Review of the Live Wrasse Fishery in Devon and Severn IFCA's District 2017–2020

Version 1.1 - March 2021

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Version control history			
Author	**Date**	**Comment**	**Version**
LH	03/02/2021	Draft report	0.1
LH, JS	05/02/2021	Amendments to initial draft	0.2
LH, JS	07/02/2021	Amendments including drafting conclusions and executive summary, mapping effort and updating methods section	0.3
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Publication note: This report is the fourth in a series of annual reviews of the Live Wrasse Fishery in Plymouth Sound to be published by Devon and Severn Inshore Fisheries and Conservation Authority, and is the first in this series to utilise updated methods adapted from Henly *et al.* (in review). At the time of writing this report, the article by Henly *et al.* is undergoing peer review for publication in *ICES Journal of Marine Science*. Henly *et al.* (in review) used an approach that standardised catch and landings per unit effort (CPUE and LPUE) data to account for ecologically-relevant drivers of CPUE and LPUE across Plymouth Sound as a whole. Their article therefore provides the first statistically robust assessment of this multispecies inshore fishery with such methods. The article uses similar data to those presented in this report; as such, many of the key results are similar, and the interpretation of these results has therefore followed that presented in Henly *et al.* (in review).

Henly, L., Stewart, J.E. and Simpson, S.D. (*in review*). Drivers and implications of change in an inshore multi-species fishery.

Cover image: Corkwing wrasse (*Symphodus melops*) observed in Devon and Severn IFCA's District

Contents

Executive Summary

A fishery for the live capture of wrasse for use as cleaner fish in Scottish salmon farms developed in the Devon and Severn Inshore Fisheries and Conservation Authority's (D&S IFCA's) District in 2015. Management was introduced in 2017 via the D&S IFCA Potting Permit Byelaw. These management measures have been adapted since their introduction based on evidence from the analysis of data collected during observer surveys in the D&S IFCA District. These previous analyses however, were unable to consider changes in catch per unit effort (CPUE) and landings per unit effort (LPUE) whilst controlling for variation that comes about as a result of geographical location an environmental variables. This report standardises monitoring data from the fishery observer surveys conducted by D&S IFCA's Environment Officers with fishing locations and environmental data obtained from external sources and identifies the main drivers of variation in CPUE and LPUE. Implications of the results for future management and sustainability of the fishery are discussed.

D&S IFCA's Environment Officers completed observer surveys on approximately 8.3% of total fishing trips in 2020, despite the difficulties posed by the COVID-19 pandemic. The majority of the data analysis for this report is based on the data from observer surveys, as these provide the species-specific data that are required for a robust assessment of the fishery. Continuing non-compliance issues have also prevented the full use of the data from the fishers' returns forms. The utility of returns data is discussed in detail in this report.

The main drivers of variation in CPUE and LPUE varied between species. There was evidence of a decline in ballan wrasse CPUE and LPUE, particularly on the landward side of the breakwater. This decline is likely driven by the relatively high retention rate of ballan wrasse in combination with specific life history and behavioural characteristics that leave the species vulnerable to overfishing. No negative year effects were seen for the other wrasse species.

There was no evidence of a decline in rock cook CPUE or LPUE across the 2017–2020 period, despite evidence of a decline highlighted in last year's report (Curtin *et al.*, 2020) that led to a prohibition on the removal of rock cook wrasse from the fishery. Rock cook CPUE and LPUE showed significant variation across broad-scale fishing areas (significantly lower in the more sheltered areas, which are protected from wind and wave exposure by the breakwater). As the majority of the observer surveys have been conducted in more sheltered locations in the last two years, it is unsurprising that the Three Year Comprehensive Review, which was not able to control for geographical variation in CPUE and LPUE, highlighted a decline in these measures over the 2017–2019 period.

Goldsinny wrasse showed seasonal variation in CPUE and LPUE across the survey season (decreasing from July to October) and lower catches were observed in locations closest to the freshwater outputs of the River Tamar. These observations agree with previously reported trends in the literature that suggest goldsinny wrasse are found in their highest densities in the summer months and away from locations that are influenced by freshwater runoff. Finally, there was a significant increase in corkwing wrasse CPUE across the 2017–2020 period, along with evidence of seasonal variation in CPUE and LPUE (increasing throughout the July – October season). The change in CRS limits in 2018 has likely benefitted the species as a lower proportion of corkwing are being landed and mature individuals of each sex are likely being protected. The seasonal variation may reflect the species' spawning season and concurrent activity levels.

It is recommended that, within the time- and resource-limited research capabilities of D&S IFCA (and the restrictions imposed by locations and timing of routine fishing activity), future monitoring of this fishery relies on observer survey data, and that future surveys should aim

to distribute survey effort evenly and consistently over time and locations. This would allow for important species-specific drivers of CPUE and LPUE, identified here, to be accounted for in future analyses, enabling robust monitoring and recommendations for management.

In addition to these monitoring recommendations, D&S IFCA's officers consider that action on the following recommendations will help to maintain the environmentally, economically and socially sustainable nature of the live wrasse fishery in D&S IFCA's District:

(i) Continue to manage the fishery as outlined in the D&S IFCA's Policy Statement and Potting Permit Conditions for the Live Wrasse Fishery (24th June 2020), except in the case of rock cook (ii, below) and ballan wrasse (iii, below), and except with regards to fishers returns forms (iv, below).

(ii) Lift the prohibition on removal of rock cook from the fishery, and reintroduce previous conservation reference size (CRS) limits to the Potting Permit Byelaw permit conditions for this species (12–23 cm).

(iii) Change the ballan wrasse CRS range from 15–23 cm to 18–26 cm (see Appendix 3 for an assessment of the impact that changes to the CRS limits may have on the retention rates of ballan wrasse in the D&S IFCA's District).

(iv) Remove the requirement for wrasse fishers to submit returns forms (without affecting future obligations under Paragraph 17 of the Potting Permit Byelaw).

1. Introduction

Since the early 1970s, inshore fisheries have developed in Norway, Scotland and Ireland for several wrasse species, namely: ballan (*Labrus bergylta*), corkwing (*Symphodus melops*), goldsinny (*Ctenolabrus rupestris*), rock cook (*Centrolabrus exoletus*), and cuckoo (*Labrus mixtus*) wrasse. These species are targeted for use as a biological control mechanism for the control of ectoparasites (Copepoda, Caligidae) in farmed Atlantic salmon (*Salmo salar*) (Bjordal, 1988, 1991; Treasurer, 1994; Tully *et al.*, 1996; Varian *et al.*, 1996). The use of wrasse as cleaners is suggested by some to be the most economical and environmentally friendly option for removal of sea lice (Treasurer, 2012; Liu and Bjelland, 2014) compared to other methods such as pharmaceutical, thermal and mechanical treatments (Roth *et al.*, 1993; Burka *et al.*, 1997; Burridge *et al.*, 2010; Overton *et al.*, 2019). Now large numbers of these cleanerfish are routinely being used in salmon aquaculture, with several million used each year in Norway alone (Darwall *et al.*, 1992; Skiftesvik *et al.*, 2014).

With the continued expansion of the Scottish salmon aquaculture industry, there has been an increase in the demand for wrasse to use as part of lice control strategies. This increased demand, and limited stocks of wrasse in Scottish waters (Rae, 2002), eventually put pressure on Scottish salmon companies to source wrasse from other locations around the UK, such as the south coast of England. Consequently, live wrasse fisheries developed in Cornwall, Devon and Dorset on the south coast of the UK in around 2015 (Davies, 2016; Street *et al.*, 2017; Gravestock, 2018).

Although wrasse are an efficient method for parasite treatment in the aquaculture sector (Bjordal, 1988, 1991; Costello and Bjordal, 1990; Skiftesvik *et al.*, 2013), the removal of large numbers of fish from wild populations poses questions regarding the sustainability and potential impacts of such exploitation on wild stocks. Furthermore, live wrasse fisheries have resulted in some conflict between stakeholders. Wrasse species are important targets for recreational sea anglers, particularly on the south coast of the UK; recreational anglers have expressed concerns over sustainability of the fishery and the consequences of removals on wrasse populations and other species within the ecosystem.

In an attempt to achieve sustainable exploitation of wrasse and avoid conflict among stakeholder groups, management measures have been developed from existing knowledge of the biology, behaviour and ecology of wrasse. Devon and Severn Inshore Fisheries and Conservation Authority (D&S IFCA) implemented management measures in June 2017, through permit conditions associated with the Potting Permit Byelaw (Clark and Townsend, 2017). These included a pot limit of 120 pots per permit holder (and an understanding that there would be up to four permit holders actively fishing for wrasse at any one time), maximum and minimum Conservation Reference Size (CRS) limits for each species, closed seasons, voluntary closed areas, and requirements for fishers to document their daily effort, landings and fishing locations. D&S IFCA's Environment Officers also carry out onboard observer surveys on a proportion of the fishing trips to record a more detailed sample of catch and landings. The data from these observer surveys have formed the basis for most of the analyses in each of the annual monitoring reports to date.

The Potting Permit Byelaw permit conditions have been adapted a number of times on the basis of analyses carried out on the data collected by D&S IFCA. These analyses provided evidence on possible improvements to management to further ensure the sustainable management of the fishery in D&S IFCA's District. For example, following analysis of observations of spawning wrasse during observer surveys in the first year of data collection, the closed season was changed to better protect spawning individuals. In addition, in 2018 the CRS limits of corkwing wrasse were changed to increase the proportion of this species that was returned to the sea, and so afford protection to greater proportions of smaller and larger individuals of the species. Following the Three Year Comprehensive Review of the fishery in D&S IFCA's District (Curtin *et al.*, 2020), D&S IFCA prohibited the removal of rock cook from the fishery due to evidence of a decline in the catches and landings per unit effort (CPUE and LPUE) across the three-year period 2017–2019. However, these previous analyses were unable to consider changes in CPUE and LPUE whilst controlling for variation that comes about as a result of geographical location an environmental variables.

Standardised CPUE and LPUE data from observer surveys can be used to obtain a relatively quick assessment of stock abundance dynamics (Metri *et al.*, 2014) and robustly assess fishery effects on target species (Henly *et al.,* in review). Standardisation of CPUE and LPUE accounts for the influence of spatial and environmental variables on catch rates, allowing for a more accurate representation of stock abundance dynamics over time (Maunder and Punt, 2004; Venables and Dichmont, 2004). It also permits identification of the variables that influence catch rates, and can therefore provide information on the ecology and population dynamics of the target species, that can help inform management decisions (Maunder and Punt, 2004). This report uses monitoring data from the fishery observer surveys conducted by D&S IFCA over the 2017–2020 period, along with environmental data obtained from external sources, to identify the main drivers of variation in CPUE and LPUE. Implications of the results for future management and sustainability of the fishery are then discussed.

2. Methods

2.1. Study system

Plymouth Sound (south west England) is an open bay with steeply sloping rocky coastlines to the east and west. The bay comprises diverse communities and habitats, including intertidal and subtidal limestone reefs and subtidal sediments. The inner Sound is sheltered by an artificial breakwater; other important geographical features include Drake's Island at the mouth of the Tamar estuary (in the north-west of the Sound) and the Mew Stone, an island to the south-east of the Sound. The River Tamar provides the dominant freshwater input into Plymouth Sound, with an annual average flow of $30m^3$ s^{-1} (Uncles *et al.*, 2015). Plymouth Sound, and the estuaries of the rivers that flow into it, are designated as a European Marine Site, with the aim to protect the prominent Annex I Habitats and Annex II species it contains (EC Birds and Habitats Directive), including reefs and their associated communities.

A Live Wrasse Fishery developed in Plymouth Sound in early 2015, and the Inshore Fisheries and Conservation Authorities (IFCAs) became aware of the fishery in 2016. Plymouth Sound falls under the jurisdiction of both Cornwall IFCA (CIFCA) and D&S IFCA, both of which have developed management measures relevant to the Live Wrasse Fishery. D&S IFCA's first management measures for this fishery came into force in July 2017 under the Potting Permit Byelaw, and included a maximum of 120 pots per permit holder, minimum and maximum conservation reference sizes (CRS) for retained wrasse, and closed fishing seasons. Several voluntary measures were also agreed, including a set of closed areas defined in collaboration with industry. The current and previous management measures are summarised in D&S IFCA's Byelaw Status and Changes Guide (Townsend, 2020).

The fishery in D&S IFCA's District comprises up to four vessels per year, each ranging from approximately five to ten metres in length. Over the course of 2017–2020, some vessels have left the fishery and been replaced by new entrants. In 2020, Vessels 3, 4 and 6 were active in D&S IFCA's District. Vessel 3 appears to have had minimal involvement in the Live Wrasse Fishery, Vessel 4 was predominantly active in CIFCA's District, but typically set a single string of approximately 40 pots in D&S IFCA's District on each trip, while Vessel 6 set up to six strings of 20 pots entirely within D&S IFCA's District on each trip. Table 1 summarises anonymised details of each vessel for context.

Fishers set strings of lightweight, rectangular wrasse parlour pots (traps) in varying numbers. All pots are manufactured by Carapax (Lysekil, Sweden), and are usually baited with crabs or bait balls to attract wrasse. Pots are designed to exclude bigger fish and are fitted with escape gaps to allow smaller wrasse to escape. Differences in fishing practices between fishers (vessels) relate to fisher preferences for fishing location, soak time (duration of trap deployment) and bait type.

__Table 1.__ Summary of vessels actively fishing for wrasse in D&S IFCA's District during 2017–2020.

Vessel number	Years active	Comments
1	2017	Fished in both CIFCA's and D&S IFCA's Districts.
2	2017 – 2019	Fished in both CIFCA's and D&S IFCA's Districts.
3	2017 – 2020	Minimal known activity in D&S IFCA's District in 2020. Fishes in both CIFCA's and D&S IFCA's Districts.
4	2017 – 2020	Predominantly fishes for wrasse in CIFCA's District. Typically uses up to two strings of pots in D&S IFCA's District on each trip.
5a	2018	Same skipper as 5a, changed vessel during 2018. Fished in both CIFCA's and D&S IFCA's Districts.
5b	2018	Same skipper as 5b, changed vessel during 2018. Fished in both CIFCA's and D&S IFCA's Districts.
6	2019 – 2020	Fishes entirely in D&S IFCA's District, typically along the eastern coastline of Plymouth Sound.

2.2. Data collection

Four classes of data were collected for this review: landings data, recorded and submitted by the fishers (Section 2.2.1), sales notes for wrasse landed in Plymouth Sound (Section 2.2.2), fishery observer surveys undertaken by D&S IFCA Environment Officers (Section 2.2.3), and supporting environmental and geographical data (Section 2.2.4).

2.2.1. Returns (total landings) data

As part of the Potting Permit Conditions introduced in 2017, and the aim to have a fully documented Live Wrasse Fishery, fishers are required to complete and submit catch return forms to D&S IFCA. Using these forms, fishers are required to record the total number of wrasse landed (caught and retained), number of strings and pots fished, and the location of fishing (1km^2 grid square) for each day of fishing. There is also space for the fishers to record the number of each individual species landed. The catch returns dataset has the potential to provide fine-scale records of wrasse removals and fishing effort and could document all retained wrasse. However, species-specific information is rarely recorded and occasionally, catch returns forms are submitted that contain only an aggregate total number of wrasse for several fishing trips, meaning it is not able to assess fine-scale day-by-day wrasse removals and fishing effort. Landings data are supplemented by transport documents supplied to the Marine Management Organisation (MMO) by the salmon farm agent, though these relate to landings across both D&S IFCA and CIFCA's Districts so it is not possible to verify the returns forms. However, by comparing these data it is possible to identify that one vessel has not been submitting returns forms in 2020.

2.2.2. Sales notes

Sales notes were obtained from the MMO, detailing the number of wrasse (overall, not species-specific) landed from Plymouth Sound by each vessel over the 2017–2020 period.

2.2.3. Fishery observer data

In order to collect a more detailed sample of catch and landings data, fishery observer surveys are completed during a sample of routine fishing operations. For each survey, the date, time and precise fishing locations (start and end points of each string hauled) are recorded using a GPS unit. Fishers provide information on the bait used (usually either crabs or krill bait balls), the number of pots per string, and the soak time of each string (recorded here as nights lie: the number of nights the pots were set for). Each wrasse caught is identified to species level and measured to the nearest 0.5 cm, with those outside the CRS range being immediately returned to the sea. Surveys are conducted between April and December each year (2017–2020), although survey effort for each month has varied between years (Curtin *et al.*, 2020). To reduce sampling bias in this analysis, we only use data from months that were surveyed across most years (July – October, though the 2020 surveys began in early August).

2.2.4. Supporting environmental and geographic variables

Bathymetric data for Plymouth Sound were obtained from the EMODnet Bathymetry Consortium Digital Terrain Model (DTM) (EMODnet, 2020); the DTM is based on a grid of 1/16 × 1/16 arc minutes of longitude and latitude (ca 115 × 115 meters), showing water depths relative to the Lowest Astronomical Tide Datum (LAT). All string start and end positions were plotted on the bathymetric data in QGIS (QGIS, 2020) and the mean depth of the two points was calculated. Strings of pots are not always laid in a straight line, and may vary in depth across the length of the string, so each string was therefore assigned to a 5 m depth band (DB) between 0–15 m. There was also a DB for areas above 0 m, which are intertidal on a low spring tide.

