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Quantifying the environmental benefits of Artificial Reefs: an investigation into the ecological effects of habitat complexity and fisheries exclusion

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ABSTRACT

Artificial reefs provide a potentially important tool in fisheries protection and ecosystem management, provided they are designed in order to maximise the survival and recruitment of commercially important species and to maintain overall ecosystem structure. However, many of the perceived benefits of artificial reef technology, particularly in northern temperate waters, remain to be quantified. A study was undertaken on a large-scale multi reef artificial reef complex and at a number of surrounding natural reef and sandy bottom sites, on the west coast of Scotland, in order to investigate two of the potential benefits of an artificial reef. Firstly, the study used *in situ* diver-conducted survey techniques to investigate the effect of habitat complexity upon variety of temperate marine species, by comparing artificial reef sites of differing complexity with natural reef sites in the surrounding area. Secondly, the study investigated the potential of an artificial reef as a fisheries protection tool through examination of the gonadal condition of the king scallop (*Pectens maximus*). Scallops were collected at the artificial reef site, where no fishing is presently occurring, and at a number of surrounding unprotected sites.

The study found that habitat complexity had some effects on the overall faunal abundance and species diversity. When the total mobile faunal abundance was examined no differences were observed between the artificial and natural reef sites. However there were marked differences between individual species with some exhibiting higher abundance at the complex artificial reef sites, e.g. the corkwing wrasse (*Crenilabrus melops*), and some species exhibiting higher abundance at the natural reef sites, e.g. the leopard spotted goby (*Thorogobius ephippiatus*). The scallop study showed that individuals sampled from the unprotected sites were in better condition than those sampled from the artificial reef sites. This was considered to be more a factor of location than protection and it is possible that the artificial reef may not be located in an area where conditions are optimal for scallop growth.

This project has shown significant ecological benefits of artificial reef deployments in northern temperate waters. However, any commercial success will be dependent on the quality of the receiving environment and the habitat complexity afforded through the design of the reefs.

1. INTRODUCTION

On a global scale, rocky subtidal habitats tend to be less common than sedimentary habitats (Nybakken, 1993), yet are estimated to account for around 35% of the coastline of the United Kingdom (JNCC, 1993). In this northern temperate region rocky reefs provide an important nursery area for a large number of marine species (Henriques & Almada, 1998), including the Atlantic cod (*Gadus morhua* Linnaeus) and the pollack (*Pollachius pollachius* Linnaeus). In addition the rocky habitat around the UK coast provides essential habitat for numerous species such as the European lobster (*Hommarus gammarus* (Linnaeus)), the velvet swimming crab (*Necora puber* (Linnaeus)) and five temperate wrasse species (Pisces: Labridae), of which the goldsinny (*Ctenolabrus rupestris* (Linnaeus)), rock cook (*Ctenolabrus exoletus* (Linnaeus)) and corkwing wrasse (*Crenilabrus melops* (Linnaeus)) have been identified as having potential commercial importance as cleaner fish in aquaculture (Darwall *et al.*, 1992; Sayer *et al.*, 1995, 1996).

In the UK a large amount of the rocky subtidal substrate tends to be exposed bedrock. This provides a low level of habitat complexity for large mobile animals and so limits the availability of refuge and foraging sites. Therefore, it can be deduced that the availability of subtidal rocky habitat is at a premium and so factors such as shelter availability (Caddy 1986), competition (Osenberg *et al.*, 1992) and predation (Williamson, 1993) can interact to produce a bottleneck in a species life cycle. Thus logic dictates that an increase in the available habitat will have a positive effect on the local populations of many species associated with rocky subtidal habitats.

1.1 ARTIFICIAL REEFS

An artificial reef has been defined by Seaman and Jensen (2000) as one or more objects, of natural or human origin deployed onto the sea floor in order to influence some aspect of the marine environment. This broad scale definition encompasses a wide range of applications of artificial reef technology including fisheries protection, environmental mitigation of pollution, coastal protection and recreational uses such as SCUBA diving and angling. As such now over 30 countries worldwide have licensed artificial reefs deployed in their territorial waters (Jenson, 2002; Seaman, 2002) and it has been postulated that artificial reefs may be a powerful tool in mitigating against the current global fisheries crisis (Baxter, 2000; Sayer, 2001).

