.toolkitfiles.co.uk/clients/25364/sitedata/files/PDFbyelaw_bottomtowedfishi.pdf

'Bottom Towed Fishing Gear' prohibits the use of any bottom towed fishing gear within sensitive areas (characterised by reef features or eelgrass/seagrass beds) in European Marine Sites throughout the district. The byelaw also states that that if transiting through a prohibited area carrying bottom towed fishing gear, all parts of the gear are inboard and above the sea. Within the Studland to Portland SAC there are two prohibited areas which cover the extent of the reef features within the site. This was based on habitat mapping data provided by Natural England and ground truthing by Southern IFCA.

4. Information about the fishing activities within the site

4.1 Activities under Consideration/Summary of Fishery

During 2015-16, a fishery for live wrasse developed within the Southern IFCA district, with exploitation partially taking place inside the Studland to Portland SAC. Wrasse are used as cleaner fish in Scottish salmon farms to remove sea-lice and as a biological alternative to the use of anti-parasitic chemical treatments. Wrasse are targeted using fish traps. The fishery operates on a seasonal basis and in 2016 and 2017 the season ran from April to October.

4.2 Technical Gear Specifications

Wrasse are predominantly targeted fish using traps, typically baited with shore crab (*Carcinus maenas*). The traps are commonly shot in short strings, with a number of traps attached to one long rope which is laid on the seabed and marked at one end with a buoy. An anchor may also be attached to one or both ends of the string. Traps will often be soaked for between 24 to 48 hours.

The traps used to catch wrasse are typically constructed of 7 mm plastic coated steel wire and covered with small meshed eel-netting, black in colour. The size and shape of the traps may vary, but typically measure 28"L x 16"W x 11"H and weight approximately 4 kg. The outside edges are wrapped with rope to protect the pot from damage through abrasion on the seabed (Figure 1). Due to the light weight nature of the traps, the majority of fishermen have fitted a metal frame to each trap to protect it from damage (Figure 2). With the addition of this metal frame, traps weigh approximately 15 kg. The position of entrances (typically two) are located on the sides of the traps and have a tapered netting entrance held open with a ring. The end or side of the trap is hinged to allow the removal of catch and bait replacement.

Figure 1. Fish trap used to catch wrasse. Source: http://en.carapax.se/creelspotstraps/cleaning-wrasse-traps/wrasse-trap

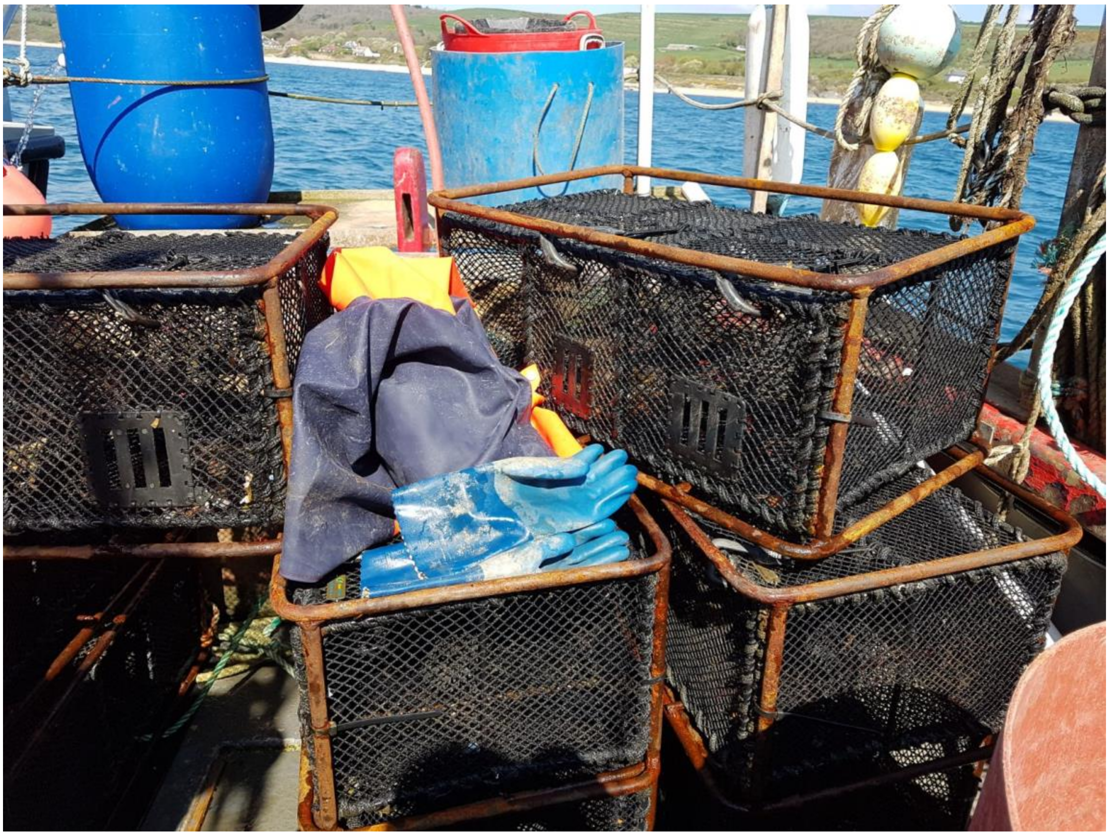

Figure 2. Fish trap used to catch wrasse fitted with metal frame and escape gaps.

4.3 Effort, Location and Scale of Fishing Activities

In 2016, approximately seven vessels operated within the fishery; all 8 metres or less in length.

Wrasse were targeted from May to October using fish traps. In 2017, the wrasse fishery commenced in early April with 8 known vessels participating within the fishery, all of which operate from Weymouth and Portland. The number of traps used by each vessel varies and the wrasse fishery guidance stipulates a pot limitation of 80 traps. Assuming 8 vessels partake in the fishery each with 80 pots, this equates to a trap density of approximately 26 pots per km^2. This is considered to be of very low to low gear intensity, depending on the definition of gear intensity used, which can vary between studies (Annex 5). Traps are arranged in strings of 5 to 10 traps. Two to three vessels that pot for wrasse also catch wrasse using rod and line. A further two vessels commenced fishing later in the season. It is anticipated the 2018 season will not commence until July and 8 vessels will take part in the trap fishery, the majority of which will solely target ballan wrasse.

Potting for wrasse occurs subtidally, although close inshore (no deeper than the 10 metre depth contour), over infralittoral rocky ground typically characterised by heavy kelp and seaweed cover. This represents the favoured habitat for wrasse species. In 2016 and 2017, fishing effort was concentrated in the Weymouth and Portland area, between Grove Point and Lulworth, with key areas from Whitenothe to Ringstead and Portland breakwater. Areas known to be fished within the SAC are illustrated in Annex 3, this area represents 1.5% of the total SAC and 0.8% of the total reef feature.

Six wrasse species occur along the south coast, four of which form the target species of the fishery. These include Corkwing (*Symphodus melops*), Goldsinny (*Ctenolabrus rupestris*), Rock cook (*Centrolabrus exoletus*) and Ballan (*Larus bergylta*). Ballan wrasse have proved to be the most popular species due to their survivability and feeding efficiency. At the beginning of the 2017 season, eight boats operating within the fishery solely targeted ballan wrasse in the size range of 12 to 28 cm. This size range later changed with the introduction of the Wrasse Fishery Guidance in June to 18 to 28 cm. Later in the 2017 season, a further two vessels started fishing for all four wrasse species. The number of wrasse caught per day varies depending on the time of year and number of traps used, with each vessel catching between a minimum of 10 and maximum of 100 wrasse. It is difficult to determine a ‘typical’ number of wrasse caught due to large daily variations, in combination with other factors (time of year, number of traps used).

5. Test of Likely Significant Effect (TLSE)

The Habitats Regulations assessment (HRA) is a step-wise process and is first subject to a coarse test of whether a plan or project will cause a likely significant effect on an EMS[10]. Each feature/sub-feature was subject to a TLSE, the results of which are summarised in Table 1.

Table 1. Summary of LSE Assessment (Subtidal bedrock reef; Subtidal boulder and cobble reef; Subtidal mussel bed on rock (Circalittoral rock; Infralittoral rock; Subtidal stony reef)).

[10] Managing Natura 2000 sites: http://ec.europa.eu/environment/nature/natura2000/management/guidance_en.htm

1. Is the activity/activities directly connected with or necessary to the management of the site for nature conservation?	No		
2. What potential pressures exerted by the gear type(s) are likely to affect the feature(s)/sub-feature(s)? Regulation 35 Conservation Advice (March 2013) / Natural England Conservation Advice (September 2017)	3. **Is the feature(s)/sub-features(s) likely to be exposed to the pressure(s) identified?**		
	Subtidal bedrock reef (Circalittoral rock; Infralittoral rock)	Subtidal boulder and cobble reef (Subtidal stony reef)	Subtidal mussel bed on rock
Physical loss – removal	OUT – The activity will not lead to the physical removal of the feature and therefore there is no direct interaction between the pressure and feature under assessment.	OUT – The activity will not lead to the physical removal of the feature and therefore there is no direct interaction between the pressure and feature under assessment.	OUT – The activity will not lead to the physical removal of the feature and therefore there is no direct interaction between the pressure and feature under assessment.
Physical loss – smothering	OUT – The activity will not lead to the physical loss of the feature through smothering and therefore there is no direct interaction between the pressure and feature under assessment.	OUT – The activity will not lead to the physical loss of the feature through smothering and therefore there is no direct interaction between the pressure and feature under assessment.	OUT – The activity will not lead to the physical loss of the feature through smothering and therefore there is no direct interaction between the pressure and feature under assessment.
Physical damage – siltation	OUT – The activity is not likely to lead to siltation and cause physical damage to features. Fish	OUT – The activity is not likely to lead to siltation and cause physical damage to features.	OUT – The activity is not likely to lead to siltation and cause physical damage to features. Fish traps are

	traps are typically deployed in areas of hard ground with limited or no fine sediment.	Fish traps are typically deployed in areas of hard ground with limited or no fine sediment.	typically deployed in areas of hard ground with limited or no fine sediment.
Physical damage – abrasion/ Abrasion/ disturbance of the substrate on the surface of the seabed	IN – The activity is likely to lead to abrasion of the feature through contact of the feature with the seabed during deployment/retrieval and any subsequent movement of gear, including ground rope, from currents or storm action. At current levels of activity, exposure of bedrock reef to physical damage through abrasion is considered to be low. Vulnerability of bedrock to physical damage is considered to be moderate. Further investigation is required to determine the severity and magnitude of this pressure.	IN – The activity is likely to lead to abrasion of the feature through contact of the feature with the seabed during deployment/retrieval and any subsequent movement of gear, including ground rope, from currents or storm action. At current levels of activity, exposure of stony reef to physical damage through abrasion is considered to be low. Vulnerability of stony reef to physical damage is considered to be low. Further investigation is required to determine the severity and magnitude of this pressure.	IN – The activity is likely to lead to abrasion of the feature through contact of the feature with the seabed during deployment/retrieval and any subsequent movement of gear, including ground rope, from currents or storm action. At current levels of activity, exposure of bedrock reef to physical damage through abrasion is considered to be low. Vulnerability of bedrock to physical damage is considered to be moderate. Further investigation is required to determine the severity and magnitude of this pressure.
Physical damage – selective extraction	OUT – The activity will not lead to physical damage through selective extraction and therefore there is no direct interaction	OUT – The activity will not lead to physical damage through selective extraction and therefore there is no direct	OUT – The activity will not lead to physical damage through selective extraction and therefore there is no direct interaction between the

	between the pressure and feature under assessment.	interaction between the pressure and feature under assessment.	pressure and feature under assessment.
Toxic contamination – introduction of synthetic/ Introduction of other substances/ Synthetic compound contamination	OUT – Insufficient activity levels to pose risk of large scale pollution event.	OUT – Insufficient activity levels to pose risk of large scale pollution event.	OUT – Insufficient activity levels to pose risk of large scale pollution event.
Toxic contamination – introduction of non-synthetic/ Hydrocarbon & PAH contamination/ Introduction of other substances	OUT – Insufficient activity levels to pose risk of large scale pollution event.	OUT – Insufficient activity levels to pose risk of large scale pollution event.	OUT – Insufficient activity levels to pose risk of large scale pollution event.
Toxic contamination – introduction of radionuclides/ Introduction of other substances/ Transition element & organo-metal contamination	OUT – Insufficient activity levels to pose risk of large scale pollution event.	OUT – Insufficient activity levels to pose risk of large scale pollution event.	OUT – Insufficient activity levels to pose risk of large scale pollution event.
Non-toxic contamination – changes in nutrient loading	OUT – The activity will not lead to any changes in nutrient loading and therefore there is no direct interaction between the pressure and feature under assessment.	OUT – The activity will not lead to any changes in nutrient loading and therefore there is no direct interaction between the pressure and feature under assessment.	OUT – The activity will not lead to any changes in nutrient loading and therefore there is no direct interaction between the pressure and feature under assessment.
Non-toxic contamination – changes in organic loading/ Organic enrichment	OUT – The activity will not lead to any changes in organic loading and therefore there is no direct	OUT – The activity will not lead to any changes in organic loading and therefore there is no direct interaction between the	OUT – The activity will not lead to any changes in organic loading and therefore there is no direct

	interaction between the pressure and feature under assessment.	pressure and feature under assessment.	interaction between the pressure and feature under assessment.
Non-toxic contamination – changes in turbidity	OUT – The activity is considered unlikely to lead to siltation and therefore is unlikely to lead to any subsequent changes in turbidity.	OUT – The activity is considered unlikely to lead to siltation and therefore is unlikely to lead to any subsequent changes in turbidity.	OUT – The activity is considered unlikely to lead to siltation and therefore is unlikely to lead to any subsequent changes in turbidity.
Non-toxic contamination – changes in salinity	OUT – The activity will not lead to changes in salinity.	OUT – The activity will not lead to changes in salinity.	OUT – The activity will not lead to changes in salinity.
Biological disturbance – introduction of microbial pathogens	OUT – The fleet operates within the local area, so the introduction of new microbial pathogens from outside the local vicinity is considered unlikely.	OUT – The fleet operates within the local area, so the introduction of new microbial pathogens from outside the local vicinity is considered unlikely.	OUT – The fleet operates within the local area, so the introduction of new microbial pathogens from outside the local vicinity is considered unlikely.
Biological disturbance – introduction of non-native species and translocation/ Introduction or spread of invasive non-indigenous species	OUT – The fleet operates within the local area, so the introduction or translocation of non-indigenous species is considered unlikely.	OUT – The fleet operates within the local area, so the introduction or translocation of non-indigenous species is considered unlikely.	OUT – The fleet operates within the local area, so the introduction or translocation of non-indigenous species is considered unlikely.
Biological disturbance – selective extraction of species/ Removal of non-target species/ Removal of target species	IN – Selective extraction refers to the removal of species or community and includes of the removal of specific species, community or key species in a biotope. It covers both the	IN – Selective extraction refers to the removal of species or community and includes of the removal of specific species, community or key species in a biotope. It covers both the	OUT – Selective extraction refers to the removal of species or community and includes of the removal of specific species, community or key species in a biotope. It covers both the removal of target and non-target

	removal of target and non-target species. The removal of fish species can have significant impacts on the structure and functioning of benthic communities. Potting targets species of wrasse, including Corkwing, Cuckoo, Goldsinny and Rock cook, but particularly Ballan wrasse, having emerged as the most popular species. Wrasse species are not mentioned in the Studland to Portland SCI Regulation 35 Conservation Advice. Despite being a mobile species, wrasse are known to be year round residents of shallow rocky areas where there is heavy kelp and seaweed cover. Wrasse species represent important predators, feeding on a variety of invertebrates and also form a prey species for other fish and birds. It is recognised that wrasse play an important ecological role in these shallow temperate rocky reef ecosystems and their removal is likely to impact on structure and	removal of target and non-target species. The removal of fish species can have significant impacts on the structure and functioning of benthic communities. Potting targets species of wrasse, including Corkwing, Cuckoo, Goldsinny and Rock cook, but particularly Ballan wrasse, having emerged as the most popular species. Wrasse species are not mentioned in the Studland to Portland SCI Regulation 35 Conservation Advice. Despite being a mobile species, wrasse are known to be year round residents of shallow rocky areas where there is heavy kelp and seaweed cover. Wrasse species represent important predators, feeding on a variety of invertebrates and also form a prey species for other fish and birds. It is recognised that wrasse play an important ecological role in these shallow temperate rocky reef	species. The removal of live wrasse as a result of potting has been assessed separately for bedrock reef. With respect to subtidal mussel bed on rock, the activity will not lead to the removal of mussels. Species associated with the sub-feature include the common starfish and very large whelks. The type of trap used to catch wrasse are highly unlikely to lead to bycatch of either starfish or large whelks. It is therefore unlikely that the activity will have a significant effect on this feature through selective extraction of species.

	functioning of associated communities. There are no regulations controlling the size or amount of wrasse caught. Previously, the market for live wrasse dictated a size between 10 and 30 cm. This changed since the introduction of maximum and minimum sizes for wrasse species outlined in the Wrasse Fishery Guidance[11]. The level of bycatch within traps is unknown. Crab and lobster pots, similar in nature to those used to catch wrasse, are known to be relatively low in incidental bycatch and non-target species are returned to the sea alive. Additionally, there is high survivability of bycatch species caught due to the shallow depths in which fishers work (less than 10 metres in depth). The removal of or damage to non-target species may occur through the mechanical impacts of potting, including contact with gear	ecosystems and their removal is likely to impact on structure and functioning of associated communities. There are no regulations controlling the size or amount of wrasse caught. Previously, the market for live wrasse dictated a size between 10 and 30 cm. This changed since the introduction of maximum and minimum sizes for wrasse species outlined in the Wrasse Fishery Guidance. The level of bycatch within traps is unknown. Crab and lobster pots, similar in nature to those used to catch wrasse, are known to be relatively low in incidental bycatch and non-target species are returned to the sea alive. Additionally, there is high survivability of bycatch species caught due to the shallow depths in which fishers work (less than 10 metres in depth). The removal of or damage to non-	

[11] https://secure.toolkitfiles.co.uk/clients/25364/sitedata/files/Wrasse-Guidance.pdf

	components, entangling of ropes and surface abrasion. Emergent fauna can be tangled, damaged or removed by setting or hauling traps. Potentially vulnerable species present in the site on bedrock reef include Dead man's fingers, Pink sea fan, Hornwrack and Ross coral. The vulnerability of bedrock reef is considered to be moderate. Further investigation is needed into the potential impacts of wrasse species, including reproductive strategies as well as the likelihood of removal of non-target species through mechanical impacts of potting.	target species may occur through the mechanical impacts of potting, including contact with gear components, entangling of ropes and surface abrasion. Emergent fauna can be tangled, damaged or removed by setting or hauling traps. Potentially vulnerable species present in the site on stony reef include Hornwrack and Ross coral. The vulnerability of stony reef is considered to be moderate. Further investigation is needed into the potential impacts of wrasse species, including reproductive strategies as well as the likelihood of removal of non-target species through mechanical impacts of potting.	
Barrier to species movement	OUT – It is unlikely that the method of fishing will present a barrier to species movement. Any bycatch retained in fish traps is returned.	OUT – It is unlikely that the method of fishing will present a barrier to species movement. Any bycatch retained in fish traps is returned.	OUT – It is unlikely that the method of fishing will present a barrier to species movement. Any bycatch retained in fish traps is returned.

Deoxygenation	OUT – The activity is not likely to lead to deoxygenation.	OUT – The activity is not likely to lead to deoxygenation.	OUT – The activity is not likely to lead to deoxygenation.
Introduction of light	OUT – The activity typically takes place during daylight hours. Any introduction of light is not likely to have any effect on the sub-feature.	OUT – The activity typically takes place during daylight hours. Any introduction of light is not likely to have any effect on the sub-feature.	OUT – The activity typically takes place during daylight hours. Any introduction of light is not likely to have any effect on the sub-feature.
Litter	OUT – It is unlikely the level of fishing activity could lead to a level of discarded fishing gear that would be at a level of concern.	OUT – It is unlikely the level of fishing activity could lead to a level of discarded fishing gear that would be at a level of concern.	OUT – It is unlikely the level of fishing activity could lead to a level of discarded fishing gear that would be at a level of concern.
Penetration and/or disturbance of the substrate below the surface of the seabed, including abrasion	OUT – Instances where subsurface penetration occurs are likely to only include anchoring. Anchoring occurs on a very infrequent as it does not commonly occur during fishing. The infrequent nature of this pressure, combined with the small area affected means it is unlikely to have a significant effect on the feature.	OUT – Instances where subsurface penetration occurs are likely to only include anchoring. Anchoring occurs on a very infrequent as it does not commonly occur during fishing. The infrequent nature of this pressure, combined with the small area affected means it is unlikely to have a significant effect on the feature.	OUT – Instances where subsurface penetration occurs are likely to only include anchoring. Anchoring occurs on a very infrequent as it does not commonly occur during fishing. The infrequent nature of this pressure, combined with the small area affected means it is unlikely to have a significant effect on the feature.
Underwater noise changes	OUT – Sub-feature is not sensitive to pressure.	OUT – Sub-feature is not sensitive to pressure.	OUT – Sub-feature is not sensitive to pressure.

Visual disturbance	OUT – Sub-feature is not sensitive to pressure.	OUT – Sub-feature is not sensitive to pressure.	OUT – Sub-feature is not sensitive to pressure.
4. What key attributes of the site are likely to be effected by the identified pressure(s)?	**Regulation 35 Conservation Advice (March 2013):** Bedrock reef: - Biotope composition of bedrock reefs - Distribution and spatial pattern of bedrock reef biotopes - Extent of representative / notable bedrock reef biotopes - Species composition of representative or notable bedrock reef biotopes - Presence and/or abundance of specified bedrock reef species Stony reef: - Biotope composition of stony reefs - Distribution and spatial pattern of stony reef biotopes - Species composition of representative or notable stony reef biotopes **Natural England Conservation Advice (September 2017):** Reefs: - Distribution: presence and spatial distribution of biological communities - Structure and function: presence and abundance of key structural and influential species - Structure: species composition of component communities		
5. What conservation objectives are likely to be effected by the identified pressure(s)?	The site's conservation objectives apply to the site and the individual species and/or assemblage of species for which the site has been classified (Qualifying features: Reefs).		