We calculated ‘distance to structure’ (Dist) for each string based on an average of the distance to the coastline or artificial structure (e.g. breakwater) of the start and end positions of each string. For strings where the start and end positions were closest to different structures, we recalculated distance to structure for each position based on the single structure which was closest to the string overall. Single depth and distance to structure estimates were used for the rare cases in which only one of the start or end positions of the string were recorded.

Each string of pots was assigned to two classes of fishing area based on (a) their position relative to the breakwater (BW: landward or seaward); and (b) the broad-scale fishing area (BA: A–E) in which they were hauled (Figure 2). Fishing areas A–E were defined based on the main fishing areas recorded during the observer surveys and the conditions of wave, wind and current exposure experienced across the Sound, based on their position relative to land, headlands, channels and the prevailing SSW winds (Uncles *et al.*, 2015). Although the 1 km^2 grid squares used for management (e.g. defining closed areas) are useful to visually monitor fishing effort within the District, they are at too fine a scale for effective use in modelling catch and landings per unit effort because (i) it is not always possible to assign surveyed strings to a single grid square, and (ii) individual grid squares have low sampling effort, leading to problems with model fitting. Conversely, no strings cross between the broad-scale fishing areas defined in Figure 2, and sampling effort in each area is sufficient to enable statistical analysis.

2.3. Data Analysis

R v3.6.1 or later (R Core Team, 2020), and PRIMER 6 with PERMANOVA + (PRIMER-E Ltd, Plymouth, UK) were used for all data analyses.

2.3.1. Calculation of $LPUE_{total}$ from catch returns data

Landings per unit effort from returns data of all wrasse species combined ($LPUE_{total}$) were calculated for each year by dividing the number of landed fish by the number of pots fished, using records from all fishing trips for which both fish and pots information were clearly recorded. In some cases, fishers specified the number of fish landed without specifying the number of pots (or *vice versa*) – these records were not included.

2.3.2. Calculation of CPUE and LPUE from observer surveys

Both catch per unit effort (CPUE) and landings per unit effort (LPUE) were calculated for each individual wrasse species from observer survey data. CPUE was calculated for each string as the number of each wrasse species caught per string divided by the number of pots in the string. No adjustment to the unit of effort was made for soak time, as a recent study highlighted no effect of nights lie on the total catch of wrasse, or of individual species (Henly *et al.*, in review). CPUE includes all wrasse, whether kept or returned to sea. LPUE was calculated as per CPUE, but using only those wrasse that were landed (kept) rather than returned (released). A proportion of each catch is returned to the sea due to being (i) damaged, (ii) dead, or (iii) not of a size that can be removed from the fishery; these returned fish were excluded from LPUE calculations. D&S IFCA specify species-specific maximum and minimum CRS (Townsend, 2020) which define the sizes that can be removed from the fishery.

2.3.3. Size structure

Differences between years in the sizes of wrasse caught during observer surveys were assessed using Kolmogorov-Smirnov (K-S) Tests. For each species of wrasse, this method tests whether the size distributions are the same in each year. However, because all wrasse were measured to the nearest 0.5 cm, the size distribution data are not suitable for use with traditional K-S tests (which require continuous data). Therefore, for these tests a 'bootstrap' approach was applied, which renders the tests insensitive to problems associated with non-continuous data. Multiple tests were performed on each dataset (e.g. 2017 data were compared with 2018, 2019 and 2020 in separate tests); therefore, the associated p-values were adjusted for multiple comparisons. The bootstrap approach and p-value adjustment were implemented using the R package 'Matching' (Sekhon, 2020).

2.3.4. Statistical modelling

Generalised linear models (GLMs) were used to assess the drivers of any change in CPUE, LPUE and average size of wrasse within the D&S IFCA District. Details of this method are presented in Appendix 1. Following Henly *et al.* (*in review*), this GLM approach has been applied in order to account for the impact of spatial and environmental variables on CPUE, LPUE and size, allowing for a more accurate representation of stock abundance dynamics and size changes over time (Venables & Dichmont 2004, Maunder & Punt 2004). Without accounting for the effects of these additional spatial and environmental variables in this way, there is a risk of either (i) incorrectly attributing apparent changes in CPUE, LPUE or size to the effects of fishing effort, or (ii) of not detecting an effect of fishing effort if it is masked by changes in other variables. This approach also permits identification of the variables that influence CPUE and LPUE and size of fish caught, and can therefore provide information on the ecology of the target species, which can help to inform management decisions (Maunder & Punt 2004; Henly *et al.,* in review).

Within this GLM-based statistical approach, day of year (DOY), year (Y), breakwater position (BW), broad area (BA), average distance to shore/ structure (Dist), depth band (DB), and bait type (Bait), were considered as potential predictors. All plausible two-way interactions between spatial/environmental (BW, BA, Dist, DB) and temporal (DOY, Y) predictors were also considered. For example, in a GLM for CPUE, a two-way interaction between Y and BA would indicate that CPUE changes between years (Y), but that the change experienced is different between broad areas. In this GLM approach all plausible combinations of variables and interactions were considered in individual models (GLMs) for each species. Model selection techniques were then applied in order to determine which model (which combination of predictor variables) is the ‘best’ model given the data (Appendix 1). The modelling, model selection and model diagnostic approaches are outlined in Appendix 1. Tukey tests, with p-values adjusted for multiple comparisons, were used to test for significant differences between levels of categorical predictors in the final models using the R package ‘multcomp’ (Hothorn *et al.*, 2020).

2.3.5. Multivariate catch composition analysis

Permutational multivariate analysis of variance (PERMANOVA) and associated pairwise tests were used to test for changes in the catch composition of wrasse between years (assessing how the community changed overall), while distance-based linear models were used to determine the drivers of change in catch composition, based on the same variables as used in the GLMs. The approaches to these methods are outlined in further detail in Appendix 2.

3. Results

3.1. Fishing Effort

There remain some challenges in interpreting fishers' returns forms due to incomplete data. However, these data suggest that fishing effort in the D&S IFCA's District has reduced over the course of 2017–2020 (Table 2). Fishing effort appears to have moved into more sheltered areas of Plymouth Sound since 2017 (Figure 1).

Fishing largely took place outside of the voluntary closed areas which were implemented in April 2018 (Figure 2). However, over the course of 2019 and 2020 a total of six incursions into a closed area in the south of Jennycliff Bay are known to have occurred (cell M12, Figure 2). These incursions occurred on days that an observer was monitoring the vessel, though it was not possible to determine the location of fishing relative to the closed area until after the fact. The fisher involved typically used six strings of pots in areas along the eastern coast of Plymouth Sound, from Batten Bay to Renney Rocks, and regularly re-shot his gear in locations near to the site of hauling; it is therefore possible that the fisher was also fishing in the closed area on other days.

Table 2. *Data from fishers' returns forms 2017–2020. Vessel 3 returned few forms which were deemed to be unreliable (Curtin et al., 2020), so these data have not been included. All reported fish, strings and numbers of pots are shown here; however, the LPUE data reported here excludes cases in which either one of the number of fish or the number of pots was not clearly recorded on the returns forms. This allows an estimate of total effort and landings while providing a more robust estimate of annual LPUE. The final column shows estimates of total wrasse landed from Plymouth Sound during 2017–2020, based on sales notes provided to the MMO by the salmon farm agent. †Sales notes data from 2017 have an associated degree of uncertainty, and so an estimate is presented based on landings data (as outlined in Davies and West, 2017).*

Year	Strings hauled	Pots hauled	Fish landed (returns forms)	LPUE	Fish landed (sales notes)
2017	1738	42548	38185	0.90	46497†
2018	585	16118	13129	0.81	39324
2019	493	11223	9084	0.81	18120
2020	336	10080	8458	0.87	16776

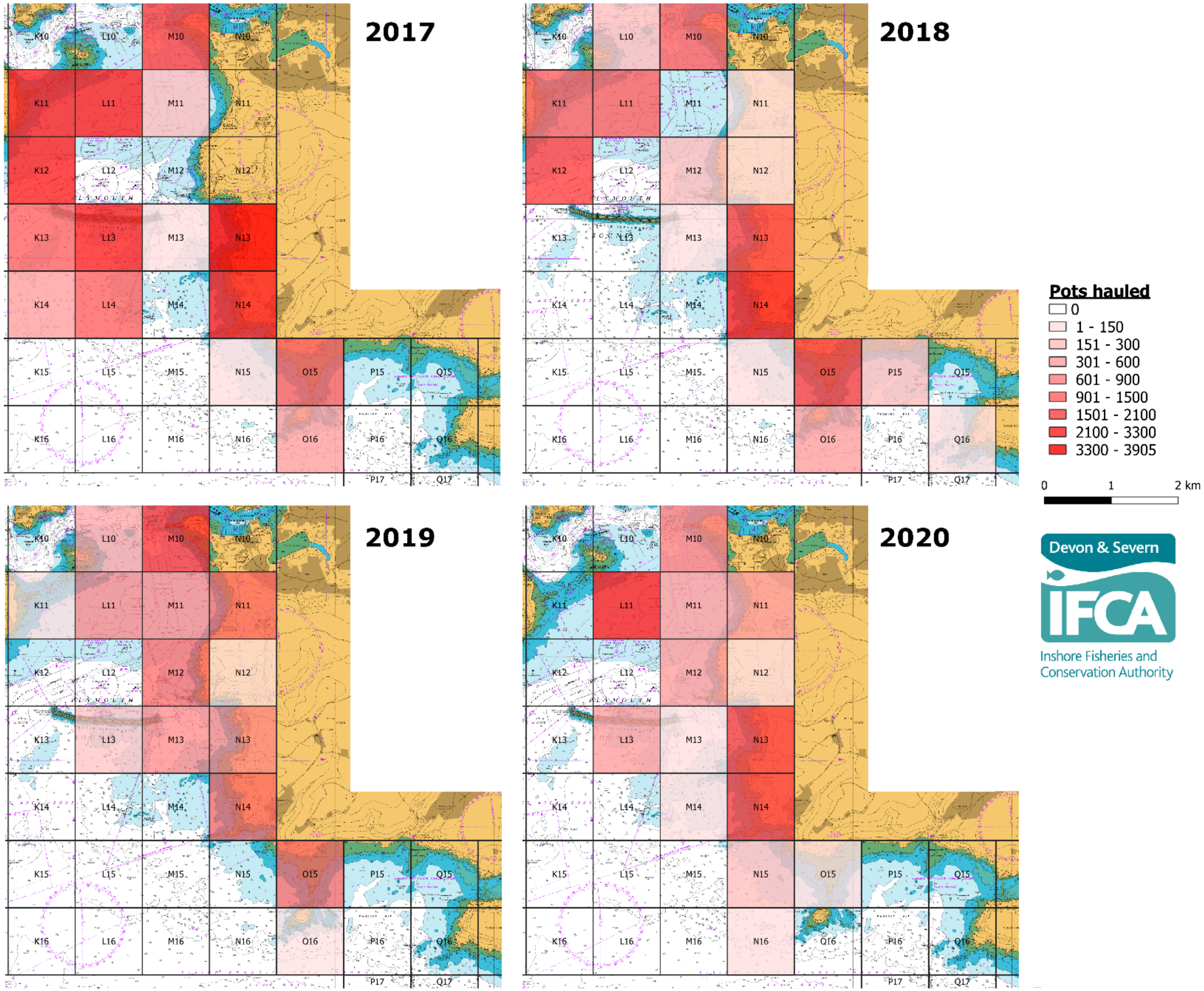

Figure 1. *Charts of Plymouth Sound showing fishing effort recorded on fishers' returns forms during 2017–2020, for all 1km^2 grid cells that fall partially or entirely within D&S IFCA's District. Data from Vessel 3 were deemed to be unreliable (Curtin et al., 2020), so have been excluded from this plot.*

3.2. Landings: Returns Forms and Sales Notes

Fishers' returns forms suggest that a total of 8458 wrasse were landed from the D&S IFCA waters of Plymouth Sound during April to December 2020 (Table 2). This represents a decline in total landings since monitoring began in 2017, which has occurred alongside an overall decrease in fishing effort (Table 2). No returns forms were available for Vessel 3, though sales notes data indicate that this fisher landed just 119 documented wrasse during 2020. These 8458 wrasse represent 50.4% of the total wrasse landed from Plymouth Sound: sales notes, supplied to the MMO by the salmon farm agent, indicate that 16,776 fish were landed (Table 2), including from CIFCA's waters in Plymouth Sound and the small amount of landings from Vessel 3. Of the fish reported in the sales notes for 2020 and transport documents, 1.8% were dead or damaged on arrival at the salmon farm. Additional mortalities may have occurred in holding pens before loading to transport, but these data are unavailable.

3.3. Survey Effort

In 2020, onboard observer surveys were risk assessed in light of the COVID-19 pandemic and it was decided that this method was unsuitable given the associated risks. Therefore, the observer surveys were carried out using a D&S IFCA RIB. With this approach, D&S IFCA observers achieved approximately 6.3% coverage of known fishing trips during 2020 (seven surveys out of 111 trips in total, 108 of which were reported by fishers). This survey effort is comparable to previous years, as a proportion of total fishing effort. In 2017, 5.5% of known fishing trips in the D&S IFCA's District had an observer onboard. This rose to 12% in 2018 but fell to 9% in 2019. The start and end locations of all strings of wrasse pots surveyed between 2017–2020 are shown in Figure 2. Each year, survey effort is planned to achieve even coverage across the active vessels. However, in practice the observer coverage varies between vessels due to periods of fishing inactivity related to vessel maintenance, cancellation of surveys due to inclement weather and difficulties in coordinating officer availability with sporadic fishing activity on an *ad hoc* basis. In 2020, the seven surveys conducted focused on two of the three active vessels: six surveys with Vessel 2 and one with Vessel 4.

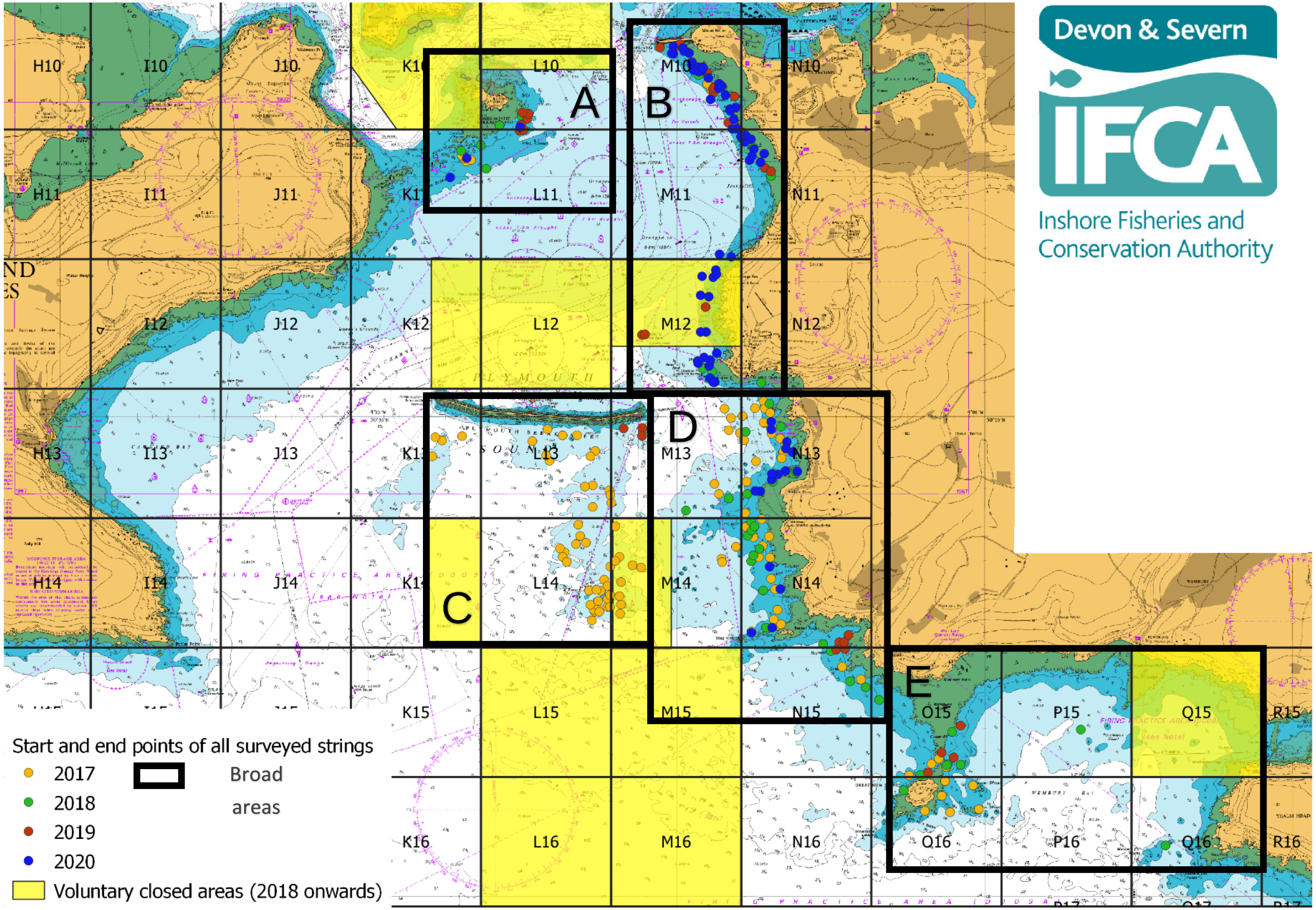

__Figure 2.__ Start and end locations of all strings of wrasse pots surveyed by D&S IFCA's officers in D&S IFCA's waters during 2017–2020, showing the voluntary closed areas implemented in 2018 and broad-scale fishing areas that were used in this analysis. The closed area shown in cell M14 was not in place during 2017, so the fishing effort in this area in 2017 was not in contravention of the voluntary closures in place at the time.

