











BRIEF COMMUNICATION

Movement patterns of temperate wrasses (*Labridae*) within a small marine protected area

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Abstract

The movement patterns of three commercially important wrasse (*Labridae*) species inside a small marine protected area (~0.15 km²) on the west coast of Norway were analysed over a period of 21 months. The mean distance between capture and recapture locations varied between 10 and 187 m, and was species and season specific. The extent of movement was not related to body size or sex. These results imply that a network of small strategically located marine protected areas can be used as management tools to protect wrasses from size- and sex-selective fishing mortality.

KEYWORDS

Centrolabrus exoletus, cleaner fish, *Ctenolabrus rupestris*, mark-recapture, movement, *Symphodus melops*, wrasse

Species in the wrasse family (*Labridae*) are important mesopredators on rocky reefs in the north-eastern Atlantic (Costello, 1991; Norderhaug *et al.*, 2005). They prey on a wide range of invertebrates and constitute an important part of the diet for larger predatory fish and seabirds (Bourlat *et al.*, 2021; Dehnhard *et al.*, 2021; Östman *et al.*, 2016). Several of these wrasses are facultative cleaners in the wild (Breen, 1996; Hilldén, 1983; Potts, 1973), a behaviour that is exploited to combat salmon louse *Lepeophtheirus salmonis* (Krøyer 1837) infestation in salmonid aquaculture in Norway and elsewhere (Henly *et al.*, 2021; Skiftesvik *et al.*, 2013). The Norwegian wrasse fishery has expanded substantially during the last decade, which has resulted in reduced abundance and body size in exploited populations of goldsinny *Ctenolabrus rupestris* (L.) and corkwing wrasse *Symphodus melops* (L.) (Halvorsen *et al.*, 2017a,b). These two species make up ~90% of reported catch (Halvorsen *et al.*, 2020). The rock cook *Centrolabrus exoletus* (L.) is also frequently caught, but is typically discarded as bycatch.

The larger Ballan wrasse (*Labrus berggylta* Ascanius 1767) is much less abundant, but is highly valued as a cleaner fish (Skiftesvik *et al.*, 2014).

The fishery for wrasse is selective by species, size and sex, which increases the risk of overfishing (Halvorsen *et al.*, 2016, 2020; Kindsvater *et al.*, 2020). Depletion of wrasses can have wider ecological consequences through top-down effects on smaller grazers and molluscs or *via* bottom-up effects on larger piscivores, since wrasses constitute an important part of the diet of seabirds and gadoid fishes (Dehnhard *et al.*, 2021; Kraufvelin *et al.*, 2020; Östman *et al.*, 2016). However, most of the coastal regions affected by this fishery lack historical population data (Halvorsen *et al.*, 2017a; Skiftesvik *et al.*, 2015), making impact assessments difficult. No-take marine protected areas (MPAs) in regions with high fishing pressure represent valuable unfished reference areas and can play an important role in providing data to support an ecosystem-based management strategy (Bourlat *et al.*, 2021; Costello, 2014; Halvorsen *et al.*, 2017a). For