As a fisheries management tool, artificial reefs are of potential importance for two reasons. Firstly, because fish tend to exhibit thigmotactic behaviour, whereby they are attracted to areas of structured habitat (Brickhill *et al.*, 2005), the artificial reef can be observed to attract fish, as demonstrated by the Acadja fishing technique used in Benin and Côte d'Ivoire (Hem & Avit, 1994). Secondly, artificial reefs tend to mitigate against trawling because there is an increase in the risk of snagging and/or damaging nets and equipment. Thus, provided an artificial reef is not over-exploited by line or static gear fisheries, it can be observed to both develop and provide protection for a typical rocky reef community and also protect an area of the seabed surrounding it. An example of this use of artificial reef technology is presently being undertaken in Hong Kong where reefs are deployed within designated Marine Protected Areas (MPAs) in an attempt to rebuild the depleted fish stocks surrounding the territory (Pitcher *et al.*, 2002; Hong Kong Artificial Reef Project, 2003).

Proposals have been made to use decommissioned oilrigs in the North Sea for the same purpose (Cripps and Aabel, 2002). However, the prospect of decommissioning oil rigs at sea remains a controversial issue. In any case the benefits of a rigs to reefs programme are likely to be negligible according to Sayer and Baine (2002) because of the concomitant loss of fishing exclusion provided by the operational rigs.

1.2 PREVIOUS RESEARCH

Research into the impact of artificial reefs upon the abundance and behaviour of fish and commercially important macroinvertebrate species has become important in order to aid the development of artificial reef design and technology. Artificial reefs have produced a large number of studies in the peer-reviewed literature and the research undertaken inevitably feeds back into their development within marine resource management. However, most of this previous research is largely qualitative in nature, due to a historical lack of formal hypothesis testing in the ecological studies of artificial reefs (Bortone, 2006). Additionally, experimental design issues such as inadequate replication of sample sites and pseudoreplicated experimental designs (Hurlbert, 1984) were also observed to be prevalent within the literature.

An example of unreplicated experimental design in artificial reef research is provided by a study by Charbonnel *et al.* (2002) on artificial reef complexity. In the study investigators used two Large Artificial Reef Units (LARUs) to study the effect of habitat complexity upon fish assemblages. One reef site was used as an experimental site, where the level of complexity was increased by filling the artificial reef module

with building materials, and one was used as a control site where the artificial reef module received no manipulation. Similarly, studies by Fabi and Fiorentini (1994) and Fujita *et al.* (1996) compared artificial reef sites with homogenous mud bottom sites and natural reefs but again in both studies only single samples of each site type were used. This is a major issue within any ecological research as these studies cannot draw conclusions beyond the isolated study sites without exceeding the error term of the experiment (Hurlbert, 1984). Therefore, these studies can be seen to add little to the literature because of their fundamental lack of replication.

The other main problem observed to recur in the peer-reviewed literature is the presence of pseudoreplication, whereby the replicated sites are not truly independent of one another. This is problematic as it is often difficult to estimate the distance between sites at which independence is achieved. Yet independence of data is essential to many of the inferential statistics used in ecology (Underwood, 1997). Examples of this are shown in the work of Bortone *et al.* (1994) in the Gulf of Mexico and by Jensen *et al.* (1994) on the Poole Bay artificial reef, UK. In the case of the former a larger scale deployment could easily have been designed as the replicate sites were all located within a 900m² area (30mx30m). In the case of the latter, the pseudoreplication was largely caused by reef design and licensing, with the whole reef situated within Poole Bay. This issue is not readily resolved as areas available for licensing can be limited (Sayer & Wilding, 2002).