3.4. Species-specific results

3.4.1. Ballan

Ballan CPUE and LPUE have shown a decline across the 2017–2020 period (Table 3, Figure 3a,c). For CPUE this decline is only significant between 2017 and 2018 (Tukey z = -2.88, p_{adj}=0.019; Figure 3a) and for LPUE, the decline was dependent on position relative to the breakwater: LPUE declines were only observed landward of the breakwater (Table 3b, Figure 3c). Ballan LPUE was significantly lower in 2018, 2019 and 2020 than in 2017 landward of the breakwater. Breakwater position was also an important predictor of ballan CPUE (Table 3, Figure 3b).

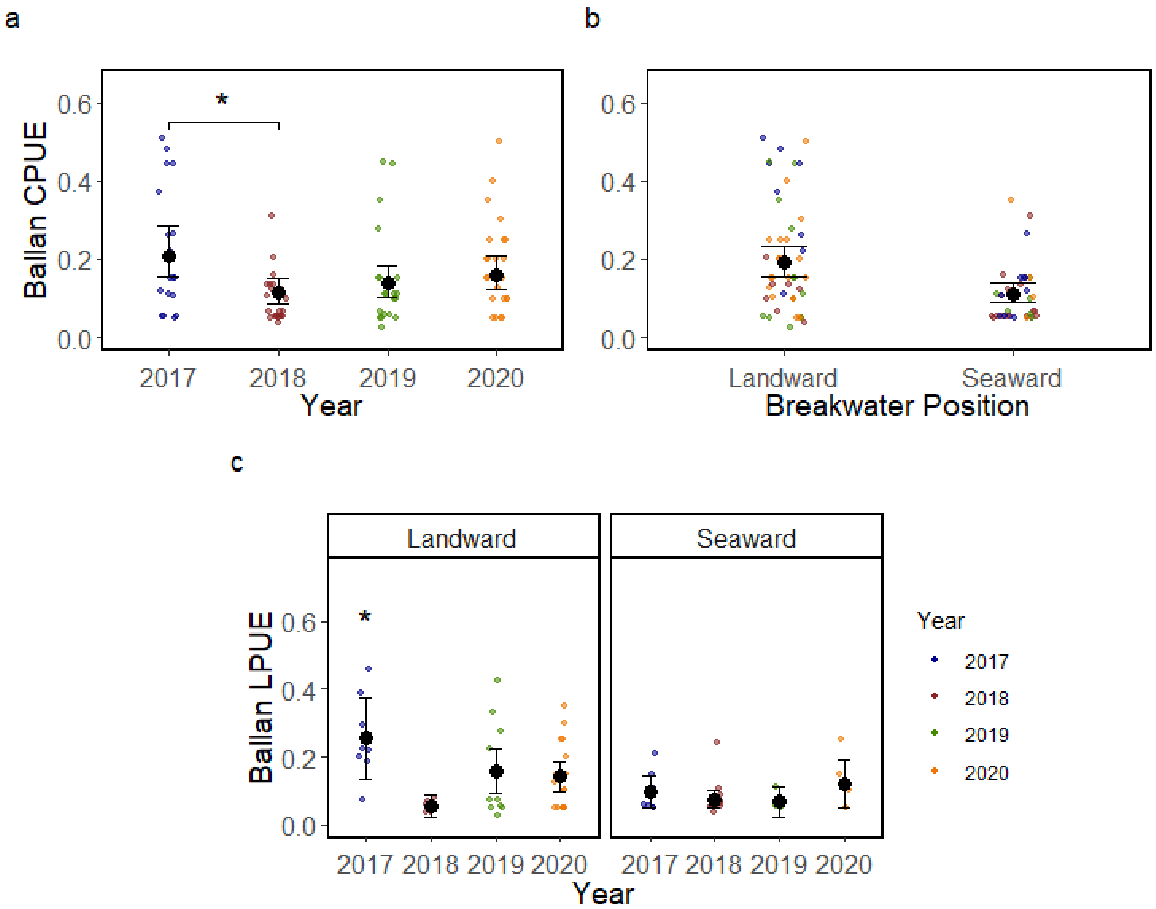

Figure 3. *Predicted effects of year and breakwater position on catch per unit effort (CPUE – total number of fish per pot; a-b) and landings per unit effort (LPUE – number of landed fish per pot; c) of ballan wrasse caught during on-board observer surveys in the Devon and Severn District between 2017–2020, as estimated by generalised linear models. Error bars represent 95% confidence intervals around the predicted means. LPUE predictions are split by breakwater position to highlight the interaction effect between these two variables. Coloured points represent raw CPUE and LPUE data per string *, **, *** denote significant differences (p<0.05, p<0.01, p<0.001) in CPUE/LPUE between all factor levels, except where specific pairwise comparisons are shown with a linking line (Tukey tests with p-values adjusted for multiple comparisons).*

Table 3. *Summary of post-hoc (Tukey) tests for differences in CPUE (a) and LPUE (b) between years for ballan wrasse. LPUE estimates are split by breakwater position to highlight the interaction effect between these two variables. *, **, *** denote significant differences (p<0.05, p<0.01, p<0.001) in ballan CPUE/LPUE between years and/or breakwater position.*

a)

Comparison	Estimate	Std. Error	z value	p_{adj}	
2017–2018	-0.6085	0.2107	-2.888	0.0197	*
2017–2019	-0.4199	0.2177	-1.928	0.2159	
2017–2020	-0.2708	0.212	-1.278	0.577	
2018–2019	0.1886	0.2088	0.903	0.8028	
2018–2020	0.3377	0.2062	1.637	0.3572	
2019–2020	0.1491	0.2007	0.743	0.8797	

b)

Comparison	Estimate	Std. Error	t value	p_{adj}	
Landward: 2018–2017	-0.15839	0.04583	-3.456	0.01031	*
Landward: 2019–2017	-0.20054	0.05225	-3.838	0.00316	**
Landward: 2020–2017	-0.18002	0.04119	-4.371	<0.001	***
Landward: 2019–2018	-0.04215	0.05225	-0.807	0.97575	
Landward: 2020–2018	-0.02163	0.04119	-0.525	0.99751	
Landward: 2020–2019	0.02052	0.04823	0.425	0.99924	
Seaward: 2018–2017	-0.09111	0.05423	-1.68	0.55407	
Seaward: 2019–2017	-0.01611	0.03615	-0.446	0.99901	
Seaward: 2020–2017	-0.03917	0.0502	-0.78	0.9795	
Seaward: 2019–2018	0.075	0.05067	1.48	0.6904	
Seaward: 2020–2018	0.05194	0.06149	0.845	0.96953	
Seaward: 2020–2019	-0.02306	0.04634	-0.498	0.99816	

The size distribution of ballan wrasse in 2017 was significantly different from that in 2020 (Figure 4a–e, Table 4a). The average size of ballan wrasse increased across years (Table 4b, Figure 4e, Figure 5). When the size data were split by breakwater position, it appears as though the increase in average size of ballan wrasse is driven by those on the landward size of the breakwater (Figure 4f, g). This is supported by the inclusion of models containing an interaction term between year and breakwater position in the set of candidate models for ballan wrasse size (Appendix 1). The percentage of all ballan wrasse caught during the July – October observer surveys that were within the CRS limits increased from 64.2% in 2017 to 71.1% in 2020, whereas the percentage below the minimum CRS limit decreased from 28.5% in 2017 to 12.4% in 2020 Table 4c). There was more variation in the percentage of caught ballan wrasse that were above the maximum CRS limit.

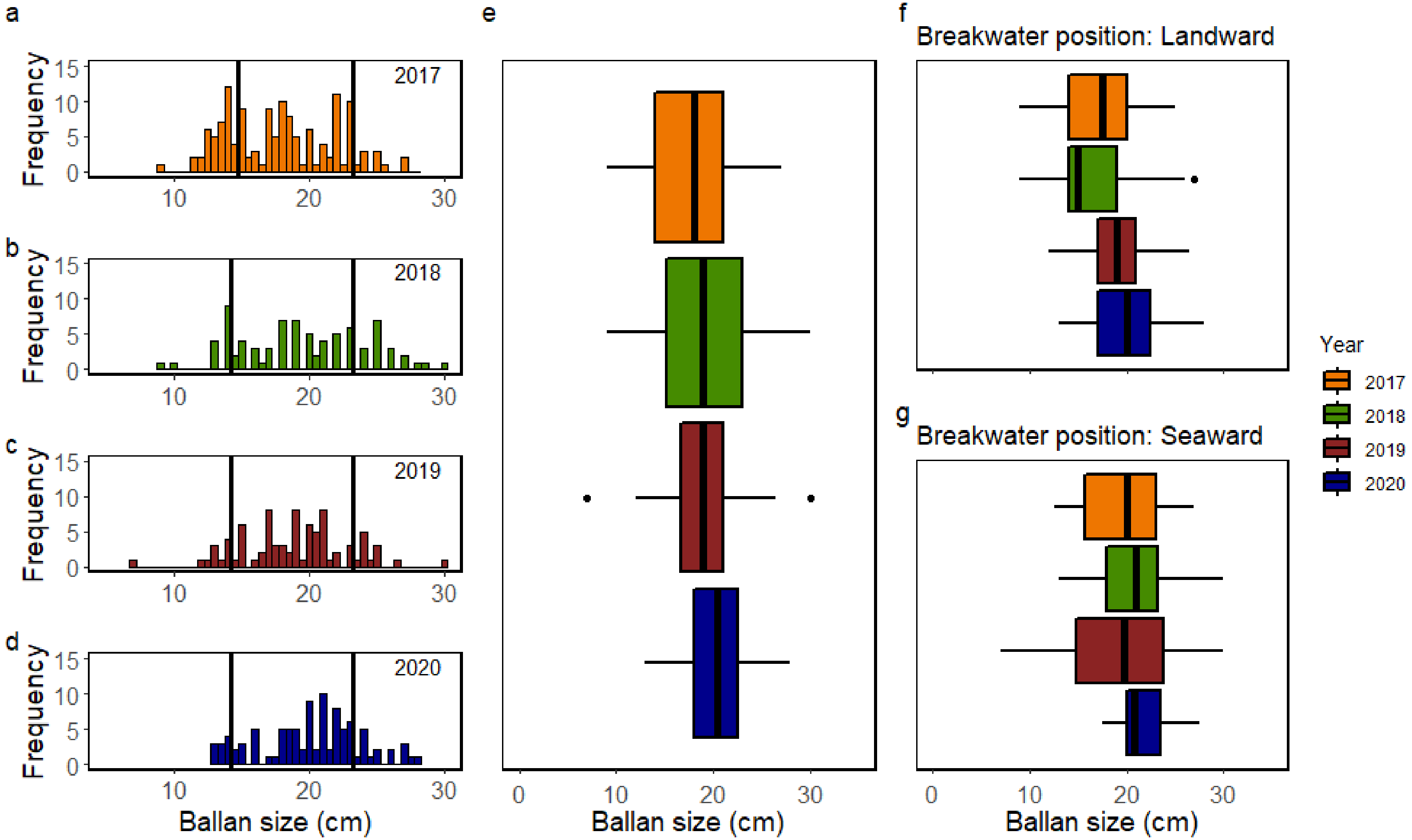

__Figure 4__. Size frequency histograms for ballan wrasse caught (regardless of whether they were retained or returned) during on-board observer surveys between July–October in (a) 2017, (b) 2018, (c) 2019, and (d) 2020. Bold, vertical black lines indicate the minimum and maximum conservation reference sizes (CRS) for ballan wrasse after implementation of the potting permit byelaw condition in July 2017. The maximum CRS for both periods was unchanged. Boxplots for total length (cm) of all ballan caught during this period(e). (f) and (g) shows these data split by position relative to the breakwater.

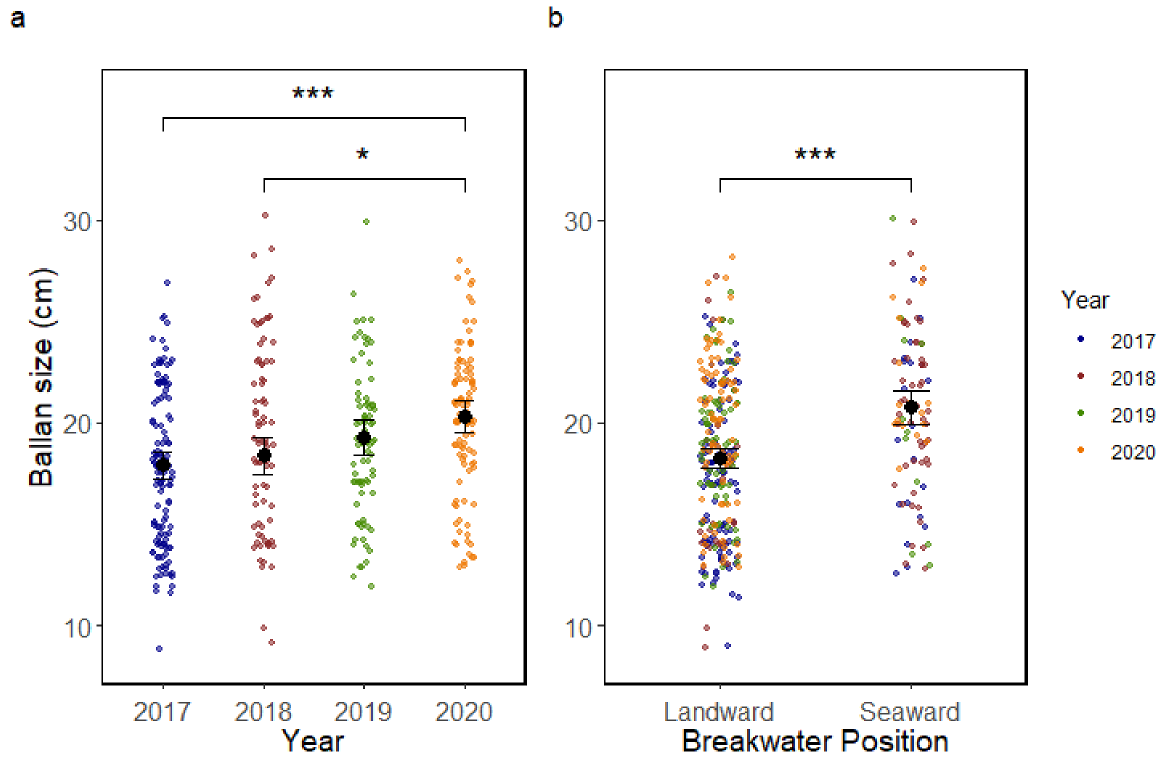

Figure 5. *Predicted effects of year (a) and breakwater position (b) on the size (total length - cm) of ballan wrasse caught during on-board observer surveys in the Devon and Severn District between 2017–2020, as estimated by generalised linear models. Error bars represent 95% confidence intervals around the predicted means. Coloured points represent raw size data per string *, **, *** denote significant differences (p<0.05, p<0.01, p<0.001) in size between all factor levels, except where specific pairwise comparisons are shown with a linking line (Tukey tests with p-values adjusted for multiple comparisons).*

Table 4. *(a) Results from bootstrapped Kolmogorov-Smirnov (K-S) tests formally comparing the distribution of ballan size (cm) between years. (b) Summary of post-hoc (Tukey) tests for differences in mean size between years and breakwater position for all ballan caught. (c) Proportions of ballan wrasse caught during the July – October observer surveys that were below, within and above the minimum (15 cm) and maximum (23 cm) conservation reference sizes (CRS). *, **, *** denote significant differences (p<0.05, p<0.01, p<0.001) in ballan size distribution/ mean size between years and/or breakwater position.*

a)

Comparison	D	p_{adj}	
2017-2018	0.201	0.067	
2017-2019	0.182	0.1192	
2017-2020	0.298	0.0012	**
2018-2019	0.149	0.3204	
2018-2020	0.148	0.3204	
2019-2020	0.171	0.2244	

b)

Comparison	Z	p_{adj}	
2017–2018	0.804	0.852	
2017–2019	2.528	0.055	
2017–2020	4.694	<0.001	***
2018–2019	1.359	0.523	
2018–2020	3.076	0.011	*
2019–2020	1.791	0.276	
Landward–Seaward	5.077	<0.001	***

c)

Year	% Below	% Within	% Above
2017	28.47	64.23	7.3
2018	20.73	57.32	21.95
2019	14.46	71.08	13.74
2020	12.37	71.14	16.49

3.4.2. Goldsinny

Neither goldsinny CPUE or LPUE showed significant change between years across the 2017–2020 period, but for both CPUE and LPUE there was evidence of a seasonal decline over the course over the main fishing season: July – October (Figure 6a, d). Goldsinny CPUE increased with distance to structure (Figure 6b) and varied between fishing areas (Figure 6c): CPUE in broad area B was significantly higher than in broad areas A and D (Table 5a). Goldsinny LPUE varied across depth bands (Figure 6e): LPUE was significantly higher in depth bands 2 and 3 (0–5 m and 5–10 m, respectively) than in depth band 1 (intertidal on spring tides) (Table 5b).