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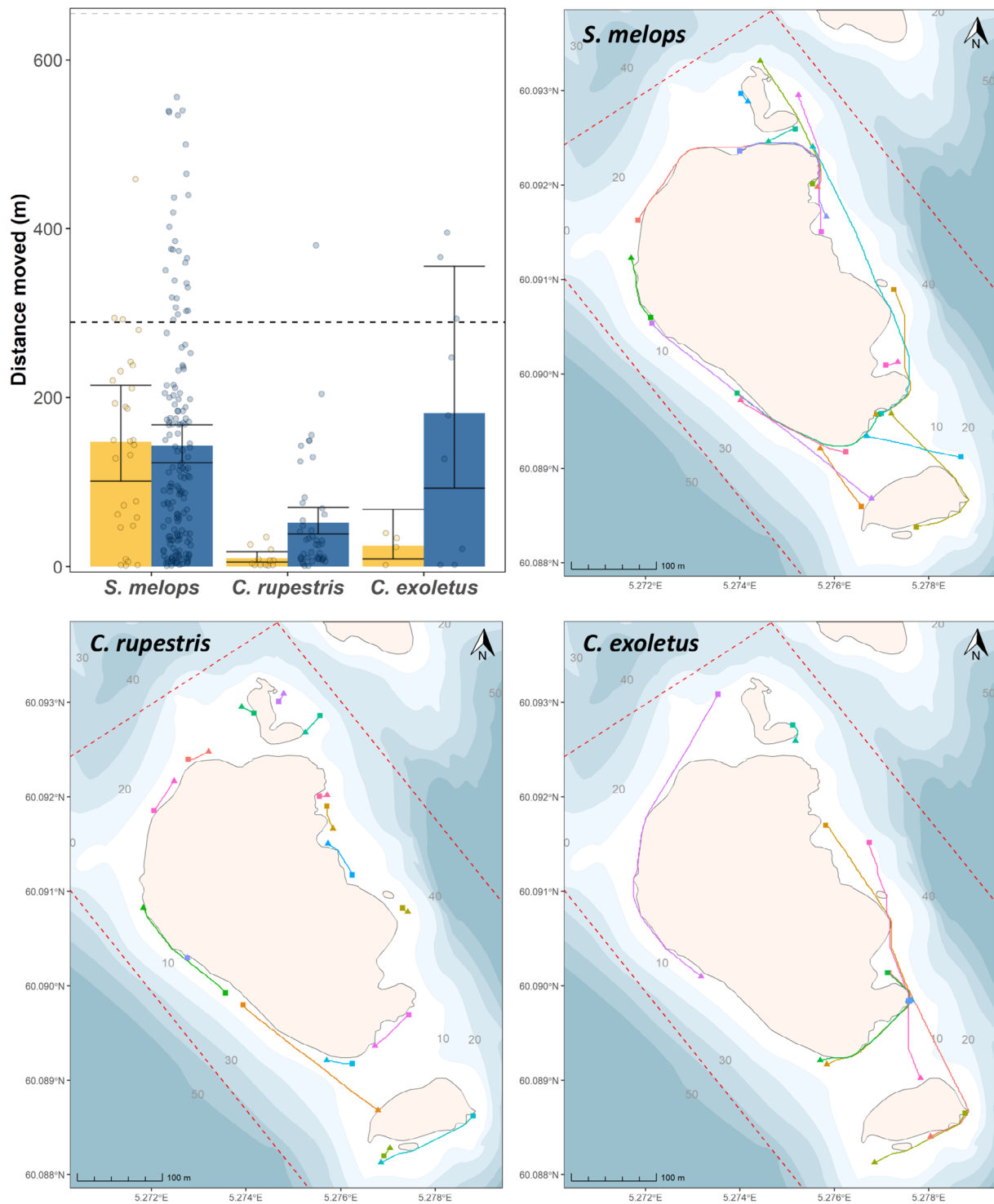


FIGURE 1 Movement of corkwing *Symphodus melops*, goldsinny *Ctenolabrus rupestris* and rock cook *Centrolabrus exoletus* inferred by capture-recapture data. The bar plot shows the model estimated mean value (bars) with corresponding 95% confidence intervals. The points represent the distance moved for each individual, jittered for clarity. The interval factor indicates whether the capture-recapture interval happened between within or between sampling years (sampling was only conducted from May to September). The dashed line shows the mean distance calculated from a random combination of all gear locations, thus indicating the expected mean distance under random movements within the study area (289 m). The maps show the study area and capture (square) and recapture (triangles), and movement trajectories (between sampling years) for randomly selected individuals for each of the three species. Colours correspond to different individuals. The dashed red line indicates the border of the marine protected area. (■) within year; (■) between year

example, length frequency distributions in and outside no-take MPAs can be used to estimate natural mortality and fishing mortality, essential parameters in stock assessment models (Wilson *et al.*, 2014). At the same time, MPAs provide the opportunity to monitor indirect effects on important prey or predators of these wrasses in the absence of fishing pressure (Bourlat *et al.*, 2021).

We characterized the movement patterns of *S. melops*, *C. rupestris* and *C. exoletus*, the three most abundant wrasses in Norway, and tested for interspecific and intraspecific differences. Movement patterns may differ between these species due to, for example, differences in life history traits, diet and vertical distribution (Halvorsen *et al.*, 2020; Sayer *et al.*, 1995, 1996). Further, males of *S. melops* build nests out of filamentous algae and provide paternal care, which may influence site-fidelity and sex-specific movements during the reproductive period of May–July (Halvorsen *et al.*, 2017b; Potts, 1985). Information about fish movement, including ranges and site fidelity, is key to guide the design of MPAs that can protect the natural phenotypic variation of these species (Halvorsen *et al.*, 2017b). We conducted a mark-recapture survey on a small island within a designated wrasse MPA in western Norway (Figure 1) during the period 2017–2019. To ensure spatially similar sampling effort, we divided the areas into 16 smaller zones and alternated between sampling odd and even numbered zones each day (eight zones per day). The location and extent of zones were determined based on natural landmarks (*e.g.*, a large rock, a small bay or other distinct change in the coastline) and distribution of suitable habitat (~hard bottom substrate, visually assessed; see Supporting Information Figure S1 for the zones shown on map).