1.3 THE PRESENT STUDY

As much of previous research has suffered from serious experimental design issues there remains a degree of official scepticism regarding the efficacy of artificial reefs in fisheries and marine resource management. As such, the present study offers an excellent opportunity to gain quantitative information of the differences in animal abundance associated with artificial and natural reefs.

The Loch Linnhe Artificial Reef provides a fully replicated artificial reef environment consisting of a large scale experimental reef complex deployed over 146 hectares of sea bed on the south side of the island of Lismore, between Branra Rock and the Eilean Dubh (Fig 1.1). The reef consists of five groups of six reef modules. Each module is constructed from 220-250 tonnes of concrete blocks of size 40 x 20 x 20cm. Within each reef group three reef modules are constructed from “simple” blocks, which are a solid, and three modules are constructed from “complex blocks,” which have two cuboidal holes running through them (Fig 1.2). These are then arranged randomly on the seabed.

Fig 1.1 Map showing the approximate location of the Loch Linnhe artificial reef within the Lynn of Lorne area. Artificial reef location marked by the open box.

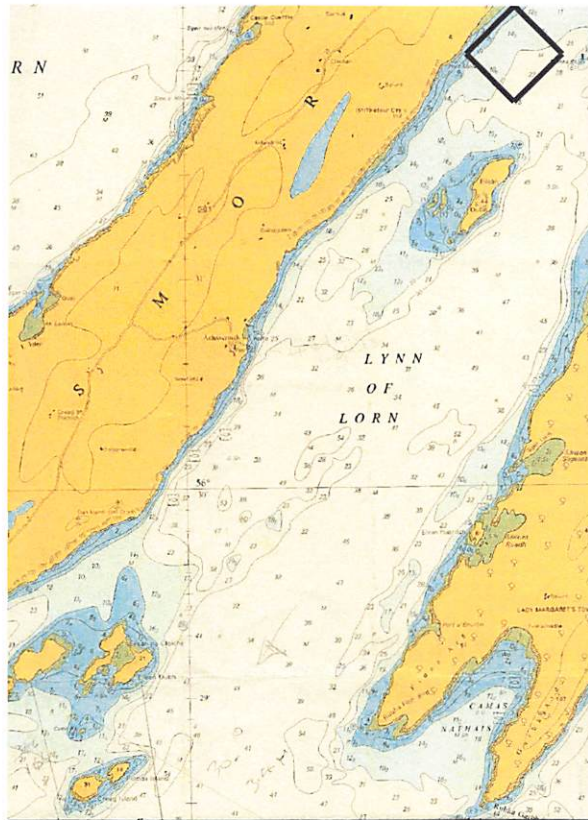


Fig 1.2 Simple (left) and complex (right) concrete reef blocks.



The complex artificial reef modules of the Loch Linnhe artificial reef have been observed to provide a more geometrically complex structure than the simple artificial reef modules or comparable natural reef sites (Rose, 2005). Also it has been observed that the complex artificial reef modules support higher levels of sessile epifaunal than the simple artificial reef modules (Beaumont, 2006). Therefore, because the complex artificial reef modules exhibit higher levels of habitat complexity in terms of exposed surface area, refuge/shelter and forage availability the hypothesis is that they support higher levels of animal abundance than simple artificial reef modules or natural reef sites. The primary aim of the present study is to use a non-destructive visual census method to examine the levels of relative abundance for each of the species listed in Table 1 on complex artificial reef sites, simple artificial reef sites, natural reef sites and a sand/mud bottomed control site. The data collected will then be used to investigate the hypothesis that increased habitat complexity provided by a complex artificial reef will support higher levels of animal abundance than a comparable simple artificial reef or natural reef.