	The objectives are to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the Favourable Conservation Status of its qualifying features, by maintaining or restoring: - The extent and distribution of qualifying natural habitats and habitats of the qualifying species - **The structure and function (including typical species) of qualifying natural habitats** - The structure and function of the habitats of the qualifying species - The supporting processes on which qualifying natural habitats and the habitats of qualifying species rely - The populations of qualifying species - The distribution of qualifying species within the site
6. Potential scale of pressures and mechanisms of effect/impact (if known)	*Fishing Activity Within the Site*[12] During 2015/16, a fishery for live wrasse developed within the Southern IFCA district. Target wrasse species include corkwing, goldsinny, rock cook and particularly ballan wrasse. Ballan wrasse have proved to be the most popular species due to their survivability and feeding efficiency, and as such formed the sole target species of 8 vessels in 2017. The wrasse are used as cleaner fish in Scottish salmon farms, to remove sea-lice and as a biological alternative to the use of anti-parasitic chemical treatments. Wrasse are targeted using fish traps with a small mesh size. Traps are operated in small strings of 5 to 10, typically from vessels less than 8 metres in length. Eight to ten vessels operate within the fishery. The fishery operates on a seasonal basis, with wrasse being targeted from May to October (now July to October). In 2016 and 2017, fishing effort was concentrated in the Weymouth and Portland area, between Grove Point and Lulworth, with key areas from Whitenothe to Ringstead and Portland breakwater *Generic Impacts*

[12] Information in this section is derived from IFCO knowledge

	The main impact pathways through which the activity may affect the designated feature are likely to be physical damage through abrasion and biological disturbance through the selective extraction. Static gear is deployed regularly on areas of bedrock reef. Physical abrasion is known to occur through the setting of pots/traps and their associated ground lines and weight, by their movement during rough weather and retrieval. If there is insufficient line when pots are deployed, it can cause the lead pot to bounce up and down on the seabed during periods of strong tides and large swell. The mechanical impacts of potting, which include weights and anchors hitting the seabed, hauling gear over the seabed and rubbing/entangling effect of ropes, can damage fragile epifauna. The greatest concerns about the potential impacts of pots/traps are those related to long-lived sessile fauna. Such taxa are more diverse and abundant in rocky-reef habitats, which is the same habitat type where wrasse are known to reside and therefore potting for wrasse takes places. Scientific research into the impacts of potting on reef habitats and associated communities has focused on crab and lobster pots. Traps used to fish for wrasse are however similar in nature to pots used crab and lobster fishing and as such results from these studies are directly relevantly. A number of studies based on the UK however have reported limits impacts of potting on sessile epifauna assemblages (i.e. Eno *et al*., 2001; Coleman *et al*., 2013; Haynes *et al*., 2014; Stephenson *et al*., 2015). Primary evidence relates mainly to specific features of these habitats, including seafans, sponges, anemones and Ross coral, and community assemblage, which have been monitored as part of experimental studies (Walmsley *et al*., 2015). The selection extraction of species refers to the removal of a species or community and includes the removal of a specific species/community/key species in a biotope. Removal of fish species can have significant impacts on the structure and functioning of benthic communities. The implications of removing wrasse, between a particular size range (see Wrasse Fishery Guidance for species-specific slot sizes), is largely unknown and the majority of scientific literature is focused on the impacts to the wrasse populations, which will differ depending on the reproductive and life-history strategies employed by each species. Potential impacts may be inferred through information relating to their ecology, biology and role within the ecosystem.

7. Is the potential scale or magnitude of any effect likely to be significant?	**Alone** Maybe	**In-Combination** N/A
8. Have NE been consulted on this LSE? If yes, what was NE's advice?		

6. Appropriate Assessment

6.1 Co-location of Fishing Activity and Site Features/Subfeature(s)

Maps of wrasse fishing location and site sub-features can be found in Annex 6. These maps reveal where fishing activity occurs in relation to the designated sub-features of the site.

6.2 Potential Impacts

It has been identified that potting for wrasse has the potential to cause an adverse impact on the features and sub-features of the Studland to Portland SAC through physical abrasion and its subsequent impact on the benthic environment and through the selective extraction of species. There are a number of factors that may influence the effect of potting of benthic habitats, including the spatial and temporal intensity of potting, technical gear type (single buoyed traps or strings of traps), the severity of weather and storm, events and the sensitivity of the effected benthic habitat (Young *et al*., 2013). Depth can also influence the effect of potting, with shallower depths potentially allowing for the greater movement of pots (Lewis *et al*., 2009).

Scientific research into the impacts of potting on reef habitats and associated communities has solely focused on those used to target on crab and lobster. Traps used to fish for wrasse are similar in nature to the pots used to crab and lobster and weigh the same, if not less. As such, results from these studies are directly relevantly and will be used in this assessment to determine the likely physical and biological disturbance caused by fish traps on the seabed (and associated communities).

6.2.1 Physical disturbance

Physical abrasion

Mechanical impacts of static gear include weights and anchors hitting the seabed which is likely to occur when the gear is set, hauling the gear over the seabed during retrieval and rubbing or entangling effects of ropes (when pots are fixed in strings) (JNCC & NE, 2011). In addition, the movement of gear may also occur over benthic habitats during rough weather or storm events (Roberts *et al*., 2010). Eno *et al*. (2001) reported that from observations of potting in Lyme Bay on rocky substrate, that when the wind and tidal streams were strong, pots tended to drag along the seabed the largest amount, especially when the wind was blowing across the tide. Anchor-weights on the end of each string of pots are typically used to prevent dragging when fishing in dynamic areas (Coleman *et al*., 2013). When deployed correctly, pots were typically observed to be static, however when there is insufficient line during deployment, it can cause the lead pot to bounce up and down on the seabed during periods of strong tides and large swell (Eno *et al*., 2001).

Lewis *et al*. (2009) investigated the impact of single-buoyed lobster traps after winter storms on coral communities in areas of hard-bottom and reef habitats in the Florida Keys, United States. Impacts were assessed after 26 wind events occurring over three winters. Traps moved when stormed sustained winds higher than 15 knots (27.8 km/h). Storms above this threshold were reported to move buoyed traps a mean distance of 3.63m, 3.21m and 0.73m per trap and affected a mean area of 4.66m^2, 2.88m^2 and 1.06m^2 per trap at depths of 4, 8 and 12 m respectively.

Young *et al*. (2013) assessed the effects of physical disturbance from potting on chalk reef communities in Flamborough Head European Marine Site. The maximum potential footprint of pots within the EMS was calculated using information of fishing effort, intensity and configuration. The maximum potential area within the SAC affected by potting per year was calculated at 2.97km^2 or 4.71% of the site. This was based on the following assumptions, which are derived from discussions with local fishermen and other information sources, include; potting intensity is at its highest in summer and halved in the winter, the number of pots fished in the EMS at any one time during the summer is 3562, each pot has a 1m^2 foot print (high estimate) and no duplicated seabed interaction, average fishing days per days of 150 and two thirds of total pots are hauled per fishing day. Survey work was also undertaken as part of the study in the Flamborough Head no-take zone (NTZ), designated in 2010, and a fished area of similar size, physical and hydrographic properties. Both areas occurred within the Flamborough Head Prohibited Trawl Area. In the fished site, a higher percentage of bare substrate (7.2%) was reported, which may imply physical abrasion from pots could be removing sessile epifauna. Reduced epifauna was however vastly reduced by adverse weather during the study which led to the seafloor being scoured within both the NTZ and fished site.

Stephenson *et al*. (2015) examined the long-term impacts of potting on benthic habitats in the Berwickshire and North Northumberland Coast European Marine Site from 2002 to 2012. The study was split up into a number of sections, one of which explored pot movement over a 23 day period using novel acoustic telemetry methods. The experimental pot configuration was made up of a string of 10 parlour pots, attached to the mainline by 2 m lengths of rope at intervals of 18 m. The end of each string was anchored with a 25 kg weight. The acoustic telemetry array allowed the position of each pot to be recorded every 1 to 5 minutes. Significant pot movements were not reported to occur daily but were detected on 6 out of 17 sampling occasions; equating to less than half of the sampling days. Significant movements occurred during neap and spring tides and at swell heights of 0-1 m and > 2 m, but not 1-2 m. Four of the six days with significant pot movement occurred during spring tides. Mean and maximum pot movement distances were slightly greater with increasingly extreme conditions, suggesting wave height and tidal height influence pot movement. The area potentially impacted by pot movements ranged between 53 and 115 m^2 per pot, with a mean of 85.8 m^2. There was no difference in the impacted area between neap and spring tides or between swell heights. The authors pointed out two aspects of the data that should be discussed, the first was lack of robustness based on the low number of significant pot movements and the second is the methodology which may under represent pot movement frequency. The conservative approach used to calculate 95% confidence intervals means only large movements will be significant as small non-significant distances are always lower than the mean error. Additionally, the mean error also means the range of possible movement is large and this means in reality the potentially impacted area may be smaller.

Gall (2016) investigated the direct physical and ecological impacts of inkwell and parlour pots on reef features within the Start Point to Plymouth Sound and Eddystone Special Area of Conservation at depths ranging from 20 to 30 metres. Sampling took place between late April and early September in 2014 and 2015. At 27 sites, a string of 4 inkwell pots and 4 parlour pots were deployed 200 m apart and were deployed in a fashion as they would under normal fishing conditions. GoPro cameras were attached to alternate pots at different angles to monitor movement and rope scour associated with the deployment and recovery of pots. Pots were left to soak for 25 minutes. The aim of the study was to quantify the mechanisms of potting interaction with the seabed and the true footprint of a pot. Additional biotic metrics were also used to quantify interactions with different taxa, including five indicator taxa known to be sensitive to fishing impact.

The study reported a haul corridor, the directly impacted area, of 3.22 ± 0.24 m^2 and a rapid haul time of 41 seconds, 20.7 seconds of which were in contact with the seabed. Pots took had an average settlement time of 3.5 seconds and once settled on the seabed, only the rims of the pot come into contact with the seabed, as opposed to the entire base and were reported to be relatively stationary during the soak period. 86% of deployments showed no movement, whilst 8% showed occasional sporadic and small movements and only one pot made significant movements throughout the soak period. Rope movement was observed for 51% of soaks, although this movement was minimal 46% of the time with only slight movements generated by the tide and no scour or species impacts.

6.2.2 Biological disturbance

6.2.2.1 Effects on non-target species

Benthic communities, including non-target epifauna, may be directly impacted by potting gear in a number of ways, including being directly struck by a pot or end-weight during deployment, through the entanglement or removal with moving pots or ropes under the influence of tidal currents or waves and through retrieval of pots which may lead to lateral dragging of the gear as it is being lifted (Coleman *et al*., 2013). The latter method is generally avoided by fishermen and is only likely to occur under the influence of wind, tide or navigational hazard which prevents vertical lift (Coleman *et al*., 2013). Up until recently there has been a paucity of scientific evidence on the impacts of static gear on benthic habitats (Walmsley *et al*., 2015). Although there is still considerably scientific literature less when compared to mobile fishing, there has been a recent rise in the number of studies investigating the impacts of potting in order to address this evidence gap. A number of the studies are still ongoing and where preliminary findings have been indicated, they have been reported here. This section will be discussed study by study.

Eno *et al*. (2001) investigated the effects of fishing with crustacean traps on benthic species in Great Britain were examined. In Scottish sea lochs, the effects of *Nephrops* creels on different sea pens were studied. In southern England (Lyme Bay) and west Wales (Greenala Point), the effects of crab and lobster pots on rocky substrates and associated communities were studied. Three species of sea pen (*Pennatula phosphorea*, *Virgularia mirabilis* and *Funiculina quadrangularis*) were all observed to bend as a result of the pressure wave generated by the sinking creel, protecting the tip of the sea pen from damage. *P. phosphorea* and *V. mirabilis* were thought to be more tolerant to disturbance than *F. quadrangularis*, although *F. quadrangularis* was found to be able to reinsert themselves after being uprooted. No lasting effects on the muddy substrate were found, although no other species were studied. In Lyme Bay and west Wales, rocky substrate habitats and associated communities appeared to be unaffected (no significant differences in abundance of species) before and after four weeks of relatively intense fishing activity (equivalent to around 1,000,000 pot hauls per km^2 per year). In west Wales, the abundance of five sponge species (*Dysidea*, *Hemimycale*, *Phorbas*, *Tethya*, Axinellids) increased significantly in experimental plots after potting, whilst in control plots no significant changes were found, except for an increase in *Dysidea* spp. *Halichondria* spp. abundance decreased significantly in control plots, but showed no significant change in experimental plots. In Lyme Bay, three out of five species (*Phallusta*, *Stelligera/Raspailia*, *Pentapora*) significantly increased in abundance in experimental plots, whilst in control plots no significant changes were found in the same three species, in addition to *Haliclona simlans*. Significant changes in *Haliclona* spp. and *Eucinella* spp. abundance (within experimental plots) could not be determined as a result of statistical limitations. *Pentapora foliacea* colony was found broken after hauling, although the cause of which is unknown and the Pink sea fan (*Eunicella verrucosa*) was observed to bend under the action of pots, but returned to an upright position once the pots had passed. The pink sea fan is slow growing and long lived and therefore considered as relatively susceptible to damage.

Sheridan *et al*. (2005) assessed the effects lobster and fish traps on coral reef ecosystems in the US Virgin Islands, Puerto Rico and Florida Keys. One part of the study was to quantify damage to corals and other structure providing organisms. Overall, a relatively small proportion (<20%) of traps set in shallow water (<30m) made contract with hard corals, gorgonians or sponges. Damage mainly occurred to hard corals and was patchy, at a scale less than the total trap footprint. In Florida Keys, habitat damage was only occasionally observed under or near traps and such limited observations did not allow for quantification of trap impacts. Habitat distribution maps revealed that only 10% are deployed over coral or sponge/gorgonian habitats, with relatively few traps found on coral habitats. In the US Virgin Islands, a significant proportion (54%) of trap locations were located within coral habitats. Unsurprisingly, diver surveys found that traps were estimated to cause damage at about 50% of traps visited, instances of damage were most relevant amount gorgonians and sponges, followed by corals.

Adey *et al*. (2007) examined the effects of fishing with *Nephrops norvegicus* creels on benthic species, in areas of soft mud, on the west coast of Scotland were examined and compared to areas of trawling and no fishing. Sampling was undertaken using towed video cameras and recordings from 2000, 2002 and 2003 were analysed. Animals were identified to the lowest possible taxonomic level and the number of species at each sampling site was recorded. A total of 142 stations were analysed and 29 species or taxonomic groups were identified. Species composition significantly differed among areas, but these differences were largely caused by variation in environmental conditions. Sea pens were used as an indicator of physical disturbance of the seabed and sea pen species *Virgularia mirabilis*, *Pennatula phosphorea* and *Funiculina quadrangularis* (and associated brittle star *Asteronyx loveni*) were all found in lower densities in the trawled areas when compared to areas fished solely by *Nephrops* creels. Despite being caught in moderate quantities by the creel fishery, high densities of *V. mirabilis* and *P. phosphorea* were observed in creel-fished areas where bycatch was greatest. High densities of *F. quadrangularis* were also observed, thus suggesting no adverse impact on these three species. Abundances of *A. loveni* in creel-fished areas were also not significantly different from no-fished zones. The portion of damaged or dead colonies of sea pen species was significantly higher in the creel-fished areas than in the trawled areas for both *F. quadrangularis* and *V. mirabilis* (10.7% and 18.6% in creel-fished areas and 5.5% and 5.4% in trawled areas, respectively). The authors however concluded this finding was contradictory and requires further investigation.

Lewis *et al*. (2009), the details of which are also discussed in section 6.2.1, reported injuries of scraping, fragmenting and dislodging sessile fauna as a result of trap movement. This resulted in significant damage to stony corals, octocorals and sponges. In areas of trap movement, sessile faunal cover reduced from 45% to 31%, 51% to 41% and 41% to 35% at depths of 4m, 8m and 12m, respectively.

Shester and Micheli (2011) quantified and compared the ecosystem impacts (discards and benthic habitat impacts) of four gear types (including lobster traps) employed in small-scale fisheries in Baja California in Mexico in areas of temperate to sub-tropical kelp forests and rocky reef. Observations were made of traps being deployed from a boat at the surface were made and to simulate the worse-case scenario of crushing of gorgonian corals, a diver lifted and forcefully dropped traps on top of gorgonian corals. Observations were also made of fishermen occasionally dragging traps and divers tried to replicate the same action that has been observed from a boat. Further simulations were achieved by divers by pulling a trap by the line over corals. After each treatment, gorgonian corals were examined for signs of skeletal damage or tissue loss. Lobster traps that were dropped onto gorgonians had minimal impact, with only one in 37 trials resulting in damage of less than 1% of the colony in the yellow gorgonian coral *Eugorgia ampla*. Lobster traps that were dragged caused damage to corals significantly more frequently than crushing, although damage was never over 5% of the skeleton. No corals were detached from the seafloor.

Coleman *et al*. (2013) studied the effects of potting on benthic assemblages, specifically sessile epifauna, in circalittoral reef habitats over a four year period following the designation of a no-take zone (NTZ) at Lundy Island in 2003. Control locations were positioned on the west coast of Lundy and on the east coast of Lundy, the latter occurring within the NTZ and for each sampling year six different sites within each location was random selectively. Differences in wave exposure, depth and substrate were present between control and NTZ locations. Control locations outside the NTZ were subject to normal levels of commercial fishing effort and those inside the NTZ were subject experimental potting of approximately 2000 pots per km^2 per year. Multivariate analyses revealed no difference in how assemblages changed over the four year period between areas subject to potting and those not fished. The study concluded no detectable effects of potting for lobster and crabs on the benthic assemblage over the time scale of the experiment. It is important to note that physical differences in NTZ and control locations are likely to complicate the detection of any changes in assemblage.

A study by Young *et al*. (2013), the details of which are also discussed in 6.2.1, consisted of a vulnerability analysis and survey work. The vulnerability analysis involved sensitivity mapping of different biotopes combined with mapping of fishing effort. A sensitivity score of 0 to 3 was assigned (0=none, 1=low, 2=moderate, 3 = high) and the following effort intensity thresholds were defined; very high (250+ pots per km^2/12 strings per km^2), high (175-250 pots per km^2/9-11 strings per km^2), moderate (100-175 pots per km^2/6-8 strings per km^2), low (50-100 pots per km^2/3-5 strings per km^2), very low (0-50 pots per km^2/0-2 strings per km^2) and none (0 pots per km^2/0 strings per km^2). Vulnerability to abrasion from potting was then defined as a function of sensitivity and exposure to fishing. Mapping revealed areas of moderate to high fishing intensity coincided with habitats of moderate sensitivity, resulting in approximately 3 km^2 considered to have high vulnerability to potting and 1 km^2 to have very high vulnerability. This analysis only applies during summer months when potting intensity it at its highest. The survey work, undertaken in in the Flamborough Head no-take zone (NTZ), designated in 2010, and a fished area, revealed a statistically significant difference in community assemblage between the NTZ and fished site was identified. A higher abundance of benthic taxa, namely Mollusca, Hydrozoa and Rhodophyta, were reported within the NTZ, the three of which accounted for 68% of the dissimilarity between the NTZ and fished site. Table 2 provides details of the differences in mean presence of different taxonomic groups. In the fished site, there was a higher percentage of bare substrate (7.2%), which may imply physical abrasion from pots could be removing sessile benthic epifauna. Contrary to expectation, the abundance of kelp species, *Sacharinna latissima*, was found to be higher in the fished site than the NTZ. The abundance of Bryozoans between sites was also found to be similar, suggesting potting pressure is unlikely to be impacting upon their abundance. The authors stated a degree of uncertainty must be associated with the survey due to unusually adverse weather conditions which occurred from January to March 2013. This led to the seafloor being scoured within both sites and subsequent reductions in epibiota across both sites. Prior to the spell of adverse weather, video footage gathered by divers' shows very high benthic cover of fauna and flora, which highlights the severity of damage. The extent of which the adverse weather influenced the outcome of the study is unknown.

Table 2. Summary of mean presence (% cover) of taxonomic groups in a no-take zone and fished area in Flamborough Head European Marine Site. Source: Young *et al*. (2013).

Site	Bryozoa	Hydrozoa	Decapoda	Mollusca	Ochrophyta	Rhodophyta
No-take zone	10.11	55.05	11.45	39.10	6.58	45.94
Fished area	13.92	36.79	8.50	29.36	20.37	31.60

Haynes *et al*. (2014) compared a dataset on the abundance of five sponge species (*Axinella dissimilis*, *Axinella infundibuliformis*, *Haliclona oculata*, *Stelligera stuposa* and *Raspailia ramosa*) from the Skomer Marine Nature Reserve collected during the autumn of 2006, 2008 and 2009, to pot density within a 50 m radius to assess the impacts of abrasion from potting. These species were identified as being susceptible to abrasion. Total species abundance and potting density (a proxy for abrasion) were tested and regression analysis revealed no significant relationship between sponge abundance and potting density. Regression analyses was also performed to examine potting density against sponge life strategy and morphotype diversity, as well as *Eucinella verrucosa* abundance (a potential indicator species for abrasion). The results reveal no significant relationship between any of these variables. Analysis of the data for testing and validation however proved inconclusive due to limited availability of suitable environmental and pressure data. The surveys were not designed to test to changes driven by a wide range of anthropogenic pressures and power to detect such changes was not a consideration of the original sampling design, meaning that existing datasets were not well suited for validation.

Stephenson *et al*. (2015; 2016) investigated the long-term impacts of potting on benthic habitats in the Berwickshire and North Northumberland Coast European Marine Site from 2002 to 2012. The study was split into a number of phases.

The first involved frequency analysis of biotopes from previously collected video footage for the purposes of condition monitoring (2002/03 and 2011), provided by Natural England, to examine if any biotope changes had occurred in relation to shellfish potting intensity. Data were extracted from previously collected video monitoring footage, undertaken in three transect corridors throughout the EMS (stratified by depth 0-10m, 10-20m, 20m+), and grouped into biotopes. These biotopes were analysed including the change in number, composition and range, to give an indication of the ecological health of the EMS. Species were recorded to the lowest taxonomic level and biotope classifications were assigned. It was hypothesised temporal changes (between 2002/03 and 2011) were related to shellfish potting intensity. Biotope richness varied slightly between years and transects, however non-significant differences were a result of rare biotopes. Biotope composition was similar between years and transects. Non-significant fluctuations in biotopes between years were attributed to natural variability and by the low frequency occurrence of rare biotopes. Overall, the number and range of biotopes was maintained between the two sampling periods (2002/03 and 2011), with the persistence of a few dominating biotopes; infralittoral kelp and circalittoral faunal and algal crust biotopes. The lack of observed change in biotopes between years meant fishing pressure as a cause of change was not investigated. Conclusions drawn from this analysis are limited due to the broad nature of biotope analysis and low number of sampling years. The methodology used did not allow for changes in abundance, species diversity or species composition of each biotope to be taken into account.

The second phase of the study involved an in depth analysis of video monitoring footage collected in 2002/03 and 2011, including changes in benthic community parameters in relation to potting intensity. Video monitoring footage, used in biotope frequency analysis (first phase of the study), was used to investigate changes in benthic community structure within specific biotopes between years, including taxonomic composition, species diversity and ecologically important species. Data was pooled and change across the whole EMS was explored to examine the effects of potting pressure. A lack of scale on the camera system used prevented collection of abundance data from the footage collected, so species presence/absence was used to describe communities. It was hypothesised that there was a link between biotic changes and potting pressure. This was tested by examining potting pressure effects on changes in benthic community structure of individual biotopes across the EMS between years (2012/03 and 2011). Potting pressure data, was categorised into two levels (low = 0 – 226 and high = 227 – 770 pots / month / km^2). The effect of potting pressure on species presence/absence between years was investigated using a mixed model. Overall, the results indicated no significant changes in species composition of biotopes within the EMS between years. Post-hoc analysis revealed the only biotope to exhibit change in species composition between years and across all transects was 'faunal and algal crusts on exposed to moderately wave-exposed circalittoral rock' (CR.MCR.EcCR.FaAlCr), thus indicating little change overall between 2002 and 2011. When incorporating 'fishing pressure' into the analysis, the same biotope exhibited an altered species assemblage and a significantly differing species composition between years. The author advised caution should be used during interpretation of results and temporal change is likely during this period, with further investigation recommended to determine specific links with pressures.

