South Arran Marine Protected Area: A baseline study

Jennifer Stark

Y3586960

UNIVERSITY of York

MSc Marine Environmental Management

2015

Word Count: 4972

Disclaimer

I hereby declare that the work and analysis reported in this dissertation is my own, except where otherwise stated (here, and under acknowledgements). Typical input, advice, support and discussion were provided by Howard Wood and Andrew Binnie of the Community of Arran Seabed Trust. I am grateful to Howard for his provision of local knowledge, expertise and experience in surveying the seabed and his help whilst conducting dive surveys. All fieldwork was conducted in accordance with the Scientific Code of Practice, Ethics, and Health and Safety Regulations of the University of York.

I declare this piece of work to be 4972 words, excluding title page, report title, acknowledgements, figures, tables and legends, reference list, appendices and this disclaimer.

Signature…… Date……

Acknowledgements

Firstly, I am extremely grateful to Howard Wood of the Community of Arran Seabed Trust (COAST) for his help, guidance and knowledge during data collection, but most of all for his time and willingness to be my dive buddy. I would like to express my gratitude to Andrew Binnie and Andy Telford of COAST, for allowing me to utilise the facilities of the office and for providing the technology required to collect data. Additionally, I would like to thank the Kilfinan Trust whose funding to COAST made this project possible. I extend my thanks to all the volunteers of COAST, particularly James Henderson for providing accommodation and Angus Robson and Alan Beasley for being so welcoming, and giving up their time to provide vital boat cover. I would like to thank Bryce Stewart and Will Hutchinson for advice on data analysis. Finally, thanks to Andrea Nelson whose proof reading was invaluable.

Abstract

Waters around Arran, Scotland, have a long history of exploitation and as a result stocks of many commercially important species have collapsed. A Marine Protected Area (MPA) has been designated around the south of the island and, prior to implementation of protection, baseline data was collected on species and diversity from twenty dive surveys. Overall 2034 individuals were identified, representing 5 Phyla: Arthropoda, Echinodermata, Chordata, Mollusca and Annelida. Cod (*Gadus morhua*), long-clawed squat lobster (*Munida rugosa*), red cushion star (*Porania pulvillus*), whiting (*Merlangius merlangus*) and harbour crab (*Liocarcinus depurator*) were the top five species. Mean fish density was 38.75 individuals per 100m^2, although no adults of commercially important species were observed. Both diver and video observations were important to identify all individuals, confirmed by analysis of agreement. This analysis provides a baseline to which subsequent surveys can be compared, thus enabling accurate tracking of changes resultant from protection.

Introduction

Home to a rich and diverse mix of marine life (WOR, 2013), oceans and seas across the globe are ideal targets for those wishing to sustain their lives through fishing (Kaiser et al., 2005). This practice, however, is largely unsustainable and due to industrialisation in the early nineteenth century (Pauly et al., 2002), followed by technological revolution in the twentieth century (Roberts, 2007), fishing pressure has significantly increased (Pauly et al., 2002). As a result, 90.1% of global fish stocks are now classed as "overfished" (28.8%) or "fully fished" (61.3%) (FAO, 2014), with significant depletions reported in populations of larger marine organisms (Pauly et al., 2002). This gross exploitation has caused a shift from fisheries which target large predatory fish (Thurstan and Roberts, 2010) to ones that catch smaller, previously spurned species (Pauly et al., 1998, 2002). Pauly et al., (1998) first labelled this phenomenon as "fishing down the food web", claiming that fishermen today now target species our ancestors once considered only good enough for bait (Roberts, 2007). One particular region where this rings true is the Firth of Clyde, Scotland, where management measures aiming to protect the marine environment are set to be implemented in a bid to recover diminished stocks whilst conserving those now targeted. This work reports vital baseline data of a specific location within the Firth of Clyde, with the overall aim of enabling accurate tracking of the area in the future.

The excessive exploitation of fish stocks, particularly over recent decades, has exhausted many marine ecosystems causing them to shift away from previously productive and diverse states (Roberts et al., 2005). Fisheries across the world have altered their target catch to match stock availability; however this comes hand-in-hand

with changes in equipment and techniques (Jennings and Kaiser, 1998). Historical fishing methods, such as nets and hooks, probably had minor adverse effects on the environment, and are arguably less invasive than current techniques (Thurstan and Roberts, 2010). Trawling and dredging are known to reduce biodiversity (Moritz et al., 2015) and although some gear is modified to increase selectivity (Wakefield et al., 2014), these fishing methods are largely indiscriminate as they scrape and scour benthic substrate (Jennings and Kaiser, 1998). Both methods affect more than just their target species, by significantly reducing habitat complexity (Kaiser et al., 2005; McIntyre et al., 2012), with dredging in particular causing marked changes in sediment topography (Currie and Parry, 1996). Three-dimensional (3D) structure afforded by complex seabed habitat is important for many species (Beukers-Stewart and Beukers-Stewart, 2009), as they provide refuge (Cacabelos et al., 2010) whilst also acting as feeding and nursery grounds (Warren et al., 2010). These habitats, such as seagrass meadows and maerl beds, often harbour high densities of commercially exploitable species such as edible crab (*Cancer pagurus*), cod (*Gadus morhua*) and scallops (*Pectinidae* spp.). Protection of species fundamentally depends on conservation of their environment and therefore protection of both flora and fauna is paramount (Thrush and Dayton, 2002).

In a bid to combat habitat destruction and overfishing, many countries across the world now utilise Marine Protected Areas (MPAs) as a management tool (Pande and Gardner, 2009). By contributing to overall protection of critical flora and fauna, MPAs can restore and maintain healthy marine environments when applied as part of a broader ecosystem-based approach to management (Wenzel, 2015). Furthermore, social and

economic opportunities can be offered by effective MPAs, such as education, research and tourism (Wenzel, 2015). Described as "a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (IUCN, 2012), MPAs are not uniform in their needs. Intentions behind establishment are usually dictated by species or habitat being conserved (Sheehan et al., 2013) and protection afforded can scale from "paper parks", where the area is designated but no legislation is enforced, to fully comprehensive No Take Zones (NTZs), where all extraction is prohibited (Slezak, 2014; Wenzel, 2015). Many MPAs across the world have reportedly been successful, most notably: Leigh Marine Reserve, New Zealand (Ballantine, 2014); Mombasa Marine National Park, Kenya (Mangi, 2007); and Georges Bank, Gulf of Maine (Murawski et al., 2000). Data gathered on these areas have shown increases in biomass, abundance, density and size of many exploited species (Gell and Roberts, 2003). Furthermore, studies have provided evidence that reserves promote habitat recovery, which in turn boosts eco-tourism, thus increasing economic gain (Sala et al., 2013). MPA success contributed to the decision by the World Parks Congress to pledge 10% of the world's oceans to be formally protected by 2020 (Juffe-Bignoli et al., 2014). Currently, 3.4% of oceans are protected; however MPA coverage is rising (Howarth et al., 2011; Juffe-Bignoli et al., 2014).

Establishment of MPAs in the United Kingdom (UK) is still in its infancy (Howarth et al., 2015a) with only 16% of national waters having some level of protection (JNCC, 2015a). Party to the European Union Habitats Directive, the focus of the UK Government is to achieve productive, healthy, clean, and biologically diverse oceans and seas (DEFRA and

MMO, 2015). This will be achieved partly through implementation of the Marine and Coastal Access Act (2009) and the Marine Act (Northern Ireland) (2013). These Acts allow for the creation of Marine Conservation Zones (MCZs), in England and Wales, and Northern Ireland, respectively, which protect nationally important marine habitats, geology and wildlife (JNCC, 2015b). In Scotland, a network of 180 designated MPAs is comprised of Nature Conservation MPAs (NCMPAs), Special Areas of Conservation (SACs), Special Protection Areas (SPAs), and Sites of Special Scientific Interest (SSSIs). This network was reinforced in 2014 when the Scottish Government utilised powers, granted through the Marine and Coastal Access Act (2009) and the Marine (Scotland) Act (2010), to officially designate 30 NCMPAs (JNCC, 2015c).

Importantly, some areas which gained protection are situated within the Firth of Clyde, an inlet of sea off the west coast of Scotland (Thurstan and Roberts, 2010). Fishing has been a key part of the Clyde's economy for centuries (Tivy, 1986) and, as a result, the area has a long history of exploitation (Thurstan and Roberts, 2010). The Firth of Clyde once boasted fertile fisheries of cod, herring (*Clupea harengus*) and haddock (*Melanogrammus aeglefinus*) (Thurstan and Roberts, 2010). However these commercially viable populations no longer exist and fisheries have been forced to shift their catch towards bottom-dwelling invertebrate species, such as the Norway lobster (*Nephrops norvegicus*), and king (*Pecten maximus*) and queen (*Aequipecten opercularis*) scallops (Pauly et al., 1998, 2002; Thurstan and Roberts, 2010). The trawling and dredging of these respective species has caused extensive modification of many sensitive habitats in the Clyde, particularly fragile maerl beds and seagrass meadows (Jennings and Kaiser, 1998). Destruction caused by trawling increased in 1984, when

the byelaw prohibiting the activity within 3 nautical miles (nm) of low-water was lifted (Thurstan and Roberts, 2010). Decline of the Clyde fisheries was witnessed by many, but two local divers from the Isle of Arran decided to take action, by founding the Community of Arran Seabed Trust (COAST).