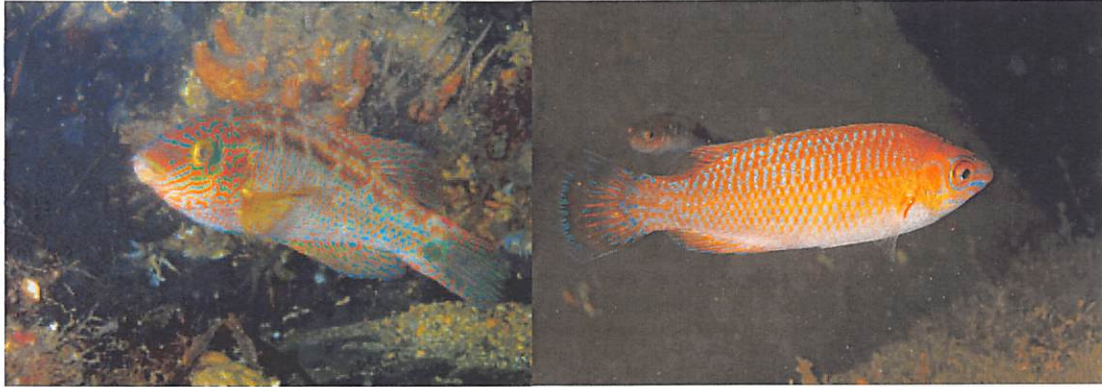
Table 1 Species List

Corkwing Wrasse	<i>Crenilabrus melops</i>	(Linnaeus)
Rock Cook	<i>Centrolabrus exoletus</i>	(Linnaeus)
Gold Sinny	<i>Centrolabrus rupestris</i>	(Linnaeus)
Ballan Wrasse	<i>Labrus bergylta</i>	Ascanius
Cuckoo Wrasse	<i>Labrus mixtus</i>	(Linnaeus)
Juvenile Cod	<i>Gadus morhua</i>	Linnaeus
Juvenile Pollack	<i>Pollachius pollachius</i>	Linnaeus
Juvenile Saithe/ Coley	<i>Pollachius virens</i>	Linnaeus
Leopard Spotted Goby	<i>Thorogobius ephippiatus</i>	(Lowe)
Brown Crab	<i>Cancer pagarus</i>	Linnaeus
Velvet Swimming Crab	<i>Necora puber</i>	(Linnaeus)
Shore Crab	<i>Carcinus maenus</i>	(Linnaeus)
European Lobster	<i>Hommarus gammarus</i>	Linnaeus
Squat Lobster	<i>Munida rugosa</i>	(Fabricius)
Edible Sea Urchin	<i>Echinus esculentus</i>	Linnaeus
Common Starfish	<i>Asterias rubens</i>	Linnaeus
Crinoid	<i>Antedon bifida</i>	(Pennant)
Tunicate	<i>Class: Ascidiacea</i>	

Fig 1.3 Photographs showing a number of the species under investigation.

a) Corkwing wrasse (*Crenilabrus melops*).

b) Rock cook (*Ctenolabrus exoletus*).



c) Leopard spotted goby (*Thorogobius ephippiatus*). d) Edible crab (*Cancer pagarus*).



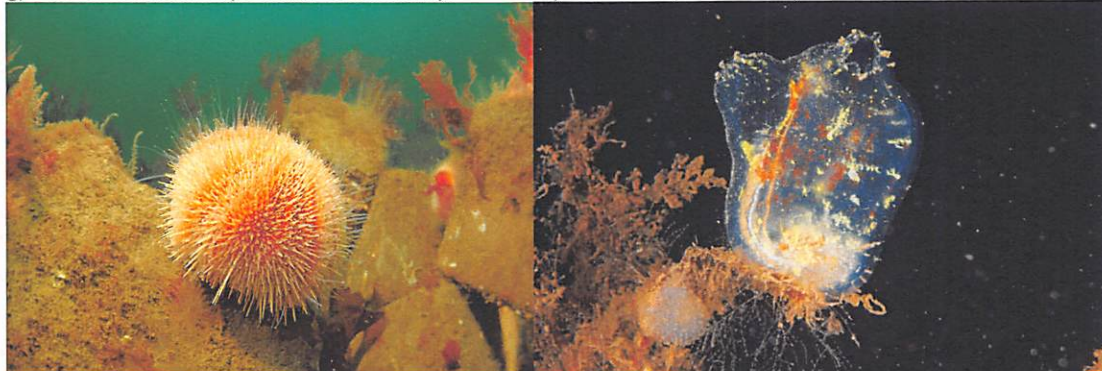
e) Velvet swimming crab (*Necora puber*).

f) Squat lobster (*Munida rugosa*).



g) Edible sea urchin (*Echinus esculentus*).

h) Tunicate (Ascidiacea).