There was little evidence to suggest that species richness within biotopes differed between years, with differences only detected in '*Laminaria hyperborea* on tide-swept infralittoral mixed substrata' (IR.MIR.KR.LhypT.Pk), Species richness did not differ in response to fishing pressure however for this biotope (IR.MIR.KR.LhypT.Pk). In three out of ten biotopes, species richness differed between levels of fishing pressure (CR.MCR.EcCr.FaAlCr, CR.MCR.EcCR.FaAlCr.Bri and CR.MCR.EcCR.FaAlCr.Flu (*Flustra foliacea* on slightly scoured silty circalittoral rock)). Greater species richness was reported at low fishing pressures in nine out of ten biotopes when compared higher fishing pressures, although not all differences were significant. The exception to this was the 'Brittlestars on faunal and algal encrusted exposed to moderately wave-exposed circalittoral rock' (CR.MCR.EcCR.FaAlCr.Bri) biotope where low species richness suggests in areas of high fishing pressure that the assemblage structure may be affected. Further information however is required and conclusions were deemed as speculative. The results suggest that biotopes most likely to be impacted by fishing pressure are deeper, faunal and algal crusts as opposed to the shallower *Laminaria* biotopes. It does however remain uncertain as to whether fishing pressure is linked to species diversity as no clear pattern in species richness between years at different fishing pressure was observed. The low number of biotopes affected and the limited temporal data do not confirm whether fishing pressure impacts the environment or not. Analysis involving the reduced list of species, chosen in relation to those which can indicate biotope sensitivity to anthropogenic impacts, revealed no changes between years. From this data, it was concluded no deterioration in 'biotope health' from 2002 – 2011 occurred; the state of health of biotopes however could not be concluded. Overall it was concluded that, despite changes in species richness and composition of the biotope FaAlCr between years, there was little evidence of change in species composition or species richness of biotopes between years and it was not fully possible to investigate the role of fishing pressure in relation to community change. Results from this research suggest that on the scale of the EMS, impacts of small scale potting on epibenthic assemblages cannot be detected against the background of natural variability.

The third phase of the study aimed to quantify small scale potting impacts on two subtidal habitat types; 'Faunal and algal crusts on exposed to moderately wave-exposed circalittoral rock' (abbreviated as FaAlCr) and Laminaria hyperborea park with foliose red seaweeds on moderately exposed lower infralittoral rock (abbreviated as Lhyp.Pk) through in-situ experimental fishing using a BACI design (Stephenson *et al*., 2016). Historic intensively (187-265 pots month^{-1} km^{-2}) and lightly (0-139 pots month^{-1} km^{-2}) fished areas were chosen and subject to the same level of experimental potting (equivalent to 10,000 pots month^{-1} km^{-2}). Three sites were selected for each fishing pressure and habitat type. Due to a lack of suitable sites Lhyp.Pk habitat was only sampled for intensively fished areas. Each site consisted of two impact areas (25 x 10 m) and one control area (5 x 10 m). Baseline data was collected by divers using photoquadrats for impact and control sites. Following this, experimental fishing began in impact sites using a single parlour pot attached to a mainline rope, anchored by two weights. Parlour pots were soaked for a minimum of 24 hours and then hauled following local commercial methods. The impact and control areas were then resampled using the same method as the baseline data. Pots were left to soak, hauled and then sampled three times in each site. Benthos from the images collected were identified and recorded and percentage cover analysis was completed. Overall changes in percentage benthos cover were the same between treatments (control and experimental fishing) in both habitats and fishing pressures. Assemblages did not differ between baseline and control treatments for all sites, habitats and fishing pressures, allowing any changes found between baseline and impact treatments and not reflected in controls to be potentially explained by experimental fishing. Whilst significant interactions between baseline and impact treatments were reported, assemblages between control baseline and control impact treatments also differed and no differences were observed between impact and control impact treatments, indicating temporal change in community composition cannot be attributed to potting impacts. Only small differences were reported in overall abundance of different species between treatments in both habitat types. Percentage cover of species did not greatly differ between pre- and post-experimental fishing in impact or control areas, with no pattern in the benthos between treatments consistent with patterns predicted to occur from potting. FaAlCr habitats subject to intensive fishing activity exhibited a greater overall diversity and abundance of large erect species than areas of low fishing intensity showing that there is no evidence community composition differences between areas of different fishing intensity is caused by potting. The lack of short-term direct impacts shown by this study infer long-term direct impacts are unlikely in the habitats examined. The fourth phase explored pot movement over a 23 day period using novel acoustic telemetry methods (Stephenson *et al*., 2015) (as discussed in section 6.2.1)

Walmsley *et al*. (2015) analysed existing literature and ongoing studies on the impacts of potting on different habitats and features as part of a project funded by the Department for Environment Food and Rural Affairs in order to provide conclusions from evidence on whether potting may compromise the achievement of conservation objectives within European Marine Sites. The review of evidence found limited sources of primary evidence specifically addressing the physical impact of potting. Studies reported no or limited significant impacts from potting on subtidal bedrock reef and subtidal boulder and cobble reef, on brittlestar beds and subtidal mud. Particular evidence gaps were identified include those which relate to certain habitats (specifically maerl, seagrass, mussel beds, subtidal mixed sediments) and pot types (i.e. whelk pots and cuttle traps). Overall, the review of evidence found that most sub-features are unlikely to be of significant concern, particularly at existing potting intensity levels and limited impacts are likely to be undetectable against natural variability and disturbance.

Gall (2016) investigated the direct physical and ecological impacts of inkwell and parlour pots on reef features within the Start Point to Plymouth Sound and Eddystone Special Area of Conservation. The methodology used and physical impacts observed are reported in section 6.2.1. During periods of rope movement (in 51% of soaks), minimal damage to taxa was observed and limited to abrasion of *A. digitatum* and *E. verrucosa*. Five instances were reported during which damage from rope contact was evident. On four occasions (3.7% of hauls), rope caught on A. digitatum; leading to abrasion and removal of 2 individuals. Direct impacts from pots were observed for 14 out of 22 identified taxa, including all five indicator species (*Alcyonium digitatum*, *Cliona celata*, *Eunicella verrucosa*, *Pentapora foliacea* and branching sponges) and individuals from six taxa were removed. Removed taxa included *Alcyonidium diaphanum, A. digitatum*, *C. celata, P. foliacea* and *D. grossularia*. Significantly more species were not damaged than damaged or removed however and in the few instances where a pot landed directly on top of an individual, *E. verrucosa* was observed to 'bounce back' once the pot had passed; supporting observations made by Eno *et al*. (2001). Although there was some level of damage and removal caused by potting impacts, the study suggested that the reef in Start Point to Plymouth Sound and Eddystone SCI is currently being maintained in a favourable condition, thus achieving the sites conservation objectives, despite the presence of potting activity.

There are a number of ongoing pieces of research into the effects of potting on benthic habitats, including Adam Rees who at the University of Plymouth, and the Agri-Food and Biosciences Institute (AFBI).

The objectives of the study being conducted by Adam Rees include assessing the level of static gear likely to have a significant impact on benthic communities and mobile organisms associated with reef habitats, assessing how different gear intensities impact populations of target species (brown crab and European lobster) (see section 4.4.3) and to assess whether areas of no fishing can lead to spillover effects into surrounding areas. All of which are based in the Lyme Bay section of the Lyme Bay and Torbay SAC (Rees *et al*., 2016). This will be achieved by manipulating potting intensity across a set number of experimental areas (16 in total). Test areas measure 500 by 500 m and are located on mixed ground or rocky reef to allow for comparison. The four potting intensities being used include no potting, low density (5 to 10 pots), medium density (15 to 25 pots) and high density (30+ pots). Intensity calculations are based on the highest density of pots, which equates to approximately 30 pots per 0.25 km^2 (120 pots per 1 km^2). Based on the assumption pots are hauled three times a week (on average), the highest density of pots equates to 19,000 pot hauls per km^2 per year. Impacts on the benthic communities and mobile species are monitored using

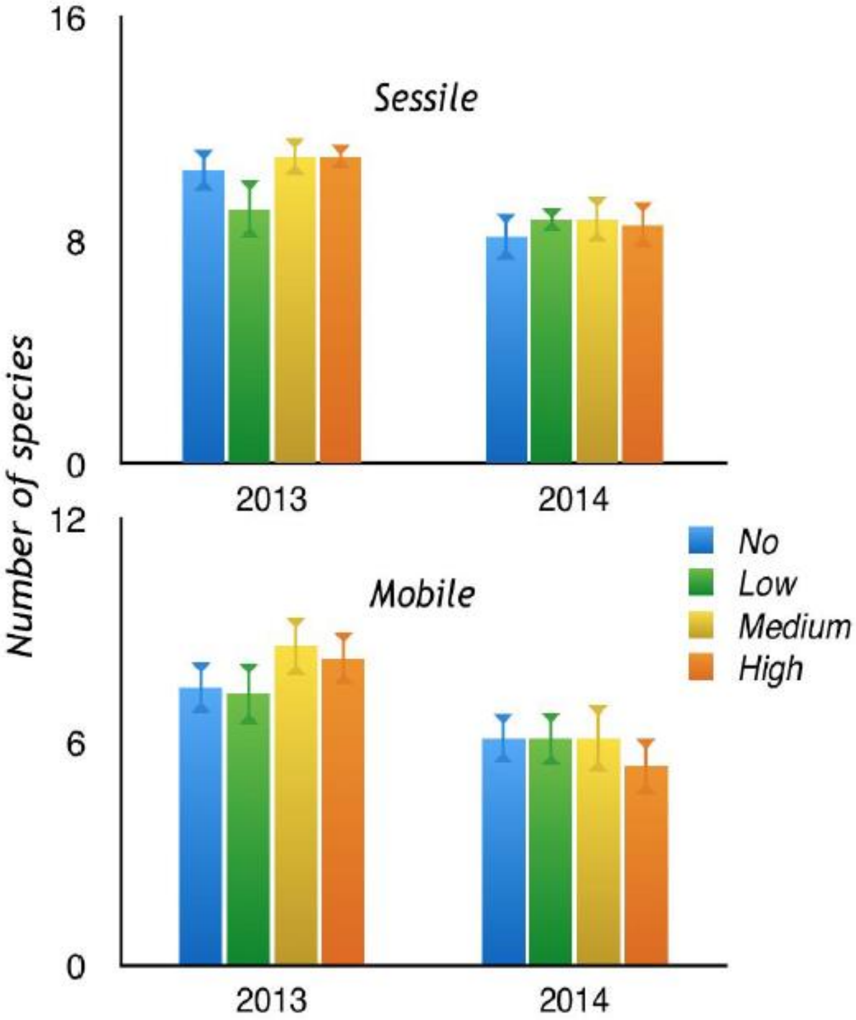

Figure 3. Changes in the number of sessile and mobile species between 2013 and 2014 in Lyme Bay, prior to and after a period of extreme weather (December 2013 to March 2014). Source: Rees, No Date.

underwater video sampling, including baited underwater video for mobile species. Data collection began in the summer of 2013 and the latest results contain information collected during summer 2013 to 2015. Adverse weather experienced during December 2013 to March 2014 interrupted the project with many of the key sessile reef features and associated mobile species being significantly reduced as a result of increased wave action from store events (Figure 3). Most reef areas were of a similar condition and represented a severely naturally disturbed state, likened to towed gear impacts and much more severe that any impacts which may occur as a result of the potting density study. Impacts from the period of adverse weather have removed any evidence of impact that the different levels of potting intensity may have started to show. As such project milestones were pushed back and an extra year was added to the project. Whilst this period of adverse weather served to interrupt the project it provides a unique opportunity to look at recovery under different fishing intensity scenarios.

Between 2013 and 2014, the overall abundance and species richness of sessile fauna was significantly reduced across all potting intensities and in 2015 remained at a consistent level showing no treatment effects. In areas of medium and high potting intensities abundance and species richness were less than 2013 levels. It is important to note that in 2013, prior to the period of adverse weather, both mean abundance and species richness were higher in areas of medium and high gear intensities than no potting and low gear intensities. Decreases in abundance between 2013 and 2014 were mirrored in the following key indicator species and species group; dead man’s finger (*Alcyonium digitatum*), Ross coral (*Pentapora fascialis*); the white sea squirt (*Phallusia mammillata*), encrusting species and large bodied erect species. Other species (Pink sea fans (*Eunicella verrucosa)* and the king scallop (*Pecten maximus*) did not exhibit a significant decline. This indicates the Pink sea fan have a tough exoskeleton and as such are more resilient to physical damage. In 2015, *P. mammillata*, a relatively fast growing species, had recovered significantly across all treatments exhibiting no treatment effect, whilst the slower growing *P. fascialis* only increased significantly in areas of no potting (similar to 2013 levels) when compared to other potting intensities. It is important to note however in areas of other potting intensity, some level of recovery was also observed. This indicates *P. fascialis* benefitted from a period of no potting, particularly in relation to its recovery. This is to be confirmed by 2016 results. Statistically, other species did not exhibit any signs of recovery but remained at a consistent level across all potting intensities.

Mobile fauna abundance and species richness declined across all treatments between 2013 and 2014 and between 2014 and 2015 increased in all treatments. Such declines may be associated with the removal of sessile reef species. Significant treatment effects were reported in areas of no potting and medium intensity potting, with higher abundances reported in both. Grouped large fish declined in all treatments (except no potting) between 2013 and 2014, remaining at similar levels in 2015 with no sign of recovery; perhaps caused by removal of key reef species which are still recovering.

The study being completed by Agri-Food and Biosciences Institute is assessing the impacts of potting on different SAC features in Northern Ireland. These include rocky reefs with sponges, *Modiolus* beds, maerl and sandbanks. The project is combining ecological data with other data sources such as fishing pressure, allowing experimental work to be extrapolated to what is occurring at a fishery scale. The project has also focused on the experimental deployment of pots with cameras and accelerometers with associated faunal analysis. Although the research is still in progress, preliminary results indicate a lack of effect on the habitats mentioned above.

6.2.2.2 Selective extraction of (target) species

The selective extraction of species refers to the removal of a species or community and includes the removal of a specific species/ community/ keystone species in a biotope. Fishing leads to the removal of certain species from an ecosystem. In the context of this assessment, potting targets four species of wrasse; Corkwing, Goldsinny, Rock cook and particularly Ballan between 10 and 30 cm in length. The mesh size used in salmon farms means that wrasse less than 10 cm (total length) are not retained and as a consequence restricts the capture of wrasse to 10 cm or greater (Treasurer, 1996; Varian *et al*., 1996; Sayer *et al*., 1996a). At the beginning of the 2017 season, eight boats operating within the fishery solely targeted ballan wrasse in the size range of 12 to 28 cm. Later in the 2017 season, a further two vessels started fishing for all four wrasse species in the size range of 10 to 30 cm. It is anticipated in the 2018 season, eight boats will operate within the fishery solely targeting ballan wrasse.

6.2.2.2.1 Population effects

The life history characteristics and reproductive strategies employed by each wrasse species is different (Table 3). This is particularly important when considering the potential population impacts of the wrasse fishery, as impacts on each species are likely to differ from one another (Skiftesvik *et al*. 2014). The following section will further explore the individual life history characteristics, reproductive strategies, ecology and genetics of each species and discuss the potential impacts on each, using scientific literature where available.

Table 3. Summary of the life history characteristics and reproductive strategies employed by five wrasse species which occur on the South Coast, not including Baillon's wrasse.

Characteristic	**Ballan wrasse** *Labrus bergylta*	**Corkwing wrasse** *Symphodus melops*	**Goldsinny wrasse** *Ctenolabrus rupestris*	**Cuckoo wrasse** *Labrus mixtus*	**Rock cook wrasse** *Centrolabrus exoletus*
Maximum age	29 years – Male 25 years – Female (Dipper *et al*. 1977)	9 years – Male (Darwall *et al*. 1992) 7+ years – Female (Sayer *et al*. 1996a)	14 years – Male 20 years – Female (Sayer *et al*. 1995)	20 years (Muus & Nielsen, 1999)	8 years – Male 9 years – Female (Darwall *et al*. 1992; Treasurer, 1994)
Maximum length	65.9 cm (IGFA, 2001)	28 cm (Quignard & Pras, 1986)	21 cm (Halvorsen *et al*. 2016)	40 cm (Bauchot, 1987)	19 cm (Skiftesvik *et al*. 2015)
Age at maturity	6-9 years – Female 6-9 years -Male (Darwall *et al*. 1992)	2-3 years – Female (Darwall *et al*. 1992) 1-3 year – Male (Uglem *et al*. 2000; Matland, 2015*)	2-3 years – Female (Darwall *et al*. 1992) 3 years – Male (Matland, 2015*)	2 years – Female 6-9 years – Male (Darwall *et al*. 1992)	2 years – Female (Darwall *et al*. 1992; Matland, 2015*) 2 years – Male (Matland 2015*)
Size at maturity	16-18 cm – Female 28 cm – Male (Darwall *et al*. 1992)	7-10 cm (Fishbase; Darwall *et al*. 1992) 9 cm – Female 14 cm – Male (Matland, 2015*)	9.5 cm (Darwall *et al*. 1992) 8 cm – Females 9 cm – Males (Matland, 2015*)	16 cm – Female 24 cm – Male (Darwall *et al*. 1992)	9 cm – Males 8.5 cm – Females (Matland, 2015*)
Spawning period (Atlantic)	April – August (Darwall *et al*. 1992)	April – September (Darwall *et al*. 1992)	April – September (Darwall *et al*. 1992)	May – July (Darwall *et al*. 1992)	May – August (Darwall *et al*. 1992)
Reproductive strategy	Hermaphrodite (Darwall *et al*. 1992)	Gonochoristic (Darwall *et al*. 1992)	Gonochoristic (Darwall *et al*. 1992)	Hermaphrodite (Darwall *et al*. 1992)	Gonochoristic (Darwall *et al*. 1992)

* Figures reported from Matland (2015) represent 'critical age' and 'critical length' which is the point at which 50% of the sample are sexually mature.

Ballan wrasse

Ballan wrasse attain the greatest size and age of all above-mentioned five wrasse species. The species is a monandric protogynous sequential hermaphrodite, meaning the species starts life as a female and a percentage of which change into male with no sex reversal thereafter (Sjolander *et al*., 1972; Darwall *et al*., 1992; Leclercq *et al*. 2014). Males are territorial and within each territory a dominant male will guard a harem of several females, whom which the male will mate with (Sjolander *et al*., 1972; Hillden, 1984; Darwall *et al*., 1992; Muncaster *et al*. 2010; Leclercq *et al*. 2014). During the spawning period benthic eggs are laid over temporary nests, built by female fish (Hillden, 1984; Darwall *et al*.1992; Muncaster *et al*. 2010).

The change in sex is believed to be largely driven by social cues (Leclercq *et al*. 2014) and also associated with size, although a greater size is not necessarily a prerequisite for sexual inversion (Muncaster *et al*. 2013). When held in captivity, it was observed that the removal of large males induced smaller females within the group to change sex (Halvorsen, 2016). Sexual inversion may also be triggered when a harem of females becomes too large for the male to dominate (Muncaster *et al*. 2013). The size and age at which sex inversion takes place has been reported to occur over a wide range and differs between studies (Table 4) (Villegas-Ríos *et al*. 2013a), with current literature indicating sex change generally occurs before reaching 40 cm (total length) and not until after 6 years in age (Muncaster *et al*. 2013). Muncaster *et al*. (2013) reported the timing of sexual inversion after the conclusion of the spawning period.

Table 4. Age and size range of sexual inversion in ballan wrasse (*Labrus bergylta*) from five studies.

Study	Age Range (years)	Size Range (cm)	Location
Quignard, 1966	-	27-41	French side of the English Channel
Dipper *et al*., 1977	5-14	25-43	Isle of Man
Villegas-Ríos *et al*. 2013a	7.4-11.8*	36-47.2*	Galicia, NW Spain
Muncaster *et al*. 2013	-	28-42 (median 36)	Western Norway
Leclercq *et al*., 2014	6-13	28.2-37.2	Scotland & Norway

* This study reports sexual inversion figures for two morphotypes; plain and spotted. Plain individuals were estimated to undergo sex change at 7.4 years and 36 cm and spotted individuals at 11.8 years and 47.2 cm.

Potential implications of removing ballan wrasse of a certain size (12 to 28 cm) can be inferred from life history parameters and existing literature. At the preferred size of removal, a proportion of the individuals (12-18 cm) caught will be female ballan wrasse that have not had the chance to spawn. Individuals from 18-28 cm are likely to be female ballan wrasse who have had the chance to spawn at least once. Based on the reported size range at which sexual inversion occurs (Table 4), the majority of individuals caught are likely to be female with the potential for a small number of males. Muncaster *et al*. (2013) reported 60% of fish caught during the breeding season (late April to June) were female. Leclercq *et al*. (2014) reported a potential trend towards fewer older and larger females which may lead to a phenotypic alteration in the age and size of sexual inversion. The targeting of one sex over another may also lead to potential impacts on breeding and subsequent recruitment (Muncaster *et al*. 2013).

Corkwing wrasse

Corkwing wrasse are a gonochoristic species, meaning the species has distinct sexes which do not change (Dipper & Pullin, 1979; Darwall *et al*., 1992). Male corkwings are very territorial and attract females into their nests, where they may lay batches of eggs (Costello *et al*., 1991; Skiftesvik *et al*.

2015). Nests are built by the male corkwing, typically using seaweed in a rock crevice (Costello *et al*., 1991). The species has two distinct male strategies; the majority build a complex nest and guard the eggs. A small proportion develop as 'accessory' or 'sneaker' males which mimic females and perform sneak spawning, whereby they pair with a female in a dominant male's territory or join a spawning pair (Warner & Robertson, 1978; Costello, 1991; Uglem *et al*., 2000; Halvorsen, 2016). At the same age, sneaker males are generally smaller and can make up between 3 and 20% of the population (Uglem *et al*., 2000). Maturation of females tends to occur earlier than in males and because of this males are typically larger than females of the same age (from 1 to 4 years) (Quignard, 1966; Dipper, 1976; Darwall *et al*., 1992; Treasurer, 1994).

Potential implications of removing wrasse of a certain size (>10 cm) can be inferred from life history parameters and existing literature. At the preferred size of removal, it is likely that the vast majority of females will have had a chance to reproduce before being caught and therefore those being removed are likely to be sexually mature. Males are reported to achieve sexual maturity at 14 cm and therefore those below this size are unlikely to have had a chance to reproduce before being caught. The reproductive biology of corkwing wrasse (i.e. larger size of nesting males than females and sneaker males) make the species vulnerable to size selective harvest (Darwall *et al*., 1992; Sayer *et al*., 1996a; Uglem *et al*., 2000; Halvorsen, 2016). Halvorsen *et al*. (2016) reported significantly larger body sizes for nesting males than females and sneaker males, with the largest differences in the northernmost populations on the western coast of Norway. This sexual size dimorphism was caused by a fast growth and delayed maturation in nesting males compared to females and sneaker males (Halvorsen *et al*., 2016).