The size distribution of goldsinny wrasse varies across the 2017–2020 period (Figure 7a–e, Table 6a). The proportion of all goldsinny wrasse caught during the July – October observer surveys that were within the CRS limits varied between 23.7% and 37% across the four-year period, whereas the proportion below the minimum CRS limit varied between 62.8% and 76.3% (Table 6c). No goldsinny were caught that were above the maximum CRS limit (Table 6c, Figure 7a–d).

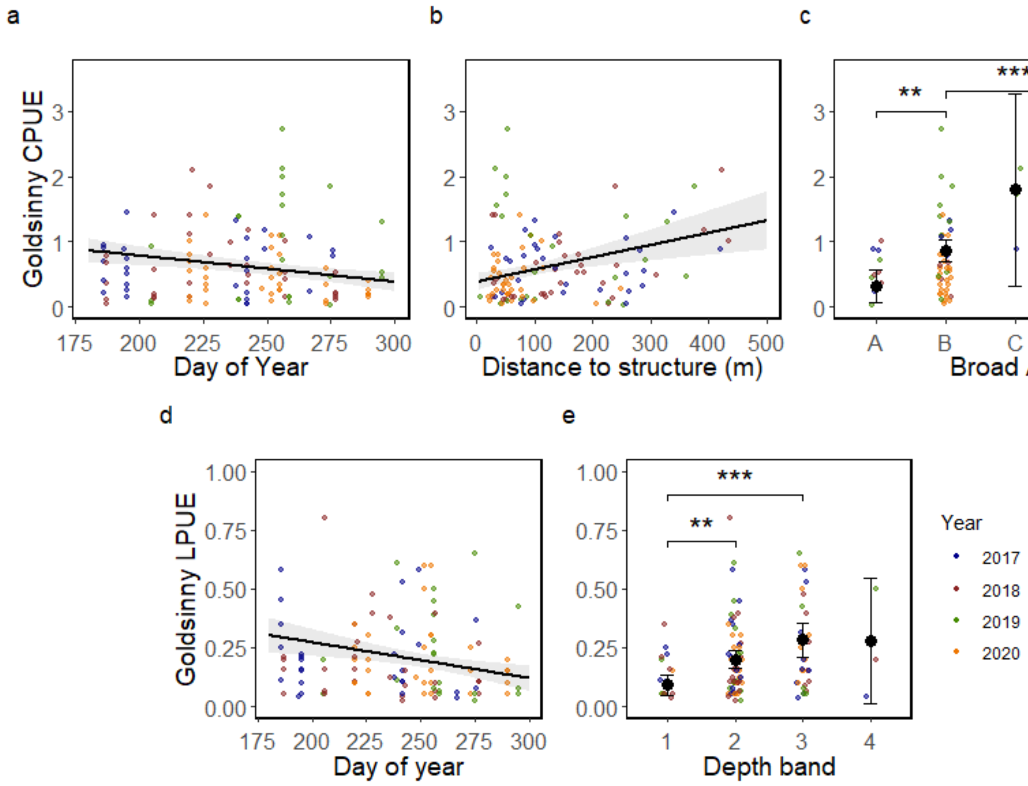

Figure 6. *Predicted effects of day of year, distance to structure and broad area (a–c) on catch per unit effort (CPUE – total number of fish per pot) and, day of year and depth band (d–e) on landings per unit effort (LPUE – number of landed fish per pot; c) of goldsinny wrasse caught during on-board observer surveys in the Devon and Severn District between 2017–2020, as estimated by generalised linear models. Shaded areas and error bars represent 95% confidence intervals around the predicted effects/ means. Coloured points represent raw CPUE and LPUE data per string *, **, *** denote significant differences (p<0.05, p<0.01, p<0.001) in CPUE/LPUE between all factor levels, except where specific pairwise comparisons are shown with a linking line (Tukey tests with p-values adjusted for multiple comparisons).*

Table 5. *Summary of post-hoc tests for differences in CPUE between broad areas (a) and LPUE between depth bands (b) for goldsinny wrasse. *, **, *** denote significant differences (p<0.05, p<0.01, p<0.001) in goldsinny CPUE/LPUE between years/depth bands.*

a)	Comparison	Estimate	Std.Error	z	p_{adj}	
	A–B	0.535	0.164	3.254	0.007	**
	A–C	1.477	0.758	1.948	0.247	
	A–D	0.086	0.151	0.567	0.975	
	A–E	0.216	0.172	1.259	0.672	
	B–C	0.942	0.75	1.262	0.67	
	B–D	-0.449	0.087	-5.135	<0.001	***
	B–E	-0.318	0.124	-2.576	0.058	.
	C–D	-1.392	0.744	-1.872	0.285	
	C–E	-1.261	0.748	-1.685	0.39	
	D–E	0.131	0.1	1.311	0.638	

b)	Comparison	Estimate	Std.Error	z	p_{adj}	
	DBand1–DBand2	0.11	0.03	3.618	0.001	**
	DBand1–DBand3	0.191	0.043	4.453	<0.001	***
	DBand1–DBand4	0.189	0.136	1.385	0.473	
	DBand2–DBand3	0.082	0.04	2.065	0.142	
	DBand2–DBand4	0.079	0.137	0.578	0.931	
	DBand3–DBand4	-0.003	0.14	-0.019	0.999	

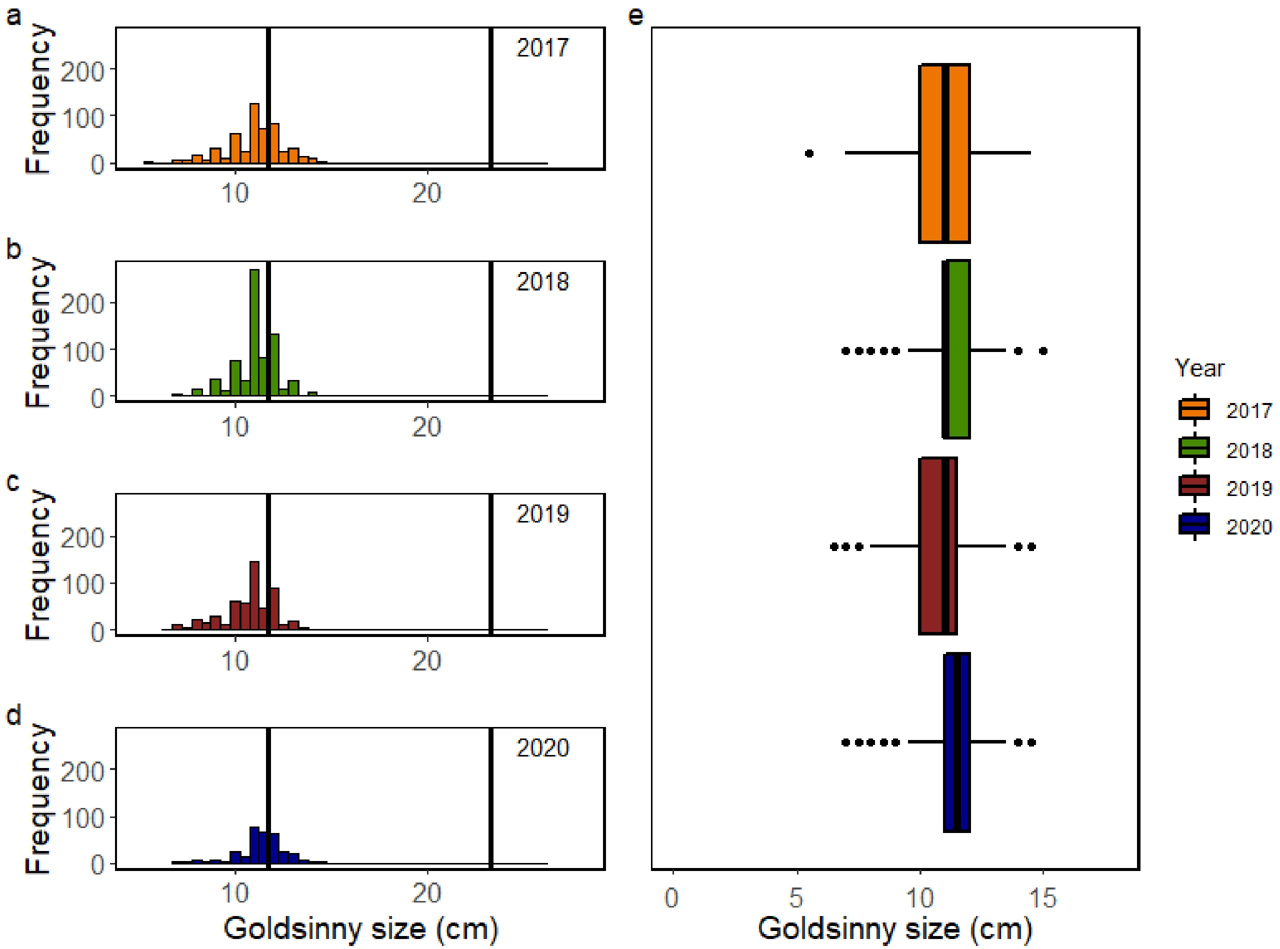

Figure 7. *Size frequency histograms for goldsinny wrasse caught (regardless of whether they were retained or returned) during on-board observer surveys between July–October in (a) 2017, (b) 2018, (c) 2019, and (d) 2020. Bold, vertical black lines indicate the minimum and maximum conservation reference sizes (CRS) for goldsinny wrasse. (e) Boxplots for total length (cm) of all goldsinny caught during this period.*

Table 6. *(a) Results from bootstrapped Kolmogorov-Smirnov (K-S) tests formally comparing the distribution of goldsinny size (cm) between years. *, **, *** denote significant differences (p<0.05, p<0.01, p<0.001) in goldsinny size distribution between years. (b) Proportions of goldsinny wrasse caught during the July – October observer surveys that were below, within and above the minimum (12 cm) and maximum (23 cm) conservation reference sizes (CRS).*

a)

Comparison	D	p_{adj}	
2017-2018	0.069	0.024	*
2017-2019	0.122	<0.001	***
2017-2020	0.128	<0.001	***
2018-2019	0.146	<0.001	***
2018-2020	0.197	<0.001	***
2019-2020	0.251	<0.001	***

b)

Year	% Below	% Within	% Above
2017	69.25	30.75	0
2018	73.7	26.3	0
2019	76.25	23.75	0
2020	62.81	37.19	0

3.4.3. Rock Cook

Rock cook CPUE did not show a significant change between years across the 2017–2020 period and LPUE did not change significantly across the 2017–2019 period (before the implementation of the prohibition on removal from the fishery), but both CPUE and LPUE varied across broad-scale fishing areas (Figure 8). The fishing areas yielding the highest and lowest rock cook CPUE were relatively consistent with those yielding the highest and lowest LPUE. Rock cook CPUE was significantly lower in area A than areas C, D and E (Table 7a, Figure 8a), and LPUE was significantly lower in areas A and B than in area E (Table 7b, Figure 8b).

The size distribution of rock cook varies across the 2017–2020 period (Figure 9a–e, Table 8a). The proportion of all rock cook caught during the July – October observer surveys that were within the CRS limits varied between 20.1% and 25.72% between 2017–2019 and increased to 45.56 in 2020 (Table 8c). No rock cook were caught that were above the maximum CRS limit (Table 8c, Figure 9a–d).

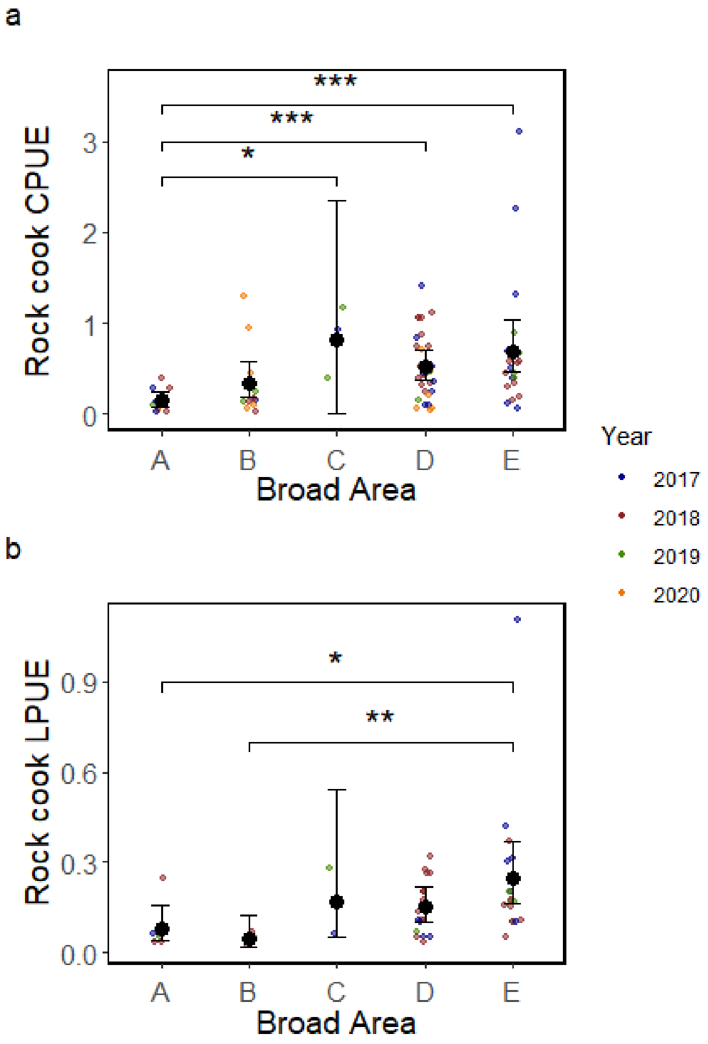

Figure 8. *Predicted effects of broad area on catch per unit effort (CPUE – total number of fish per pot; a) and landings per unit effort (LPUE – number of landed fish per pot; b) of rock cook caught during on-board observer surveys in the Devon and Severn District, as estimated by generalised linear models. CPUE is estimated over the 2017–2020 period, and LPUE Is assessed over the 2017–2019 period (before the implementation of the prohibition on removal of rock cook from the fishery). Error*

*bars represent 95% confidence intervals around the predicted means. Coloured points represent raw CPUE and LPUE data per string *, **, *** denote significant differences (p<0.05, p<0.01, p<0.001) in CPUE/LPUE between all factor levels, except where specific pairwise comparisons are shown with a linking line (Tukey tests with p-values adjusted for multiple comparisons).*

Table 7. *Summary of post-hoc tests for differences in CPUE (a) and LPUE (b) between broad areas for rock cook wrasse. *, **, *** denote significant differences (p<0.05, p<0.01, p<0.001) in rock cook CPUE/LPUE between broad areas. CPUE is estimated over the 2017–2020 period, and LPUE Is assessed over the 2017–2019 period (before the implementation of the prohibition on removal of rock cook from the fishery).*

a)	**Estimate**	**Std.Error**	**z**	p_{adj}	
A–B	0.836	0.387	2.16	0.182	
A–C	1.734	0.592	2.932	0.025	*
A–D	1.265	0.319	3.968	<0.001	***
A–E	1.56	0.338	4.617	<0.001	***
B–C	0.898	0.592	1.518	0.532	
B–D	0.428	0.319	1.344	0.647	
B–E	0.724	0.338	2.142	0.189	
C–D	-0.469	0.549	-0.855	0.907	
C–E	-0.174	0.561	-0.31	0.998	
D–E	0.296	0.257	1.152	0.766	

b)	**Estimate**	**Std.Error**	**z**	p_{adj}	
A–B	-0.533	0.578	-0.924	0.878	
A–C	0.737	0.667	1.104	0.789	
A–D	0.618	0.383	1.617	0.463	
A–E	1.113	0.388	2.87	0.03	*
B–C	1.27	0.746	1.703	0.409	
B–D	1.152	0.507	2.27	0.141	
B–E	1.647	0.512	3.219	0.009	**
C–D	-0.118	0.607	-0.194	0.999	
C–E	0.377	0.611	0.617	0.97	
D–E	0.495	0.273	1.814	0.343	