Established in 1995, gaining charity status in 2011, COAST is a community-led marine conservation foundation, whose vision is to restore the waters around Arran and the rest of the Clyde to their once productive states (COAST, 2012). Thirteen years of campaigning by COAST led to the creation, in 2008, of a 2.67km^2 NTZ in Lamlash Bay, situated in the south-east of Arran (Howarth et al., 2015b). This closure has been studied in detail, with reports indicating a greater number of species, individuals and overall biodiversity within the NTZ (Howarth, 2011, 2012; Howarth et al., 2015). Due to the success of Lamlash Bay NTZ, and in a bid to protect a greater range of habitats, COAST submitted a 3rd Party proposal to designate the South Arran NCMPA to the Scottish Government in 2012 (COAST, 2014). The area was officially designated in 2014 (Scottish Government, 2014a) with specific aims of protecting and restoring seagrass and maerl beds, along with other sensitive habitats and species that have functional importance as nursery and breeding grounds (COAST, 2014). COAST argued that restoration of habitats and species required exclusion of all destructive fishing practices (COAST, 2014), however the exact level and scope of protection that will be afforded by the South Arran MPA is yet to be confirmed (Scottish Government, 2015a). The most recent talks closed in August 2015, with legislation aimed to be implemented in November 2015. However, on September 9th, the Scottish Government announced a delay in enforcing fishing restrictions, in order for opponents of MPAs to voice their

concerns (BBC, 2015). Although enforcement is essential, the delay in regulations has allowed vital baseline data to be gathered prior to implementation of official protection.

To effectively demonstrate the ecological effects of MPAs (Jones et al., 1992; Pande and Gardner, 2009), information regarding biota before reserve establishment is essential (Edgar and Barrett, 1999). Without this baseline data (i.e. data gathered from an area ahead of change; Kingsford and Battershill, 1998), there is no "before" evidence to which future data, gathered once formal regulations are enforced, can be compared (Pande and Gardner, 2009). This study is therefore imperative, as data collected after designation, but preceding legislation implementation, has been analysed and will provide a vital baseline of the quality of benthic marine life around the South Arran MPA.

This investigation will quantify the abundance and distribution of mobile species around the South Arran MPA. Specifically, the following was examined: (1) distribution and abundance of species within 5 select phyla; (2) survey site species diversity; and (3) exploratory analysis to assess the appropriateness of methods employed.

Methodology

Study Site

The Isle of Arran is the largest island, with an area of 432km^2, in the Firth of Clyde, situated off the west coast of Scotland (Axelsson et al., 2009). In 2014, the waters surrounding the south of the island were officially designated as an MPA (Scottish Government, 2014a). Boundaries of the 282km^2 area run from Drumadoon Point on the west, at a 3nm range, round to Corriegills Point on the east (Fig. 1), inclusive of Lamlash Bay NTZ (COAST, 2012). The MPA encompasses a variety of different substrates (Howarth et al., 2015a). Studies have reported that seabed habitats in the shallows range from maerl beds and macroalgal communities to large seagrass meadows (Duncan, 2003), whereas the majority of substrate found at the deeper, outer edges of the MPA is reportedly burrowed mud (Scottish Government, 2014b).

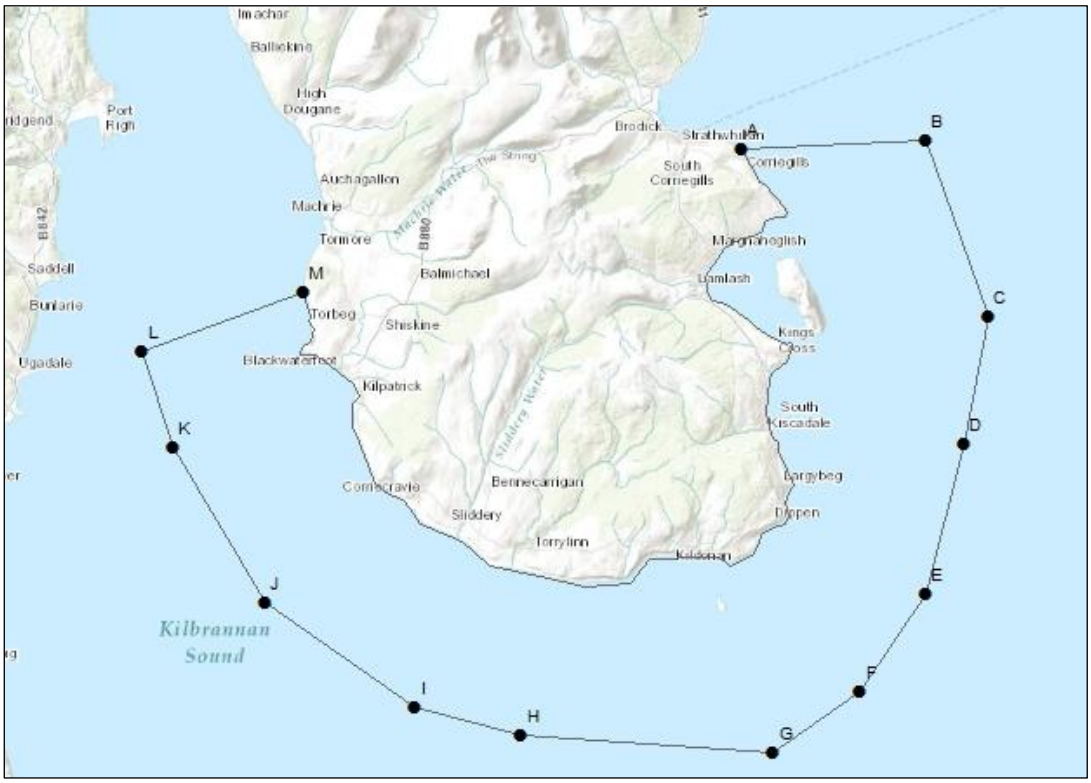

Figure 1: Map showing the south Isle of Arran, with the boundaries of South Arran MPA marked. Co-ordinates of boundaries markers found in Appendix 1. Map generated using ArcMap and ArcGIS.

Dive Surveys

Throughout July and August 2015, dive surveys were conducted within the South Arran MPA from the 6.5m commercially coded Rigid Inflatable Boat (RIB), Deep Blue. Twenty 2 x 50m transects were completed (Fig. 2), in depths no greater than 26m so recreational SCUBA diving decompression limits would not be exceeded (Howarth, 2012). Sites were selected at random, however they were chosen to be representative of the inshore habitats of the MPA and also depended on weather and wind conditions of the day.

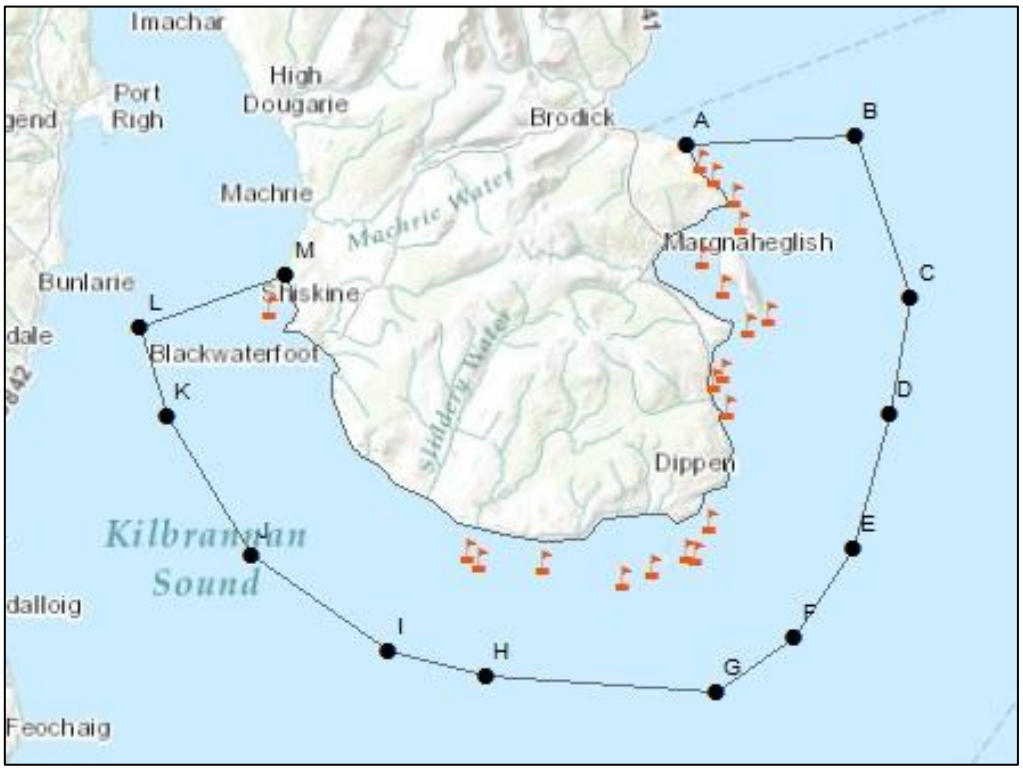

Figure 2: Map showing the south Isle of Arran, with the boundaries of South Arran MPA marked. The red flags indicate the twenty survey sites from this study Map generated using ArcMap and ArcGIS.