The selective removal of larger fish, most likely to be dominant territorial males, are likely to affect social structure (influence on the frequency of sneaker males), reduce egg survival (through the removal of nest-guarding males), lead to biased sex ratios (in favour of females) and decrease the average size and age at first maturity (Darwall *et al*., 1992; Halvorsen, 2016). Halvorsen *et al*. (2016) reported the 12 cm minimum legal-size limit in Norway led to different levels of protection for nesting males, females and sneaker males due to differences in body size and failed to protect any mature nesting populations in five out of 8 populations. Further investigation by Halvorsen *et al*. (2016) found dominant nesting males to have a higher vulnerability of capture, regardless of body size, with a possible explanation related to physiological or behavioural differences between sexes. Halvorsen *et al*. (2017) also investigated differences in catch per unit effort, size, age and sex ratio of goldsinny and corkwing wrasse of populations within marine protected areas (MPAs) (not subject to fishing) and control areas (open to fishing). Catch per unit effort of individuals above the minimum size limit was higher in three out of the four MPAs. The relative difference between the two areas ranged from -16% to 92%. The size and age of individuals within MPAs were significantly greater than in control areas. No differences in sex ratio between the two areas were reported.

In the Irish wrasse fishery, Darwall *et al*. (1992), Deady *et al*. (1993) and Varian *et al*. (1996) reported a decline in catch per unit effort (CPUE) for corkwing in years following exploitation (Sayer *et al*., 1996a). More specifically, Darwall *et al*. (1992) reported a reduction in males greater than 13 cm in length in the second year of sampling, potentially suggesting the depletion of large males. Like Halvorsen *et al*. (2016), Darwall *et al*. (1992) noted catches of corkwing were male biased and that males were on average larger than females.

Goldsinny wrasse

Like corkwing, goldwinny wrasse are a gonochoristic species and have 'accessory' males who mimic females and perform sneak spawning (observed in two thirds of spawnings) (Hillden, 1981; Darwall *et al*., 1992). Although the males maintain territories, spawning occurs within the water column as the eggs of goldsinny wrasse are pelagic, as opposed to benthic eggs of the other temperate wrasse species (Hillden, 1981; Darwall *et al*., 1992). Males will often use their territory to spawn, as well as for foraging (Hillden, 1981). During the spawning period a single male will spawn with several females, despite a 50:50 sex ratio (Hillden, 1981). Females will then stay within the vicinity of the males territory with which they have spawned (Hillden, 1981).

Potential implications of removing wrasse of a certain size (>10 cm) can be inferred from life history parameters and existing literature. At the preferred size of removal, all individuals should have had the chance to reproduce before being caught and therefore all individuals being removed will be sexually mature. Like corkwing wrasse, male goldsinny wrasse have a tendency to grow at a slightly greater rate and size selective harvesting of these individuals is likely to influence age structure and sex ratios (Sayer *et al*., 1996b; Varian *et al*., 1996; Halvorsen *et al*., 2016).

Halvorsen *et al*. (2017) investigated differences in catch per unit effort, size, age and sex ratio of goldsinny and corkwing wrasse of populations within MPAs (not subject to fishing) and control areas (open to fishing). Catch per unit effort of individuals above the minimum size limit was 33% to 65% higher in MPAs. Goldsinny were not significantly older or larger within MPAs relative to control areas and no differences in sex ratio between the two areas was reported. Goldsinny is smaller in size, when compared to the other wrasse species, and appears to benefit from the minimum size limit (11 cm), which applies outside of the MPAs.

In the Irish wrasse fishery, Darwall *et al*. (1992) and Deady *et al*. (1993) reported a decline in catch per unit effort (CPUE) for goldsinny in years following exploitation.

Cuckoo wrasse

Cuckoo wrasse are diandric protogynous hermphrodites (Costello, 1991). This means that only a proportion of females change into males (Costello, 1991). Sexual inversion is associated with a distinct change in colour (Dipper & Pullin, 1979) and is reported to occur after reaching a certain size (Irving, 1998), over four years of age (Costello, 1991) or between 7 and 13 years (Irving *et al*., 1998). Quignard (1966) reported all individuals over 29 cm and 10 years of age to be males. Sex change may also been influenced the sex ratios in the local population, with most having more females than males (Naylor, 2005). Males will build and guard a nest (Costello, 1991).

Potential implications of removing wrasse of a certain size (>10 cm) can be inferred from life history parameters. This species is not targeted and the wrasse fishery guidance states all live Cuckoo wrasse should be returned to the fishery immediately. If the species were targeted the potential implications would be similar to ballan wrasse. At the current size of removal a large proportion of individuals (10-16 cm) would be immature female cuckoo wrasse who have not had a chance to spawn. Individuals over 16 cm are likely to be mature females who have had the chance to spawn at least once. There is limited information surrounding the size range of sexual inversion. Darwall *et al*. (1992) reported sexual maturity of males at 24 cm. In this case, there would be the potential to remove a proportion of males from the population. The fishery would therefore likely remove immature females, mature females and mature males. Like ballan wrasse, this would have likely implications for the timing of sexual inversion and the targeting of different sexes could lead to potential impacts on breeding and subsequent recruitment.

Rock cook wrasse

Rock cook wrasse are believed to be a gonochoristic species, with no evidence of sex change (Dipper, 1987). It is the least studied of the five above-mentioned wrasse species so details of its reproductive strategy are not well known. Eggs of the rock cook are sticky and benthic, like other wrasse species, except for goldsinny, and so it is believed the male may build a nest (Costello, 1991).

Potential implications of removing wrasse of a certain size (>10 cm) can be inferred from life history parameter and existing literature. Similar to goldsinny wrasse, at the preferred size of removal, all individuals should have had the chance to reproduce before being caught and therefore all individuals being removed will be sexually mature. Like corkwing and goldsinny wrasse, males grow faster than females (Taki, 1974) and size selective harvesting of these individuals is likely to influence age structure and sex ratios (Sayer *et al*., 1996b; Varian *et al*., 1996; Halvorsen, 2016).

All species

There is the potential for sex-selective harvesting to take place in all above-mentioned five species. The sexual size dimorphism associated with all gonochrositic species, particularly corkwing, means the fishery will lead to the removal of larger males. The hermphroditic nature of ballan and cuckoo wrasse, combined with greater sizes at sexual maturity, mean larger mature and smaller immature females are removed, with some concern over the removal of males in cuckoo wrasse. The greater size of sexual inversion for ballan wrasse however means this is less of a concern for this species.

Size selective harvesting has a variety of implication related to population dynamics, demography and reproduction (Halvorsen, 2016). Firstly, it can truncate age and size distributions (Halvorsen, 2016). The depletion of older and larger individuals, particularly more fecund females, can influence recruitment and the ability to adapt to a changing environment (Longhurst, 2002; Hixon *et al*., 2014). In species with parental care (corkwing, cuckoo and ballan male wrasse), selective removal of those which exhibit this trait can directly influence the level of offspring survival (Suski *et al*., 2003; Sutter *et al*., 2012). Additionally, changes in sex ratio can also lead to sperm or egg limitation and impact on mating behaviour (i.e. reduction in encountering mates) and sexual selection (Rowe & Hutchings 2003; Alonzo & Mangel 2004; Kendall & Quinn 2013).

The varied life histories and reproductive strategies employed by all the different wrasse species mean that fishing is likely to effect each differently (Skiftesvik *et al*., 2014). There are however a number of potential issues common to all species. The first is the demand for wrasse as cleaner fish coincides with their spawning season in spring and early summer (Costello, 1991). Skiftesvik *et al*. (2014) reported that fishing during the summer leads to a higher incidence of wounds and greater mortality, with female corkwing believed to be particularly vulnerable. The survival rate (75% mortality) of wrasse captured in June (i.e. during the spawning season) and subsequently kept in tanks was much lower than those captured in September (5% mortality). This led the authors to conclude that wrasse should be protected during the spawning season.

The second issue relates to the territorial behaviour and high level of site fidelity exhibited by all above-mentioned five wrasse species (Costello, 1991; Skiftesvik *et al*., 2014). Villegas-Ríos *et al*. (2013b) reported a home range of 0.091±0.031 km^2 (91,000 m^2), with a core area of 0.019±0.006 km^2 for ballan wrasse. Other studies have reported territory sizes of 300 m^2 during the spawning period (Sjolander *et al*., 1972). Despite difference between studies, in relative terms this is still a small range and demonstrates the sedentary behaviour associated with this species (Villegas-Ríos *et al*., 2013b). Territory sizes for other species are smaller than that reported for ballan wrasse. Hillden (1981) reported an average territory size of 1.4 m^2 for goldsinny with no change in the size or form during the study period (May to September). The territory size for corkwing is around 10 m^2 (Sjolander *et al*., 1972), although individuals do travel up to 50 metres away from nesting sites (Potts, 1985).

A high site fidelity and small home ranges/ territories can lead to local depletion and limited potential for replenishment from nearby populations (Halvorsen, 2016). The size structure of the wrasse population will be an indicator of fishing intensity (Shepherd *et al*., 2010). If populations are genetically isolated from one another, there is likely to be a strong selection for slower growing and smaller individuals in populations within heavily fished areas (Skiftesvik *et al*., 2014). In addition to this, populations with poor genetic diversity are often associated with inbreeding, reduced fitness and less evolutionary potential (Frankham, 2002; D'Arcy *et al*., 2003). Such implications may be true for corkwing wrasse and goldsinny wrasse as populations have been shown to be genetically differentiated along the cost of Norway (Sundt & Jorstad, 1998; Knutsen *et al*., 2013), but less so for cuckoo and ballan wrasse with studies revealing genetically differentiated populations on a much larger spatial scale between the Atlantic and Scandinavia (Robalo *et al*., 2011; D'Arcy *et al*., 2013). This can be attributed to the relatively long planktonic larval stages observed in ballan and cuckoo wrasse which is likely to lower the level of genetic differentiation between neighbouring areas (D'Arcy *et al*., 2013).

Small MPAs (<0.5 km^2) can afford effective and long-term protection for species with high site fidelity and small home ranges/territories, like those exhibited by the above-mentioned five wrasse species (Morel *et al*., 2013). Halvorsen *et al*. (2017) explored the potential use of small MPAs (0.6-5.3 km^2), as no take-zones, as a management tool for the protection of targeted wrasse species (goldsinny and corkwing wrasse) in Norway. The study reported a greater prevalence of individuals above the minimum size limits for both species and concluded small MPAs have potential as a tool for maintaining natural population sizes and structure.

6.2.2.2.2 Ecosystem-wide effects

Rocky reefs and their associated algal cover form at least one, if not the only habitat, of all above-mentioned five wrasse species (Costello, 1991). Although there are differences in the level of exposure and depths favoured by each species (Costello, 19991; Skiftesvik *et al*., 2015). Along the Norwegian coast, wrasse make up the most numerous fish within shallow water communities (Halvorsen, 2016), although their importance in such a complex coastal ecosystem is unclear (Skiftesvik *et al*., 2014).

In order to identify possible wider ecosystem effects their removal could have on this habitat type it is important to establish their role and position within the food web. Wrasse are considered to belong to a functional group known as 'coastal mesopredatory fish' (Bergström *et al*., 2016). Coastal mesopredatory fish are defined as mid-trophic level demersal and benthic species with a diet consisting predominantly of invertebrates (Bergström *et al*., 2016). Mesopredatory fish serve as a food source for higher trophic levels (i.e. piscovorous fish) and perform a regulating function on lower trophic levels (Sieben *et al*. 2011; Baden *et al*. 2012; Östman *et al*. 2016; Bergström *et al*., 2016). Thus their abundance is highly likely to have important effects on other parts of the ecosystem web due to their central role within it (Bergström *et al*., 2016).

Wrasse graze on animal growth found on seaweeds and rocks and are important predators of hard-shelled animals, such as crustaceans and molluscs, leading to a diverse diet and making all species carnivorous (Costello, 1991; Sayer *et al*. 1995;1996a; Deady & Fives 1995). Dietary studies have revealed that decapods, predominantly *Cancer pagurus* and *Carcinus maenas*, represent a key food category for ballan wrasse (Dipper *et al*., 1977), whilst one of the main food categories for corkwing wrasse is gastropods molluscs; *Gibbula umbilicalis* and *Helcion pellucidum* in particular (Sayer *et al*., 1996a). The diet of rock cooks have been found to be dominated by bivalve molluscs and amphipods (Sayer *et al*., 1996a) and dominant food items for goldsinny, as well as larger corkwing, including mussels and barnacles (Deady & Fives, 1995; Sayer *et al*., 1995). The removal of wrasse, in their role as grazers and predators of epifaunal species, can lead to top-down effects (Bergström *et al*., 2016). Top-down effects include a loss of grazing control, whereby wrasse feed upon epifaunal species which in turn graze on algal species (Bergström *et al*., 2016). A loss of grazing control, caused by the removal of wrasse species, can therefore lead to an increase in epifaunal growth and subsequent increases in the grazing of algal species.

In coastal areas of temperate regions, an important example of the loss of grazing control is the overgrazing of algal assemblages (particularly kelp forests) by sea urchins, whose populations have increased as a result of fisheries-related decline in predatory fish (Figueiredo *et al*. 2005). This concern has recently been cited by Coghlan *et al*. (2017) over the removal of wrasse for cleaner fish in salmon farms. Figueirdo *et al*. (2005) assessed the importance of sea urchins in the diets of ballan wrases in the Azores and found that echinoderms, particularly echinoids, were the second most important prey group and accounted for 41.5% (by weight) of all identified food items and the importance of this prey group increased with fish size. Prior to this study, the importance of echinoderms in the diet of ballan wrasse had not been recorded. The study concluded that ballan wrasse are likely to provide a very significant contribution to the control of sea urchin populations within the Azores and that a reduction in the mean size of fish (often a consequence of fishing) may lead to a signfiicant decline in sea urchin predation and subsequent sea urchin profileration and overgrazing. Another study, on the diet of corkwing wrasse on the west coast of Scotland, reported sea urchin spines in over 5% of individuals examined; much less than the reported for ballan wrasse in the Azores (Sayer *et al*., 1996a).

A number of studies have examined the relationship between wrasse predation on epifaunal invertebrate grazers of brown macro algae found in rocky areas in New Zealand. Using mesocosm experiments, Perez-Matus and Shima (2010) investigated the interaction of two wrasse species, *Notolabrus celidotus* and *N. fucicola* and found both species had a positive indirect effect on the giant kelp, *Macrocystis pyrifera*, through the consumption and behavioural change of amphipods, respectively. Overall, the presence of the *N. celidotus* and *N. fucicola* led to a 5-fold and 2-fold decrease, respectively, in the number of grazing marks (Perez-Matus & Shima, 2010). Newcombe and Taylor (2010) conducted similar mesocosm experiments using *N. celidotus* and three species of brown macroalgae; *Ecklonia radiata*, *Carpophyllum flexuosum* and *C. maschalocarpum*. The study reported a reduction (to 7-20% of predator-free densities) in epifaunal grazing on algae species as a result of predation. When epifaunal densities were reduced (artificially or by fish predation), algal biomass was greater (due to less damage) but more heavily fouled. When predatory fish were not present, macroalgae sustained increased damage and biomass was reduced to 21-74% of epifauna-free algal biomass. In the study a trophic cascade was apparent, as the addition of predator led to a reversal in the decline of primary producer biomass caused by herbivores (Newcombe & Taylor, 2010). The results of the study were not found to be consistent with field surveys of varying fish densities.

The above studies demonstrate the potential importance of top down control of epibenthic grazers and how the removal of wrasse might lead to potential trophic cascades. The applicability of these studies and their results however must be considered with caution, particularly with respect to study conducted by Figueirdo *et al*. (2005). This is due to the likely differences in epifaunal assemblages found in the Azores and found on the south coast of the UK, and thus the importance of echinoderms as a component of the species diet is likely to be less considerable.

Wrasse also serve as a prey species for gadoids, sea birds and mammals (seals and otters) (Steven 1933; Nedreaas *et al*. 2008; Helfman *et al*., 2009; Smale, 2013). At low abundances of piscivores, the distribution of coastal meopredatory fish and piscivores is tightly coupled (Bergström *et al*. 2016). A reduction in wrasse is therefore likely to lead to subsequent reduction and/or and change in the distribution of species which feed on them. Halvorsen (2016) reported goldsinny growth rates to be negatively related to population and the abundance of coastal cod. This demonstrates that the potential implications of wrasse removal are likely to be complex.

6.2.2.2.3 Cleaning behaviour

There is relatively limited information surrounding the wild cleaning behaviour of ballan, corkwing, goldsinny, cuckoo and rock cook wrasse and field observations of the behaviour is rare (Costello, 1991). A number of early observations were made of rock cooks cleaning behaviour of ballan wrasse and grey mullet (*Chelon labrosus*) in the wild (Potts, 1973; Costello, 1991). These was confirmed by later observations made by Henriques and Almada (1997) at Arrabida, Portugal. Rock cook were observed to clean a total of 12 species, with corkwing and ballan wrasse being the most frequently cleaned (Henriques & Almada, 1997). From this study, it was reported that rock cook wrasse are facultative cleaner fish, with cleaning acts representing 7% of all feeding acts that were observed and an incidence rate of 11 per hour per host; similar to the number reported for tropical fish (12 acts per hour per host) (Grutter, 1995).

Similar early observations were made of wild goldsinny cleaning behaviour on the Swedish coast (Hillden, 1983), Lough Hyne, Ireland (Hutcherson, 1990) and Black Sea (Darkov & Mochek, 1980). The former two were involved the cleaning of ballan wrasse (Costello, 1991). Hillden (1983) showed goldwinny wrasse to be a facultative cleaner fish. A total of 24 cleaning acts were observed over a 6 year period (1975-1981) (Hillden, 1983).

Anecdotal observations of wild cleaning behaviour on the south-coast of the UK have also been made by Naylor (2005) who noted rock cook and goldsinny wrasse acting as cleaner fish on larger wrasse (i.e. Ballan wrasse), including the removal of parasites from their flanks, sometimes in small groups. Certain locations act as ‘cleaning stations’ where cleaning behaviour is regularly observed. Such locations include boilers on shallow-water wrecks, cleaning stations.

In aquaria, corkwing, goldsinny and rock cook were recorded to exhibit cleaning behaviour (Potts, 1973; Samuelsen, 1981). The species cleaned by wrasse varied and included plaice, black bream, red bream, mackerel, goldsinny wrasse, ballan wrasse and angler fish (Costello, 1991). The early observations of wrasse cleaning behaviour, made in the wild and aquaria stimulated interest in their use as cleaner fish in the salmon farming industry as a way to control ectoparasites (Henriques & Almada, 1997). Introductory experiments in tanks and aquaria found that goldsinny, rock cook and female cuckoo wrasse as facultative cleaners of lice infested salmon (Bjordal, 1988; Bjourdal *et al*., 1991). Additional observations of cleaning behaviour of juvenile ballan wrasse and cuckoo wrasse were made by Potts (Bjordal, 1991). The observations of cleaning behaiour obtained in fish farms and other captive conditions are likely to be poor predictors of behaviour of the same wrasse in nature and vice versa (Henriques & Almada, 1997).

Cleaning behaviour of fish is widely recognised as an integral part of maintaining overall reef health by removing parasites and cleaning damaged tissue from fish and other marine organisms (Natural

England, 2017). The removal of significant numbers of wrasse could have adverse impacts of species that require cleaning, and subsequently the overall health of the reef (Natural England, 2017). The facultative cleaning behaviour of rock cock and goldsinny wrasse and limited observation of cleaning behaviour in the wild however implies the cleaning behaviour carried out by the different wrasse species is poorly understood within the ecosystem. Evidence from tropical ecosystems demonstrates the role of cleaning behaviour of certain wrasse species (summarised in section 6.2.2.2.4), but further investigation is necessary to better understand its role and importance within temperate ecosystems.

6.2.2.2.4 Evidence of cleaning behaviour in tropical ecosystems

In tropical systems, parasitic sea lice have been shown to have a number of deleterious effects on coral reef fish (i.e. Finley & Forester, 2003; Grutter *et al*., 2011). Over a 5 month field study, Finley and Forester (2003) reported a significant reduction in growth (66%) and gonad mass (68%) and increase in mortality by a factor of 1.8 in the bridled goby, *Coryphopterus glaucofraenum*, as a result of a copepod microparasite infecting the gills. Parasitism was associated with an increase in gill ventilation rate and subsequent reductions in feeding. Similarly, Grutter *et al*. (2011) reported increased respiration (35% higher oxygen consumption rate) in resting juvenile damselfish (*Pomacentrus amboinensis*) parasitized with one gnathiid isopod, as well as reductions swimming speed, with parasitized individuals ceasing to swim before uninfected individuals in 77% of trials. When placed into their natural setting, parasitized individuals disappeared first in 67% of trials, thus potentially leading to an indirect affect on the successful establishment of juvenile fishes as they move from the pelagic environment to reefs.

The presence of cleaner fish in tropical reef systems has been shown to have a significant effect on the abundance parasitic sea lice (Grutter, 1996). Grutter (1996) examined the cleaning behaviour of Blue streak cleaner wrasse (*Labroides dimidiatus*) on the Blackeye thicklip wrasse (*Hemigymnus melapterus*) infected with gnathiid isopods at Lizard Island, Great Barrier Reef. Based on predation rate and time spent inspecting the host fish, it was estimated *L. dimidiatus* removed 61±5 per day; 6 times the number of gnathiids found per individual host fish (11±3). Such a high level of removal occurs due to the high infection rates of gnathiids, with gnathiid abundance shown to double in less than 6 days. As such, the high predation rate relative to the number of gnathiids on fish and their infection rate demonstrate *L. dimidiatus* have a significant effect on gnathiid abundance on infected host fish.

It may be expected, as shown from the deleterious effects sea lice can have on coral reef fish, the lack of cleaner fish may have negative implications on host fish populations. In a long-term study (over 8.5 years) conducted at Lizard Island, Waldie *et al*. (2011) reported a shift in size distribution to smaller damselfishes (Pomacentridae) in areas free of cleaner wrasse (*L. dimidiatus*). The same study also revealed implications on the overall coral reef fish community. Significant changes in community parameters were also observed, with a reduction in the abundance (37%) and richness (23%) of resident fishes in areas free of cleaner wrasse. Similar reductions in abundance (23%) and species richness (33%) of visitor fishes were also observed. Bshary (2003) reported similar findings, with significant declines in fish diversity 4 to 20 months after the removal of *L. dimidiatus* from patch reefs at Ras Mohammed National Park, Egypt. The immigration or experimental addition of cleaner wrasse led to a significant increase in fish diversity within 2 to 4 weeks, with increases most pronounced for visitor fishes. These studies demonstrate cleaner fish in tropical ecosystems can be of great ecological importance and are key for maintaining local reef diversity.

Further benefits of cleaner fish have been reported with respect to reductions in stress levels as a result of tactile stimulation from physical contact with cleaner fish (Bshary *et al*., 2007; Soares *et al*., 2011). Soares *et al*. (2011) reported significantly lower levels of cortisol in surgeonfish when stimulated by moving models, compared with control fish with access to stationary models. Bshary *et al*. (2007) reported similar findings in two client species (*Chromis dimidiata* and *Pseudanthias squamipinni*). Using cortisol levels as an indicator, a reduction in short term stress response to capture, transport and one-hour confinement in small aquaria occurred when in the presence of cleaner organisms (cleaner wrasse and shrimp). A reduction in stress response as a result of cleaner fish therefore indicates those with no access to cleaning organisms may be less fit.