The second aim of the present study is to investigate how the closure of the area surrounding the Loch Linnhe artificial reef to commercial fishing activities has affected the soft bottomed fauna within the surrounding seabed. Kaiser *et al.* (2000, 2002) observed that when fishing regulation excluded demersal trawl and dredge fishing from an area it had a positive impact upon the benthic habitat, with concomitant improvement in the condition of many benthic organisms. Investigation of the reproductive investment of a relatively sedentary organism, for example the king scallop (*Pectens maximus* (Linnaeus)), can provide an indicator of benthic habitat condition. This can be achieved by sampling individuals from areas protected from commercial exploitation and comparing the relative gonadal and somatic masses of these individuals with those of individuals obtained from areas not afforded the same protection. Therefore, this study will examine if there is a significant difference between the reproductive investment of scallops found both in the surrounding the artificial reef site and at a number of other sites in the Firth of Lorne, without the same level of protection from trawling.

The study required diving operations to be undertaken. As such all fieldwork was conducted under the Health and Safety Executive's Diving at Work Regulations, 1997 (HSE books, 1998a) using the Scientific and Archaeological Accepted Code of Practise (ACoP) (HSE books, 1998b). All diving was conducted within the No Decompression Limits of the Bühlmann '86 Decompression Tables (Lippmann & Mitchell, 2005).

2. MATERIALS & METHODS

2.1 COMPARISON OF ARTIFICIAL & NATURAL REEFS

Visual census of a number of artificial reef stations, natural reef stations and a control sand/mud bottom station was carried out using scientific diving techniques. The experiment took place between August 2005 and June 2006, with all sites being visited and surveyed on a monthly basis. Six stations were selected on the Loch Linnhe artificial reef (three complex reef modules and three simple reef modules) and these were compared against three natural reef stations and one sand/mud bottom control station in the Lynn of Lorne area (Table 2).

Table 2: Study Sites.

Site	Station	No. of Transects
Complex Artificial Reef	Loch Linnhe Artificial Reef B2c	2
Complex Artificial Reef	Loch Linnhe Artificial Reef C2c	2
Complex Artificial Reef	Loch Linnhe Artificial Reef D1c	2
Simple Artificial Reef	Loch Linnhe Artificial Reef B1s	2
Simple Artificial Reef	Loch Linnhe Artificial Reef C2s	2
Simple Artificial Reef	Loch Linnhe Artificial Reef D2s	2
Natural Reef	Creag Isles	2
Natural Reef	Saulmore Point	2
Natural Reef	Eilean Mhor	2
Control	Dunstaffnage Bay	4

Experimental Sites

The artificial reef site was located upon the Loch Linnhe Artificial Reef (fig1.1). Out of the available 32 reef modules, three constructed from complex blocks and three constructed from simple blocks were chosen at random. In order to maintain independence only one complex and one simple reef module were chosen at random

from three of the five reef groups, Groups B, C and D respectively. Thus the complex artificial reef stations selected were B2c (56°32.113'N, 5°27.330'W), C2c (56°32.119'N, 5°27.103'W) and D1c (56°32.222'N 5°26.933'W). The simple artificial reef stations selected were B1s (56°32.102'N, 5°27.373'W), C2s (56°32.099'N, 5°27.076'W) and D2s (56°32.224'N, 5°26.880'W).