Each transect was marked out using a lightly leaded 50m line, which was lowered carefully down to the seabed from the boat. The line was set along known depth contours, to prevent the transect depth differing by no more than 3 meters. The far end was weighted down by a 3kg plough anchor and marked at the surface by a buoy. The

near end of the line was held in place by a 2kg weight and was attached to the boat so it was kept taught without dragging on the seabed. If winds or tides did not permit the boat to sit on point, it would be unattached with the start of the transect instead being marked by a surface buoy.

All surveys were conducted by two experienced research divers whose role remained the same throughout, to ensure consistency and exclude inter-observer bias. Diver 1 utilised a Canon EOS7D camera with a Tokina Fisheye 10-17 lens, housed inside a Nauticam waterproof housing. Attached to this were two subsea lights and two LED-strobe lights. This whole unit was mounted to the centre of a 1m wide stainless steel bar, the two ends of which had a waterproof laser pointer attached. Diver 2 carried an underwater slate and pencil, upon which common species most likely to occur were written, to aid recording (Fig. 3).

Figure 3: Picture showing diving equipment. Camera, lights and strobes are circled in red, lasers in green and the slate is highlighted by a pink rectangle. Photograph courtesy of COAST.

After descending to the seabed, Diver 1 would set the camera's white balance according to the depth and light, whilst Diver 2 recorded the depth and substrate at the start of the transect. Divers proceeded simultaneously along the right hand side of the line, carrying out their respective duties. Diver 1 video recorded the 50m transect, pausing every 5m to take a photograph, saved in both JPEG and RAW. Consistency of diver position and video recording was maintained by ensuring the left hand laser was focused upon the transect line, whilst the right hand one marked exactly a meter to the right. Diver 2 was positioned on the right hand side of Diver 1, just out of camera shot. Whilst swimming, Diver 2 would tally all target species that passed through the one meter marker of the lasers (Fig. 4). Upon completion of the fifty meters, which took up to 11 minutes to complete depending on current strength, the substrate and depth would be re-recorded. Divers returned on the opposite site of the line whilst utilising the same technique, thus covering a 100m^2 survey area per transect.

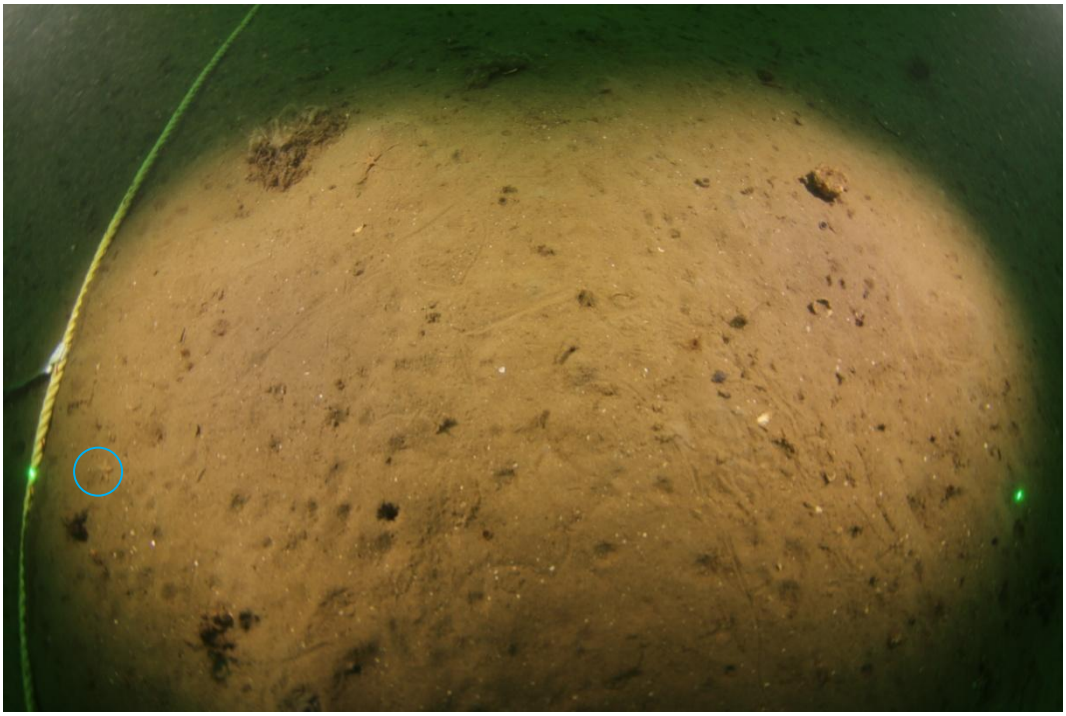

Figure 4: Snapshot from a survey dive. The green dots are the laser pointers. The left hand one is aligned on the transect line, whilst the other marks a meter to the right. As an example as to what would be counted, a sand star has been highlighted in blue. Photograph courtesy of COAST.

Upon completion of each dive, the co-ordinates of each end of the transect were recorded. This was conducted by pulling in the lightly buoyed line, attached to relevant end, until it was vertical so latitudes and longitudes could be read (Appendix. 2) using the boat's GPS device. The leaded line was then carefully pulled up, with the RIB moving slowly forward using motor propulsion to ensure no dragging on the seabed occurred from either the leaded line or the light anchors.

Video Analysis

All videos recorded on each dive were uploaded to a computer and analysed frame-by-frame. Species identified belonged to one of five previously selected taxonomic phyla; Arthropoda, Echinodermata, Chordata, Mollusca and Annelida. All individuals viewed in the survey area were counted, including any with more than 50% of its area within the markers. All organisms were identified to the most detailed taxonomic level possible. The number of each species was then compared to diver observations to ensure accuracy of count, with all observed species counted in analysis regardless of identification method. Seasearch survey forms were filled out (Appendix. 3) for each transect, with the SACFOR (Superabundant, Abundant, Common, Frequent, Occasional, Rare) scale used to estimate species abundance giving an indication as to what to expect in each survey.

Data Analysis

Species composition

The number of species within a given phylum were identified, and individuals of each were counted across all transects. The total number of individuals per species was

recorded from which the mean of each species per 100m^2 could be established. This also allowed for cumulative species curves to be plotted, to illustrate the number of species found at each transect site with regard to their contribution to site richness.

Simpson's Diversity Index

To describe the diversity of a given transect, whilst accounting for differences in species richness and evenness across sites, Simpson's Diversity Index (SDI) was calculated for each. First, Simpson's Index (D) was calculated using the equation from Figure 5a, before SDI (commonly known as Inverse Simpson's) was calculated using the equation in Figure 5b (Magurran, 1988). The SDI was used in preference to SI as interpretation is easier, with higher SDI's equating to higher diversity (Magurran, 1988).

(a) $$D = \frac{\sum n_i(n_i - 1)}{N(N-1)}$$ (b) $$SDI = 1 - D$$

Figure 5: (a) Simpson's Index (D), where n_i = number of individuals in the *i*th species and N = the total number of individuals; (b) Simpson's Diversity Index (SDI).

Phyla distributions, populations and densities

Analysis of each phylum was undertaken to examine the following: (1) the number of transects in which each were present; (2) number of species per phylum; (3) number of individuals per phylum; (4) mean density of phylum across transects; and (5) further observations.

Analysis of Tools

The method used to count individual species in this study is highly sensitive, with data gathered by a diver and subsequent video analysis. During data analysis it became apparent that some individuals were detected on the dive but not the video, and vice versa.

To assess the appropriateness of methods used in this study, an exploratory analysis was conducted to determine the extent of agreement between diver and video identification. This was conducted thorough calculation of Cohen's kappa (Cohen, 1960), a measure of agreement beyond chance alone (Bland, 2008). Kappa values can be interpreted using guidelines from Landis and Koch (1977), cited in Bland (2008) (Table. 1). All analyses were conducted using Microsoft Office Excel 2010.

Table 1: Level of agreement determined by given value of kappa.

Value of kappa	**Strength of agreement**
< 0.20	Poor
0.21 – 0.40	Fair
0.41 – 0.60	Moderate
0.61 – 0.80	Good
0.81 – 1.00	Very Good

Results

Species composition

In the area surveyed, 41 species were identified across 5 different phyla: Arthropoda (9), Echinodermata (10), Chordata (14), Mollusca (7), and Annelida (1) (Table. 2). A total of 2034 individuals were recorded, of which 62.6% were accounted for by the top five dominant species. These were determined, in descending order, to be: cod (*Gadus morhua*), long-clawed squat lobster (*Munida rugosa*), red cushion star (*Porania pulvillus*), whiting (*Merlangius merlangus*) and harbour crab (*Liocarcinus depurator*).