There were five sampling periods during which wrasse were collected using fyke nets; 1: August/September 2017, 2: May 2018, 3: July 2018, 4: September 2018 and 5: May 2019. The fyke nets were set perpendicular to the shore, left overnight and hauled the next day. Passing fish encountering the leader of the fyke net are guided into the cod-end, thus capture depends on fish movement. Each zone was visited three times during the first sampling period. Two different types of fyke nets were used in equal proportions: a short type specialized for catching cod (5 m single leader, 55 cm diameter entrance ring and leader mesh size of 30 mm) and a longer, more fine-meshed type specialized for wrasses (leader: 7.8 m leader, entrance ring 70 cm diameter, 11 mm mesh size). The fine-meshed fyke nets captured a wider size range of the target species and therefore it was decided to continue with this type only in later sampling periods, during which each zone was sampled twice. Data from an additional sampling period carried out with pots in September 2018 (2 weeks after the previous fyke net sampling) was also included in the analysis. These were standard wrasse pots baited with 40–80 g frozen prawns, *Pandalus borealis* (Krøyer, 1838) (pots were two-chambered, 70 × 40 × 28 cm, 11 mm mesh size, 60 × 90 mm elliptical entrances, 12 mm wide escape openings). Although the use of baited gear may influence the movement of reef fish (Bacheler *et al.*, 2018), we assumed that this had little influence on our results given a balanced distribution of gear in the sampling area. Inclusion of pot-data improved statistical power and the main conclusions (*e.g.*, significance

of model coefficients) were not affected if pot-data was excluded. The sampling effort was distributed as described for fyke nets and the traps were soaked overnight.

A handheld GPS (Garmin GPSMAP 78 s, Garmin International Ltd, Olathe, Kansas, USA) was used to record the position of the gear. On capture, all fish were identified to the species level and a handheld reader (BioMark HPR, Boise, Idaho 83702, USA) was used to check for the presence of a passive integrated transponder tag (PIT, 2,12 × 12 mm; RFID Solutions, Stavanger, Norway). Wrasses were visually sexed and measured for total length to the nearest millimetre, and those with a total length of ≥10 cm were PIT-tagged following the procedure described for *S. melops* in Halvorsen *et al.* (2017b). No fish were tagged during the pot sampling. All fish were carefully released at the site of capture. The method of capture, tagging and handling of live fish complies with the Norwegian Food Safety Authority's animal welfare laws, guidelines and policies as approved by the Norwegian Animal Research Authority (Project IDs: 8715 and 15307).

We PIT-tagged 3721 wrasse and recaptured 265 (195 *S. melops*, 57 *C. rupestris* and 13 *C. exoletus*; Supporting Information Table S1). Capture-recaptures from the *same* sampling period were excluded. All data analysis was conducted using R v. 4.03 (R Core Team, 2020). The R-package *gdistance* (van Etten, 2017) was used to calculate the shortest waterway distance between all capture-recapture positions (Kersula & Seitz, 2019). We first converted our map polygon to a raster with 1818 × 1381 cells (1 × 1 m), with each assigned a value corresponding to land (0) or sea (1). Movement between cells was constrained by a transition matrix with up to 16 possible directional movements to neighbouring cells, while restricting movement to cells in the sea. The distance moved by an individual was calculated as the least-cost path distance between the cell of capture and the cell of recapture. Since sampling was restricted to a relatively small area, we estimated the expected mean movement under scenarios of completely free and random movement between all of the capture locations. To this end, the full list of coordinates of gear placements was duplicated 10 times and then randomly paired. The mean distance between these simulated random movements was 289 m (maximum 655 m). We assumed that there was minimal migration in or out of the study system, which is surrounded by waters of at least 20 m depth (Figure 1). The shortest distance to the nearby island was ~85 m (from the northernmost land point), but this stretch of water has a sandy bottom, which is an unfavourable habitat for wrasse such that they are unlikely to occupy or traverse it (Costello, 1991; Gjørseter, 2002; Sayer *et al.*, 1993).