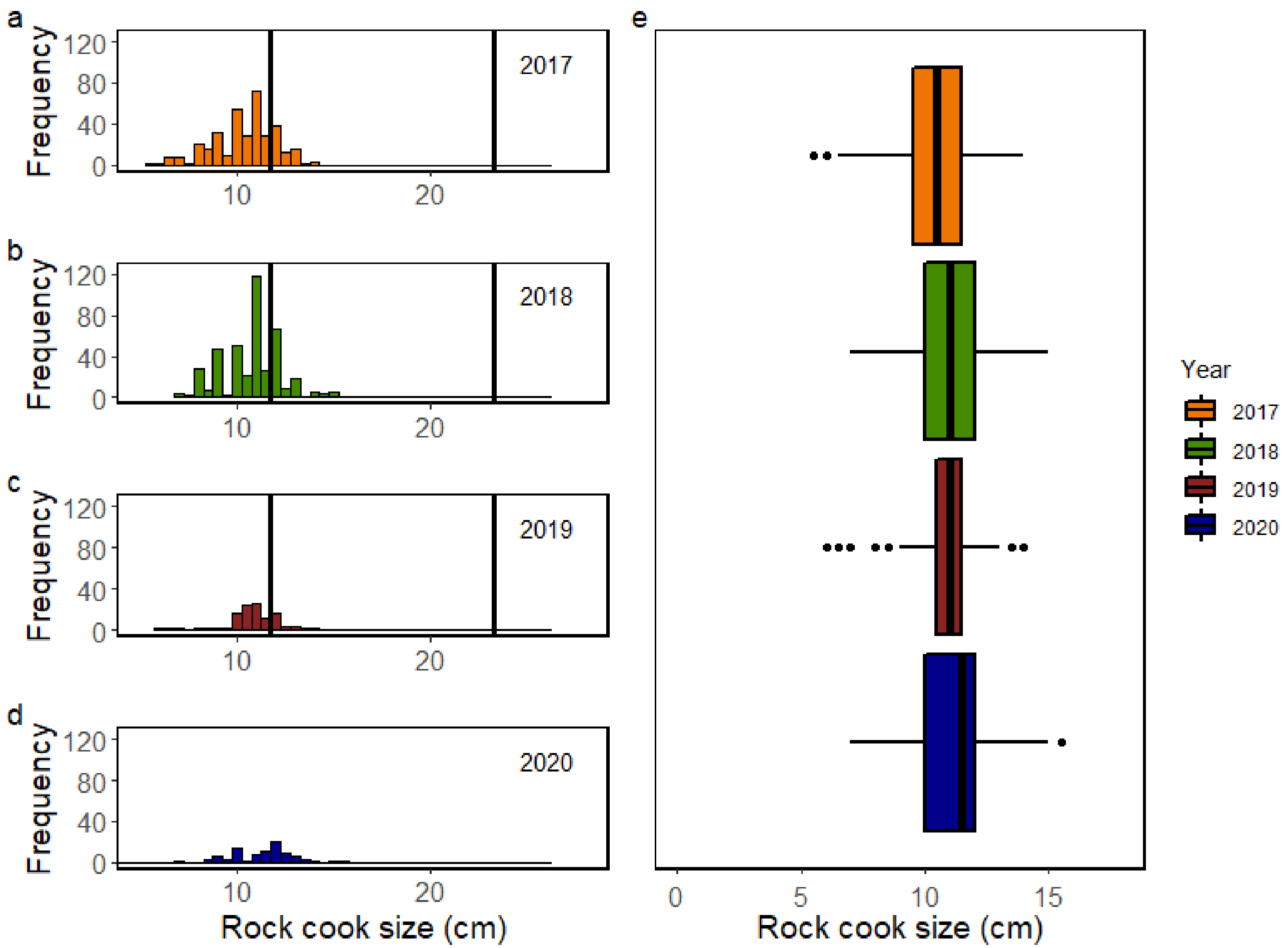

Figure 9. *Size frequency histograms for rock cook caught (regardless of whether they were retained or returned) during on-board observer surveys between July–October in (a) 2017, (b) 2018, (c) 2019 and (d) 2020. Bold, vertical black lines indicate the minimum and maximum conservation reference sizes (CRS) for rock cook. No rock cook were landed in 2020 following implementation of the revised Potting Permit Byelaw permit conditions. (e) Boxplots for total length (cm) of all rock cook caught during this period*

Table 8. *(a) Results from bootstrapped Kolmogorov-Smirnov (K-S) tests formally comparing the distribution of rock cook size (cm) between years. *, **, *** denote significant differences (p<0.05, p<0.01, p<0.001) in rock cook size distribution between years. (b) Proportions of rock cook caught during the July – October observer surveys that were below, within and above the minimum (12 cm) and maximum (23 cm) conservation reference sizes (CRS).*

a)

Comparison	D	p_{adj}	
2017-2018	0.117	0.003	**
2017-2019	0.200	<0.001	***
2017-2020	0.296	<0.001	***
2018-2019	0.153	0.007	**
2018-2020	0.256	<0.001	***
2019-2020	0.253	0.002	**

b)

Year	% Below	% Within	% Above
2017	79.89	20.11	0
2018	74.28	25.72	0
2019	78.07	21.93	0
2020 †	54.44	45.56	0

† No rock cook were removed from the fishery in 2020, but the proportions of catch below, within and above the previous CRS range have been presented for context.

3.4.4. Corkwing

Corkwing CPUE has increased over the 2017–2020 period: CPUE in 2019 and 2020 was significantly higher than in 2017 and 2018 (Figure 10b, Table 9). Corkwing LPUE however, did not show significant change across the 2017–2020 period. There was evidence of a seasonal increase in both CPUE and LPUE over the course over the main fishing season: July – October (Figure 10a, c).

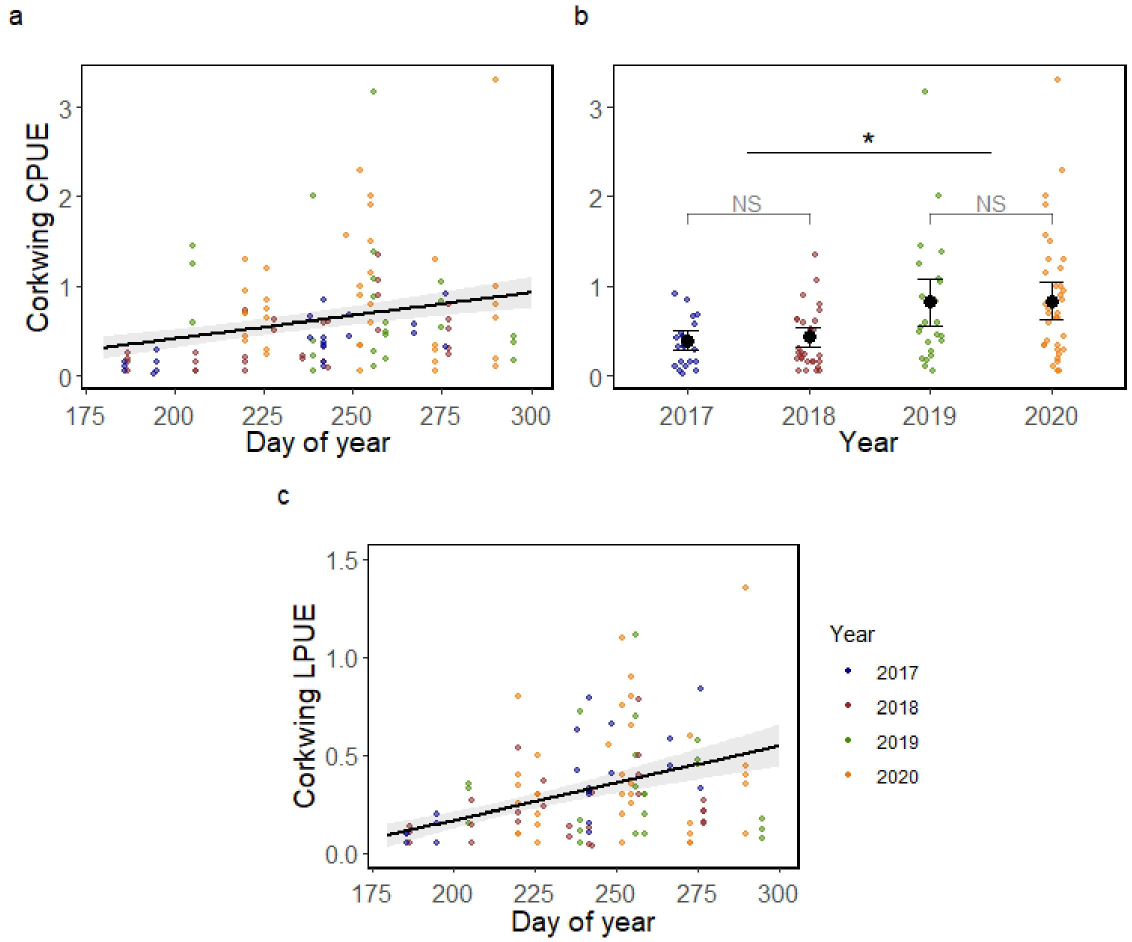

Figure 10*. Predicted effects of day of year and year (a–b) on catch per unit effort (CPUE – total number of fish per pot) and, day of year (c) on landings per unit effort (LPUE – number of landed fish per pot) of corkwing wrasse caught during on-board observer surveys in the Devon and Severn District between 2017–2020, as estimated by generalised linear models. Shaded areas and error bars represent 95% confidence intervals around the predicted effects/ means. Coloured points represent raw CPUE and LPUE data per string *, **, ***, NS denote significant differences (p<0.05, p<0.01, p<0.001, no significant difference) in CPUE/LPUE between all factor levels, except where specific pairwise comparisons are shown with a linking line (Tukey tests with p-values adjusted for multiple comparisons).*

*__Table 9:__ Summary of post-hoc tests for differences in CPUE between years for corkwing wrasse. *, **, *** denote significant differences (p<0.05, p<0.01, p<0.001) in corkwing CPUE between years.*

	Estimate	Std. Error	z value	p_{adj}	
2017–2018	0.041	0.05	0.815	0.834	
2017–2019	0.429	0.142	3.024	0.011	*
2017–2020	0.442	0.119	3.717	<0.001	***
2018–2019	0.389	0.141	2.754	0.025	*
2018–2020	0.401	0.118	3.403	0.003	**
2019–2020	0.013	0.167	0.077	0.999	

The size distribution of corkwing varied across the 2017–2020 period, but the size distribution in 2017 was similar to that in 2020, and the size distribution in 2018 was similar to that in 2019 (Figure 11a–e, Table 10a). The proportion of all corkwing caught during the July – October observer surveys that were within the CRS limits before the changes to the CRS limits on 13th August 2018 was 94.3% in 2017 and 85.0% before the change in 2018. This proportion was reduced to between 45.1% and 49.9% in the years following (Table 10b). In 2017 and before the change in CRS limits in 2018, no corkwing wrasse were caught above the maximum CRS (23 cm), however, once the maximum CRS was decreased to 18 cm, between 96.9%–15.7% of the catch was above the maximum CRS (Table 10b). The proportion of corkwing wrasse caught below the minimum CRS also increased to between 34.4%–47.44% after the minimum CRS changed from 12 cm to 14 cm.

*__Table 10.__ (a) Results from bootstrapped Kolmogorov-Smirnov (K-S) tests formally comparing the distribution of corkwing size (cm) between years. *, **, *** denote significant differences (p<0.05, p<0.01, p<0.001) in corkwing size distribution between years. (b) Proportions of corkwing caught during the July – October observer surveys that were below, within and above the minimum and maximum conservation reference sizes (CRS). Proportions of corkwing caught were split by day of year (d<225/ d≥225) in 2018 to reflect the differing CRS ranges in the proportions of wrasse caught. Under the old potting permit byelaw conditions for 2017 and the period before 13 August 2018 (d<225) minimum and maximum CRS were 12 cm and 23 cm, respectively. Following this period (d>225, 2019 and 2020) corkwing minimum and maximum CRS were changed to 14 cm and 18cm, respectively.*

a)

Comparison	D	p_{adj}	
2017-2018	0.201	<0.001	***
2017-2019	0.182	<0.001	***
2017-2020	0.298	0.434	
2018-2019	0.149	0.434	
2018-2020	0.148	<0.001	***
2019-2020	0.171	<0.001	***

b)

Year	% Below	% Within	% Above
2017	5.74	94.26	0.00
2018 $_{d<225}$	15.00	85.00	0.00
2018 $_{d>225}$	46.23	46.85	6.92
2019	47.44	45.12	7.44
2020	34.35	49.92	15.73

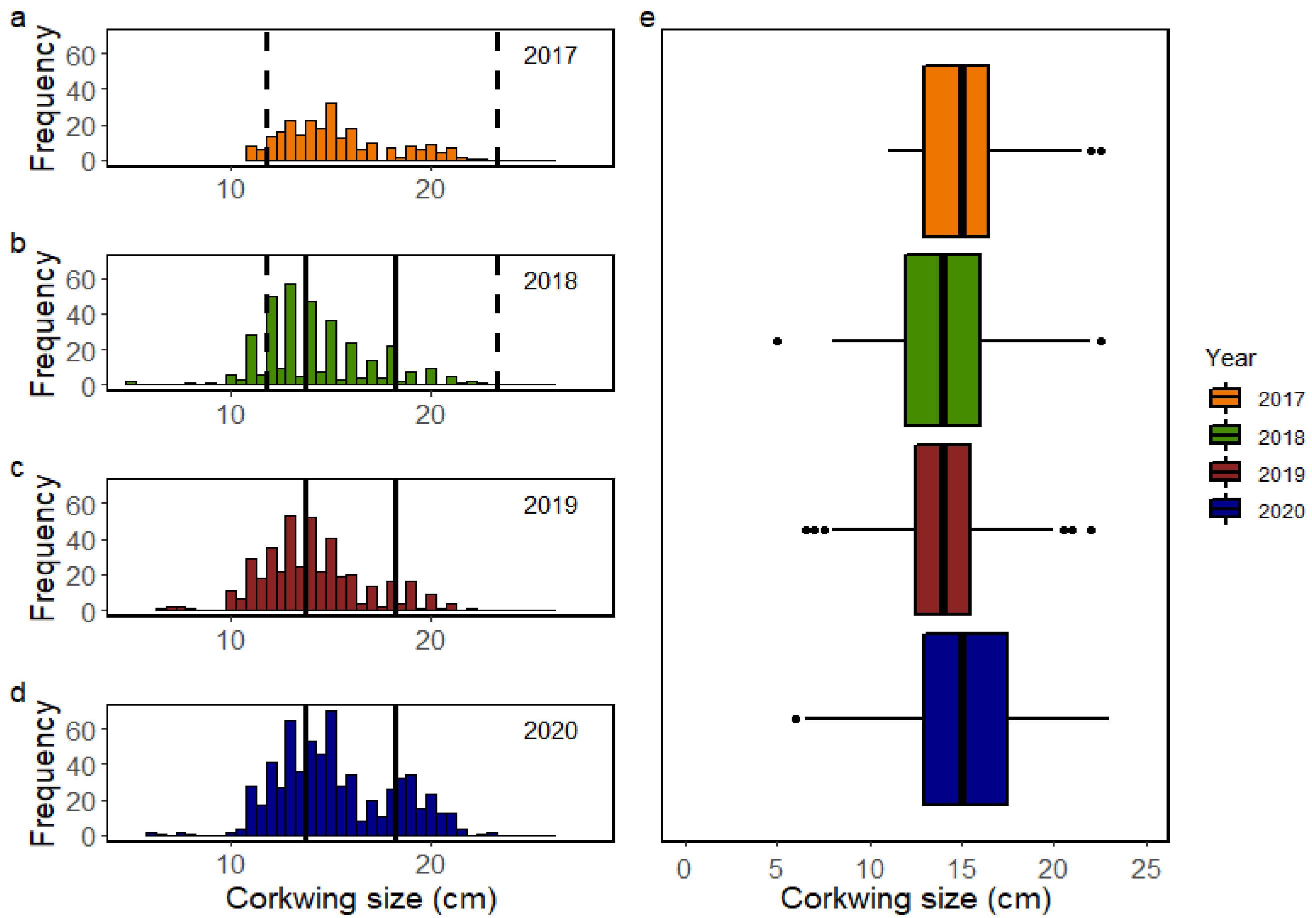

Figure 11. *Size frequency histograms for corkwing caught (regardless of whether they were retained or returned) during on-board observer surveys between July–October in (a) 2017, (b) 2018, (c) 2019, and (d) 2020. Bold, vertical black lines indicate the minimum and maximum conservation reference sizes (CRS) for corkwing after implementation of the new potting permit byelaw conditions on 13 August 2018. The dashed, black vertical lines indicate the minimum and maximum CRS for corkwing wrasse under the old potting permit byelaw conditions for the period before 13 August 2018. Boxplots for total length (cm) of all corkwing caught during this period (e) Boxplots for total length (cm) of all corkwing caught during this period*

3.4.5. Cuckoo

Cuckoo wrasse are not targeted by the fishery and therefore all individuals caught are returned to sea. Catches of this species are typically low in D&S IFCA's District (Curtin *et al.*, 2020), and none were caught during observer surveys in 2020. See Curtin *et al.* (2020) for a summary of observer survey data for the 2017–2019 period.

3.5. Catch Composition

The catch composition of wrasse caught and recorded for the 2017–2020 period during the on-board observer surveys is shown in Figure 12. Catch composition varied significantly between years (PERMANOVA; pseudo-F = 6.81, $p < 0.001$; see Table 11 for pairwise tests), but the variables that explained the most variation in catch composition were, in order of importance, broad-scale fishing area and day of year.