Each transect was dominated by just one or two given species, as indicated by the steep gradients of the cumulative species curves (Fig. 6). The extremely steep incline observed at Whiting Bay Jetty (Site 3 – a large seagrass meadow) is accounted for by juvenile cod, which comprise 88.32% (n=137) of the total individuals observed at this site. Organisms here were represented by 10 species, meaning that it was not the least rich site; Lagg Burn (Site 5) was the least species rich (n=7). Site 20, East of the NTZ Navigation Buoy, just outwith Lamlash Bay NTZ, was found to have the greatest species richness, containing 48.78% (n=20) of the total number of species identified over all transects.

Table 2: Mean density of individuals of each species observed across all 20 transects. Density is reported per 100 m^2 (± Standard Error of each species). Where cells contain a '-' no common name was available. When only juveniles of a given species were observed, entries are marked with '*'.

Taxonomic Phyla	Latin Name	Common Name	Overall Species Density (per 100m^2)
Arthropoda	*Munida rugosa*	Long-clawed squat lobster	13 (1.47)
	Liocarcinus depurator	Harbour crab	6.4 (0.50)
	Pagurus bernhardus	Common hermit crab	4 (0.42)
	Inachus spp.	Small spider crab	2.85 (0.39)
	Cancer pagurus	Edible crab	1.05 (0.25)
	Porcellana platycheles	Broad-clawed porcelain crab	0.85 (0.60)
	Hyas araneus	Great spider crab	0.8 (0.56)
	Necora puber	Velvet swimmer crab	0.7 (0.42)
	Pagurus prideaux	Anemone hermit crab	0.3 (0.33)
Echinodermata	*Porania pulvillus*	Red cushion star	11.6 (0.96)
	Henricia oculata	Bloody henry starfish	2.05 (0.46)
	Asterias rubens	Common starfish	1.75 (0.27)
	Luidia ciliaris	Seven armed starfish	1.65 (0.52)
	Marthasterias glacialis	Spiny starfish	1.45 (0.47)
	Echinus esculentus	Edible sea urchin	1.3 (0.59)
	Astropecten irregularis	Sand sea star	1.05 (0.36)
	Crossaster papposus	Common sunstar	0.95 (0.30)
	Ophiura ophiura	Common sand brittle star	0.15 (0.39)
	Solaster endeca	Purple sunstar	0.1 (0.32)
Chordata	*Gadus morhua**	Cod*	22.6 (1.30)
	*Merlangius merlangus**	Whiting*	10.1 (1.34)
	*Melanogrammus aeglefinus**	Haddock*	2.95 (0.63)
	Ctenolabrus rupestris	Goldsinny wrasse	0.75 (0.59)
	Gobiusculus flavescens	Two-spotted goby	0.65 (0.69)
	Callionymus lyra	Dragonet	0.45 (0.20)
	Pleuronectes platessa	Plaice	0.4 (0.21)
	Pollachius spp.	Pollack	0.25 (0.50)
	Pholis gunnellus	Butterfish	0.2 (0.26)
	Syngnathus acus	Greater pipefish	0.15 (0.28)
	Microstomus kitt	Lemon sole	0.1 (0.32)
	Taurulus bubalis	Long spined scorpion fish	0.05 (0.22)
	Hyperoplus lanceolatus	Great sand eel	0.05 (0.22)
	Scyliorhinus canicula	Dogfish	0.05 (0.22)
Mollusca	*Aequipecten opercularis*	Queen scallop	5.65 (0.63)
	Pecten maximus	King scallop	3.6 (0.68)
	Janolus cristatus	-	0.1 (0.22)
	Eledone cirrhosa	Curled octopus	0.05 (0.22)
	Polycera faeroensis	-	0.05 (0.22)
	Coryphella browni	-	0.05 (0.22)
	Pleurobranchus membranaceus	Highland dancer	0.05 (0.22)
Annelida	*Myxicola infundibulum*	Eyelash worm	1.4 (0.80)

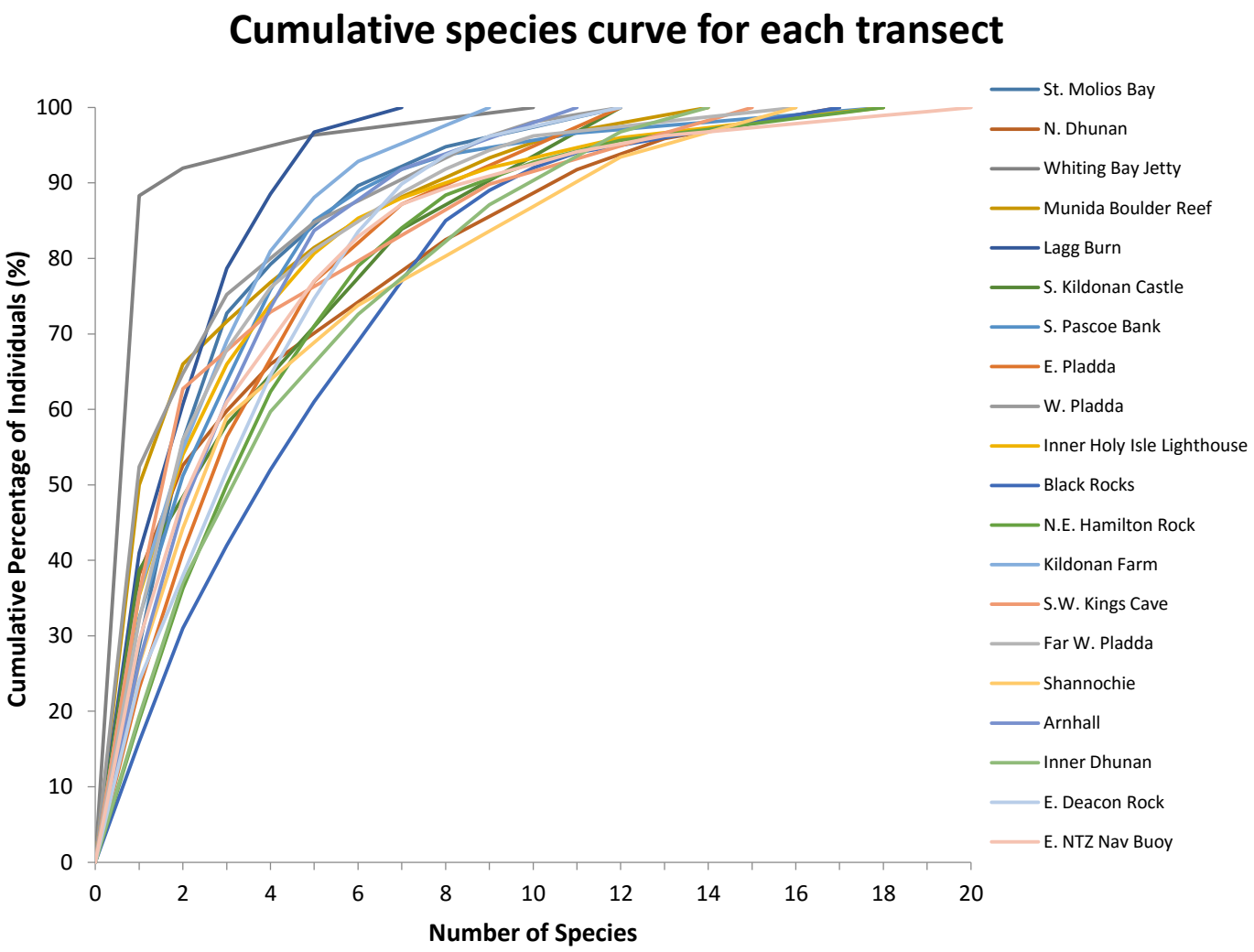

Figure 6: Cumulative species curves for each individual transect conducted at each survey site around the South Arran MPA.

Simpson's Diversity Index

The Simpson's Diversity Index (SDI) of each transect site was calculated, giving an overall mean SDI across all sites of 0.80 (where higher values represent greater diversity). Thus, the overall diversity of all twenty transects was found to be quite high. As shown in Figure 7, all transects had an SDI of 0.70 or more, apart from Whiting Bay Jetty (SDI=0.22). Notably Black Rocks and Inner Dhunan had extremely high SDI's at 0.91 and 0.90, respectively.