A generalized linear model (glm; gamma-distribution, log-link) was used to test for interspecific differences in the distance moved (m) between capture and recapture locations. The model included two factors and their interaction: **Species** (*C. rupestris*, *C. exoletus*, *S. melops*) and **Interval** (*within year*, including those captured and recaptured in May–July–September 2018, and *between years*, including those captured and recaptured in different sampling years). This distinction is important because wrasses are territorial during spring and summer but hibernate or migrate to deeper water during winter

(Costello, 1991; Hilledén, 1981; Sayer, 1999). Intraspecific differences in movement patterns were assessed using glm, including sex and body size as covariates. Model selection was carried out from among eight models, defined *a priori* (including the null model), by comparing their Akaike Information Criterion values, corrected for small sample sizes (AIC_c) (Burnham & Anderson, 2004). The highest ranked models did not include sex or size for any of the species (Supporting Information Table S2). However, we acknowledge that the small sample sizes for *C. rupestris* and *C. exoletus* provide limited statistical power for detecting fine-scale differences. For both of these species, models including sex ranked second in both species ($\Delta AIC_c < 2$), so we cannot exclude the possibility of subtle sexual differences in movement that could have been detected with a larger data set. The underlying statistical assumptions in the glm models (homogeneity of variance, normally distributed residuals) were assessed by graphical inspection of residuals plotted against fitted values and covariates.

The distance moved between capture-recapture locations depended on species and sampling interval (glm; species \times year effect, likelihood ratio test $L = 18.73$, d.f. = 2, $P < 0.001$). *S. melops* and *C. rupestris* moved less than would be expected if they were moving unconstrained between all sampling locations [289 m vs. glm estimated means (*sampling interval*), *S. melops* 147 m (*within year*) and 143 m (*between years*), *C. rupestris* 10 m (*within year*) and 54 m (*between years*); Figure 1 and Table 1]. Thus, the small study area ($\sim 0.15 \text{ km}^2$) appears to provide sufficient space for natural movements for these two species. *C. exoletus* had the highest mean distance moved (189 m, *between years*), but the 95% confidence intervals were wide and overlapped the mean estimates for *S. melops* and for the simulated random movements (289 m). Thus, we acknowledge that a larger study area and sample size is necessary for assessing the movement of *C. exoletus* over a winter (*between years*). Nevertheless, like *C. rupestris*, the *C. exoletus* moved significantly shorter distances in the summer (25 m, *within year*).

It is possible that the longer between-year movements seen in *C. exoletus* and *C. rupestris* are related to territory relocation following hibernation in deeper waters (Hilledén, 1981). The depth ranges of *C. rupestris* (0–40 m) and *C. exoletus* (0–50 m) are wider than for

S. melops (0–18 m; Halvorsen *et al.*, 2020), and the two former species also show a seasonal shift in distribution towards deeper waters from June to October, which is not seen for *S. melops* (Figure 4 in Halvorsen *et al.*, 2020). This is consistent with diving observations at 5, 10 and 18 m, where *C. exoletus* and *C. rupestris* were only observed in deep locations (10 and 18 m) during winter (February–March), while both had the highest abundance at 5 m depth from May to June (Skog *et al.*, 1994). *S. melops* was rarely observed at 10 m and never at 18 m in that survey. These studies suggest that *C. rupestris* and *C. exoletus* have a higher scope for vertical movements, while *S. melops* are more confined to horizontal movements when searching for food or other resources and might remain in the same home range for winter hibernation. The very high residency of *C. rupestris* is consistent with previous studies: individuals remain in the same territories/refuges across several months and even consecutive years (Hilledén, 1981; Sayer, 1999). Acoustic telemetry would be useful to obtain more detailed insights into home-range and habitat use of these species, for instance in revealing the location and duration of winter hibernation. Relatively small acoustic tags are now available (e.g., V5, VEMCO, Nova Scotia, Canada), but they would probably be too large for the majority of *C. rupestris* and *C. exoletus* caught in this study. Moreover, these small tags have a relatively short battery life (months) which would limit the applicability of this method in comparing movement between different seasons in the same study.