Three natural reef stations were selected in the Lynn of Lorne area, at the Creag Isle (CI) (56°28.655'N, 5°30.950'W), Eilean Mhor (EM) (56°27.348'N, 5°26.034'W) and Saulmore Point (SP) (56°27'N, 5°24'W). These stations were selected as they were in a similar depth range as the artificial reef modules and rose up from a sand/mud bottom in a similar fashion to the artificial reef site. However, obtaining suitable natural reef stations was problematic because most of the reefs in the locality were composed of exposed bedrock and so tended to be continuous over a very large area and range of depths.

The control site selected was an area of flat sand/mud seabed of similar depth to the reef site. The site selected was below the outer admiralty buoy in Dunstaffnage Bay (AB) (56°27'N, 5°26'W).

Sampling Technique

The sampling method selected for this study was the belt transect technique of underwater visual census, first adopted by Brock (1954). The technique involves the observer moving along a transect of known length and width, marked by a rope boundary, and counting the number of individuals of a given species encountered within its boundary over a standardized time, as recommended by Samoily (1997).

The belt transect was chosen for this work as it was observed by Kimmel (1985) to provide a high level of accuracy in assessing fish abundance. Also it was considered that the belt transect technique would be more effective in the limited visibility of temperate waters than the line transect technique described in the “Reef Fish Watch 2000 Protocol” (Ormond 2000) or the stationary count technique described by Bortone *et al.* (2000).

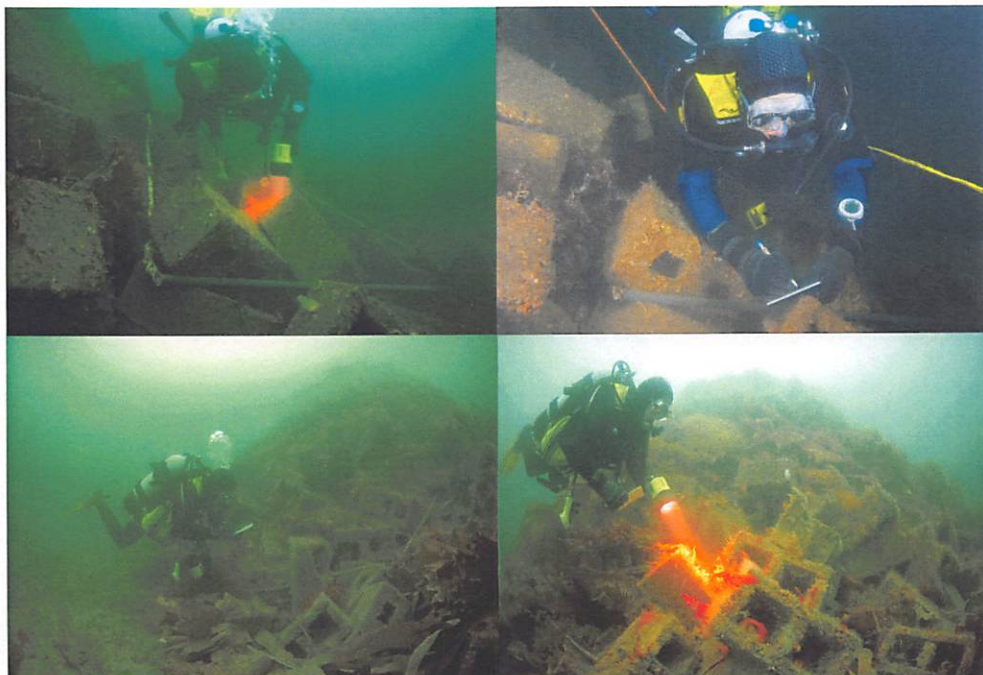
Transect size was partly determined on the merits of previous studies of fish and benthic megafauna conducted on the west coast of Scotland (Wilding & Sayer, 2002; Magill & Sayer 2004). Additionally, transect size was affected by logistical restrictions of construction and ease of deployment and recovery by a team of two divers. Therefore, the increased precision provided by a large survey area (Sayer & Poonian, 2006) enclosed within a transect had to be balanced against the previously mentioned logistical considerations.