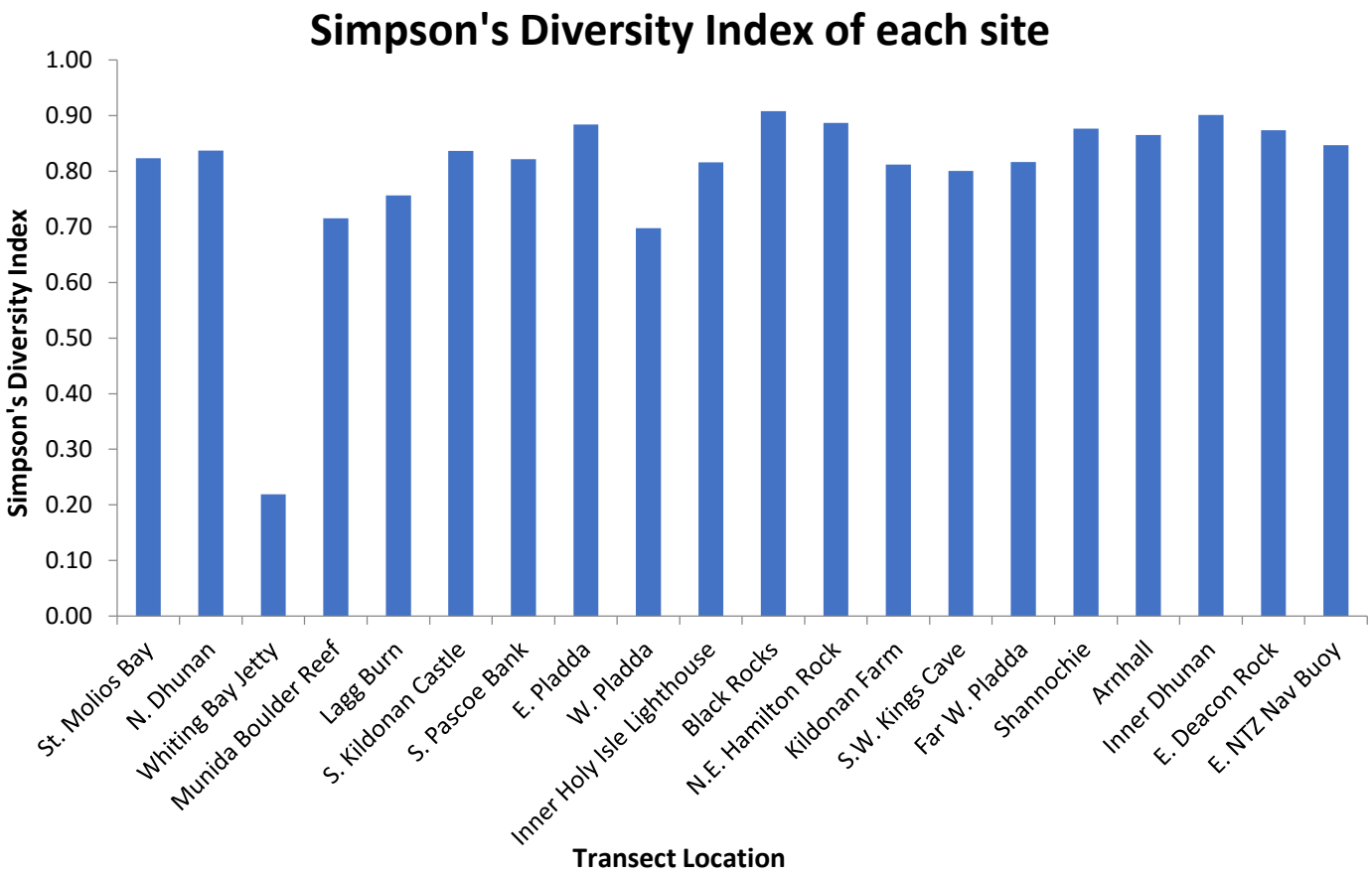

Figure 7: Simpson's Diversity Index of each transect site conducted around the South Arran MPA. The higher the value, the more diverse the site.

Chordata abundance

Chordata (fish) were observed in all twenty transects. In total 14 different fish species were represented by 775 individuals (Table. 3). Across all 20 transects a mean density of 38.75 (±SE 1.31) individuals per 100m^2 was found. As shown in Figure 8, transect site 3, Whiting Bay Jetty, showed the highest fish density of 130 individuals per 100m^2 (over 5 species). In comparison site 6, South Kildonan Castle, was found to have the lowest fish density of all twenty transects, at 1 individual per 100m^2.

Table 3: Total number of individuals of each fish species recorded across all 20 transects, with juveniles denoted by '*'. The contribution of each species to the overall fish abundance (n = 775) is given as a percentage, as is the number of transects in which each species was observed.

Species	Total Number of Individuals Observed	Percentage of total fish population (%)	Proportion of transects in which observed (%)
Butterfish (*Pholis gunnellus*)	4	0.52	15
Cod* (*Gadus morhua*)	452	58.32	75
Dogfish (*Scyliorhinus canicula*)	1	0.13	5
Dragonet (*Callionymus lyra*)	9	1.16	40
Goldsinny wrasse (*Ctenolabrus rupestris*)	15	1.94	20
Great sand eel (*Hyperoplus lanceolatus*)	1	0.13	5
Greater pipefish (*Syngnathus acus*)	3	0.39	10
Haddock* (*Melanogrammus aeglefinus*)	59	7.61	45
Lemon sole (*Microstomus kitt*)	2	0.26	5
Long spined scorpion fish (*Taurulus bubalis*)	1	0.13	5
Plaice (*Pleuronectes platessa*)	8	1.03	35
Pollack (*Pollachius spp.*)	5	0.65	5
Two-spotted goby (*Gobiusculus flavescens*)	13	1.68	10
Whiting* (*Merlangius merlangus*)	202	26.06	35

Cod were seen in 75% of transects, haddock in 45% and whiting in 35% of transects. Respectively, these accounted for 58.32%, 7.61% and 26.06% of the total number of individual fish. Therefore, 91.99 % of all fish observed were juveniles of commercially important species.

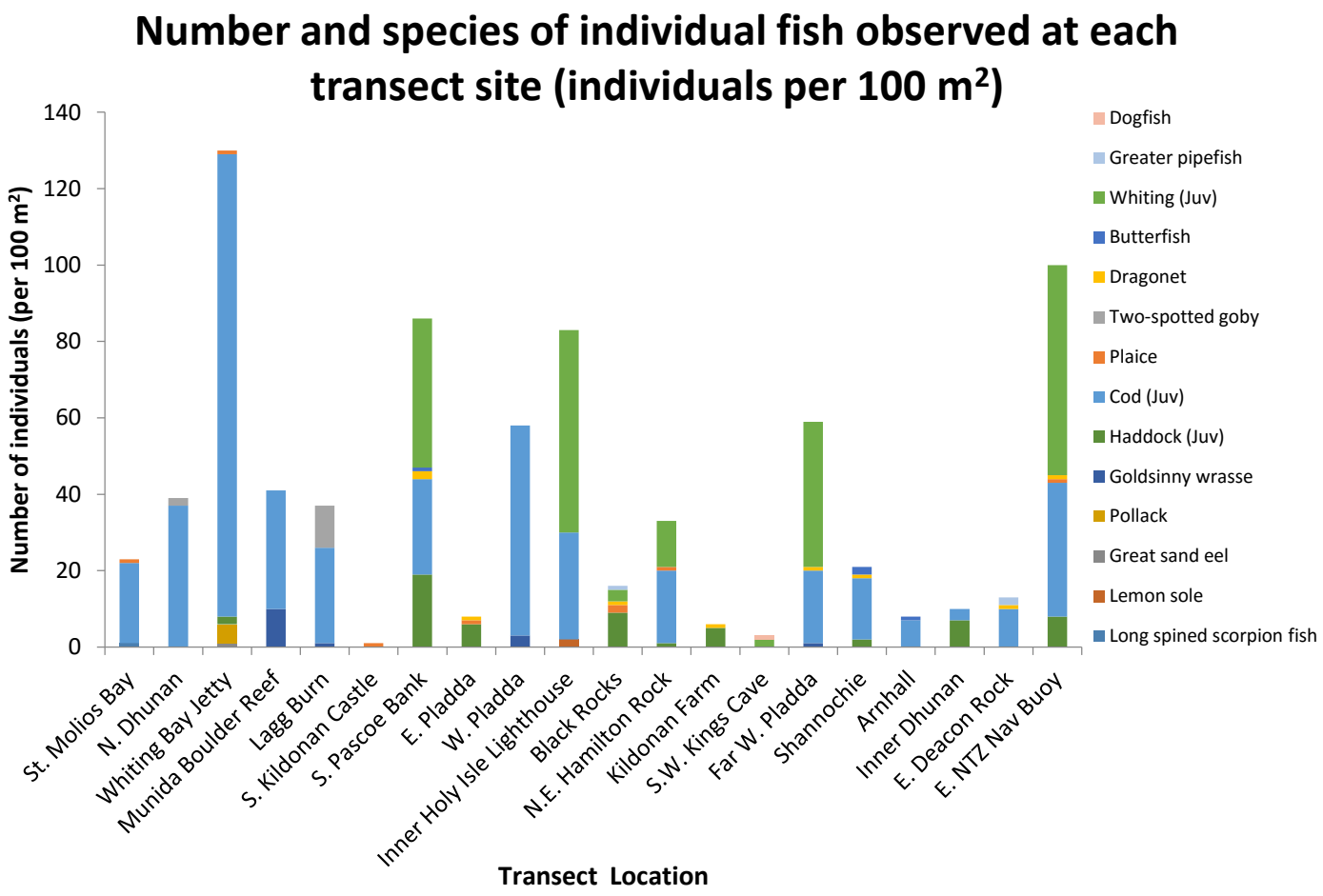

Figure 8: Total number of individual fish, and representative species observed at each transect site around the South Arran MPA.