6.2.3 Sensitivity

6.2.3.1 Sensitive species

A number of studies used indicator species, perceived to be sensitive to potting, to detect change as a result of potting impacts, whilst others use community assemblage (Young *et al*., 2013). Such species are often sessile and are diverse and abundant in rocky reef habitats, where crab and lobster potting commonly takes place. Epifauna on subtidal rock include erect and branching species which can be characterised by slow growth and as such are vulnerable to physical disturbance (Roberts *et al*., 2010). There is a risk that static gear could cause cumulative damage to such species, with some being more resilient to the effects of fishing than others, and the recovery of more vulnerable species from such impacts likely to be slow (Roberts *et al*., 2010; JNCC & NE, 2011). The ability of fauna to resist impacts of static gear will depend on the species and degree of impact will depend on intensity and duration (Roberts *et al*., 2010). Recovery of species will depend on the life-history characteristic of species affected, including the ability to repair or regenerate damaged parts and the ability of larvae to recolonise the habitat (Roberts *et al*., 2010). Typical species include axinellid sponges, pink sea fan (*Eunicella verrucosa*) and Ross coral (*Pentapora foliacea*) (Roberts *et al*., 2010). Other potential vulnerable species in the North East Atlantic include dead men's fingers (*Alcyonium digitatum*) and various erect branching sponges (e.g. *Axinella* spp., *Raspalia* spp.) (Coleman *et al*., 2013).

MacDonald *et al*. (1996) assessed the fragility and recovery potential of different benthic species to determine their sensitivity to fishing disturbance. Recovery represents the time taken for a species to recover in a disturbed area and fragility represents the inability of an individual or colony of the species to withstand physical impacts from fishing gear. Recovery was scored on a scale of 1 to 4 (1 – short, 2 – moderate, 3 – long and 4 – very long) and fragility was scored on a scale of 1 to 3 (1 – not very fragile, 2 – moderately fragile and 3 – very fragile). The scores assigned to potentially vulnerable species in the Studland to Portland SAC are detailed in Table 5. The table also includes sensitivity information assigned by MarLIN in relation to physical disturbance and abrasion. Please note that the sensitivity ratings assigned by MarLIN are based on a single dredging event, the force of which is likely to be greater in magnitude than the impacts caused by potting. Also note this is not an exhaustive list of potentially vulnerable species, these were selected based on those listed by MacDonald *et al*. (1996) on rocky ground and which also occur within the Studland to Portland SAC.

Table 5. Likely sensitivity of some species (which occur within the Studland to Portland SAC) to disturbance caused by an encounter with fishing gear on rocky ground scored by MacDonald *et al*. (1996) and MarLIN (in relation to physical disturbance and abrasion). Low intensity gears include pots, gill nets and longlines. Fragility is derived from personal knowledge of species structure and recovery values were derived from a review of literature on life-histories of the species. Source: MacDonald *et al*. (1996) and www.marlin.ac.uk/).

		MacDonald *et al*. (1996)			**MarLIN**		
Species	**Common name**	**Fragility**	**Recovery**	**Sensitivity (for low intensity gear)**	**Intolerance**	**Recoverability**	**Sensitivity**
Eucinella verrucosa	Pink sea fan	3	3	24	Intermediate	Moderate	Moderate
Pentapora foliacea	Ross coral	3	2	16	High	Moderate	Moderate
Alcyonium digitatum	Dead man's fingers	1	2	5	Intermediate	High	Low
Halichondria panicea[1]	Breadcrumb sponge	1	1	3	Intermediate	High	Low
Laminaria hyperborea	Tangle or cuvie (brown algae / seaweed)	2	2	11	Intermediate	Moderate	Moderate
Flustra foliacea	Hornwrack	2	2	11	Intermediate	High	Low
Nemertesia antennina	Sea beard (a hydroid)	2[a]	1	5	Intermediate[2]	High	Low
Pomatoceros sp.[3]	A tubeworm	1	1	3	-	-	-

[1]*Halichondria* sp. is listed in Regulation 35 Conservation Advice but only sensitivity scores for this species is available; [2]Sensitivity scores for *Nemertesia ramosa*; [3]Sensitivity scores for *Pomatoceros triqueter*

6.2.3.2 Sensitivity analyses

A number of recent studies have endeavoured to map the sensitivity of habitats to different pressures (Tillin *et al*., 2010) and fishing activities (Hall *et al*., 2008).

Tillin *et al*. (2010) developed a pressure-feature sensitivity matrix, which in effect is a risk assessment of the compatibility of specific pressure levels and different features of marine protected areas. The approach used considered the resistance (tolerance) and resilience (recovery) of a feature in order to assess its sensitivity to relevant pressures (Tillin *et al*., 2010). Where features have been identified as moderately or highly sensitive to benchmark pressure levels, management measures may be needed to support achievement of conservation objectives in situations where activities are likely to exert comparable levels of pressure (Tillin *et al*., 2010). In the context of this assessment, the relevant pressures likely to be exerted are surface abrasion, removal of target species and removal of non-target species. All features have medium to high sensitivity to the removal of non-target species, not sensitive to medium sensitivity for the removal of target species and the sensitivity to surface abrasion ranged between low too high for moderate energy circalittoral rock, high for fragile sponge and anthozoan communities on subtidal rocky habitats and medium for blue mussel beds and moderate energy infralittoral rock (Table 6). The hard outer shell of mussels buys some protection against physical impacts although their shells can be broken by direct pressure (Roberts *et al*., 2010). The denser aggregations of *Mytilus edulis* mean the species is likely to be less sensitive to abrasion than mussel species including *Modiolus modiolus* (Walmsley *et al*., 2015). It is important to note that generally there is low confidence in these assessments.

Hall *et al*. 2008 aimed to assess the sensitivity of benthic habitats to fishing activities. A matrix approach was used, composed of fishing activities and marine habitat types and for each fishing activity sensitivity was scored for four levels of activity (Hall *et al*., 2008). The matrix was completed using a mixture of scientific literature and expert judgement (Hall *et al*., 2008). The type of fishing activity chosen was ‘static gear (fishing activities which anchor to the seabed)’ as this best encompassed the fishing activity under consideration. Rock with erect and branching species appears to be the most sensitive to higher gear intensities compared with rock with low-lying and fast growing faunal turf and shallow subtidal rock with kelp which were considered to have a sensitivity level of no higher than medium (Table 7). Under boulder communities on lower shore and subtidal boulders and cobbles and biogenic reef on sediment and mixed substrate were the least sensitive with low sensitivity to heavy, moderate and light gear intensities.

Table 6. Sensitivity of SAC features to pressures identified by Tillin *et al*. (2010). Confidence of sensitivity assessment is included in brackets.

	Pressure		
Feature	Surface abrasion: damage to seabed surface features	Removal of non-target species	Removal of target species
Fragile sponge and anthozoan communities on subtidal rocky habitats	High (Low to High)	High (Low)	Not sensitive (Low)
Moderate energy infralittoral rock	Medium (Low)	Medium (Low)	Medium (Low)
Moderate energy circalittoral rock	Low to High (Low)	Medium to High (Medium)	Not sensitive to Medium (Medium)
Blue Mussel beds (including intertidal beds on mixed and sandy sediments)	Medium (Low)	Medium (High)	-

Table 7. Sensitivity of SAC features to different intensities (high, medium, low, single pass) of static gear (fishing activities which anchor to the seabed) as identified by Hall *et al*. (2008).

Habitat Type	**Gear Intensity***			
	Heavy	Moderate	Light	Single pass
Shallow subtidal rock with kelp	Medium	Low	Low	Low
Rock with low-lying and fast growing faunal turf	Medium	Medium	Low	None
Rock with erect and branching species	High	High	Medium	None
Biogenic reef on sediment and mixed substrate (includes *Mytilus*)	Low	Low	Low	None
Under boulder communities on lower shore and shallow subtidal boulders and cobbles	Low	Low	Low	None

*Heavy - >9 pairs of anchors/area 2.5nm by 2.5nm fished daily, Moderate- 3- 8 pairs of anchors/area 2.5nm by 2.5nm fished daily, Light - 2 pairs of anchors/area 2.5nm by 2.5nm fished daily, Single - Single pass of fishing activity in a year overall

6.3 Site Condition

Natural England provides information on the condition of designated sites and describes the status of interest features. This is derived from the application of 'Common Standards Monitoring Guidance' which is applied to a subset of 'attributes' of site features as set out in the sites' Regulation 33/35 Conservation Advice document. Feature condition influences the Conservation Objectives in that it is used to determine whether a 'maintain' or 'recover' objective is needed to achieve the target level for each attribute. Natural England's current process for conducting condition assessments for marine features was developed due to requirements to report on condition of Annex 1 features at the national level in 2012/13 under Article 17 of the Habitats Directive. Since then, the methods have been reviewed and Natural England are actively working to revise this process further so that it better fulfils obligations to inform management actions within MPAs and allows them to report on condition. In light of this revision to the assessment methods, the condition assessments for the features of European Marine Sites have not been made available in the timeframe required under the revised approach.

An indication as to the condition of the site is available from the Regulation 35 Conservation Advice which states that 'video and photographic analysis (Axelsson *et al*., 2011) combined with extensive diver survey data indicate that the majority of the reef habitat within the site is of excellent quality and structure'.

6.4 Existing Management Measures

- **Bottom Towed Fishing Gear Byelaw** – prohibits bottom towed fishing gear over sensitive reef features within the Lyme Bay portion of the Studland to Portland SAC.

- **Vessel Used in Fishing Byelaw** – prohibits commercial fishing vessels over 12 metres from the Southern IFCA district. The reduction in vessel size also restricts the type of gear that can be used and the level of static gear that can be worked.

6.5 New Management Measures

Alongside Cornwall and Devon and Severn IFCA, Southern IFCA developed a 7-point 'Fishery Guidance' plan in June 2017 which involves a number of voluntary measures[13]. The plan includes:

- A range of species specific maximum and minimum sizes have been developed in order to maintain recruitment into the fishery through aligning minimum sizes above the size of sexual maturity. The maximum size will serve to maintain a balanced population structure through protecting the larger established family groups from capture. Maximum sizes are particularly effective at protecting the longer-lived and larger growing wrasse species which employ a hermaphrodite reproductive strategy.

- No take zones are believed to afford effective and long-term protection for species with high site fidelity and small home ranges/territories, like those exhibited by local wrasse species (Morel *et al*., 2013). A series of no take zones and no potting zones have been developed within the Southern IFCA district, in many cases overlapping with the boundaries of Marine Protected Areas. Approximately 60% of fishable areas (i.e. those less than 10 metres depth) are no take zones. In addition, popular sites for recreational sea fishing have been included as no take zones in order to reduce conflict between users and to ease the pressure on wrasse populations in these areas.

- Pot depth restrictions (>10 m) to protect the survivability of catches. Approximately 90% of the SAC is deeper than 10 metres. Survivability of wrasse species is negatively correlated with the depth from which they are fished. Individuals brought up over 10 metres water depth are visually affected by the change in pressure and take longer to recover.

- Effort restriction through a pot limitation of 80 traps per vessel. A limit of 80 traps represents both a proportionate limit for the small inshore fishing vessels (8 metres or less in length) involved in the fishery and a level of gear intensity which is high unlikely to lead to any significant interaction with the seabed through abrasion (supported by the results of potting impact studies). Aligning the pot limitation with the capacity of participating vessels ensures the fishery continues to be economically viable for these smaller fishing vessels, whilst safeguarding against the participation of any larger higher impact vessels. The density of pots, based on a pot limitation of 80 traps used by a maximum of 8 vessels operating within the fishery and within the area known to be fished, has been calculated at approximately 26 pots set per km^2. This corresponds to a 'very low' to 'low' fishing gear intensity (Annex 6). Results from potting impact studies (i.e. Eno *et al*., 2001; Shester & Micheli, 2011; Coleman *et al*., 2013; Young *et al*., 2013; Haynes *et al*., 2014; Stephenson *et al*., 2015; 2016; Gall, 2016; Rees *et al*., 2016) infer the impacts of potting on temperate rocky habitats are negligible or limited in extent, particularly at such low densities.

- A fishing closed season from April to June (inclusive) has also been introduced to protect wrasse populations during their peak spawning period.

[13] https://secure.toolkitfiles.co.uk/clients/25364/sitedata/files/Wrasse-Guidance.pdf

- Monthly fishermen catch returns detailing the quantities of species caught, fishing location and fishing effort. This provides valuable information to better understand the exploitation of the fishery.

- Biosecurity and husbandry is related to the storing and transporting of live fish and seawater, following appropriate biosecurity and husbandry measures to prevent the mixing of genetic structure and transport of disease, parasites and non-native species.

The Fishery Guidance aims to protect the long-term sustainability of wrasse populations within the Southern IFCA District and maximise the enjoyment of the species by other users, notably recreational sea anglers, divers and snorkelers.

The fishery guidance plan has been developed alongside the local fishermen and with the scientific literature in mind. In line with byelaw making guidance, Southern IFCA hopes for an 'industry-led' approach for the management of the wrasse fishery in order to secure long-term sustainability. Southern IFCA have discussed the fishery guidance plan with local fishermen and salmon farm representatives. Should the approach prove ineffective or significant changes occur within the fishery, Southern IFCA will introduce regulatory measures to address the issue of wrasse fishery management.

6.6 Monitoring

A number of monitoring activities will accompany the 7-point 'Fishery guidance' plan and help to assess its short-term success. In collaboration with a range of partners including Natural England and industry operators, Southern IFCA has commenced a programme of study to improve our understanding of the fishery and its effects on the marine environment. Research techniques include the collection of fishery catch data, catch sampling and the development of a PhD.

Fishery catch data (quantities of species caught, fishing location and fishing effort) is obtained through monthly catch returns provided by fishermen. Catch sampling involves Southern IFCA officer going onboard wrasse fishing vessels on an ad hoc basis, recording and measuring the catch of each pot (including target wrasse and bycatch species). The data collected will provide valuable information on the level of exploitation of the fishery, population structure of the catchable population and the selectivity of gear type (traps) used. Improved monthly catch return forms for 2018 will allow for the differentiation of effort and level of removal between areas, including within and outside of the SAC. The fishery catch data and catch sampling data collected in 2017, alongside landings data from buyers, was analysed as part of one of Southern IFCAs 2017 internship programme projects; 'Wrasse Fishery Assessment'. This involved calculating catch per unit effort from catch return data and undertaking size length frequency analysis from catch sampling data. The internship also involved a literature search on wrasse biology. The report from this internship is presented in annex 7.

Going forward, catch sampling will be undertaken as part of a three-year research project run by the University of Exeter, in collaboration with Natural England and Southern IFCA. The project aims to investigate the functional role of wrasse within the inshore reef systems on the south west coast of England. The project will focus on assessing their role as cleaner fish species and mesopredators within the food web, looking at their behavioural and feeding ecology. It will aim to quantify the importance of the role wrasse play on reefs and therefore what the impacts of their removal will be on these habitats. In addition, the project will aim to develop a broad scale view of the population structure of wrasse in the region and improve our knowledge on elements of their general ecology such as sex ratios, and spawning seasons. The results from this project will be made available to the three south-west IFCAs in order to better inform their management, in addition to being used by Natural England in their conservation advice packages, where applicable.

Information gathered through these monitoring activities and regular compliance patrols, as well as work undertaken by other organisations, including Cefas, will allow Southern IFCA to closely monitor the fishery, particularly fishing effort and landings. This information will be reviewed when appropriate i.e. at the end of the fishing season or when new evidence becomes available. The review of information from monitoring activities will form part of a feedback process to identify if there is a need for assessment, and depending on the outcome, initiate a review of management. This feedback process will be outlined in a standalone Monitoring and Control Plan and provides a framework for adaptive management.

Table 8: Summary of Impacts

The potential pressures, associated impacts, level of exposure and mitigation measures are summarised in table 8. Only relevant attributes identified through the TLSE process have been considered here.

Feature	Sub feature(s)/ Supporting habitat(s)	Attribute	Target	Potential Pressure(s) and Associated Impacts	Nature and Likelihood of Impacts	Mitigation measures[14]
Reefs	Bedrock reef	Biotope composition of bedrock reefs; (Updated Conservation Advice September 2017: Structure: species composition of component communities)	Maintain the full variety of bedrock reef biotopes identified for the site to an established baseline, subject to natural change.	Abrasion and disturbance to the surface of the seabed was identified as a potential pressure. Benthic communities can be directly impacted by potting gear through crushing, entanglement or removal, when gear is being deployed, hauled or under the influence of currents or waves which can involve lateral dragging. Epifauna on subtidal rocky habitats include erect and branching species which often have slow growth and are vulnerable to physical disturbance.	Currently 10 commercially licensed vessels, all less than 8 metres in length, fish for wrasse using fish traps, with 2 to 3 vessels also using rod and line. Not all fish within the Studland to Portland SAC. Fishing for wrasse is limited to less than 10 metres depth and is not conducted throughout the entire site, thus only occurring over a relatively small area. The number of traps worked by each vessel is restricted to 80 per vessel, as per the Wrasse Fishery Guidance. Existing scientific literature and ongoing studies suggest that the impact of potting on benthic	Vessel Used in Fishing byelaw prohibits commercial fishing vessels over 12 metres from the Southern IFCA district. The reduction in vessel size also restricts the level of traps that can be worked. Fishery Guidance Plan is currently being developed. The plan will involve traps limitations.

[14] Detail how this reduces/removes the potential pressure/impact(s) on the feature e.g. spatial/temporal/effort restrictions that would be introduced.

				There is a relative paucity of scientific evidence on the impacts of potting on benthic communities when compared when mobile gear. No data currently exists for fish traps used to catch wrasse however potential impacts have been inferred from studies investigating the impacts of pots used to catch crab and lobster, as both traps/pots are similar in nature. Existing literature infers that impacts of potting on temperate rocky habitats are negligible or limited in extent, especially when compared to impacts resulting from periods of adverse weather conditions (i.e. Eno *et al*., 2001; Shester & Micheli, 2011; Coleman *et al*., 2013; Young *et al*., 2013; Haynes *et al*., 2014; Stephenson *et al*., 2015). Preliminary results from ongoing studies are also in agreement (Rees *et al*., 2016, AFBI).	communities is negligible or limited in extent. Damage to benthic habitats by adverse weather conditions have been reported to be far in access of that caused by potting (Rees *et al*., 2016).	
Reefs	Bedrock reef	Distribution and spatial pattern of bedrock	Maintain the distributio n and	Addressed above.	Addressed above.	Addressed above.

		reef biotopes; (Updated Conservation Advice September 2017: Distribution: presence and spatial distribution of biological communities)	spatial pattern of bedrock reef biotopes identified for the site, to an established baseline, allowing for natural change.			

Reef	Bedrock reef	Extent of representative / notable bedrock reef biotopes (Updated Conservation Advice September 2017: Distribution: presence and spatial distribution of biological communities)	No change in the extent of the *Mytilus edulis* biotopes, from an established baseline, allowing for natural change.	In addition to information provided above, the preliminary findings of an ongoing study by AFBI into the impacts of potting on *Modiolus* beds, suggest a lack of effect.	In addition to information provided above, the preliminary findings of an ongoing study by AFBI into the impacts of potting on *Modiolus* beds, suggest a lack of effect.	In addition to measures stated above, the Southern IFCA conduct an annual survey to monitor the presence of mussel spat within designated fished areas. The survey is used to inform an appropriate assessment which is required for potential impacts of the proposed mussel seed fishery to be assessed in view of the integrity and conservation objectives of the site. The survey involves the collection of underwater camera footage which is analysed for mussel density, presence of megafauna, presence/absence of mussel spat and mussel average length, as well as mapping extent of mussel beds in the area. The most recent survey highlighted a lack of mussel presence as a result of recent storm activity and highlighted the ephemeral nature of mussel beds.
Reef	Bedrock reef	Species composition of representative or notable bedrock reef	No decline in bedrock reef biotope quality due to changes in species	The selective extraction of the target species has the potential to lead to a change in the biomass of listed biotope representative and/notable species. Principally brown algal species; *Laminaria*	Currently 10 commercially licensed vessels, all less than 8 metres in length, fish for wrasse using fish traps, with 2 to 3 vessels also using rod and line. Not all fish within the Studland to Portland SAC.	Vessel Used in Fishing byelaw prohibits commercial fishing vessels over 12 metres from the Southern IFCA district. The reduction in vessel size also restricts the level of traps that can be worked.

		biotopes (Updated Conservation Advice September 2017: Structure: species composition of component communities)	composition or loss of notable species, from an established baseline, allowing for natural change.	*hyperborea*, *Saccorhiza polyschides* and *Zanardinia typus*. The removal of wrasse, as an epibenthic grazer, may have indirect effects on algal biomass due to a loss top-down control on epifaunal growth and subsequent overgrazing (Bergström *et al*., 2016). A number of studies based in New Zealand reported a positive indirect effect of wrasse species on macroalgae, with significant increases in biomass in the presence of fish predators (Perez-Matus & Shima, 2010; Newcombe & Taylor, 2010). Any potting impacts, as a result of abrasion, on this attribute are addressed above.	Fishing for wrasse is limited to less than 10 metres depth and is not conducted throughout the entire site, thus only occurring over a relatively small area. The number of traps worked by each vessel is restricted to 80 per vessel, as per the Wrasse Fishery Guidance. Currently, ballan wrasse are the predominant target species of the fishery, solely targeted by 8 out 10 vessels in 2017. If other wrasse species are caught by these vessels they are returned. Two vessels, which commenced fishing in the latter half of the 2017 season, targeted four wrasse species (Corkwing, Goldsinny, Rock Cook, Ballan). Therefore the removal of the other four wrasse species (other than ballan wrasse) is likely to be less. It is unknown if the functional role of each species is interchangeable, however all wrasse species are considered to belong to the functional group 'coastal mesopredatory fish' and to some extent are likely to perform similar functional roles as mid-trophic level demersal and benthic species consisting predominantly of invertebrates. As such, the less	The fish traps used by fishermen are fitted with escape gaps. This should allow for wrasse below the targeted size to escape from the trap and eliminate the stresses associated with being hauled and subsequent handling.

					targeted wrasse species may continue to exert some form of top-down control on epifaunal growth and grazing. The limited area in which the activity takes place (~1.5% of the total SAC and ~0.8% of the total reef feature) and small number of boats which undertaken the activity, it is unlikely that the activity will lead to the level of removal where algal biomass is significantly affected. Whilst existing literature is helpful in indicating the potential for overgrazing in the absence of/ reduction in wrasse species, no such studies have been conducted in the UK. This means there is difficulty in determining the potential severity of such potential impacts as existing literature is based on different species and therefore warrants further investigation.	
Reef	Bedrock reef	Presence and/or abundance of specified bedrock reef species (Updated Conservation Advice September	Maintain presence and/or abundance of species from an established baseline, allowing	Abrasion and disturbance to the surface of the seabed is addressed above. The selective extraction of species was identified as a potential pressure. Four wrasse species are targeted by the fishery.	Currently 10 commercially licensed vessels, all less than 8 metres in length, fish for wrasse using fish traps, with 2 to 3 vessels also using rod and line. Not all fish within the Studland to Portland SAC. Fishing for wrasse is limited to less than 10 metres depth and is not conducted throughout the entire	Vessel Used in Fishing byelaw prohibits commercial fishing vessels over 12 metres from the Southern IFCA district. The reduction in vessel size also restricts the level of traps that can be worked. The fish traps used by fishermen are fitted with

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		2017: Structure and function: presence and abundance of key structural and influential species)	for natural change.	Ballan wrasse form predominant target species of the fishery, solely targeted by 8 out of 10 of vessels. The size range of wrasse targeted by the fishery ranges from 10 to 30 cm (12 to 28 cm for ballan wrasse). Such removal impacts directly on wrasse populations through size-selective harvesting (Halvorsen, 2016) and on the wider ecosystem due to the central position wrasse hold within the food web as a mesopredatory fish (Bergström *et al*., 2016). Studies have highlighted concerns surrounding the direct impacts size-selective harvesting may have on wrasse populations, particularly the removal of larger more fecund sexually mature adults, as well as immature individuals, depending on the species and associated impacts on population structure and reproduction (Darwall *et al*., 1992; Deady *et al*. 1993; Varian *et al*., 1996; Muncaster *et al*., 2013;	site, thus only occurring over a relatively small area. This leaves a significant proportion of the MPA not subject to fishing which are likely to act a refuge areas and a potential source of replenishment for fished areas. The number of traps worked by each vessel is restricted to 80 per vessel, as per the Wrasse Fishery Guidance. The measure of this attribute is the presence and/abundance of specified bedrock reef species. The recently updated Studland to Portland SAC Conservation Advice mention two wrasse species under the general description of the 'Reef' feature however wrasse species do not feature in the list of key species associated with associated the reef biotope, in either the updated Conservation Advice or Regulation 35 Conservation Advice packages. Therefore the main concerns surrounding this attribute are related to the potential indirect impacts on algal growth related to the removal of wrasse species. The comments attached to this attribute however do specify 'the species selected should serve an important role in the structure and function of	escape gaps. This should allow for wrasse below the targeted size to escape from the trap and eliminate the stresses associated with being hauled and subsequent handling.