Two transects were deployed at random at each natural and artificial reef station, in a depth range of 10-18m below Chart Datum. Also four transects were deployed randomly at the control site. The transects were constructed from 12mm leaded rope and 15mm plastic pipe and were 6 metres in length and 1.5 metres in width (Fig 2.1), with each transect being surveyed at a speed of $1.5\text{m}^2\text{min}^{-1}$. A species list was drawn up comprising of 17 different species of marine organisms and one taxonomic grouping (Table 1). The number of individuals observed within each belt transect was recorded upon a dive slate, as the diver swam along the transect (Fig 2.2).

Fig 2.1 Belt Transect prior to deployment. The transect would be rolled up in the manner of a scroll and thus could be deployed with ease by a team of two divers working together.



Fig 2.2 A diver conducts the visual census by swimming slowly along the length of the transect and periodically recording the number of individuals of each species observed within the transect.



Temperature

Temperature was recorded at each site during each individual survey from August 2005- June 2006 using a SUNNTO Gekko dive computer. This did not provide a particularly sensitive means of recording temperature, but it allowed the general trend of temperature change over the study period to be observed (Fig 2.3). This trend was then used to categorise the monthly datasets into four seasonal regimes. The seasonal regimes imposed upon the data are recorded in Table 3.

Fig 2.3 Scatterplot with Lowess smoothed trendlines showing the general trend and variation in the seawater temperature ($^{\circ}\text{C}$) over the study period (August 2005- June 2006).

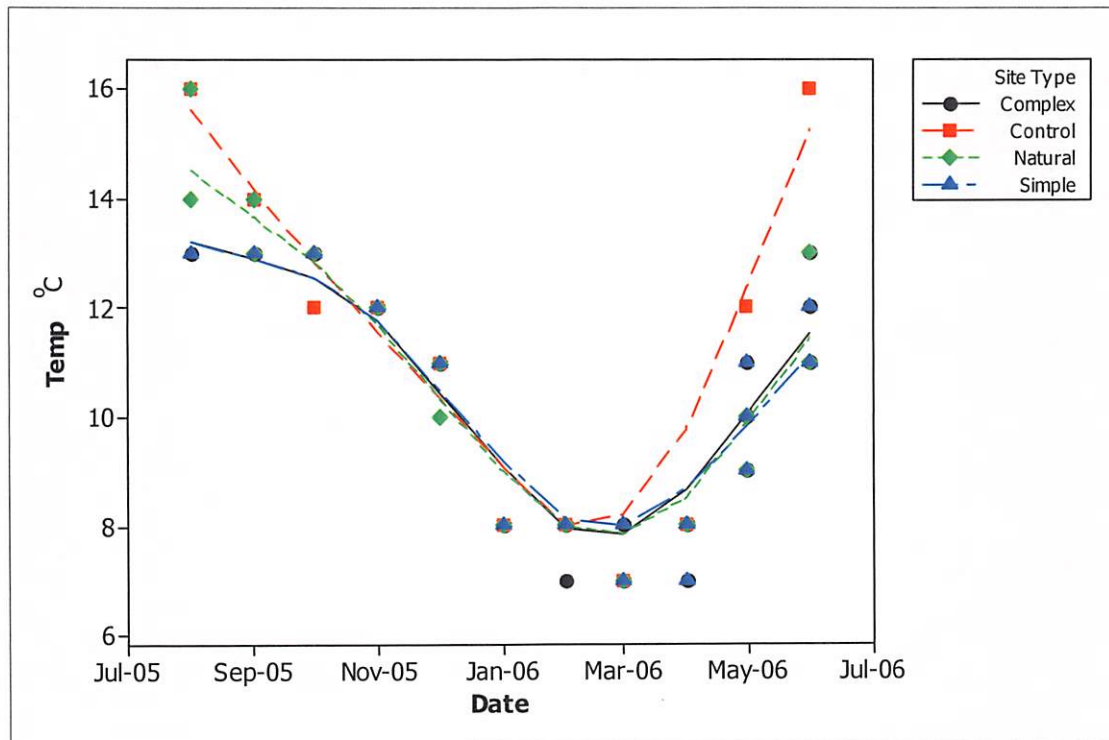


Table 3: Table showing the seasonal regimes imposed upon the data, and the mean seawater temperature recorded for each season.