Arthropoda abundance

Arthropoda (crustaceans) were observed in 19 of the 20 transects. In total 9 different species were represented by 599 individuals, with a mean density of 3.33 (±SE 1.01) individuals per 100m^2. Long-clawed squat lobsters were counted in 63.16% of transects in which species were observed, edible crabs in 57.89% and velvet swimmer crabs in 31.58%. Respectively these accounted for 43.41%, 3.51% and 2.34% of the total number of individual crustaceans, thus 49.26% of individuals were from commercially exploited species.

Echinodermata abundance

Echinoderms were present in all twenty transects. In total 10 different species were represented by 441 individuals, with a mean density of 2.21 (±SE 0.77) individuals per 100m^2. Red cushion stars accounted for the majority (52.61%) of individuals observed and was present in 80% of transects.

Mollusca abundance

Molluscs were present in 17 of 20 transects. In total 7 different species were represented by 191 individuals, with a mean density of 9.55 (±SE 0.76) individuals per 100m^2. King scallops were seen in 88.24% of transects where molluscs were observed, as were queen scallops, accounting for 37.7% and 59.16% of all individuals, respectively. Therefore, commercially fished molluscs accounted for 96.86% of this phylum.

Annelida abundance

Only one Annelida (worm) was observed in all surveys, and was present in 30% of transects conducted. Overall 28 individuals were counted, with an average of 1.4 (±SE 0.80) individuals per 100m^2.

Analysis of Tools

During analysis it became apparent that some species were detected on the dive but not the video, and vice versa. In order to illustrate this, Cohen's kappa was calculated for three species: long-clawed squat lobster, edible crab and king scallop (Fig. 9). These

tables show that a significant number of individuals were seen on video analysis which had been missed by the diver: 18/57 (32%) edible crab, 11/72 (15%) king scallop, and 153/260 (59%) long-clawed squat lobster. In addition, the diver could record species not seen in the video; 1/72 king scallops and 3/4 (75%) of nudibranchs (kappa not calculated due to small sample size). When methods were assessed for edible crabs and king scallop, moderate agreement was determined. However, for long-clawed squat lobsters, fair agreement was reported.

Edible crab		Video		
		Yes	No	Total
Diver	Yes	39	0	39
	No	18	0	18
	Total	57	0	57

kappa = 0.41

King scallop		Video		
		Yes	No	Total
Diver	Yes	60	1	61
	No	11	0	11
	Total	71	1	72

kappa = 0.46

Long-clawed squat lobster		Video		
		Yes	No	Total
Diver	Yes	107	0	107
	No	153	0	153
	Total	260	1	260

kappa = 0.29

Figure 9: Tables of observations of Edible crab, King scallop and Long-clawed squat lobster. Values show the number of individuals identified via each method, along with the calculated kappa.

Discussion

Twenty dive surveys were completed at randomly selected inshore sites around the South Arran MPA in July and August 2015. Video recordings of each transect were analysed to identify mobile species present in each $100m^2$, providing an estimate of species composition within the MPA.

In contrast to a similar study analysing surveys conducted in 2014, where 30 species were observed (Hutchinson, 2015), this study found an overall species richness of 41 across the same five phyla. Furthermore, 2034 individuals were identified in this investigation, whereas Hutchinson (2015) counted 1372. These differences could be due to a number of reasons. Firstly, the 2015 transects were conducted at different sites around the MPA, thus direct comparisons cannot be made. Discrepancies between years can be accounted for by spatial heterogeneity, commonly reported in regard to the marine environment (Benedetti-Cecchi et al., 2003; Jelinski. 2015). Furthermore, the methodology between studies varied slightly as Hutchinson (2015) relied entirely on desk-based video analysis. Ultimately, this could mean that some inconspicuous species which may be noticed by a diver could have been missed. This point was addressed in this investigation through exploratory analysis, thought to be the first of its kind regarding these methods, quantifying agreement between diver and video identification of species. Kappa values for three exploratory analyses were moderate to fair, indicating that both diver and video analysis is required in order to ensure collection of all important data (if agreement was very high this might allow replacement of one method, say diver tally recording, with video analysis). It is therefore recommended that future studies employ this method in order to ensure a fair comparison between "before" and "after" management data, unless future researchers can adjust for the high

sensitivity of this technique. Both spatial heterogeneity and limited data mean that it is not safe to conclude that diversity or richness has significantly increased since 2014.

The importance of continued monitoring and adequate protection of the South Arran MPA should not be underestimated. The Firth of Clyde is well known for the substantial decline of once abundant fish stocks, a consequence of intensive fishing, by-catch and habitat degradation (Thurstan and Roberts, 2010; Heath and Speirs, 2012.). A positive finding was the identification of many juvenile fish species of commercial importance in many of the transects; cod, haddock and whiting. The findings from this study support evidence, provided by other authors (Cullen-Unsworth et al, 2014; Lilley and Unsworth, 2014), that cod have enhanced growth and survival within seagrass habitats, with 121 individuals counted in one seagrass habitat alone. Their presence, along with other juvenile species, reiterates the need for protection of potential nursery habitats. Seagrass meadows and maerl beds around the south of Arran are highly degraded, with Howarth et al., (2015a) reporting at least 90% of maerl within the Lamlash Bay NTZ as dead. Despite the presence of juveniles in many of the transects, no adults of these species were observed. This highlights that it is not just nursery grounds which need protected, but also structurally complex habitats that support larger, more mature individuals (Warren et al., 2010). Planned management measures in the South Arran MPA may exclude dredging and limit trawling, with an aim of exerting bottom-up control, through conservation of habitat, then species (Scottish Government, 2014c). It is hoped marine ecosystems will recover, thus returning to their once productive and diverse states that will benefit all invested (Thurstan and Roberts, 2010).

Data presented here report species diversity of sites to be relatively high. Notably,

Whiting Bay Jetty had a lower SDI than all other sites (0.22). This area is known for having some of the largest seagrass beds in the Firth of Clyde (SNH, 2015) and, due to the 3D habitat this affords, a high SDI was expected. The unexpected result was likely due to one species, juvenile cod, accounting for 88.32% of the total individuals in the transect, whereas in all others it took at least 4 species to reach this level. With regards to species richness one site, Lagg Burn, is noteworthy. This site had a low number of species (n=7), however this may be due to the structural complexity of the site. It was composed largely of boulders and red tufting algae, providing refuge for many study species, meaning counts are likely to be underestimates. Interestingly, the most species rich site was located right next to the boundary of the Lamlash Bay NTZ. Many MPA success stories indicate increases in stock biomass, abundance and fecundity (Gell and Roberts, 2003), resulting in organisms moving out-with protection as areas reach carrying capacity. This so called “spillover” has been reported by studies looking specifically at Lamlash Bay NTZ (Axelsson et al., 2009), and results presented here are consistent with such.

Despite this report providing key baseline data, limitations in study design, technique and application must be addressed. Research was conducted over a relatively short time scale, and therefore only select analysis could be done. To provide a better baseline of the South Arran MPA, it is recommended that sessile species, flora and substrate be recorded. Due to the provision of video recordings from this study, this could be done with no additional data collection by future researchers, to expand the baseline description of the MPA. The sensitivity of using two observation methods has been discussed and further analysis could be carried out on this to determine agreement in observations for each species, which was beyond the scope of this study.

Further restrictions of this study included the lack of repeat surveys, which would provide a better baseline of each site. Furthermore surveys of control areas outwith the MPA boundaries were not undertaken, with this unavoidable omission being due to the time available for surveying and the prioritising of the MPA. Control area evidence would measure the recovery of the area and spillover, as would evidence from burrowed mud near the boundary lines, and data from the west of the MPA. Finally, it is unknown how diver presence affected the abundance of organisms observed in the transects. Some organisms visibly crossed over the survey line as divers passed, leading to the risk that some may have been counted upon the return thus leading to artificially enhanced baselines. Mobile species may also have left the transect to avoid divers, leading to under-recording of species, although presumably this would occur in all subsequent observations.

Conclusion

Following the designation of a network of 30 MPAs last year, the future for Scottish waters is promising. However, until management measures are enforced, this is not ensured. The immediate future regarding the legislation implementation in the South Arran MPA is uncertain. If management measures are enforced as planned this coming November, prohibiting scallop dredging throughout and limiting trawling activity to the outer areas of the MPA (Scottish Government, 2015b), then results presented here would provide the "before" data to which the "after" can be compared, thus providing scientific evidence that any observed changes are a result of legislation being enforced. Alternatively, there may be further consultations and a delay in management implementation of protection, providing time for more baseline data to be collected and analysed. The latter scenario, however, leaves sensitive habitats and vulnerable stocks within the MPA at risk from destructive fishing for longer.