				Leclercq *et al*., 2014; Halvorsen, 2016; Halvorsen *et al*., 2016; 2017). Other concerns directly influencing wrasse populations include targeting wrasse during the spawning season (Skiftesvik *et al*., 2014) and potential for local depletion associated with their small home ranges/ territories and high site fidelity (Halvorsen, 2016). The removal of wrasse and any subsequent impacts on population structure and reproduction may have wider ranging ecosystem impacts. Wrasse are defined as mesopredatory fish whose diet consists primarily of invertebrates (Bergström *et al*., 2016). This functional group serve as a food resource for higher trophic levels and perform a regulatory function on lower trophic levels, including a top-down control on epifauna which graze upon algal species (Bergström *et al*., 2016).	biological community'. Wrasse may be considered to fulfil these criteria based on their central role within the ecosystem. Wrasse are not protected by specific UK legislation, are not listed as a designated feature for Special Area of Conservation or Marine Conservation Zones and are not considered to be keystone species, nor characterising species of any reef community (Natural England, 2017). Advice from Natural England (2017) does however consider wrasse should be assessed in the same way as crab and lobster when undertaking this assessment. Currently, ballan wrasse are the predominant target species of the fishery, solely targeted by 8 out 10 vessels in 2017. If other wrasse species are caught by these vessels they are returned. Two vessels, which commenced fishing in the latter half of the 2017 season, targeted four wrasse species (Corkwing, Goldsinny, Rock Cook, Ballan). Therefore the removal of the other four wrasse species (other than ballan wrasse) is likely to be less. It is unknown if the functional role of each species is interchangeable, however all wrasse species are considered to	

				The removal of wrasse, as an epibenthic grazer, may therefore have indirect effects on algal biomass due to a loss top-down control on epifaunal growth and subsequent overgrazing (Bergström *et al*., 2016). A number of studies based in New Zealand reported a positive indirect effect of wrasse species on macroalgae, with significant increases in biomass in the presence of fish predators (Perez-Matus & Shima, 2010; Newcombe & Taylor, 2010). In recently updated Conservation Advice (September 2017) wrasse species (Ballan wrasse, Goldsinny wrasse) are mentioned under the general description of the 'Reef' feature when describing the diverse suite of mobile species supported by the feature. Wrasse are not however not listed within either the updated Conservation Advice or Regulation 35 Conservation Advice Packages. There are	belong to the functional group 'coastal mesopredatory fish' and to some extent are likely to perform similar functional roles as mid-trophic level demersal and benthic species consisting predominantly of invertebrates. As such, the less targeted wrasse species may continue to exert some form of top-down control on epifaunal growth and grazing. The limited area in which the activity takes place (~1.5% of the total SAC and ~0.8% of the total reef feature) and small number of boats which undertaken the activity, it is unlikely that the activity will lead to a significant removal of wrasse and subsequent reductions in algal biomass. The MarLIN web page for the 'Grazed *Laminaria hyperborea* park with coralline crusts on lower infralittoral rock biotope' highlights mechanisms that control sea urchin aggregations are poorly understood but have been attributed to top down urchin predators (cod, lobsters). Large scale urchin barrens within the North East Atlantic are limited to the North Norwegian and Russian Coast. Within the UK, urchin grazed	

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				a number of representative and/notable species however which have the potential to be indirectly impacted by the removal of wrasse. Principally brown algal species; *Laminaria hyperborea*, *Saccorhiza polyschides* and *Zanardinia typus*. The MarLIN web page for the 'Grazed *Laminaria hyperborea* park with coralline crusts on lower infralittoral rock biotope' highlights kelp (Laminaria hyperborea) biotopes are partially reliant on low or no populations of sea urchins, with dense aggregations a principal threat to these biotopes in the North Atlantic. Intense urchin grazing can lead to a shift from kelp dominated biotopes to those characterised by coralline encrusting algae, with subsequent reductions in biodiversity. Concerns have also been raised regarding the potential reduction in	biotopes are generally localised to a few regions in North Scotland and Ireland. They are also a listed biotope of the Studland to Portland SAC. 'Urchin barrens' are however not presently an issue within the UK, however relatively low urchin grazing has been found to the control the depth distribution of *L. hyperborea* which can negatively impact on recruitment of the species and reduce understory community abundance and diversity. Such issues, with respect to urchin barrens, are not highlighted as an issue within the Studland to Portland SAC Conservation Advice. The sensitivity of the biotope 'Grazed *Laminaria hyperborea* park with coralline crusts on lower infralittoral rock' is considered as having medium sensitivity to the removal of target and non-target species. Literature from tropical reef systems highlight the deleterious effect sea lice can have on the health of coral reef fish species, the importance of cleaner fish with respect to the removal of sea lice and the potential ecological importance of cleaner fish and their role in maintaining local reef diversity.	

				cleaning behaviour associated with the removal of wrasse and subsequent impacts on the overall health of the reef system (Natural England, 2017).	Whilst existing literature is helpful in indicating the potential for overgrazing in the absence of/ reduction in wrasse species and potential implications of reductions in cleaning behaviour, no such studies have been conducted in the UK. This means there is difficulty in determining the potential severity of such potential impacts as existing literature is based on different species and therefore warrants further investigation.	
Reef	Stony reef	Biotope composition of stony reefs (Updated Conservation Advice September 2017: Structure: species composition of component communities)	Maintain the full variety of biotopes identified for the site to an established baseline, subject to natural change.	Biotopes identified for stony reef are all circalittoral, so therefore are unlikely to be subject to fishing for wrasse as this is limited to <10m. Otherwise addressed under bedrock reef.	Biotopes identified for stony reef are all circalittoral, so therefore are unlikely to be subject to fishing for wrasse as this is limited to <10m. Otherwise addressed under bedrock reef.	Addressed under bedrock reef.
Reef	Stony reef	Distribution and spatial pattern of stony reef biotopes (Updated	Maintain the distribution and spatial pattern of	Biotopes identified for stony reef are all circalittoral, so therefore are unlikely to be subject to fishing for wrasse as this is limited to <10m.	Biotopes identified for stony reef are all circalittoral, so therefore are unlikely to be subject to fishing for wrasse as this is limited to <10m.	Addressed above under bedrock reef.

		Conservation Advice September 2017: Distribution: presence and spatial distribution of biological communities)	stony reef biotopes identified for the site, to an established baseline, allowing for natural change.	Otherwise addressed under bedrock reef.	Otherwise addressed under bedrock reef.	
Reef	Stony reef	Species composition of representative or notable stony reef biotopes (Updated Conservation Advice September 2017: Structure: species composition of component communities)	No decline in stony reef biotope quality due to change in species composition or loss of notable species, from an established baseline, allowing for natural change.	Biotopes identified for stony reef are all circalittoral, so therefore are unlikely to be subject to fishing for wrasse as this is limited to <10m. In addition, it is unlikely that any of the representative or notable species listed will be indirectly or directly affected by the removal of wrasse species.	Biotopes identified for stony reef are all circalittoral, so therefore are unlikely to be subject to fishing for wrasse as this is limited to <10m. In addition, it is unlikely that any of the representative or notable species listed will be indirectly or directly affected by the removal of wrasse species.	Addressed above under bedrock reef.

7. Conclusion[15]

In order to conclude whether fishing for wrasse using fish traps is likely to have an adverse effect on the integrity of the Studland to Portland SAC, it was necessary to assess whether the impacts of the activity are likely to hinder the site's conservation objectives, namely:
"Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the Favourable Conservation Status of its Qualifying Features, by maintaining or restoring:

- The extent and distribution of qualifying natural habitats
- The structure and function (including typical species) of qualifying natural habitats, and
- The supporting processes on which the qualifying natural habitats rely."

A review of research and scientific literature focused on the potential impacts of pressures identified through the test of likely significant effect (TLSE) process (abrasion, removal of non-target species and removal of target species). These potential impacts were then assessed against relevant attributes (biotope composition, distribution and spatial pattern of biotope, extent of representative/ notable biotopes, species composition of representative/ notable biotopes and presence and/or abundance of specified species) (see table 8), also identified through the TLSE process.

With regards to abrasion and associated removal of non-target species, research into the impacts of potting on benthic habitats has shown there is a relative paucity of scientific evidence when compared with the impacts of mobile gear. The number of studies completed in recent years on the impacts of potting in rocky habitats has however increased and additional studies are ongoing in order to address this evidence gap. Existing literature (i.e. Eno *et al*., 2001; Shester & Micheli, 2011; Coleman *et al*., 2013; Young *et al*., 2013; Haynes *et al*., 2014; Stephenson *et al*., 2015; 2016; Gall, 2016) and preliminary results from ongoing studies (Rees *et al*., 2016; AFBI) infer the impacts of potting on temperate rocky habitats are negligible or limited in extent, especially when compared to impacts resulting from periods of adverse weather (Young *et al*., 2013; Rees, No Date). Gall (2016) concluded the reef in Start Point to Plymouth Sound and Eddystone SCI is being maintained in favourable condition, thus achieving the sites conservation objectives, despite the presence of potting activity.

With regards to the removal of target species, the reproductive strategies and life history of the four targeted wrasse species, combined with studies on past or current wrasse fisheries, were used to determine the likely direct impacts on wrasse populations. Research highlighted that the wrasse fishery is size-selective and therefore leads to the removal of certain groups from the population. For ballan wrasse, the key target species, the preferred size (prior to the introduction of the wrasse fishery guidance) at which individuals were targeted could to the removal of both immature

[15] If conclusion of adverse effect alone an in-combination assessment is not required.

and larger mature females. For the smaller gonochoristic species (corkwing, goldsinny and rock cook), mature individuals, particularly males were particularly vulnerable to removal. There was therefore the potential for size- and thus sex-selective harvesting to take place in all four species which has a variety of implications related to population dynamics, demography and reproduction (Halvorsen, 2016). Following the introduction of the minimum and maximum sizes, as part of the wrasse fishery guidance, these impacts will be eliminated or reduced. The measures are designed to allow individuals to reproduce at least once before being removed from the fishery by aligning minimum sizes with sexual maturity and to protect wrasse species with complex reproductive strategies. Despite this, there is still the potential for size- and thus sex-selective harvesting to take place. When considering the direct impacts on wrasse populations in the context of the relevant attributes identified, whilst mentioned under the general description of the 'Reef' feature in the recently updated Conservation Advice (September 2017), wrasse species do not appear within any species list within the updated Conservation Advice or Regulation 35 Conservation Advice packages. Nor does wrasse appear as any designated feature of either a Special Area of Conservation or Marine Conservation Zone (Natural England, 2017). Therefore, these direct impacts on wrasse populations, whilst important for the long-term sustainability of the fishery, are not directly relevant in the context of this assessment and the attributes against which these impacts are assessed.

The indirect impacts arising from the removal of wrasse and any subsequent changes in population and reproduction were also considered. There is a lack of evidence surrounding the ecological function of wrasse species and subsequent wider ecosystem impacts resulting from the removal of wrasse species. As such, best available evidence, on the diet of wrasse species and the ecosystem/trophic interactions studied in other temperate reef habitats, was used to infer the potential for wider ranging impacts on the ecosystem. Research highlighted concerns surrounding the removal of wrasse as an epibenthic grazer, due to the potential for indirect effects on algal biomass as a result of reduced top-down control on epifaunal growth and subsequent overgrazing (Bergström *et al*., 2016). Studies based on reef systems in other temperate locations reported positive indirect effects of wrasse species on macroalgae, with significant increases in algal biomass in the presence of fish predators (Perez-Matus & Shima, 2010; Newcombe & Taylor, 2010). These indirect impacts have the potential to affect a number of notable and representative brown algal species; *Laminaria hyperborea*, *Saccorhiza polyschides* and *Zanardinia typus*.

The wrasse fishery has taken place seasonally from April to October, however the start of the season will commence from July in 2018 with the introduction of the new wrasse fishery guidance. At the beginning of 2017 season 8 vessels engaged in the fishery, with a further two vessels engaging later in the season, in the area surrounding the Studland to Portland SAC, although all vessels are not believed to fish within the SAC. In 2018, it anticipated 8 vessels will take part in the fishery and as per the wrasse fishery guidance the fishery will not commence until July. Fishing for wrasse is limited to less than 10 metres in depth and is not conducted throughout the entire site, thus only covering a small area (equating to ~1.5% of the total SAC area). This leaves a significant portion of the MPA to act as refuge ages and potential source of replenishment for fished areas. Currently, the dominant species targeted is ballan wrasse, *Labrus bergylta* (solely targeted by 8 out 10 vessels in 2017). If other species of wrasse are caught by these vessels they are returned and, as considered to belong to the 'coastal mesopredatory fish' functional group and thus to some extent may perform similar functional roles, are likely to continue to exert some form of top-down control on epifaunal growth and grazing.

It is Southern IFCAs duty as the competent and relevant authority to manage damaging activities that may affect site integrity and lead to deterioration of the site. Based on the scale of the fishery (described above), the relatively small area subject to fishing (~1.5% of the total SAC area and ~0.8% of the total reef feature) and targeting of only predominantly one wrasse species, it is unlikely that the potential indirect effects on algal growth of notable/ representative brown algae will occur at levels significant enough to have an adverse effect on site integrity and therefore will not hinder the sites conservation objectives. This is further supported by the lack of scientific evidence to suggest potting is likely to have an adverse effect on reef features.

The wrasse fishery guidance, introduced in June 2017, outlines a wide range of different measures and as such means the fishery is subject to the greatest number of restrictions in the Southern IFCA district. The guidance is aimed at ensuring the long-term sustainability of the fishery and such as will directly benefit wrasse populations by preventing over-exploitation, through a number of measures, particularly minimum and maximum sizes, no take zones, closed season and pot depth restrictions. By protecting wrasse populations from over-exploitation this will, in turn, lead to indirect wider ecosystem benefits. In particular, the fishery guidance will help to safeguard against the potential impacts related to the ecological function (i.e. cleaning behaviour, role as epibenthic grazer) and wider ecosystem (i.e. overall reef health). As such, it significantly reduces potential risks associated with uncertainties surrounding these effects on ecological function. It is hoped uncertainties will be addressed during the 3-year research project being undertaken.

The wrasse fishery is newly emerging and therefore is likely to be subject to changes in fishing effort and location as it becomes established. A limited number of new participants may enter the fishery. A number of measures in the fishery guidance will help to mitigate against a number of these changes, such as pot limitations, pot depth restrictions and no take zones. As outlined in section 6.6, information gathered through monitoring activities, regular compliance patrols and monthly catch returns, also part of the fishery guidance plan, will allow for Southern IFCA to closely monitor the fishery, particularly fishing effort and landings. A feedback process for reviewing information resulting from monitoring activities, any new evidence relevant to this gear/feature interaction and the need for assessment will be detailed in a standalone Monitoring and Control Plan and as such will provide a framework for adaptive management.

8. In-combination assessment

No adverse effect on the reef feature/sub-features of Studland to Portland SAC was concluded for the effect of fishing for wrasse using fish traps alone within the SAC. This activity currently occurs in the Studland to Portland SAC alongside other fishing activities and therefore requires an in-combination assessment.

No commercial plans and projects were found to occur within or potentially affect the Studland to Portland SAC. Potential projects were considered and screened out[16].

There is the potential for wrasse potting activity to have a likely significant effect when considered in-combination with other fishing activities that occur within the site. These are outlined in section 8.1. Any fishing activities that were screened out as part of the revised approach assessment process will not be considered (see Studland to Portland SAC screening summary for details of these activities), except for rod and line due to the overlap in target species. In the Studland to Portland SAC, commercially licensed fishing vessels are known to utilise a number of different gear types and are engaged in multiple fishing activities (i.e. potting, netting and longlining) and this, whilst dividing effort between gear types, may lead to cumulative impacts different to those of a single fishing activity.

8.1 Other fishing activities

Fishing activity	Potential for in-combination effect
Potting (crab & lobster/ whelk)	The location of crab and lobster potting and wrasse potting may overlap, as the target species of both activities inhabit areas of rocky reef. Having said this, crab and lobster potting typically concentrated over areas of circalittoral rock whilst wrasse potting is focused on the infralittoral, thus limiting any spatial overlap. Both activities have the potential to lead to physical abrasion with the seabed. Existing scientific literature and ongoing studies suggest that the impact of potting on benthic communities is negligible or limited in extent. As such, the potential for any in-combination effects with respect to physical abrasion is very limited. The activities target different species in different areas of rocky reef (as described above) and therefore there are not likely to be any in-combination effects with respect to the selective extraction of target species. Wrasse are a bycatch species in crab and lobster pots and are often retained as pot bait, although not specifically targeted. As crab and lobster pots are set deeper than 10 metres in depth the survival of wrasse caught as bycatch are often compromised and the survival of wrasse returned is questionable as their swim bladders are often 'blown out' reducing their chances of survivability. The use of wrasse as bait for crab and lobster pot fisheries has been a longstanding practice. The non-target nature and low level of this practice has not been previously raised as a concern. It is not believed that this practice is occurring as a level significant enough to lead to in-combination effects with the live wrasse fishery.

[16] Please refer to the 'Dorset MPAs In-Combination Assessment – Other Plans & Projects' document.

	The level of fishing effort associated with wrasse potting is low (8 vessels) when compared to crab and lobster potting (up to 30 vessels). A majority of vessels engaged in wrasse potting, either also engage in crab and lobster potting, or previously engaged in crab and lobster potting, prior to the development of the wrasse fishery. This means that fishing effort of many vessels is split between the two types of potting or switched from one to the other. In conclusion, there are unlikely to be any in-combination effects with crab and lobster potting due to the low impact of gear, relatively low fishing effort, limited spatial overlap and separate target species.
Demersal netting/ longlining	Netting and longlining largely occurs inshore outside of the SAC boundary. There are limited sightings of netting and longlining and the do not appear to largely overlap with the areas subject to wrasse potting activity. If the two were to overlap, it is likely to occur over reef features or on the boundary of reef features. Netting and longlining has potential to lead to physical abrasion with the seabed however the area affected is small. Unlike potting, which has evidence to support the activity has a negligible or no impact on reef features, there is a severe lack of evidence to suggest netting or longlining has any impact. Based on this, the activities combined are unlikely to lead to a significant effect. The activities target different species and therefore there are not likely to be any in-combination effects with respect to the selective extraction of species. The level of fishing effort associated with wrasse potting is low (8 vessels) when compared to demersal netting and longlining (up to 20 vessels). A number of vessels engaged in wrasse potting, are likely to also engage in netting and/or longlining, or previously engaged in netting and/or longlining, prior to the development of the wrasse fishery. This means that fishing effort of many vessels is split between the two activities or switched from one to the other. In conclusion, there are unlikely to be any in-combination effects with demersal netting and longlining, due to the unlikely overlap between the two activities, low impact of the gear, relatively low fishing effort and separate target species.
Pelagic longlining	Longlining only occurs on the fringes of the site and therefore potential for spatial overlap is limited. Pelagic longlining has very limited potential for contact with the seabed and therefore the likelihood of physical abrasion is negligible. The very low potential for physical abrasion with respect to pelagic longlining and the lack of evidence to suggest negative impacts potting, mean the two activities in-combination are likely to lead to a likely a significant effect. The two activities target different species and therefore there are no in-combination effects with respect to the selective extraction of species.

	In addition, the level of fishing effort associated with pelagic longlining (up to 5 vessels) and wrasse potting (8 vessels) is low. In conclusion, there are unlikely to be any in-combination effects with pelagic longlining, due to the very low impact of pelagic longlining, low fishing effort and separate target species.
Commercial diving	Commercial diving may overlap spatially with potting activity over reef features. Commercial diving however is a very low impact activity and has very limited potential for physical abrasion, with the area affected likely to be negligible. The very low potential for physical abrasion with respect to commercial diving and the lack of evidence to suggest negative impacts of potting, mean the two activities in-combination are likely to lead to a likely significant effect. The two activities target different species and therefore there are no in-combination effects with respect to the selective extraction of species. In addition, the level of fishing effort associated with commercial diving (up to 5 vessels) and wrasse potting (8 vessels) is low. In conclusion, there are unlikely to be any in-combination effects with commercial diving, due to the very low impact of commercial diving, low fishing effort and separate target species.
Rod and line	Two to three vessels who engage in wrasse potting are also known to target wrasse using rod and line. The area in which the activity takes place is typically separate from the area used for wrasse potting and as such there is limited spatial overlap between the two fishing methods. There is a view that the number of vessels using rod and line to target wrasse may increase, which may increase the overlap between the two fishing methods, however this increase is likely to correspond with a reduction in the number of traps fished. This is supported by the lower number of traps used by vessels who use both methods to target wrasse when compared to vessels solely using pots. A reduction in the number of traps fished is compensated by increased catches using rod and line. The two activities are used to target the same species, however there is likely to be no in-combination effects as the areas used for each are currently distinct from one another. Due to the small home ranges/territories of wrasse, the two activities are likely to target separate populations. In areas where the two methods have the potential to overlap there is unlikely to be a higher removal of wrasse, as an increase in rod and line is likely to be accompanied by a reduction in the use of traps to target wrasse. In addition, the level of fishing effort is very low (only two vessels) and as both vessels engage in potting and rod and line they can only undertake one activity at any one time.