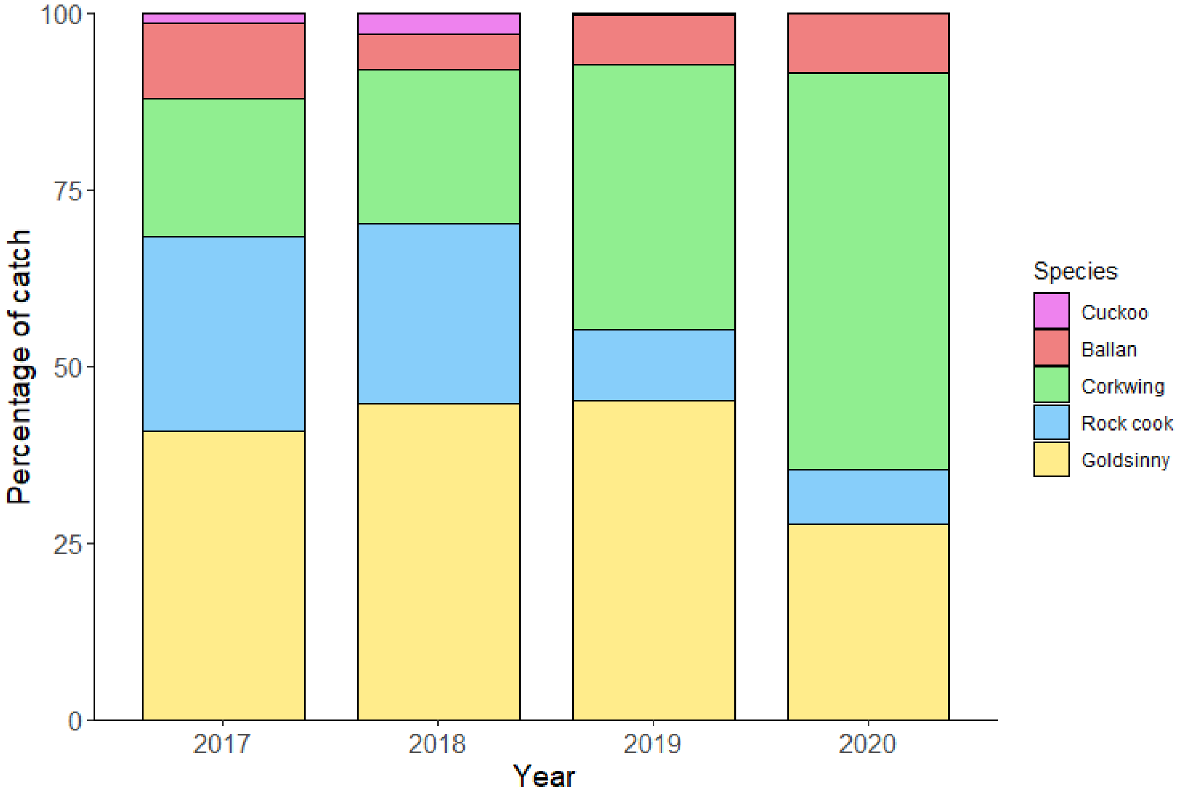

Figure 12. *Composition of wrasse catches for the 2017-2020 period. Taken from data obtained during the observer surveys between July – October in the D&S IFCA District only.*

Table 11 *Pairwise comparisons (PERMANOVA) testing for differences in catch composition between years (2017–2020). *, **, *** denote significant differences (p<0.05, p<0.01, p<0.001) in catch composition between years.*

Comparison	t	p_{perm}	
2017-2018	1.109	0.312	
2017-2019	2.228	0.001	**
2017-2020	3.264	<0.001	***
2018-2019	2.366	0.003	**
2018-2020	3.852	<0.001	***
2019-2020	1.6723	0.033	*

4. Discussion

This report assesses the sustainability of the Live Wrasse Fishery in the Devon & Severn Inshore Fisheries and Conservation Authority's (D&S IFCA) District. Using monitoring data from the fishery observer surveys conducted by D&S IFCA's Environment Officers during the fishing season, along with environmental data obtained from external sources, the main drivers of variation in catches and landings per unit effort (CPUE and LPUE) are assessed and implications of the results for future management are discussed. This report will also feed into a review of the relevant Habitats Regulations Assessments regarding interactions between fish traps and relevant features within Plymouth Sound & Estuaries SAC and Tamar Estuaries Complex SPA. These HRAs were most recently reviewed in 2020, following the Three Year Comprehensive Review of the Live Wrasse Fishery (Curtin *et al.,* 2020). Natural England recommended that the HRAs be reviewed after one year to allow D&S IFCA to incorporate findings from ongoing monitoring.

4.1. Ballan wrasse

Catch and landings per unit effort (CPUE and LPUE) of ballan wrasse declined during the 2017–2020 period, but for LPUE the decrease was only evident on the landward side of Plymouth breakwater. The decrease in CPUE of ballan wrasse was only statistically significant between 2017–2018, whereas LPUE of ballan wrasse have remained significantly lower than the 2017 level in 2018–2020. This is despite the fact that the majority of the fishing effort over the last two years (2019–2020) has been based on the landward side of the breakwater, where CPUE and LPUE are significantly higher. Declines in CPUE and LPUE are likely driven by the high retention rate of ballan wrasse (between 57% and 71% of the total catch), as well as the specific behavioural and life history traits, discussed below, that render this species potentially vulnerable to fishing pressure, even under the management measures enacted by D&S IFCA to date.

In the D&S IFCA's District, there is a closed fishing season for wrasse (1st May – 15th July, revised from the previous closed season of 1st April – 30th June on 12th April 2018). The closed fishing season was introduced to protect wrasse during their spawning season. Wrasse should be protected during their spawning season not only to allow spawning individuals to contribute to recruitment before they are removed from the fishery (Skiftesvik *et al.*, 2014), but also to endure there is sufficient survival of eggs once they have been laid. For benthic spawning wrasse species, such as ballan wrasse, eggs may take up to 16 days to hatch after they have been laid in nests, which males guard (Potts, 1985; Darwall *et al.*, 1992). Removal of nest-guarding individuals during this period may significantly reduce egg survival (Darwall *et al.*, 1992). Evidence from Ireland and Norway suggests that the ballan wrasse spawning season occurs between April–August (Darwall *et al.*, 1992; Artüz, 2005; Muncaster *et al.*, 2010), but in Galicia, Spain, ballan wrasse have been observed spawning from January–April (Villegas-Ríos *et al.*, 2013). The timing of spawning clearly varies over space and, without a robust assessment of ballan wrasse spawning seasons on the south coast of the UK, it is difficult to ensure that spawning individuals are being sufficiently protected. There is currently limited evidence of the timing of the spawning season for ballan wrasse on the south coast of the UK, particularly in the south west region, however the best available evidence from the literature has been used as a basis for the current management measures.

Ballan wrasse are protogynous sequential hermaphrodites: individuals develop from females into males after a number of years (Dipper and Pullin, 1979; Muncaster *et al.*, 2010). In any fishery it is necessary to protect a proportion of mature individuals of both sexes in order to ensure enough breeding individuals survive to breed and support the population. Protecting both mature females and males is more complicated in sequential hermaphroditic species such as ballan wrasse. This is because measures such as the length at sexual maturity and length at sexual inversion (female to male change) often vary across the species' distribution, and the length at sexual inversion can change within a location as a result of social cues, i.e. the absence of functional males (Dipper and Pullin, 1979). For ballan wrasse specifically, estimates for the length at which sexual inversion is *first seen* in a population range from 22–35 cm (Dipper et al. 1977, Muncaster et al. 2013, Villegas-Ríos et al. 2013, Leclercq et al. 2014). However, researchers have also recorded the length at which 50% of ballan wrasse have undergone sexual inversion. This length is termed the 'L_{50}', and is between 34–36 cm (Villegas-Ríos *et al.*, 2013; Leclercq *et al.*, 2014), but could be as large as 47 cm for the spotted colour morphs (Villegas-Ríos *et al.*, 2013). This reproductive system therefore leaves the population vulnerable to sex-selective fishing if appropriate size restrictions are not in place. The current conservation reference size (CRS) limits for ballan wrasse in D&S IFCA's District are 15–23 cm. Estimates in the literature suggest that the length at sexual maturity in female ballan wrasse is between 16–18 cm. Therefore, it is likely that the fishery is highly selective towards mature female ballan wrasse that are below the L_{50}. The disproportionate removal of one particular sex could result in a shift in sex ratio and have consequences for future recruitment and breeding (Muncaster *et al.*, 2010), particularly as eggs are more likely than milt to be the limiting factor in reproduction. Targeting a large proportion of the mature female size classes of protogynous fish below the L_{50} has the potential for a negative effect of fishing pressure, even at relatively low levels of removal (Alonzo and Mangel, 2004).

4.2. Goldsinny wrasse

CPUE and LPUE of goldsinny wrasse did not change significantly between years, suggesting a lack of a fishery effect on this species. Similarly, though there were significant changes in goldsinny size distributions across years, the average size of goldsinny wrasse remained relatively stable, and the variation observed in goldsinny size likely reflects natural variation. Relatively few goldsinny caught in Plymouth Sound exceed 12 cm in length, which is the minimum CRS for goldsinny. Consequently, only the very largest size classes are landed. Male goldsinny reportedly mature at 9cm, while females mature at 8 cm (Matland, 2015). Therefore, the current minimum CRS is likely to be protecting individuals that have the potential to spawn and restock the population.

Many of the drivers of goldsinny CPUE and LPUE are consistent with previous research into this species. For example, goldsinny wrasse occur at lower densities at sites that are influenced by freshwater runoff (Sayer *et al.*, 1993); in the present analysis, goldsinny CPUE is significantly lower in broad area A: near Drake's Island and the area closest to the freshwater input from the River Tamar. Likewise, shallower areas are likely to be influenced more by freshwater runoff than deeper areas (Sayer *et al.*, 1993), and goldsinny LPUE was found to be significantly lower in shallower waters than in deeper waters. There appears to be significant seasonal variation in goldsinny CPUE and LPUE, which were highest at the start of the July – October season and declined as the season progressed. This supports observations by Sayer *et al*. (1993), who found highest densities of goldsinny in the summer months, with declines in the numbers of actively swimming goldsinny from October onwards. It is thought that seasonal changes in goldsinny activity (and hence catchability) may be influenced by both water temperature and photoperiod (Darwall *et al.*, 1992; Sayer *et al.*, 1993; Thangstad, 1999; Gjøsaeter, 2002).

4.3. Rock cook

Rock cook CPUE and LPUE did not change significantly between years; this supports the work of Henly *et al.* (in review). D&S IFCA's Three Year Comprehensive Review of the Live

Wrasse Fishery (Curtin *et al.*, 2020) reported a significant decline in rock cook CPUE and LPUE in the D&S IFCA's District over the 2017–2019 period. However, the Three-Year Comprehensive Review could not account for other potential drivers of variation in CPUE and LPUE (e.g. geographic location of fishing, environmental and seasonal variables) and also used data from differing time periods within each year. For example, data from April–June were included in the 2017 data set and data from November – December included in the 2019 dataset despite no other data being available for those months in other years. The current analysis has shown that CPUE and LPUE of some wrasse species vary with day of year, so interannual comparisons must compare equivalent time periods (e.g. July–October each year – months for which data are available for all years).

More significantly for rock cook however is the effect of fishing location. In the current analysis, both rock cook CPUE and LPUE showed significant variation across broad-scale fishing areas. Rock cook CPUE was lower in the more sheltered areas (broad areas A and B, which are protected by the breakwater from the prevailing SSW winds). This supports Skiftesvik *et al.* (2015) who showed that rock cook in Norway in were relatively less abundant in sheltered locations. As the majority of the observer surveys have been conducted in more sheltered locations in the last two years, it is unsurprising that the Three Year Comprehensive Review, which did not control for geographical variation in CPUE and LPUE, highlighted a negative year effect. Within the time- and resource-limited research capabilities of D&S IFCA (and the restrictions imposed by locations of routine fishing activity), future surveys should aim to distribute survey effort evenly and consistently over time and locations, accounting for the important species-specific drivers of CPUE and LPUE identified here, to achieve robust monitoring and recommendations for management.

There were significant changes in rock cook size distributions between years, suggestive of a gradual increase in size of rock cook caught in Plymouth Sound. This is despite the fact that only the largest rock cook caught were landed prior to the prohibition on retaining rock cook in D&S IFCA's District, suggesting that the size selectivity has not had detrimental effects. However, it should be noted that, due to changes in fishing location over time, catches of rock cook were relatively low in recent years, so comparisons of size distributions may not be robust. Relatively few rock cook caught in Plymouth Sound exceed 12 cm in length, which is the minimum CRS for this species. Consequently, only the very largest size classes are landed. Male rock cook reportedly mature at 9cm, while females mature at 8.5 cm (Matland, 2015). Therefore, the current minimum CRS is likely to be protecting individuals that have the potential to spawn and restock the population.

4.4. Corkwing wrasse

Corkwing CPUE increased across the 2017–2020 period, yet there was no significant change in corkwing LPUE over this time. The proportion of caught corkwing that were smaller than the minimum CRS increased significantly over 2017–2020, suggesting that the increase in CPUE across the 2017–2020 period was driven by an increase in the number of wrasse in the size classes that were being returned to the sea. Prior to the 2018 change in corkwing CRS range, only a small proportion of corkwing caught in Plymouth Sound were returned to the sea: just 5.7 – 15.0% were below the minimum CRS (12 cm), while none were above the maximum CRS (23 cm). Unlike ballan wrasse, corkwing wrasse are not sequential hermaphrodites and mature earlier (between 1–3 years old; ballan: 6–9 years) (Darwall *et al.*, 1992; Halvorsen *et al.*, 2016). Male corkwings, which build nests and provide parental care, grow faster, mature later and attain larger sizes than females (Potts, 1974; Darwall *et al.*, 1992; Sayer *et al.*, 1996; Halvorsen *et al.*, 2016). The suggested length at maturity for corkwing wrasse is ~10 cm (Darwall *et al.*, 1992), which is less than the minimum CRS limit before the change in 2018. This may explain why, despite the high retention rate of corkwing wrasse in 2017 (~94%), corkwing CPUE remained stable between 2017–2018. Corkwing CPUE increased in 2019, which is likely to be a result of the changes in CRS measures. The minimum CRS was increased to 14 cm, likely increasing the probability that more of the recently sexually matured males (which tend to be larger; Halvorsen *et al.*, 2016) were returned to sea. Additionally, decreasing the maximum CRS to the more biologically-relevant 18 cm affords protection to large and more fecund individuals of both sexes, aiding recruitment (Birkeland and Dayton, 2005).

Corkwing CPUE and LPUE increased with day of year, from July to October, which may partly reflect changing activity levels (and hence differing levels of catchability) of corkwing within each year. For example, nesting male corkwing wrasse exhibit high site fidelity during the spawning season as they are occupied by nest building and territory defence (Potts, 1985; Darwall *et al.*, 1992; Halvorsen *et al.*, 2016). It is possible that there is reduced likelihood of catching (and landing) nesting male corkwing wrasse (thus reducing CPUE and LPUE overall) during the spawning season. The corkwing spawning season is likely to occur prior to or early in the survey period presented here: timing of spawning can vary between April–September, with Norwegian populations showing a peak in spawning activity in June (Darwall *et al.*, 1992; Skiftesvik *et al.*, 2015).

4.5. Assemblage-level changes

There was an apparent change in catch composition between years, but this is explained by differences in spatial patterns of fishing effort between years and seasonal variation in wrasse catches within years. The abundance and/or catchability of some wrasse species vary seasonally and also vary significantly over space (shown in the species-specific results in Section 3.4). The location of fishing effort has changed across the 2017–2020 period (Figure 1), so the change in catch composition across years is likely to be a result of a change in fishing location.

4.6. Compliance with the Fully Documented Fishery

During 2020, D&S IFCA's observer surveys achieved up to 6.3% coverage of known days fished. This represents seven surveys from a total of 111 known fishing trips (108 trips were reported by fishers, four of which also had an observer present, three additional trips had an observer present but were not reported by fishers). Over the course of monitoring the fishery during 2017–2020 there have been difficulties in arranging observer surveys due to inclement weather and mechanical issues causing survey cancellations, and difficulty aligning limited officer time with sporadic fishing activities. In 2020, this was further complicated by the COVID-19 pandemic. D&S IFCA officers managed to continue the fishery observer surveys by conducting surveys from D&S IFCA's RIB from early August until early October, though COVID-19-related concerns caused the cessation of surveys from early October. Aside from Vessel 3, fishers are typically very welcoming of officers surveying their activities, whether on-board or using D&S IFCA's RIB. For 2020, the true coverage of fishing effort with observer surveys is likely to be lower than 6.3% due to underreporting of days fished by Vessel 3 throughout 2020. Vessel 3 has a history of non-cooperation regarding arranging onboard surveys as well as not supplying returns forms.

Wrasse fishers are required under the Potting Permit Byelaw to 'provide any relevant fisheries information required by the Authority for the discharge of its functions'. Therefore, in order to comply with the fully documented fishery, fishers are required to submit weekly returns forms complete with a set of pre-determined and pre-notified information regarding each day of fishing activity. These forms were provided to all fishers in advance of the fishing season by the SEO and DCO. Below is an outline of the required information and the completion or otherwise of the data provided by fishers during 2020. Forms were typically not submitted weekly, but were instead submitted for two or more weeks of activity at a time.

Requirement 1: Dates of fishing trips

> To the best of D&S IFCA's knowledge these are reported accurately. No returns were provided by Vessel 3.

Requirement 2: Locations of strings of pots (based on 1 km grid squares on a map of Plymouth Sound)

> This was reported fully on 76% of reported fishing trips (82 out of 108). However, evidence suggests that this self-reporting is not always accurate. On four occasions in 2020, a fisher submitted a returns form for a day on which an observer was also surveying their catch. This allows for a comparison of fisher self-reporting of location with known locations of their gear based on GPS records noted by the observer. For these four trips, the fisher correctly reported the location of their gear for only 66% of strings hauled.

Requirement 3: Number of strings hauled and number of pots hauled

> This was reported fully for 69% of the reported fishing trips conducted in 2020. This poor recording makes it impossible to calculate landings per unit effort for the misreported trips. The misreporting in this case includes instances in which the fisher recorded having fished 5 strings (on the returns form for that trip), whereas the D&S IFCA observer recorded data from 6 strings from that trip.