Regardless of the immediate outcome, on-going monitoring of the South Arran MPA is imperative to its success, and possible island-round expansion. It is hoped that the baseline work reported here, and in Hutchinson (2015), will allow vital comparisons to be drawn in future studies of the area. Continued monitoring will provide scientific backing to any observed changes, thus allowing informed decisions regarding regulations. Although impending legislation is a step in the right direction until all destructive fishing methods are prohibited, the "P" remains absent from the South Arran MPA.

References

Axelsson, M., Dewey, S., Plastow, L., and Doran, J. (2009). *Mapping of marine habitats and species within the Community Marine Conservation Area at Lamlash Bay*. Scottish Natural Heritage Commissioned Report No. 346.

Ballantine, B. (2014). Fifty years on: Lessons from marine reserves in New Zealand and principles for a worldwide network. *Biological Conservation*, 176, 297-307.

BBC. (2015). *Short delay to enforcing Marine Protected Area rules*. [Online]. Available at: www.bbc.co.uk/news/uk-scotland-highlands-islands-34202480 [Accessed 9 September 2015].

Benedetti-Cecchi, L., Bertocci, I., Micheli, F., Maggi, E., Fosella, T., and Vaselli, S. (2003). Implications of spatial heterogeneity for management of marine protected areas (MPAs): examples from assemblages of rocky coasts in the northwest Mediterranean. *Marine Environmental Research*, 55, 429-458.

Beukers-Stewart, B.D., and Beukers-Stewart, J.S. (2009). *Principles for the Management of Inshore Scallop Fisheries around the United Kingdom*. Report to Natural England, Scottish Natural Heritage and Countryside Council for Wales. Marine Ecosystem Management Report no. 1, University of York, 58pp. [Online]. Available at: http://www.researchgate.net/profile/Bryce_Beukers-Stewart/publication/260898429_Principles_for_the_Management_of_Inshore_Scallop_Fisheries_around_the_United_Kingdom/links/0f3175329cdc26836a000000.pdf [Accessed 12 September 2015].

Bland, M. (2008). Cohen's kappa. [Online]. Available at: https://www-users.york.ac.uk/~mb55/msc/clinimet/week4/kappash2.pdf [Accessed15 September 2015].

Cacabelos, E., Olabarria, C., Incera, M., and Troncoso, J.S. (2010). Effects of habitat structure and tidal height on epifaunal assemblages associated with macroalgae. *Estuarine, Coastal and Shelf Science*, 89, 43-52.

Cohen, J. (1960). A coefficient of agreement for nominal scales. *Educational and Psychological Measurement*, 20, 37-46.

Community of Arran Seabed Trust (COAST). (2012). *Proposed Marine Protected Area, South Coast of Arran*. [Online]. Available at: http://www.arrancoast.com/website_pdf/COAST_Arran_South_MPA_proposal_9May_FinalP.pdf [Accessed 20 July 2015].

Community of Arran Seabed Trust (COAST). (2014). *South Arran Marine Protected Area*. [Online]. Available at: http://www.arrancoast.com/campaigns/south-arran-marine-protected-area [Accessed 18 July 2015].

Cullen-Unsworth, L.C., Nordlund, L.M., Paddock, J., Baker, S., McKenzie, L.J., and Unsworth R.K.F. (2014). Seagrass meadows globally as a coupled social-ecological system: Implications for human wellbeing. Marine Pollution Bulletin, 83(2), 387-397.

Currie, D., and Parry, G. (1996). Effects of scallop dredging on a soft sediment community, a large-scale experimental study. *Marine Ecology Progress Series*, 134, 131-150.

Department for Environment Food and Rural Affairs (DEFRA)., and Marine Management Organistation (MMO). (2015). *2010 to 2015 government policy: marine environment*. [Online]. Available at: https://www.gov.uk/government/publications/2010-to-2015-government-policy-marine-environment/2010-to-2015-government-policy-marine-environment [Accessed 1 September 2015].

Duncan, C. (2003). *Lamlash Bay Seasearch. Interim report by the Marine Conservation Society to Scottish Natural Heritage*. [Online]. Available at: http://www.arrancoast.com/science/arran_seasearch_report_2003.pdf [Accessed 2 September 2015].

Edgar, G.J., and Barrett, N.S. (1999). Effects of the declaration of marine reserves on Tasmanian reef fishers, invertebrates and plants. *Journal of Experimental Marine Biology and Ecology*, 242, 107-144.

Food and Agriculture Organization of the United Nations (FAO). (2014). The *State of World Fisheries and Aquaculture – Opportunities and Challenges*. [Online]. Available at: http://www.fao.org/3/a-i3720e.pdf [Accessed 2 September 2015].

Gell, F.R., and Roberts, C.M. (2003). Benefits beyond boundaries: the fishery effects of marine reserves. *TRENDS in Ecology and Evolution*, 18(9), 448-455.

Heath, M.R., and Speirs, D.C. (2012). Changes in species diversity and size composition in the Firth of Clyde demersal fish community (1927-2009). *Proceedings of the Royal Society B*, 279, 543-552.

Howarth, L.M. (2012). *Exploring the fishery and ecological effects of Lamlash Bay No-Take Zone – Science report for COAST April 2012.* [Online]. Available at: http://arrancoast.com/website_pdf/Exploring_the_fishery_and_ecological_effects_of_Lamlash_Bay_No_Take_Zone_Howarth_Uni_York_April2012.pdf [Accessed: 27 July 2015].

Howarth, L.M., Wood, H.L., Turner, A.P., and Beukers-Stewart, B.D. (2011). Complex habitat boosts scallop recruitment in fully protected marine reserve. *Marine Biology*, 158, 1767-1780.

Howarth, L.M., Pickup, S.E., Evans, L.W., Cross, T.J., Hawkins, J.P., Roberts, C.M., and Stewart, B.D. (2015a). Sessile and mobile components of a benthic ecosystem display mixed trends within a temperate marine reserve. *Marine Environmental Research*, 107, 8-23.

Howarth, L.M., Roberts, C.M., Hawkins, J.P., Steadman, D.J., and Beukers-Stewart, B.D. (2015b). Effects of ecosystem protection on scallop populations within a community-led temperate marine reserve. *Marine Biology,* 162, 823-840.

Hutchinson, W. (2015). *Don't take the "P" out of MPAs: Identifying conservation priorities for the South Arran Marine Protected Area*. Unpublished: University of York. MSc

International Union for Conservation of Nature (IUCN). (2012). *When is a Marine Protected Area really a Marine Protected Area*. [Online]. Available at: http://www.iucn.org/?uNewsID=10904 [Accessed 2 September 2015].

Jelinski, D.E. (2015). On a landscape ecology of a harlequin environment: the marine landscape. *Landscape Ecology*, 30, 1-6.

Jennings, S., and Kaiser, M.J. (1998). The effects of fishing on Marine Ecosystems. *Advances in Marine Biology*, 34, 201-352.

Joint Nature Conservation Committee (JNCC). (2015a). *Contributing to a Marine Protected Area Network*. [Online]. Available at: http://jncc.defra.gov.uk/page-4549 [Accessed 4 September 2015].

Joint Nature Conservation Committee (JNCC). (2015b). *Marine Conservation Zones*. [Online]. Available at: http://jncc.defra.gov.uk/page-4525 [Accessed 4 September 2015].

Joint Nature Conservation Committee (JNCC). (2015c). *Marine Conservation Zones*. [Online]. Available at: http://jncc.defra.gov.uk/page-5269 [Accessed 4 September 2015].

Jones, G.P., Cole, R.C., Battershill, C.N. (1992). Marine reserves: do they work? *Proceedings of the Second International Temperate Reef Symposium*, 29-44.

Juffe-Bignoli, D., Burgess, N.D., Bingham, H., Belle, E.M.S., de Lima, M.G., Deguignet, M., Bertzky, B., Milam, A.N., Martinez-Lopez, J., Lewis, E., et al. (2014). *Protected Planet Report 2014*. UNEP-WCMC: Cambridge, UK.

Kaiser, M.J., Attrill, M.J., Jennings, S., Thomas, D.N., Barnes, D.K.A., Brierley, A.S., Polunin, N.V.C., Raffaelli, D.G., and Williams, P.J. le Be. (2005). *Marine Ecology: Processes, Systems and Impacts.* Oxford University Press, Oxford.

Kingsford, M., and Battershill, C. (1998). *Studying Temperate Marine Environments. A Handbook for Ecologist*s. Canterbury University Press.

Landis, J.R., and Koch, G.G. (1977). The measurement of observer agreement for categorical data. *Biometrics*, 33, 159-174.

Lilley, R.J., and Unsworth, R.K.F. (2014). Atlantic Cod (*Gadus morhua*) benefits from the availability of seagrass (*Zostera marina*) nursery habitat. *Global Ecology and Conservation*, 2, 367-377.

Magurran, A.E. (1988). *Ecological Diversity and Its Measurement*. Princeton University Press, United States of America.

Mangi, S. (2007). *Successes and disappointments of MPAs in the Western Indian Ocean: the case of Mombasa Marine Park and Reserve*. In: Edwards, A.J., Hooper, T.,L. (Eds). (2009). *Proceedings of the workshop on a regional perspective on MPAs in the Western Indian Ocean, Rodrigues Island, Mauritius, 9-14 May 2007*. Newcastle University, Newcastle Upon Tyne and Marine Education Trust, Collompton, UK.