Body size did not affect movement for any of these species (this study), nor in the larger *L. bergylta* (Villegas-Ríos *et al.*, 2013). This implies that these temperate wrasses maintain the size and location of the home range as they grow and age. Thus, the low mobility and affinity to shallow water is probably a strong determinant of population structure for these wrasses. The Norwegian coastline is rugged, with deep fjords and numerous rocky reefs and islands (Simensen *et al.*, 2021), which implies a fine-scaled structure of many isolated and demographically closed populations because of natural barriers. This is probably a major reason for the considerable differences in growth, age and abundance that have been observed at spatial scales of less than 5 km for these wrasses, and this has been related to variation in fishing pressure (Halvorsen *et al.*, 2017a; Olsen *et al.*, 2019). Hence, rebuilding a natural species composition and size distribution

Predictors	Log distance (m)		
	Estimate	Standard error	P value
(Intercept)	4.99	0.19	<0.001
Species (<i>C. rupestris</i>)	−2.7	0.35	<0.001
Species (<i>C. exoletus</i>)	−1.78	0.55	0.001
Interval (between years)	−0.03	0.21	0.889
Species (<i>C. rupestris</i>) \times interval (Between years)	1.69	0.39	<0.001
Species (<i>C. exoletus</i>) \times interval (between years)	2.02	0.65	0.002
Observations: 265.	(195 <i>S. melops</i> , 57 <i>C. rupestris</i> , 13 <i>C. exoletus</i>)		
R ² Nagelkerke	0.310		

TABLE 1 Summary of the generalized linear model (gamma distribution, log link) testing for differences in distance moved between species and sampling interval (within or between years), showing estimates (log-scale), standard errors and the associated P values (bold italics denote significance)

in isolated populations may be slow if there is no exchange of juveniles or adults with nearby areas.

MPAs are well suited to protect coastal species from overfishing (Fernández-Chacón *et al.*, 2015; Sørtdalen *et al.*, 2020). For *S. melops*, a very small MPA (600 m coastline, comparable to the size of our study area) can provide protection for *S. melops* with moderate rates of spill-over (fishing mortality MPA 6%–9% vs. fished 36%–40%; Halvorsen *et al.*, 2017b). Since movement does not depend on size or sex (this study), a network of such small MPAs may be an effective approach to ensure that intensively fished regions support productive wrasse populations with natural sex ratios and can also potentially benefit fisheries through spill-over. This is something that should be assessed in future studies, along with dispersal and connectivity patterns during the egg and larval stages. Such information can be used to inform the optimal spatial design of an MPA network that also ensures sufficient exchange of recruits between fished areas and neighbouring MPAs (Baetscher *et al.*, 2019; Harrison *et al.*, 2020). On the other hand, adult spill-over rates should be minimal if MPAs are to be used as unfished reference areas for scientific monitoring. This can be achieved if the MPA boundaries align with natural barriers to movement for wrasses (depth and/or larger stretches of sandy bottom), as is the case for the MPA in this study, or if MPAs are considerably larger, as in earlier studies where MPAs have been used to assess the impact of wrasse fisheries (Halvorsen *et al.*, 2017a; four MPAs, mean size 2.8 km²) and (Bourlat *et al.*, 2021; single MPA, 2.6 km²).

These observations on wrasse movement are relevant for another potential challenge in this fishery; the survival of smaller wrasses discarded as bycatch (Halvorsen *et al.*, 2017b; Skiftesvik *et al.*, 2014). In Norway, fishers are obligated to return any bycatch immediately at sea close to shore, although not necessarily at the site of capture (Halvorsen *et al.*, 2020). The extent of this practice has not yet been assessed, but we highlight the potential of local negative consequences, such as higher predation risk for individuals that attempt to home or, in case of settlement in a new location, of increased competition for territories or food. This may be a more serious problem for *C. rupestris*, for which smaller individuals can make up almost half of the catch (Halvorsen *et al.*, 2020). Furthermore, the very high residency of *C. rupestris* implies that it would be practically difficult to release those caught as bycatch within their home range. Homing ability should therefore be further investigated, along with consequences for fitness and social interactions if translocated wrasses establish in a new area.

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AUTHOR CONTRIBUTIONS

K.T.H., A.B.S. and L.A.V. conceived and planned the study. K.T.H., T.L., N.A., A.B.S., C.D. and R.M.B. conducted the fieldwork. K.T.H. and T.L. analysed the data. K.T.H. drafted the first version of the manuscript.

All authors contributed to the write-up and approval of the final version.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of this article.

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