Requirement 4: Number of wrasse retained on board per trip

> This is reported for 83% of trips (90 out of 108 trips). The wrasse caught on the remaining 18 trips were reported as aggregate totals across six different submissions (ie. on six occasions, the total wrasse landed from 2–4 individual trips was recorded as a single total). In addition, for one fishing trip the D&S IFCA observer recorded 33 wrasse retained on board whereas the fisher subsequently reported 25 fish landed on the returns form for that trip. Even if eight fish were discarded in the harbour prior to landing, this under-reports removals from the population.

Overall, compliance with the returns forms aspect of the Fully Documented Fishery is relatively low, which prevents thorough examination of the returns data. The main advantage to accurate returns data would be the availability of fine-scale information on wrasse landings over time. Fortunately, this information is available on transport documents provided by the salmon farm agent, though admittedly at a coarser temporal resolution (approximately every week or fortnight, sometimes monthly), rather than daily (though fishers do not always report daily totals). Given the issues of low compliance and inaccurate reporting, the primary value of these returns forms has been in aiding D&S IFCA's understanding of the spatial distribution of fishing effort in each year.

D&S IFCA's officers have reviewed the requirement to submit returns forms, and have identified two further constraints associated with these data, which apply even to fully-completed returns data: (i) the spatial scale of reporting of wrasse catches means that it is not possible to estimate the numbers of wrasse caught in each grid cell (since total wrasse retained are reported for the trip, not for each string) and, critically, (ii) recent analyses have demonstrated that robust monitoring and management of this fishery requires species-specific data on catch and landings per unit effort, which are not available from these fishers' returns forms. Species-specific data are only available from the observer surveys carried out by D&S IFCA's officers, which have provided a four-year dataset collected with standardised methods that is therefore comparable with future data collected by observers.

D&S IFCA's officers would therefore recommend removing the requirement for fishers to submit returns forms, which would reduce the associated administrative and time cost of monitoring, and allow greater focus on monitoring via observer surveys. The observer surveys provide much richer and more reliable data, and are especially efficient when carried out from D&S IFCA's RIB; using the RIB as an observer platform reduces the time taken to conduct each survey, is seen as safer than surveys on board fishing vessels, and can be effectively combined with other patrol and enforcement work.

5. Conclusions and Recommendations

The four wrasse species that are routinely landed in the D&S IFCA District each respond differently to fishing pressure, geographical variables and environmental variables, which highlights the importance of considering the species separately for management purposes. In highlighting the main drivers of variation in CPUE and LPUE, the importance of including geographical and environmental variables (particularly fishing location) is clear. Wrasse catches and landings can vary significantly over small spatial scales within the D&S IFCA's District, likely due to the differing habitat types or conditions experienced in the areas (e.g. exposure or freshwater influence). If these environmental variables are not taken into account throughout survey design and data analysis, incorrect conclusions may be drawn regarding fishery effects and sustainability.

It is recommended that, within the time- and resource-limited research capabilities of D&S IFCA (and the restrictions imposed by locations and timing of routine fishing activity), future monitoring of this fishery relies on observer survey data, and that future surveys should aim to distribute survey effort evenly and consistently over time and locations. This would allow for important species-specific drivers of CPUE and LPUE, identified here, to be accounted for in future analyses, enabling robust monitoring and recommendations for management.

In addition to these monitoring recommendations, D&S IFCA's officers consider that action on the following recommendations will help to maintain the environmentally, economically and socially sustainable nature of the live wrasse fishery in D&S IFCA's District:

(i) Continue to manage the fishery as outlined in the D&S IFCA's Policy Statement and Potting Permit Conditions for the Live Wrasse Fishery (24th June 2020), except in the case of rock cook (ii, below) and ballan wrasse (iii, below), and except with regards to fishers returns forms (iv, below).

(ii) Lift the prohibition on removal of rock cook from the fishery, and reintroduce previous conservation reference size (CRS) limits to the Potting Permit Byelaw permit conditions for this species (12–23 cm).

(iii) Change the ballan wrasse CRS range from 15–23 cm to 18–26 cm (see Appendix 3 for an assessment of the impact that changes to the CRS limits may have on the retention rates of ballan wrasse in the D&S IFCA's District).

(iv) Remove the requirement for wrasse fishers to submit returns forms (without affecting future obligations under Paragraph 17 of the Potting Permit Byelaw).

References

Alonzo, S. H., and Mangel, M. 2004. The effects of size-selective fisheries on the stock dynamics of and sperm limitation in sex-changing fish. Fishery Bulletin, 102: 1–13.

Artüz, M. L. 2005. Age and growth of the ballan wrasse in the Sea of Marmara.

Birkeland, C., and Dayton, P. K. 2005. The importance in fishery management of leaving the big ones. Trends in Ecology & Evolution, 20: 356–358.

Bjordal, A. 1988. Cleaning symbiosis between wrasses (Labridae) and lice infested salmon (Salmo salar) in mariculture.: 8.

Bjordal, Å. 1991. Wrasse as cleaner-fish for farmed salmon. 17-28. Underwater Association for Scientific Research. https://imr.brage.unit.no/imr-xmlui/handle/11250/109019 (Accessed 7 February 2021).

Burka, J. F., Hammell, K. L., Horsberg, T. E., Johnson, G. R., Rainnie, D. J., and Speare, D. J. 1997. Drugs in salmonid aquaculture – A review. Journal of Veterinary Pharmacology and Therapeutics, 20: 333–349.

Burridge, L., Weis, J. S., Cabello, F., Pizarro, J., and Bostick, K. 2010. Chemical use in salmon aquaculture: A review of current practices and possible environmental effects. Aquaculture, 306: 7–23.

Clark, S., and Townsend, N. 2017. Devon and Severn IFCA Potting Permit Byelaw.

Costello, M., and Bjordal, Å. 1990. How good is this natural control on sea-lice? Fish Farmer, 13: 44–46.

Curtin, S., Henly, S., and Stewart, J. 2020. Three Year Comprehensive Review of the Live Wrasse Fishery in Devon and Severn IFCA's District. Devon and Severn Inshore Fisheries and Conservation Authority, Brixham, Devon.

Darwall, W. R. T., Costello, M. J., Donnelly, R., and Lysaght, S. 1992. Implications of life-history strategies for a new wrasse fishery. Journal of Fish Biology, 41: 111–123.

Davies, S. 2016. *A review of wrasse ecology and fisheries interactions*. D&S IFCA, Brixham, Devon.

Davies, S. and West, E. (2017). Live Wrasse Fishery in Devon and Severn IFCA District. D&S IFCA, Brixham, Devon.

Dipper, F. A., and Pullin, R. S. V. 1979. Gonochorism and sex-inversion in British Labridae (Pisces). Journal of Zoology, 187: 97–112.

EMODnet. 2020. EMODnet Bathymetry Viewing and Download service. https://portal.emodnet-bathymetry.eu/ (Accessed 9 February 2021).

Gjøsaeter, J. 2002. Distribution and density of goldsinny wrasse (Ctenolabrus rupestris) (Labridae) in the Risør and Arendal areas along the Norwegian Skagerrak coast. Sarsia, 87: 75–82. Taylor & Francis.

Gravestock, V. 2018. HRA - Studland to Portland SAC - Fish traps. SIFCA, Poole, Dorset.

Halvorsen, K. T., Sørdalen, T. K., Durif, C., Knutsen, H., Olsen, E. M., Skiftesvik, A. B., Rustand, T. E., *et al.* 2016. Male-biased sexual size dimorphism in the nest building corkwing wrasse (Symphodus melops): implications for a size regulated fishery. ICES Journal of Marine Science, 73: 2586–2594.

Hartig, F., and Lohse, L. 2020. DHARMa: Residual Diagnostics for Hierarchical (Multi-Level / Mixed) Regression Models. https://CRAN.R-project.org/package=DHARMa (Accessed 26 January 2021).

Hothorn, T., Bretz, F., Westfall, P., Heiberger, R. M., Schuetzenmeister, A., and Scheibe, S. 2020. multcomp: Simultaneous Inference in General Parametric Models. https://CRAN.R-project.org/package=multcomp (Accessed 26 January 2021).

Leclercq, E., Grant, B., Davie, A., and Migaud, H. 2014. Gender distribution, sexual size dimorphism and morphometric sexing in ballan wrasse Labrus bergylta. Journal of Fish Biology, 84: 1842–1862.

Liu, Y., and Bjelland, H. vanhauwaer. 2014. Estimating costs of sea lice control strategy in Norway. Preventive Veterinary Medicine, 117: 469–477.

Matland, E. C. 2015. The biological indicators and temporal spawning habits of wrasse (Family: Labridae) from Sunnhordland. The University of Bergen. https://bora.uib.no/bora-xmlui/handle/1956/10421 (Accessed 6 February 2021).

Maunder, M. N., and Punt, A. E. 2004. Standardizing catch and effort data: a review of recent approaches. Fisheries Research, 70: 141–159.

Metri, C. B., Perez, J. A. A., Metri, C. B., and Perez, J. A. A. 2014. A LPUE (landing per unit effort) analysis of the trawl fishery for the coastal shrimps Artemesia longinaris and Pleoticus muelleri off southern Brazil. Brazilian Journal of Oceanography, 62: 235–245. Instituto Oceanográfico da Universidade de São Paulo.

Muncaster, S., Andersson, E., Kjesbu, O. S., Taranger, G. L., Skiftesvik, A. B., and Norberg, B. 2010. The reproductive cycle of female Ballan wrasse Labrus bergylta in high latitude, temperate waters. Journal of Fish Biology, 77: 494–511.

Mundry, R., and Nunn, C. L. 2009. Stepwise Model Fitting and Statistical Inference: Turning Noise into Signal Pollution. The American Naturalist, 173: 119–123. The University of Chicago Press.

Overton, K., Dempster, T., Oppedal, F., Kristiansen, T. S., Gismervik, K., and Stien, L. H. 2019. Salmon lice treatments and salmon mortality in Norwegian aquaculture: a review. Reviews in Aquaculture, 11: 1398–1417.

Potts, G. W. 1974. The colouration and its behavioural significance in the corkwing wrasse, *Crenilabrus melops*. Journal of the Marine Biological Association of the United Kingdom, 54: 925–938.

Potts, G. W. 1985. The Nest Structure of the Corkwing Wrasse, Crenilabrus Melops (Labridae: Teleostei). Journal of the Marine Biological Association of the United Kingdom, 65: 531–546. Cambridge University Press.

QGIS. 2020. QGIS Geographical Information System. Open Source Geospatial Foundation Project. http://qgis.org.

R Core Team. 2020. R: A language and environment for statistical computing. R Foundation For Statistical Computnig, Vienna, Austria.

Rae, G. H. 2002. Sea louse control in Scotland, past and present. Pest Management Science, 58: 515–520.

Richards, S. A. 2008. Dealing with overdispersed count data in applied ecology. Journal of Applied Ecology, 45: 218–227.

Richards, S. A. 2015. Likelihood and model selection. http://dx.doi.org/10.1093/acprof:oso/9780199672547.003.0004 (Accessed 26 January 2021).

Roth, M., Richards, R. H., and Sommerville, C. 1993. Current practices in the chemotherapeutic control of sea lice infestations in aquaculture: a review. Journal of Fish Diseases, 16: 1–26.

Sayer, M. D. J., Gibson, R. N., and Atkinson, R. J. A. 1993. Distribution and density of populations of goldsinny wrasse (*Ctenolabrus rupestris*) on the west coast of Scotland. Journal of Fish Biology, 43: 157–167.

Sayer, M. D. J., Treasurer, J. W., and Costello, M. J. 1996. Wrasse: biology and use in aquaculture. Fishing News Books. https://agris.fao.org/agris-search/search.do?recordID=US9618464 (Accessed 5 February 2021).

Sekhon, J. S. 2020. Matching: Multivariate and Propensity Score Matching with Balance Optimization. https://CRAN.R-project.org/package=Matching (Accessed 26 January 2021).

Skiftesvik, A. B., Bjelland, R. M., Durif, C. M. F., Johansen, I. S., and Browman, H. I. 2013. Delousing of Atlantic salmon (Salmo salar) by cultured vs. wild ballan wrasse (Labrus bergylta). Aquaculture, 402–403: 113–118.

Skiftesvik, A. B., Blom, G., Agnalt, A.-L., Durif, C. M. F., Browman, H. I., Bjelland, R. M., Harkestad, L. S., *et al.* 2014. Wrasse (Labridae) as cleaner fish in salmonid aquaculture – The Hardangerfjord as a case study. Marine Biology Research, 10: 289–300.

Skiftesvik, A. B., Durif, C. M. F., Bjelland, R. M., and Browman, H. I. 2015. Distribution and habitat preferences of five species of wrasse (Family Labridae) in a Norwegian fjord. ICES Journal of Marine Science, 72: 890–899.

Street, K., Trundle, C., Jenkin, A., and Naylor, H. 2017. Live Wrasse Fishery Investigations 2016 - 2017, A Report to Cornwall IFCA Authority Meeting December 2017. CIFCA, Hayle, Cornwall.

Thangstad. 1999. Spatial and temporal distribution of three wrasse species (Pisces: Labridae) in Masfjord, western Norway: habitat association and effects of environmental variables. University of Bergen. https://bora.uib.no/bora-xmlui/bitstream/handle/1956/1242/Hovedoppgave-thangstad.pdf?sequence=1 (Accessed 6 February 2021).

Townsend, N. 2020. Devon and Severn IFCA's Byelaw Status and Changes Guide. Version 5. D&S IFCA, Brixham, Devon. Available at https://www.devonandsevernifca.gov.uk/Resource-library/G-Authority-Communications-Publications [under Information Guides (Text)].

Treasurer, J. W. 1994. The distribution, age and growth of wrasse (Labridae) in inshore waters of west Scotland. Journal of Fish Biology, 44: 905–918.

Treasurer, J. W. 2012. Diseases of north European wrasse (Labridae) and possible interactions with cohabited farmed salmon, *Salmo salar* L. Journal of Fish Diseases, 35: 555–562.

Tully, O., Daly, P., Lysaght, S., Deady, S., and Varian, S. J. A. 1996. Use of cleaner-wrasse (Centrolabrus exoletus (L.) and Ctenolabrus rupestris (L.)) to control infestations of Caligus elongatus Nordmann on farmed Atlantic salmon. Aquaculture, 142: 11–24.

Uncles, R. J., Stephens, J. A., and Harris, C. 2015. Physical processes in a coupled bay–estuary coastal system: Whitsand Bay and Plymouth Sound. Progress in Oceanography, 137: 360–384.

Varian, S., Deady, S., and Fives, J. 1996. The effect of intensive fishing of wild wrasse populations in Lettercallow Bay, Connemara, Ireland: implications for the future management of the fishery. *In* pp. 100–118.

Venables, W. N., and Dichmont, C. M. 2004. GLMs, GAMs and GLMMs: an overview of theory for applications in fisheries research. Fisheries Research, 70: 319–337.

Villegas-Ríos, D., Alonso-Fernández, A., Domínguez-Petit, R., and Saborido-Rey, F. 2013. Intraspecific variability in reproductive patterns in the temperate hermaphrodite fish, Labrus bergylta. Marine and Freshwater Research, 64: 1156–1168. CSIRO PUBLISHING.

Appendix 1 – Methods of LPUE, CPUE and size distribution analyses using Generalised Linear Models

Generalised Linear Models (GLMs) are essentially a flexible form of ‘linear regression’, which is a statistical method that describes change in one variable (the response) as a function of change in one or more predictors. In linear regression, the response variable* (e.g. LPUE) is assumed to be normally distributed (a histogram of the data resembles a bell-shaped curve) and has a linear relationship with the predictors (i.e. a graph would show a straight-line relationship between the response and predictors). Generalised linear models are a more flexible extension of this approach, which allow for the modelling of non-normal response variables (whose plots do not look like an ordinary bell-shaped curve), and allow for non-linear relationships between the response and predictors.

* Strictly speaking, it is the ‘residuals’ that are assumed to be normally distributed, but an exploration of this aspect of statistical techniques is beyond the scope of this report.

GLMs were used to assess changes in species-level LPUE, CPUE and fish length (size) over the 2017–2020 period, and to investigate the drivers of change. Following Henly *et al.* (*in review*), this GLM approach has been applied in order to account for the impact of spatial and environmental variables on CPUE, LPUE and size, allowing for a more accurate representation of stock abundance dynamics and size changes over time (Venables & Dichmont 2004, Maunder & Punt 2004). Without accounting for the effects of these additional spatial and environmental variables in this way, there is a risk of either (i) incorrectly attributing apparent changes in CPUE, LPUE or size to the effects of fishing effort, or (ii) of not detecting an effect of fishing effort if it is masked by changes in other variables. This approach also permits identification of the variables that influence CPUE, LPUE and size of fish caught, and can therefore provide information on the ecology of the target species, which can help to inform management decisions (Maunder & Punt 2004; Henly *et al.,* in review).

Within this GLM-based statistical approach, day of year of fishing trip (DOY), year (Y), fishing location relative to the breakwater (BW), broad scale fishing area (BA), average distance to shore/ structure (Dist), depth band (DB), and bait type (Bait), were considered as potential predictors. All plausible two-way interactions between spatial/environmental (BW, BA, Dist, DB) and temporal (DOY, Y) predictors were also considered. For example, in a GLM for CPUE, a two-way interaction between Y and BA would indicate that CPUE changes between years (Y), but that the change experienced is different between broad areas.