McIntyre, F., Fernandes, P.G., and Turrell, W.R. (2012). *Clyde Ecosystem Review*. Scottish Marine and Freshwater Science, 3 (3). [Online]. Available at: http://www. /Resource/0039/00395239.pdf [Accessed 8 July 2015].

Moritz, C., Grave, D., Savard, L., McKindsey, C.W., Brêthes, J.-C., and Archambault, P. (2015). No more detectable fishing effect on Northern Gulf of St Lawrence benthic invertebrates. *ICES Journal of Marine Science*, 72 (6).

Murawski, S.A., Brown, R., Lai, H.-L., Rago, P.J., and Hendrickson, L. (2000). Large-scale closed areas as a fishery-management tool in temperate marine systems: The Georges Bank experience. Buletin of Marine Sciences, 66(3), 775-798.

Pande, A., and Gardner, J.P.A. (2009). A baseline biological survey of the proposed Taputeranga Marine Reserve (Wellington, New Zealand): spatial and temporal variability along a natural environmental gradient. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 19, 237-248.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R. And Torres Jr., F. (1998). Fishing Down Marine Food Webs. *Science*, 279 (5352), 860-863.

Pauly, D., Christensen, V., Guénette, S., Pitcher, T.J., Sumaila, U.R., Walters, C.J., Watson, R., and Zeller, D. (2002). Towards sustainability in world fisheries. *Nature*, 418, 689-695.

Roberts, C.M. (2007). *The Unnatural History of the Sea: The past and future of humanity and fishing*. Island Press, London.

Roberts, C.M., Hawkins, J.P., and Gell, F.R. (2005). The role of marine reserves in achieving sustainable fisheries. *Philosophical Transactions of the Royal Society B*, 360, 123-132.

Sala, E., Costello, C., Dougherty, D., Heal, G., Kelleher, K., Murray J.H., Rosenberg, A.A., and Sumaila, R. (2013). A general business model for marine reserves. *PLoS One*, 8(4).

Scottish Government. (2014a). *Environmental Protection Marine Management: South Arran Nature Conservation Marine Protected Area Order, 2014*. [Online]. Available at: http://www.gov.scot/Resource/0045/00456829.pdf [Accessed 22 July 2015].

Scottish Government. (2014b). *Marine Scotland – 2014 Consultation on the Management of Inshore Special Areas of Conservation and Marine Protected Area – Maps.* [Online]. Available at: http://www.gov.scot/Resource/0046/00462926.pdf [Accessed 2 September 2015].

Scottish Government. (2014c).*MPA Designation Orders – South Arran Designation*. [Online]. Available at: http://www.gov.scot/Topics/marine/marine-enviornment/mpanetwork/developing/DesignationOrders/ARRDOrder [Accessed 28 August 2015].

Scottish Government. (2015a). *Marine Protected Areas (MPAs).* [Online]. Available at: www.gov.scot/Topics/marine/marine-environment/mpanetwork [Accessed 2 September 2015].

Scottish Government. (2015b). *Partial Business and Regulatory Impact Assessment*. [Online]. Available at: http://www.gov.scot/Resource/0047/00479001.pdf [Accessed 28 August 2015].

Scottish Natural Heritage (SNH). (2015). *South Arran Marine Protected Area*. [Online]. Available at: http://www.snh.gov.uk/protecting-scotlands-nature/protected-areas/national-designations/mpas/mpa-arr/ [Accessed 28 August 2015].

Sheehan, E.V., Cousens, S.L., Nancollas, S.J., Stauss, C., Royle, J., and Attrill, M.J. (2013). Drawing lines at the sand: Evidence for functional vs. visual reef boundaries in temperate Marine Protected Areas. *Marine Pollution Bulletin*, 76, 194-202.

Slezak, M. (2014). *Conservation report reinforces fears over "paper parks".* New Scientist, Issue 2996, November 2014. [Online] Available at: https://www.newscientist.com/article/dn26552-conservation-report-reinforces-fears-over-paper-parks/ [Accessed 12 September 2015].

Thrush, S.F., and Dayton, P.K. (2002). Disturbance to Marine Benthic Habitats by Trawling and Dredging: Implications for Marine Biodiversity. *Annual Review of Ecology and Systematics*, 33, 449-473.

Thurstan, R.H., and Roberts, C.M. (2010). Ecological Meltdown in the Firth of Clyde, Scotland: Two Centuries of Change in a Coastal Marine Ecosystem. *PLoS ONE*, 5(7).

Tivy, J. (1986). The geography of the estuary and Firth of Clyde. *Proceedings of the Royal Society of Edinburgh B (Biological Sciences),* 90, 7-23.

Wakefield, C.D., Blight, S., Dorman, S.R., Denham, A., Newman, S.J., Wakeford, J., Molony, B.W., Thomson, A.W., Syers, C., and O'Donoghue, S. (2014). *Independent observations of catches and subsurface mitigation efficiencies of modified trawl nets for endangered, threatened and protected megafauna bycatch in the Plibara Fish Trawl Fishery*. Fisheries Research Report No. 244. Department of Fisheries, Western Australia. 40pp.

Warren, M.A., Gregory, R.S., Laurel, B.J., and Snelgrove, P.V.R. (2010). Increasing density of juvenile Atlantic (Gadus morhua) and Greenland cod (G. ogac) in association with spatial expansion and recovery of eelgrass (Zostera marina) in a coastal nursery habitat. *Journal of Experimental Marine Biology and Ecology*, 394, 154-160.

Wenzel, L. (2015). *Framework for the National System of Marine Protected Areas of the United States of America*. National Marine Protected Areas Center. [Online]. Available at: http://marineprotectedareas.noaa.gov/nationalsystem/framework/final-mpa-framework-0315.pdf [Accessed 2 September 2015].

World Ocean Review (WOR). (2013). World Ocean Review 2: The Future of Fish – The Fisheries of the Future. [Online] Available at: http://worldoceanreview.com/wp-content/downloads/wor2/WOR2_english.pdf [Accessed 2 September 2015].

Appendix 1

Table showing co-ordinates of the boundary points of the South Arran MPA.

Co-ordinate Label	Latitude (N)	Longitude (W)
A	55° 34.324	05° 06.509
B	55° 34.479	05° 00.259
C	55° 31.115	04° 58.162
D	55° 28.679	04° 58.982
E	55° 25.844	05° 00.291
F	55° 23.982	05° 02.496
G	55° 22.824	05° 05.414
H	55° 23.158	05° 13.914
I	55° 23.685	05° 17.506
J	55° 25.682	05° 22.576
K	55° 28.648	05° 25.687
L	55° 30.469	05° 26.702
M	55° 31.602	05° 21.301

Appendix 2

Site number, name and start and end co-ordinates of all 20 surveys conducted.

Transect Number	Transect Name	Start Latitude (N)	Start Longitude (W)	End Latitude (N)	End Longitude (W)
1	North St. Molios Bay	55° 31.350	05° 05.039	55° 31.368	05° 05.064
2	North Dhunan	55° 33.988	05° 05.845	55° 34.000	05° 05.900
3	Seagrass , Whiting Bay Jetty	55° 29.373	05° 05.393	55° 29.384	05° 05.440
4	Munida Boulder Reef	55° 25.607	05° 16.996	55° 25.580	05° 16.980
5	Lagg Burn	55° 25.804	05° 14.476	55° 25.806	05° 14.433
6	South Kildonan Castle	55° 25.738	05° 06.054	55° 25.760	05° 06.025
7	South Pascoe Bank	55° 30.795	05° 03.390	55° 30.772	05° 03.374
8	East Pladda	55° 25.802	05° 06.423	55° 25.821	05° 06.467
9	West Pladda	55° 25.458	05° 07.676	55° 25.481	05° 07.688
10	South-West Inner Holy Isle Lighthouse	55° 30.537	05° 04.146	55° 30.561	05° 04.110
11	14m Shoal Black Rocks	55° 28.802	05° 04.830	55° 28.831	05° 04.834
12	North-East Hamilton Rock	55° 33.267	05° 04.635	55° 33.297	05° 04.692
13	Kildonan Farm	55° 26.416	05° 05.516	55° 26.400	05° 05.543
14	South-West Kings Cave	55° 30.934	05° 21.845	55° 30.895	05° 21.848
15	Far West Pladda	55° 25.230	05° 08.773	55° 25.247	05° 08.724
16	Shannochie	55° 25.546	05° 11.728	55° 25.527	05° 11.706
17	Arnhall, Whiting Bay	55° 29.577	05° 05.056	55° 29.554	05° 05.036
18	Inner Dhunan	55° 33.709	05° 05.338	55° 33.679	05° 05.322
19	East Deacon Rock	55° 31.988	05° 05.804	55° 31.959	05° 05.823
20	East Red Buoy, Outside NTZ	55° 32.711	05° 04.351	55° 32.676	05° 04.376