9. Summary of consultation with Natural England

Consultation	Date submitted	Response from NE	Date received
HRA Studland to Portland SCI – Fish traps – v1.1 (First draft)	17/05/2017	Recommended amendments	23/08/2017
HRA Studland to Portland SAC – Fish traps – v1.5 (Amended draft)	26/02/2018	Amendments accepted and further amendments recommended.	26/03/2018

10. Integrity test

It can be concluded that the activities in this Habitat Regulations Assessment (fish traps), alone (at current levels) or in-combination with other activities, do not adversely affect the reef feature/sub-features of the Studland to Portland SAC.

As outlined in section 6.5, Southern IFCA has introduced a range of management measures in the form of the Wrasse Fishery Guidance for the district’s wrasse fishery in order to ensure the long-term sustainability. These measures are likely to limit fishing effort and lead to an overall reduction in fishing mortality.

Southern IFCA has begun a series of monitoring activities and is working alongside Natural England on a PhD to assess the wider impacts of the wrasse fishery and develop our understanding of the species’ functional role.

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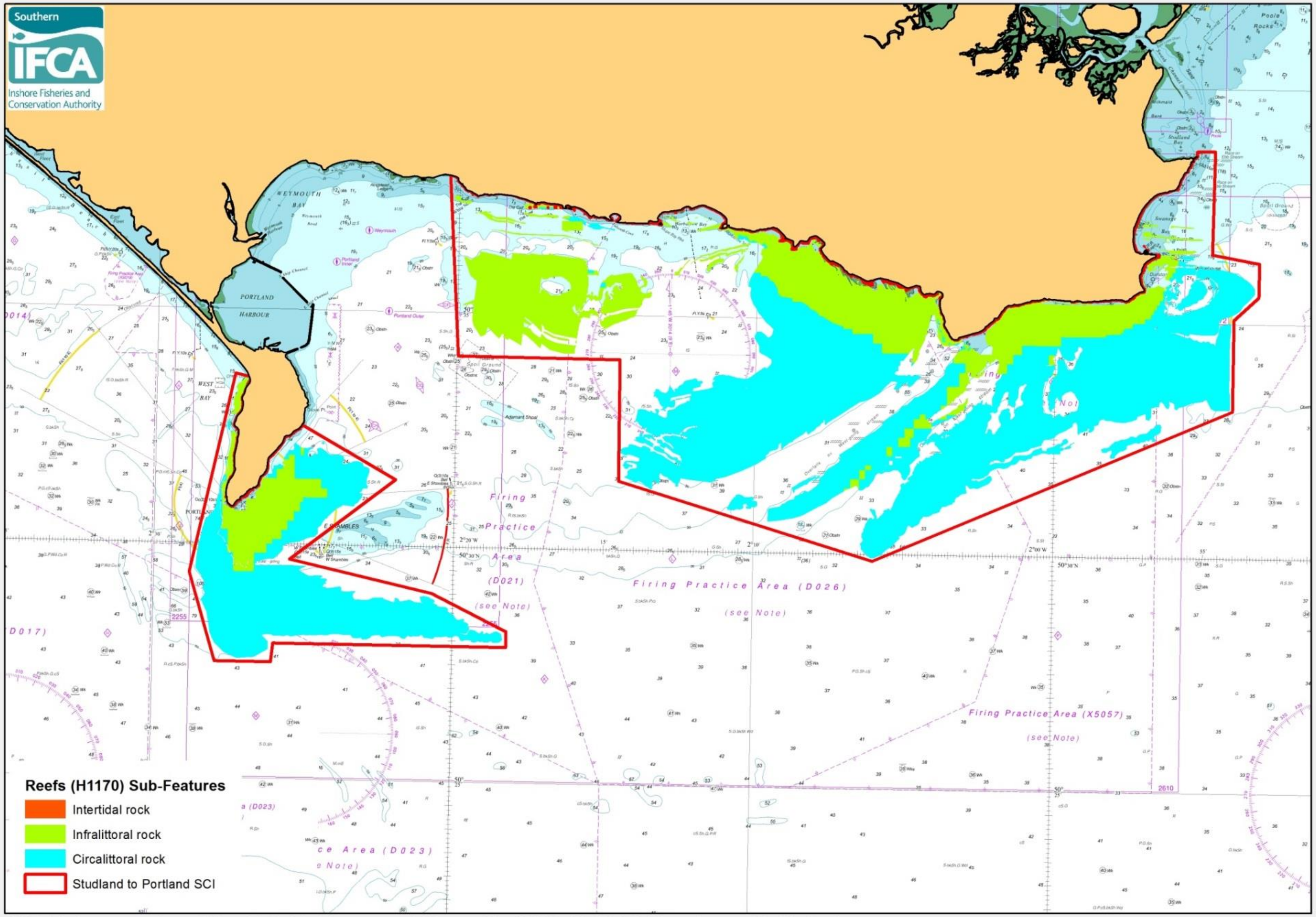
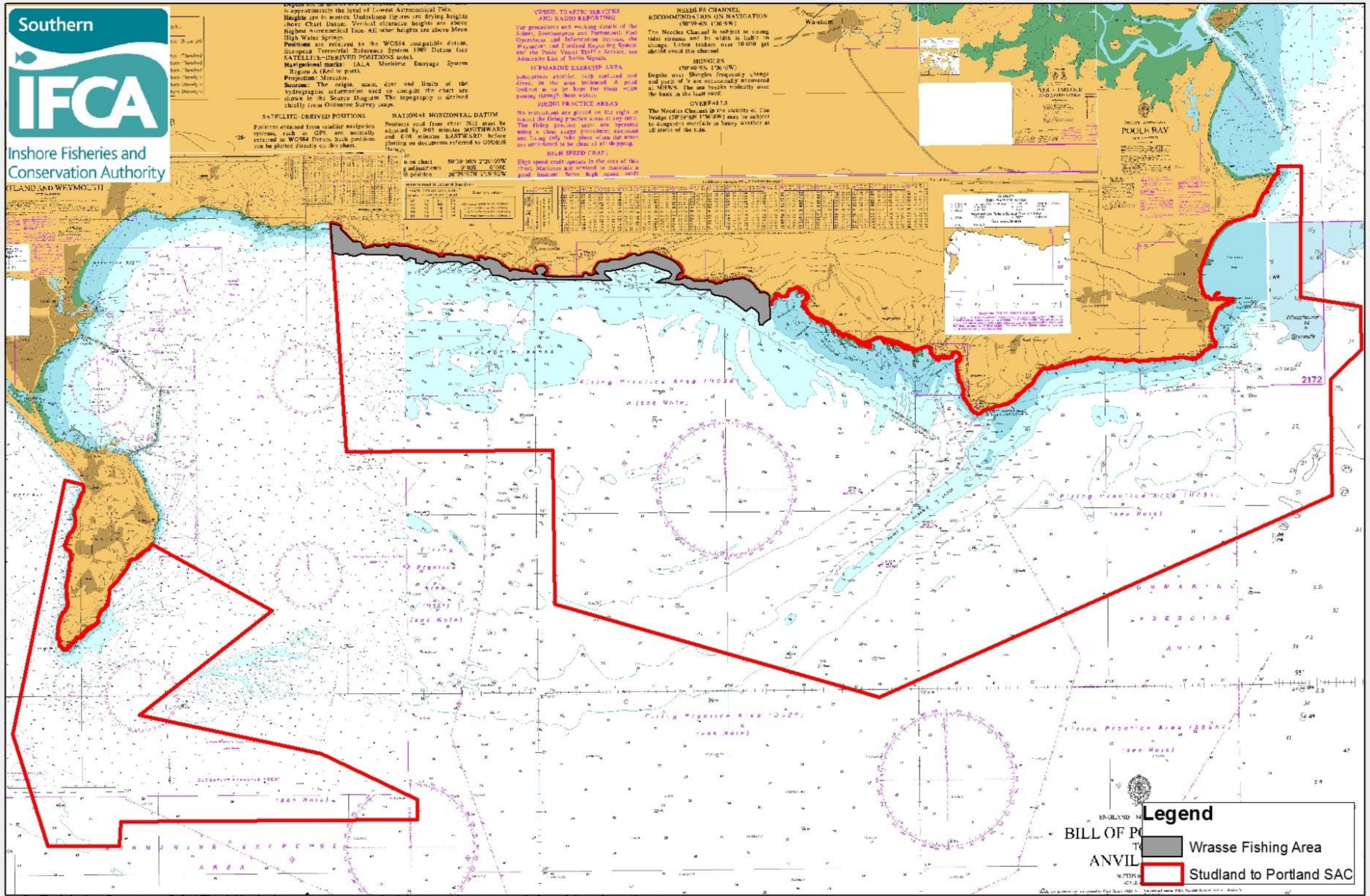
Annex 2: Site feature/sub feature map for Studland to Portland SAC.


Southern
IFCA
Inshore Fisheries and Conservation Authority
Reefs (H1170) Sub-Features
Intertidal rock
Infralittoral rock
Circalittoral rock
Studland to Portland SCI
PORTLAND HARBOUR
WEYMOUTH BAY
Firing Practice Area (D026)
Firing Practice Area (X5057)
(see Note)

Annex 3: Fishing activity map using the known area of potting for wrasse (using information provided by local fishermen) in the Studland to Portland SAC.


Southern
IFCA
Inshore Fisheries and Conservation Authority
POOLE BAY
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ANVIL
Legend
Wrasse Fishing Area
Studland to Portland SAC

Annex 4: Natural England advice on the management of the emerging wrasse fishery

Date: 23 February 2017
Our ref: Wrasse management support letter to IFCA

NATURAL ENGLAND

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By email only, no hard copy to follow

Dear Simon,

Re: Management of emerging wrasse fishery

Natural England's statutory purpose is to ensure that the natural environment is conserved, enhanced, and managed for the benefit of present and future generations, thereby contributing to sustainable development. Through this letter Natural England would like to offer their support to Southern IFCA's proposal to develop suitable management measures in collaboration with the Cornwall, and Devon and Severn IFCAs, for an emerging wrasse fishery observed to occur across the three IFCA districts on the south-west coast of England.

Within the Dorset portion of the Southern IFCA district Natural England has become aware of the development of a fishery targeting wrasse species (principally Ballan wrasse - *Labrus bergylta*, Rock cook - *Centrolabrus exoletus*, Corkwing wrasse - *Symphodus melops*, Goldsinny wrasse - *Ctenolabrus rupestris*). During the course of the 2016 fishing season (April – November) an increase in fishing intensity has been observed, both anecdotally by local stakeholders and also by Southern IFCA.

The emergence of this wrasse fishery has been attributed to an increased demand for cleaner fish species by the Scottish salmon producers, as an environmentally-friendly alternative to anti-parasitic chemical salmon treatments. With the continued growth of the salmon farming industry and the high infection rates of salmon observed within sea pens, the demand for wrasse is likely to increase.

The intrinsic value of wrasse, in particular to the ecology of inshore reefs has been highlighted by the important ecosystem function they play as a cleaner species. Cleaner fish are widely recognised as an integral part of maintaining the overall health of reef systems through the removal of parasites and by cleaning damaged tissue from fish and other marine organisms. The removal of significant numbers of wrasse could have unwanted negative impacts on animals that require cleaning, and therefore the overall health of the reef. The position wrasse occupy within the food web as both predators and prey species, in addition to their complex reproductive biology, their territorial nature and characteristic small home ranges, indicates that their removal in large numbers could seriously impact wrasse populations at a local level. It is therefore clear that further evidence is needed in order to fully understand the consequences of this fishery on reef systems.

Wrasse are not directly protected by specific UK legislation, and are not listed as a designated feature of either Special Areas of Conservation (SACs, as a European Marine Site, (EMS)), or Marine Conservation Zones (MCZ). They are not currently considered to be keystone species, nor characterising species of any reef communities (as defined by Marine Habitat Classification for Britain and Ireland (v15.03)). However, as a territorial and residential species, wrasse could be considered as part of the faunal component for particular reef communities (e.g. infralittoral rock), and therefore it is Natural England's view that wrasse should be considered in the same way as crabs and lobsters when undertaking a Habitat Regulations Assessment (HRA) or MCZ Assessment. This is because both of these mobile groups are associated with specific habitat types and provide specific ecological roles within those habitats.

As potting for these species currently occurs in the Studland to Portland Site of Community Importance (SCI), its potential impacts on their features will need to be assessed. The HRA should consider the indirect impacts of potting for wrasse on the reef feature, and also the impact of the removal of the target species (i.e. wrasse) as a group associated with the reef communities. There is some uncertainty around this latter point and further evidence is required to assess the functional role wrasse have on the condition of Annex 1 reef features. Natural England is currently reviewing this information and will provide updated advice in due course.

It is recommended that the management of wrasse should also be considered through broader legislative means, such as the Natural Environment and Rural Communities Act (2006) Section 40, and the Marine Strategy Framework Directive (MSFD). In the case of wrasse, these two pieces of legislation should act as an incentive to implement a management strategy that leads to the conservation of wrasse and their sustainable use as a fishery resource. Natural England therefore recognises and welcomes the approach taken by Southern IFCA to manage wrasse under the legislation currently available.

There is no existing evidence that points to the sustainable levels at which wrasse can be removed, and therefore this raises concerns that a further likely increase in fishing intensity could have negative consequences on reef systems. We therefore welcome the proactive approach proposed by Southern IFCA to investigate ways in which to manage potential impacts on wrasse populations. One such way of managing potential impacts could be through the introduction of appropriate minimum and maximum size limitations for each species, to prevent recruitment overfishing and protect the spawning stock. Information on their reproductive biology currently represents the best available evidence on wrasse, and therefore Natural England believe that until more information on the removal of wrasse is known, implementing a size slot would be a sensible way to proceed. We would also support the introduction of measures to fully document the fishery through the collection of fisheries data. Developing this baseline would further our understanding of the fishery and provide a sound foundation on which future management can be developed.

With the impacts surrounding the fishery uncertain, Natural England is keen for a collaborative and consistent approach where possible among the three south-western IFCAs. A joined up approach would find common ground in terms of the data collected and management implemented, while also facilitating data sharing, coordination (non-duplication) of research, and the exchange of information from the industry.

Natural England would like to work with Southern IFCA during the development of these measures, through providing support and/or advice as required. On agreement of the management measures, and to ensure that wrasse are afforded some protection, we would support their implementation before the start of the next fishing season, which is anticipated to commence in March/April 2017.

If you have any queries relating to the content of this letter please do not hesitate to contact me using the details provided below.

Yours Sincerely

A Narozanski

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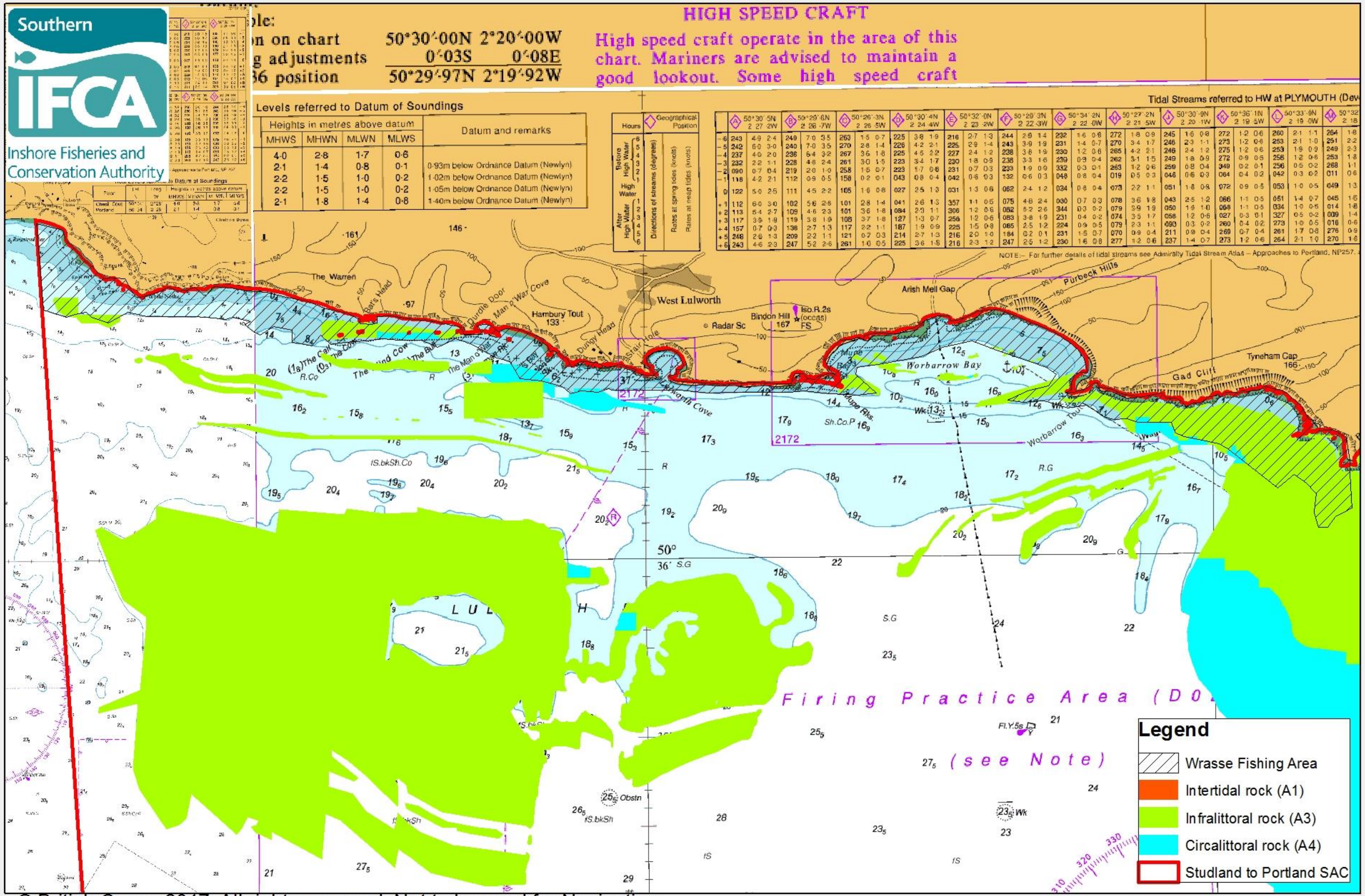
Annex 5. Gear intensity thresholds defined by different potting impact studies.

Study	Gear Intensity Thresholds
Eno *et al*., 2013	Heavy — Lifted daily, more than five pots per hectare (i.e. 100m by 100m) (equivalent to over 182,500 pot hauls per km^2 per year) Moderate — Lifted daily, two to four pots per hectare (equivalent to 73,000– 182,500 pot hauls per km^2 per year) Light — Lifted daily, less than two pots per hectare (equivalent to less than 73,000 pot hauls per km^2 per year) Single — Single accidental fishing event of a string
Young *et al*., 2013	Very high - 250+ pots per km^2/12 strings per km^2 High - 175-250 pots per km^2/9-11 strings per km^2 Moderate - 100-175 pots per km^2/6-8 strings per km^2 Low - 50-100 pots per km^2/3-5 strings per km^2 Very low - 0-50 pots per km^2/0-2 strings per km^2 None - 0 pots per km^2/0 strings per km^2
Stephenson *et al*., 2016	Low – 0 – 139 pots $month^{-1}$ km^{-2} (equivalent to 4170 pot hauls $month^{-1}$ km^{-2}, assuming 30 hauls per month) Medium – 140 – 187 pots $month^{-1}$ km^{-2} High – 188 – 265 pots $month^{-1}$ km^{-2} (equivalent to 7950 pot hauls $month^{-1}$ km^{-2}, assuming 30 hauls per month)
Rees *et al*., 2016	Low – 5 – 10 pots 0.25 km^{-2} (equivalent to 20-40 pots per km^{-2}) Medium – 15 – 25 pots 0.25 km^{-2} (equivalent to 60-100 pots per km^{-2}) High – 30+ pots 0.25 km^{-2} (equivalent to 120 pots per km^{-2})

Annex 6: Co-location of fishing activity using the known area of potting for wrasse (using information provided by local fishermen) and site feature(s)/sub-feature(s) in the Studland to Portland SAC.


Southern IFCA
Inshore Fisheries and Conservation Authority
HIGH SPEED CRAFT
High speed craft operate in the area of this chart. Mariners are advised to maintain a good lookout. Some high speed craft
Levels referred to Datum of Soundings
Heights in metres above datum
MHWS
MHWN
MLWN
MLWS
Datum and remarks
Tidal Streams referred to HW at PLYMOUTH
The Warren
West Lulworth
Hambury Tout
Arish Mell Gap
Purbeck Hills
Worbarrow Bay
Tyneham Cap
Gad Cliff
Radar Sc
Bindon Hill
Firing Practice Area
(see Note)
Legend
Wrasse Fishing Area
Intertidal rock (A1)
Infralittoral rock (A3)
Circalittoral rock (A4)
Studland to Portland SAC

Annex 7: 'Wrasse Fishery Assessment' Internship Report by Larisa Lewis (August 2017)

<u>Assessing the new wrasse fishery in Southern IFCA District</u>

Since 2016, Southern IFCA has seen the development of a live-capture wrasse fishery in the district for the use in Scottish salmon farms as cleaner fish (Davies, 2016). The Authority is aware of a small number of fishermen that have been consistently operating over this period, with the majority of activity concentrated in Weymouth Bay and around Portland. Wrasse fishermen use small open fishing vessels (<8m length) to deploy and retrieve strings of up to ten baited pots. Captured wrasse are kept in cages in the harbour until they are collected and transported up to Scotland. These fishing methods have been found to be relatively small-scale and non-detrimental to the integrity of the site which has SAC status (Walmsley *et al*., 2015; Davies 2016), and furthermore demonstrates an economically viable small-scale industry (Riley, 2017). However, there is no information regarding the wrasse population demographics nor an estimate of the annual wrasse landings from the Southern District as a whole.

Similar fisheries have emerged prior to this in other districts such as Cornwall IFCA in 2015, and it is probable that these fisheries will continue to emerge as the demand for effective bio-controls such as wrasse continues (D'Arcy, 2013). Cultivating wrasse themselves in aquaculture has been considered (see Karlsbakk *et al*., 2013, Skiftesvik *et al* 2013) yet no large-scale alternative to live-capture is currently in action. Despite the wide-spread nature of these fisheries throughout the UK and Europe, information regarding the impacts on stock abundance and population demographics is still limited. For example, these fisheries have been established in Norway since 1990 (Skiftesvik, 2014) and there have been numerous studies attempting to quantify the effects on wild wrasse populations, though the majority have focussed on goldsinny and corkwing wrasse.

The five wrasse species found in the UK (ballan, goldsinny, corkwing, cuckoo and rockcook) were recognised to have distinct life histories (Skiftesvik *et al*. 2015) which led to the development of species-specific size restrictions and voluntary recommendations in the form of a 7-point 'Fishery Guidance' plan (taken from Gravestock, 2017):

- Species-specific minimum and maximum size limits, with the aim of maintaining population size frequency distributions and promoting recruitment (protecting immature individuals and older more fecund individuals)
- A series of no take zones for wrasse, located within sections of marine protected areas (including Studland to Portland SAC)
- Pot depth restrictions (>10 m)
- Effort restriction through pot limitations
- Seasonal closures
- Monthly fishermen catch returns
- Biosecurity compliance

Importantly, the new fishery in the Southern IFCA District is primarily solely targeting ballan wrasse (*Labrus bergylta*), due to their resilience in both transport and salmon-cages; making them the most effective species for their role in aquaculture (*Leclercq, 2014*). Therefore, it is necessary to consider the potential implications of the removal of this individual species. Firstly, as ballan are protogynous hermaphrodites (Davies, 2016), they are exposed to risks of sex-selective harvesting causing skewed sex-ratios and shifts in size structure (Villegas-Rios *et al*. 2013a). A size restriction was set at 160-280mm to allow caught individuals to have reached maturity and therefore had an opportunity to reproduce prior to being caught. This would also reduce the possibility of removing large, mature males (Gravestock, 2017) which would change social structure and compromise egg survival; as guarding is a male role (Darwall *et al*. 1992). Conversely, size restrictions could affect the population demographics by promoting the survival of small, slower maturing females and very large males.