In this GLM approach all plausible combinations of variables and interactions were considered in individual models (GLMs) for each species. Model selection techniques were then applied in order to determine which model (which combination of predictor variables) is the ‘best’ model given the data. Tukey tests, with p-values adjusted for multiple comparisons, were used to test for significant differences between levels of categorical predictors in the final models using the R package ‘multcomp’ (Hothorn *et al.*, 2020).

Detailed modelling and model selection approach

For each response variable (CPUE for each species, LPUE for each species and size of ballan wrasse), a candidate set of models was sought that were consistent with the data, in addition to a ‘null model’ (which contained no predictor variables). Then to select the most appropriate model from this set of candidate models, AIC (Akaike’s Information Criterion) was used as the model selection criterion. Though the model with the lowest AIC is likely to be the most parsimonious, AIC is only an estimate of parsimony. Therefore, following Henly *et al.* (in review) and Richards (2008), certain other models were considered as well. First, models that generated AIC values with $\Delta AIC \leq 6$ were determined and then, to prevent unsupported, overly-complex models being selected, models from the candidate set that were more complex versions of other selected models were removed (Richards, 2008). Where this process failed to identify a single 'best' model, biological inference was based on the model with fewest terms (or lowest AIC where models had the same number of terms), following Richards (2015). This approach allowed all good candidate models to be compared, and permitted consideration of other important variables that would be excluded using methods such as stepwise selection (Mundry and Nunn, 2009). Then, comparing the final model to the null model essentially allows for assessment of whether the models are performing better than random (i.e. whether the predictor terms are useful in predicting the response variable).

Biological inference based on selected models

Following selection of the most parsimonious model for each response variable (CPUE, LPUE or size), the GLM output was used to identify changes in the response variable over the 2017–2020 period. For cases in which a model outperforms the associated null model (based on AIC), this is widely considered to be sufficient evidence that the predictor variables are useful in predicting change in the response variable. However, *p*-values associated with individual model terms are presented, as these may be more familiar to readers of this report. *P*-values < 0.05 essentially indicate that the model terms are significant predictors of change in the response.

Model assessment

Model diagnostics were checked based on visual and statistical assessment of scaled model residuals, using the 'DHARMa' R package (Hartig and Lohse, 2020).

Detailed AIC analyses and model results

This section reports comparisons (based on AIC) of the GLMs for each response variable (CPUE for each species and LPUE for each species).

Table S1.1: Drivers of variation in catch and landings per unit effort (CPUE and LPUE [and average size (cm)]), for ballan (a, b, [c]), goldsinny (d, e), rock cook (f, g) and corkwing (h, i), summarising AIC analyses for all candidate GLMs (M_1, M_2,…M_n) with $\Delta AIC \leq 6$. M_{AIC} denotes the best AIC model and M_{final} denotes the selected, most parsimonious model. Also presented for comparison is the null model (M_{null}). Parameter estimates (with standard errors) are shown for the intercept (β_0), bait type (Bait), broad area (BA), breakwater position (BW), depth band (DB), distance to shore/ structure (Dist), day of year (DOY), tidal range (TR), year (Y) and interaction terms for BW:Y and Dist:DOY. DB was fitted as an ordered factor with orthogonal polynomial contrasts, so parameter estimates are presented for the linear (DB_L), quadratic (DB_Q) and cubic (DB_C) terms. BA was fitted as zero-sum contrasts, whereas BW and Y are presented as treatment contrasts. k is the number of parameters and LL is the log-likelihood of the model. All models fitted with gamma error distribution and identity link function unless marked with *, where a log link function was used

a) Ballan CPUE candidate models

Model	β_0	$Bait_{Crabs}$	$BW_{Seaward}$	Y_{2018}	Y_{2019}	Y_{2020}	$BW{:}Y_{Sea2018}$	$BW{:}Y_{Sea2019}$	$BW{:}Y_{Sea2020}$	k	LL	ΔAIC
M_{AIC}	0.317 (0.081)	0.038 (0.020)	-0.240 (0.082)	-0.233 (0.083)	-0.133 (0.088)	-0.148 (0.083)	0.215 (0.036)	0.139 (0.091)	0.211 (0.096)	10	101.47	0
M_1	0.355 (0.080)	–	-0.240 (0.083)	-0.241 (0.084)	-0.171 (0.087)	-0.168 (0.084)	0.219 (0.089)	0.150 (0.092)	0.214 (0.098)	9	100.90	1.00
M_2*	-1.610 (0.240)	0.316 (0.188)	-0.602 (0.162)	-0.544 (0.215)	-0.158 (0.259)	-0.120 (0.227)	–	–	–	7	98.51	1.92
M_{final}*	-1.320 (0.179)	–	-0.541 (0.156)	-0.609 (0.211)	-0.420 (0.218)	-0.271 (0.212)	–	–	–	6	96.95	3.04
M_{null}	0.160 (0.013)	–	–	–	–	–	–	–	–	2	82.45	24.04
M_{null}*	-1.830 (0.083)	–	–	–	–	–	–	–	–	2	82.45	24.04

b) Ballan LPUE candidate models

Model	β_0	$BW_{Seaward}$	Y_{2018}	Y_{2019}	Y_{2020}	$BW{:}Y_{Sea2018}$	$BW{:}Y_{Sea2019}$	$BW{:}Y_{Sea2020}$	k	LL	ΔAIC
M_{final}	0.256 (0.059)	-0.158 (0.064)	-0.201 (0.062)	-0.097 (0.068)	-0.113 (0.064)	-0.179 (0.067)	-0.067 (0.076)	-0.135 (0.077)	9	97.66	0
M_{null}	0.128 (0.012)	–	–	–	–	–	–	–	2	82.14	17.03

c) Ballan Size candidate models

Model	β_0	$BW_{Seaward}$	BA_B	BA_D	BA_E	Y_{2018}	Y_{2019}	Y_{2020}	k	LL	ΔAIC
M_{final}	17.255 (0.343)	2.544 (0.501)				0.479 (0.596)	1.370 (0.542)	2.403 (0.512)	6	-1064.78	0
M_1	17.283 (0.399)		-0.103 (0.543)	2.112 (0.693)	2.993 (0.735)	0.406 (0.607)	1.409 (0.588)	2.527 (0.587)	8	-1064.15	2.7
M_{null}	18.89 (0.205)								2	-1088.76	39.9

d**) Goldsinny CPUE candidate models**

Model	β_0	BA_A	BA_B	BA_C	BA_D	Dist	DOY	k	LL	ΔAIC
M_{final}	0.787 (0.153)	-0.463 (0.191)	0.072 (0.165)	1.014 (0.596)	-0.377 (0.156)	0.196 (0.056)	-0.123 (0.034)	8	-43.66	0
M_{null}	0.628 (0.050)	–	–	–	–	–	–	2	-61.19	23.06

e) Goldsinny LPUE candidate models

Model	β_0	BA_A	BA_B	BA_C	BA_D	DB_L	DB_Q	DB_C	Dist	DOY	Dist:DOY	k	LL	ΔAIC
M_{AIC}	0.209 (0.034)	–	–	–	–	0.146 (0.089)	-0.047 (0.071)	-0.001 (0.038)	0.008 (0.016)	-0.038 (0.014)	0.036 (0.017)	8	78.75	0
M_{final}	0.213 (0.035)	–	–	–	–	0.145 (0.092)	-0.056 (0.072)	-0.013 (0.040)	–	-0.047 (0.015)	–	6	76.39	0.72
M_1	0.217 (0.030)	-0.131 (0.050)	0.036 (0.037)	0.132 (0.109)	-0.045 (0.034)	–	–	–	0.033 (0.020)	-0.023 (0.016)	0.048 (0.018)	8	77.38	4.73
M_{null}	0.207 (0.016)	–	–	–	–	–	–	–	–	–	–	2	68.83	7.84

f) Rock cook CPUE candidate models

Model	β_0	$Bait_{Crabs}$	BA_A	BA_B	BA_C	BA_D	$BW_{Seaward}$	DB_L	DB_Q	DB_C	Dist	DOY	Y_{2018}	Y_{2019}	Y_{2020}
M_{AIC}*	-2.146 (0.357)	–	–	–	–	–	1.858 (0.408)	–	–	–	0.206 (0.105)	–	0.167 (0.450)	-0.005 (0.514)	1.547 (0.490)
M_1*	-2.197 (0.365)	–	–	–	–	–	2.036 (0.435)	–	–	–	–	0.131 (0.078)	0.286 (0.465)	0.043 (0.530)	1.308 (0.474)
M_2	0.183 (0.041)	–	–	–	–	–	0.439 (0.071)	0.112 (0.084)	-0.168 (0.080)	-0.143 (0.075)	–	–	–	–	–
M_3*	-0.943 (0.190)	0.159 (0.230)	-1.396 (0.223)	-0.092 (0.238)	0.914 (0.411)	0.172 (0.182)	–	–	–	–	0.265 (0.113)	–	–	–	–
M_4*	-0.864 (0.134)	–	-1.429 (0.283)	-0.174 (0.251)	0.875 (0.414)	0.249 (0.191)	–	–	–	–	0.253 (0.116)	0.077 (0.115)	–	–	–
M_{final}*	-0.847 (0.140)	–	-1.079 (0.254)	-0.243 (0.254)	0.655 (0.430)	0.186 (0.189)	–	–	–	–	–	–	–	–	–
M_{null}*	-0.697 (0.116)	–	–	–	–	–	–	–	–	–	–	–	–	–	–
M_{null}	0.498 (0.058)	–	–	–	–	–	–	–	–	–	–	–	–	–	–

Model	$BW{:}Y_{Sea2018}$	$BW{:}Y_{Sea2019}$	$BW{:}Y_{Sea2020}$	k	LL	ΔAIC
M_{AIC}*	-0.431 (0.515)	-0.214 (0.605)	-2.467 (0.595)	10	-5.98	0
M_1*	-0.611 (0.532)	-0.540 (0.616)	-2.439 (0.606)	10	-6.56	1.14
M_2	–	–	–	6	-10.7	1.43
M_3*	–	–	–	8	-9.88	1.79
M_4*	–	–	–	8	-9.91	3.85
M_{final}*	–	–	–	6	-12.6	5.23
M_{null}*	–	–	–	2	-23.26	18.55
M_{null}				2	-23.26	18.55

g) Rock cook LPUE candidate models

Model	β_0	$Bait_{Crabs}$	BA_A	BA_B	BA_C	BA_D	$BW_{Seaward}$	DOY	k	LL	ΔAIC
M_{AIC}*	-3.164 (0.375)	0.531 (0.353)	–	–	–	–	1.094 (0.320)	0.272 (0.148)	5	47.9	0
M_{final}*	-2.128 (0.172)	–	-0.387 (0.310)	-0.920 (0.404)	0.350 (0.479)	0.232 (0.225)	–	–	6	48.34	1.12
M_1	0.070 (0.022)	–	–	–	–	–	0.124 (0.037)	–	3	45.27	1.26
M_{null}*	-1.771 (0.147)	–	–	–	–	–	–	–	2	39.48	10.84
M_{null}	0.170 (0.025)	–	–	–	–	–	–	–	2	39.48	10.84

h) Corkwing CPUE candidate models

Model	β_0	$Bait_{Crabs}$	BA_A	BA_B	BA_C	BA_D	Dist	DOY	Y_{2018}	Y_{2019}	Y_{2020}	k	LL	ΔAIC
M_{Final}	0.396 (0.055)	–	–	–	–	–	–	0.148 (0.0.029)	0.041 (0.050)	0.429 (0.142)	0.442 (0.119)	6	-36.28	0
M_1*	-1.425 (0.183)		1.312 (0.244)	0.329 (0.172)	-2.122 (0.439)	0.185 (0.164)	-0.225 (0.092)	–	0.130 (0.214)	0.865 (0.227)	0.809 (0.236)	10	-34.99	5.41
M_2*	-1.216 (0.248)	-0.220 (0.178)	1.279 (0.241)	0.317 (0.171)	-2.157 (0.434)	0.219 (0.163)	-0.224 (0.091)	–	0.021 (0.220)	0.716 (0.251)	0.692 (0.251)	11	-34.16	5.75
M_{null}	-0.626 (0.057)	–	–	–	–	–	–	–	–	–	–	2	-57.26	33.95
M_{null}*	-0.468 (0.091)	–	–	–	–	–	–	–	–	–	–	2	-57.26	33.95

i) Corkwing LPUE candidate models

Model	β_0	DOY	k	LL	ΔAIC
M_{final}	-0.337 (0.024)	0.107 (0.017)	3	26.04	0
M_{null}	0.330 (0.026)	–	2	18.06	13.96

Literature Cited (Appendix 1)

Crawley, M.J. 2007. The R Book. John Wiley and Sons Ltd, Chichester, United Kingdom, 942 pp.

Hartig, F., and Lohse, L. 2020. DHARMa: Residual Diagnostics for Hierarchical (Multi-Level / Mixed) Regression Models. https://CRAN.R-project.org/package=DHARMa (Accessed 26 January 2021).

Henly, L., Stewart, J.E. and Simpson, S.D. (in review). Drivers and implications of change in an inshore multi-species fishery.

Richards, S.A. 2008. Dealing with overdispersed count data in applied ecology. *Journal of Applied Ecology*, 45: 218–227.

Appendix 2 – Multivariate Catch Composition Analysis

Permutational multivariate analysis of variance (PERMANOVA) and associated pairwise tests were used to test for changes in the catch composition of wrasse between years (assessing how the community changed overall). A Bray-Curtis similarity matrix using square root-transformed CPUE data for each species served as input for the analyses. The PERMANOVA design used year as a fixed factor and the analysis used 9,999 permutations.

Distance-based linear models (DISTLM) were applied to assess the combinations of the 7 core variables (excluding interaction terms) that most parsimoniously predict total catch composition using the model selection process outlined in Appendix 1. DISTLM is a non-parametric procedure that performs distance-based analysis on a linear model for any dissimilarity matrix (Anderson et al. 2008). The Bray-Curtis dissimilarity matrix was used to obtain a Euclidean distance measure of dissimilarity between pairs of samples for square root-transformed CPUE data for each species (including cuckoo wrasse, which was not included in the main CPUE and LPUE analyses). The purpose of DISTLM is to perform a permutational test for the multivariate null hypothesis of no relationship between two matrices on the basis of any distance measure of choice, using permutations of the observations.

All DISTLMs were restricted to a maximum of five predictors to avoid overfitting, running 9,999 tests by permutation. Predictor variables were standardized to zero mean and unit variance prior to multivariate analyses. The final selection of model variables balanced parsimony with low AIC values (see Appendix 1).

Appendix 3 – Ballan CRS review

Under the current CRS limits of 15–23 cm, up to 71% of ballan caught in pots were retained during 2017–2020. As highlighted in the report, targeting this size range with high retention rates risks highly sex-selective fishing and removal of mature females from the population, potentially limiting reproductive potential and affecting population growth. The evidence is therefore in support of a review of the CRS limits for ballan wrasse. Scenarios of CRS changes have been presented below for context.

Alternative scenarios were presented and recommended to D&S IFCA's Byelaw and Permitting Sub-Committee in the previous version of this report, however this report recommends a revised CRS range of 18–26 cm. This change came about following an additional review of the literature on ballan wrasse life history and management measures, and in consultation with the salmon farm agent who deals with this fishery. D&S IFCA's Officers believe this revised CRS range is the best approach to balancing the environmental, social, and economic aspects of the fishery. This revised CRS range was suggested verbally in a presentation to the Byelaw and Permitting Sub-Committee (February 2021), at which this management recommendation was voted upon.

Table S3.1a shows the proportion of ballan below, within and above the current CRS limits (15 – 23 cm). Table S3.1b shows, for comparison, these proportions under an hypothetical CRS change scenario applied to the catches from 2017–2020: 18–26. The latter generally allows for a larger proportion of the catch to be returned to sea and, potentially, to subsequently breed. Based on previous research into the size at sexual maturity and sexual inversion in this sequentially hermaphroditic (sex-changing) species, it appears likely that increasing the minimum CRS from 15 cm to 18 cm would allow for a greater proportion of sexually mature females to remain in the population. By also increasing the maximum CRS from 23 cm to 26 cm would shift some of the removal pressure onto a proportion of the male ballan wrasse, whilst also allowing an economically viable number of wrasse to be retained by fishers.

Table S3.1. *Proportion of ballan wrasse caught in Plymouth Sound in 2017–2020 that (a) are below, within and above the current CRS range (15–23 cm), and (b) would be below, within and above the CRS range with an hypothetical recommended CRS CRS range of 18–26 cm.*

(a) *Current* CRS limits: 15–23 cm			
Year	**% Below**	**% Within**	**% Above**
2017	28.47	64.23	7.3
2018	20.73	57.32	21.95
2019	14.46	71.08	14.46
2020	12.37	71.14	16.49
(b) CRS *scenario* 1: 18–26 cm (recommended)			
Year	**% Below**	**% Within**	**% Above**
2017	49.64	48.90	1.46
2018	34.15	59.75	6.10
2019	38.55	59.04	2.41
2020	22.68	72.26	5.06

Shifting the CRS range for ballan wrasse from that which is currently stipulated in the Potting Permit Byelaw permit conditions (15–23 cm) to a higher and more conservative CRS range of 18–26 cm would likely result in lower future retention rates (e.g. Table S3.1a vs Table S3.1b). A review of the scientific literature, as presented in the full report, indicates that this may improve the reproductive potential and population growth of this species.