Furthermore, the resultantly small home ranges from the highly territorial behaviour combined with low genetic diversity between local sites (Villegas-Rios *et al*. 2013a), compared with other wrasse species, leaves them extremely vulnerable to threats such as disease when experiencing fishing pressure. Ecosystem-wide effects of the fishery have been considered to be low; with the potting techniques mirroring those of lobster fisheries and therefore, when correctly deployed causing minimal damage to reefs or species of special interest (Gravestock, 2017). Consequences of removing ballan from their role in top-down grazer control is likely to be mitigated by the presence of the other wrasse species (Halvorsen, 2016). However, as ballan are the largest wrasse species, complications for the health of the ecosystem could still arise as studies have shown reduced effectiveness with a decrease in mean fish size (Figueiredo *et al*. 2005).

Past literature is useful guidance for the development of assessment, however, in order to understand the potential implications of targeting ballan wrasse from a no doubt complex ecosystem, extensive surveying and research will need to be undertaken in order to ascertain the sustainability of this fishery.

It is evident that immediate comprehensive assessments are necessary to understand the impact of this fishery. Already, requirements for long-term and short-term strategies have been identified and outlined by Devon & Severn IFCA in Ross (2017), with the end goal of establishing a Maximum Sustainable Yield. Short-term strategies focus on establishing a relationship between fishing pressure and stock abundance (Ross, 2017), with the identification of the assessment complexities that this fishery faces due to the nature of fishery and the ecology of the target species; small home ranges and therefore effects of hyperstability or hyperdepletion (Ross, 2017).

This report will therefore focus on the current monitoring undertaken by Southern IFCA, providing the foundations for a stock assessment. Data collection has thus commenced ranging from surveys aboard fishing vessels, to landings and catch return data provided by buyers and fishermen respectively. These data will be examined for their strength in providing a short-term stock assessment and further local wrasse population demographics, which will ultimately inform future research direction and management decisions.

Materials and methods

Sampling took place during the months of June and July 2017, though was limited by adverse weather conditions, so overall only three potting surveys were carried out aboard local fishing vessels in Weymouth Bay. The area surveyed is characterised by bedrock supporting red algae with an approximate depth range of 5-20m (Axelsson *et al*., 2011); agreeing with previously described wrasse habitat of rocky reefs (Dipper, 2001).

Sampling techniques were those used commercially by the fishermen; hauling pots (72Lx40Wx28H) baited with shore crab (*Carcinus maenas*) and a soak time of 24-48 hours. Wrasse species and by-catch were identified, measured by total body length and released. Due to regulations set out in the 7-point plan by Southern IFCA, pot depth did not exceed 10m. There were no catch mortalities found upon hauling pots, however predatory seabirds were observed feeding on the fish being returned, therefore efforts were made to reduce this. These included releasing the fish very close to the side of the vessel or allowing recovery from depth by keeping fish in sheltered tanks with release at the end of the survey. Duration of sampling varied depending on the number of strings hauled, as some vessels had only 6 strings whilst others hauled up to 9 strings.

Additional data was provided through fishermen's logbooks to give an estimate of Catch per Unit Effort (CPUE) in Weymouth, whilst landings data from the buyers provided an idea of the export of Ballan wrasse from the Southern IFCA District, from April to July 2017.

The data presented below focusses solely on ballan wrasse, though all by-catch were identified and measured; for location and size information regarding other species e.g. lobsters, please see the raw data. Metadata for each survey can also be found within processed and original datasheets.

Results & Discussion

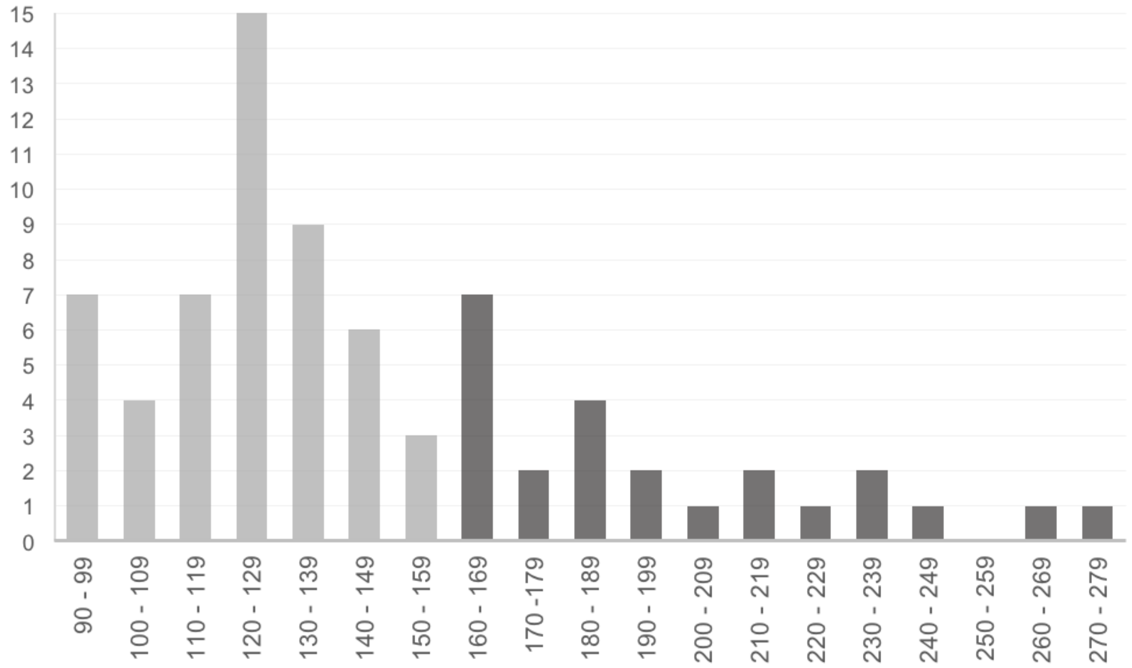
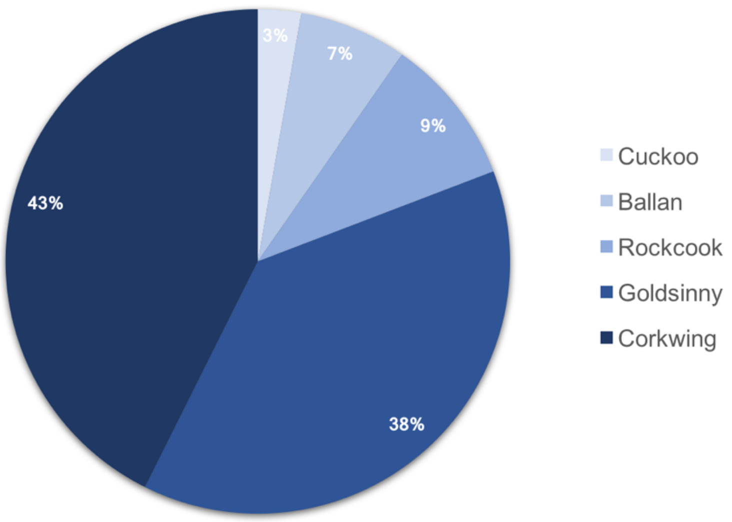


Figure 1. Range frequencies for the different size classes (mm) of sampled ballan wrasse for all three surveys in Weymouth during June – July 2017. Dark columns represent the wrasse which fell within the 160-280mm size restriction and were subsequently retained by the fishermen.



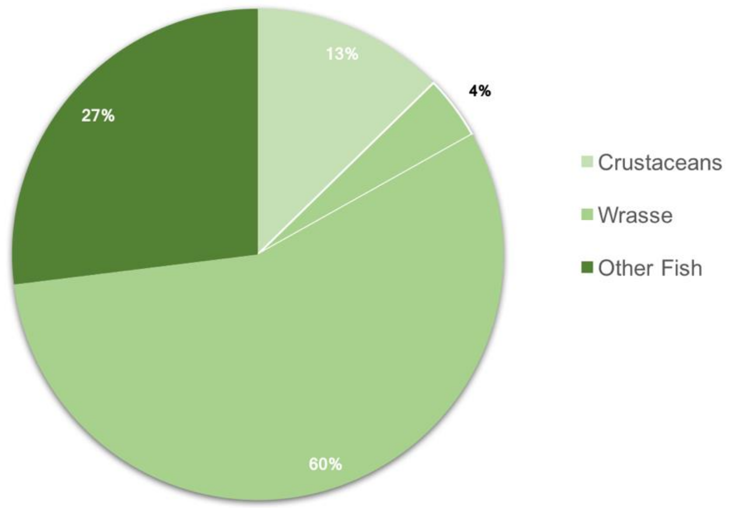


Figure 2. The species composition of the pots sampled in Weymouth Bay, using data collected from the three surveys during June – July 2017. The top chart shows the species composition for wrasse only; which makes up 60% of the total species composition shown in the bottom chart.

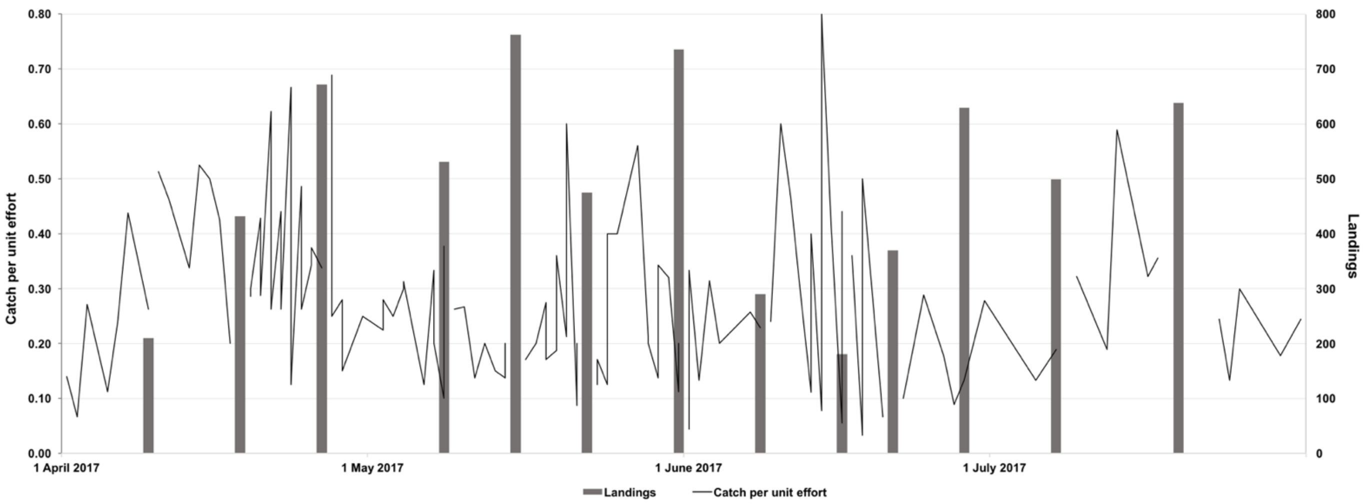


Figure 3. Potting catch per unit effort of one fisherman in Weymouth, and also the total landings for Weymouth (which includes potting and pole and line catch methods) during April – July 2017.

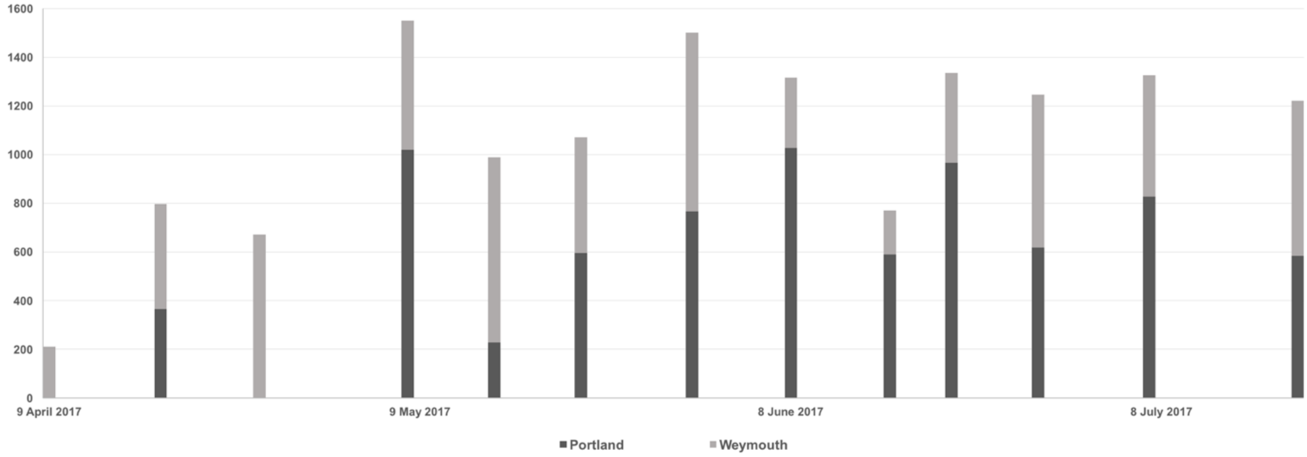


Figure 4. Total landings for the Southern IFCA District for April – July 2017

Over the three surveys, 75 Ballan wrasses were caught. However, due to the size restrictions, only 24 of these were retained by the fishermen - as signified by the dark columns in **Figure 1**. The minimum total length measured was 90mm and maximum was 270mm, whilst average total length was 146mm. The undersized majority of caught individuals is clearly shown in **Figure 1**, with 68% of total ballan caught measuring less than 160mm, and 29% of these falling into the size class of 120-129mm.

It could be assumed that these are immature individuals, however it would be useful to mirror techniques used by Halvorsen (2017) to age a sample of this population. This is done by euthanizing a small selection of pots and enables estimations to be made for future years stock population dynamics. Furthermore, a significant error in the data collection was not differentiating the sex of the individual measured; as past research suggests the population will be skewed towards immature females and a small number of large males (Halvorsen, 2016). Though sex could potentially be estimated from the sizes shown in **Figure 1** this is undoubtedly inaccurate, and so useful information regarding stock structure is potentially omitted. Future sampling must therefore ascertain sex when measuring individuals through distinguishing colouration and gonads. Similarly, recording whether individuals are spawning is also beneficial as during the first survey, in mid-June, three spawning corkwings were found. However, since the study focus is on ballan wrasse, future surveys should aim to take place during their spawning period, enabling the current regulations for closed season to be reviewed and made more accurate.

The distinct differences between quantity of undersized and sizeable fish also brings to attention important aspects of the life histories of ballan wrasse. Villegas-Rios *et al*. (2013) found that there are two different morphotypes of ballan wrasse in Atlantic waters, and suggested that there could be greater removal of spotted, slower maturing individuals' due to their larger mean size in comparison to the plain morphotype. Future sampling should therefore aim to distinguish between and record the presence of spotted or plain individuals. This leads onto considering different individual populations of wrasse within the Southern District; studies carried out by D'Arcy *et al*. (2013) found great population differentiation between spatially separated populations i.e. between Norway and UK. However, within regions there was considerably lower genetic diversity which may leave these populations especially vulnerable to fishing pressure (Davies, 2016). Furthermore, Villegas-Rios *et al*. (2013) suggest a link between parasitic infection and timing of sex change in ballan wrasse, so the need to understand population dynamics such as age, sex and links to genetics is evident, as reduced diversity could increase susceptibility to infection Frankham (2002). Therefore, future research into the genetic diversity of local populations using markers such as microsatellites or single nucleotide polymorphisms would be beneficial. However, as emphasised by D'Arcy *et al*. (2013) the lack of currently available markers for ballan wrasse would make this a difficult feat and would require considerable funding.

When looking at the species composition of the area; ballan wrasse made up only 7% of the total wrasse caught during the surveys, whilst corkwing (43%) and goldsinny (38%) made up the majority (**Figure 2**). Therefore, there is good evidence that, with this community, selectively harvesting ballan wrasse may not be detrimental to the wider ecosystem in terms of top-down grazer control. However, inter-specific interactions and ecological niches need to be examined because ballan wrasse could be significantly outcompeted for habitat or prey when under pressure from fishing. Alternatively, this community represents a future conservation strategy; despite the current demand for only ballan wrasse, a compromise to allow stocks to recover would be instead exploiting a highly abundant wrasse species, such as corkwing.

Due to the small amount of data collected from the three surveys, it was not feasible to test for significant differences between the species composition of locations within the bay or across dates. However, future research with a much larger data set, over a longer time scale, could potentially look into this and would allow an insight into population and community structure of wrasses in different localities and seasons. This would be especially useful if comparing the composition between fished areas and MPA's as seen in (Halvorsen, 2017), or when looking at the effect of closed seasons on the population.

So far, this sampling technique has a variety of strengths; most importantly that it is very repeatable and to an extent gives a snapshot of the population. However, due to the size restriction made on the entrance of the pots, fish greater than 280mm, which are generally males or mature females, will be excluded and therefore the population will be under-sampled to some degree. This is also true when considering saturation or the presence of escape gaps; if a pot is full of wrasse or by-catch, due to territoriality, some individuals will purposefully not enter the pots (Davies, 2016; Ross, 2017). Similarly, with escape gaps smaller individuals will also not be sampled.

Notably, only a number of pots actually had escape gaps, which is evident when looking at the large proportion of undersized fish in **Figure 1.** Escape gaps are important for reducing stress on fish which are repeatedly hauled and handled when the pots are consistently placed in their territory (Skiftesvik *et al*., 2014). This stress could have unknown long-term effects, so regulations should ensure that all pots have escape gaps**.** Overall, this exemplifies the need for a combination of sampling techniques such as baited cameras which could give a better insight to population dynamics and also species composition.

With this fishing method, there was a large number of by-catch (**Figure 2),** in total 22 species and 1796 individuals were caught over the three surveys, however ballan made up only 4% of the total catch. This could either reflect low efficiency of catch methods, hence why other techniques such as pole and line are used in conjunction with pots, or other factors may be having influence. Since the fishery effort is concentrated between April and October (Pengelly & Gravestock, 2017), the small number of sizeable individuals could be related to season; suggesting the majority have already been fished. This is emphasised further when looking at the potting catch per unit effort (CPUE), which varies substantially day to day for individual fishermen participating in the April-July surveys. There are undoubtedly a large number of factors which affect the distribution of wrasse and subsequent CPUE. Previous studies such as Darwall *et al*. (1992) have investigated correlations between wrasse abundance and temperature, whereas personal reports from fishermen imply that CPUE could be affected by the underlying tidal cycle. They believe that CPUE is often greater when potting between Spring and Neap tides rather than directly on one. Generally, weather conditions and seasons will have a strong influence, as well as competition amongst fishermen themselves (Halvorsen, 2017). With such a small data set it is difficult to recognise and give weight to trends. This highlights the need for not only collecting more detailed metadata in terms of temperature or tidal streams but also constructing a well-planned sampling schedule for a long-term dataset (> 1 year) that would coincide with different tidal cycles, seasons and overall be more consistent i.e. four times a month or one week straight of sampling.

Furthermore, **Figure 3** shows the CPUE of only one fisherman's landings (also shown on Figure 3) through potting. So, where the CPUE is low and landings are high, it is made up by the other fishermen in the area, or through different catch methods. Therefore, it would be advantageous to have catch return data from all fishermen and all fishing methods. Additionally, to improve the relevance of the catch return data and give an indication of population demographics, it could be worth implementing voluntary size measuring of retained ballan. This could be done when the fishermen are putting their wrasse into storage cages and wouldn't be too difficult or time consuming. Alternatively, Southern IFCA enforcement officers could carry out "sampling" when the fishermen return to harbour.

Overall, when considering the four months of landings data from Weymouth (**Figure 3**), there is great variability over this period. The average number of ballan wrasse bought came to 494, however there are some periods when this was significantly different. For example, in April, at the start of the season, the landings were expectedly low; only 210 ballan sold, which then increased up to the highest landing in mid-May reaching 762 ballan. In mid-June, there was a significantly large drop down to 181 ballan. It is difficult to discern whether there are any meaningful trends behind these findings, they could reflect spawning patterns, catch methods or as discussed above, be a result of an array of factors. It is worth mentioning that the number of strings per vessel increased over the period; with one fisherman going from 6 strings up to 9 (totalling 30 more pots). Pengelly & Gravestock (2017) recommended a maximum number of 8 strings per vessel, so the Authority should consider enforcing this. Further, the minimum sizes rely on the integrity of the fishermen, so the high landings could be a result of taking under-sized fish and regular enforcement will be necessary to avoid this.

The landings data shown in **Figure 4** provide a baseline for the current magnitude of exploitation; over the 4-month period a total of 14,009 ballan wrasse were exported out of the Southern District, with 7584 from Portland and 6425 from Weymouth. The Authority could potentially use this data in the future to set a landings cap for the district, as it is evident that Weymouth is not the only port in the Southern District seeing substantial fishing activity; landings in Portland are on average 195 fish higher. Consequently, future surveys should focus on both Portland and Weymouth, and engage in data collaboration with Cornwall and Devon & Severn IFCA districts, who are too undertaking survey work investigating the impacts of the wrasse fisheries. It is also worth noting from Figure 2 that lobsters make up 13% of the total catch and could potentially be used in future assessments as grounds for comparison, in terms of CPUE, with previous potting studies assessing the sustainability of lobster fisheries using catch data.

The sampling methods thus far give an accurate indication of what the fishermen themselves will be catching, and is therefore highly representative of the commercial fishery. However, when considering populations and stock assessments, as mentioned by (Ross, 2016), difficulties arise with these surveys and catch return data being spatially biased; fishermen target areas where they have highest catch and tend to move their equipment to reflect areas of high fish density. The alternative to this is carrying out systematic random sampling aboard chartered vessels as suggested by Ross (2017), which would also remove reliance on fishermen for surveying. However, this is a considerably more expensive and time consuming than the current sampling method.

Future methods of assessing the fishery outlined throughout the report include:

- Comparison of wrasse abundance and size between protected and unprotected areas within the district
- Population aging via euthanizing samples
- Genetic studies
- Time-series comparison of species composition
- Enforcing the presence of escape gaps
- Baited camera
- Devising a well-structured long-term sampling regime
- Voluntary measuring of catch, or regular sampling by Southern IFCA Officers
- Enforce maximum number of strings per vessel
- Surveying at all ports in the Southern District
- Using charted vessels

This is not necessarily a comprehensive list since techniques such as mark capture release could provide valuable information on home-ranges and ballan ecology, but may not be essential for providing a short-term, baseline stock assessment. Future assessment will therefore take these recommendations into account and decide which are the most useful and cost-effective to the long-term goal of a complete stock assessment.

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