Season	Months Included	Mean Temperature ($^{\circ}\text{C}$)	SE Mean
Summer	August 2005- October 2005	13.4	0.1
Autumn	November 2005- December 2005	11.4	0.1
Winter	January 2006- March 2006	7.6	0.1
Spring	April 2006- June 2006	10.2	0.3

Data Analysis

Initially the data collected for each species were converted from a raw data set of counts per transect to a level of abundance per unit area. This was carried out by the equation:

$$Ab_{it} = N_{it} / A_t$$

Ab_{it} = Abundance of Species i , in individuals.metre⁻², in transect t
 N_{it} = Number of individuals of species i , in transect t
 A_t = Area of transect t , in metres².

Univariate statistical analysis was carried out upon the abundance data placed within the seasonal regimes. Multivariate statistical analysis was also carried out upon the seasonal abundance data. These two forms of analysis are described separately.

For each of the four seasonal regimes the differences in species abundance between site types were examined. Prior to commencing statistical analysis the data were tested for equal variance in the <MINITAB> statistics program using Levene's test ($p > 0.05$) (Underwood, 1997). It was observed that because of the inherent problems of counting mobile fauna the data did not conform to the assumptions of homogeneity variance required for both analysis of variance and the non-parametric alternatives, e.g. Kruskal-Wallis test (Underwood, 1997). It was also observed that standard transformation procedures did not remedy this situation. Therefore, the bootstrap technique (Diaconis & Efron, 1983), a novel resampling method, was employed to remedy the situation and supply data with homogenous variance for statistical analysis.

their corresponding seasonal regime and used to test for any difference in the level of species diversity at each of the four site types under investigation, on a seasonal basis. This analysis was carried out in <MINITAB> by one way ANOVA at a significance level of $p < 0.05$ with Fisher's pairwise comparison, with data transformed accordingly to fit Levene's test (Table 4).

Table 4: Transformation required to normalise the variance of the Shannon-Weiner Diversity Index data for each season.

Season	Transformation Required	Levene's Test p-value
Summer	None	0.774
Autumn	Rank	0.082
Winter	None	0.119
Spring	None	0.390

For the statistical analysis the following assumptions were made during the course of the study. Firstly, it was necessary to assume that the diver did not have a significant influence upon the abundance of fish and other mobile fauna within the experimental area (Buxton and Smale, 1989). Secondly, it was assumed that because the belt transects were of relatively short length they would mitigate against the changes in the detectability of target species caused by changes in the visibility of the water. The final assumption deemed necessary for the experiment was to mitigate against poor weather restricting access to the study sites. As such, a range of sample sites were selected exhibiting different levels of exposure and it was assumed that any variability between the sites would be controlled by adequate replication.

2.2 REPRODUCTIVE INVESTMENT OF SCALLOPS

Specimens of the king scallop (*Pectens maximus*) were collected by SCUBA diving during the months of December 2005 and March 2006. Sampling occurred in the soft sediment at the Loch Linnhe Artificial Reef Site, where no commercial fishing has been observed to occur over the last five years, and in soft sediment surrounding the Creag Isles, in Ardnamucknish Bay and surrounding the Glas Eilean, in the Mouth of Loch Creran. These latter three sites were not provided with any protection from commercial fishing. Sampling occurred during two isolated periods when filled gonads could be expected (Duinker & Nylund, 2002) in order to control for the seasonal spawning behaviour of this species (Pazos, *et al.* 1996). At each site ten specimens were collected during each period of sampling.

Gonadosomatic Index

Scallops were placed into a freezer, at -19°C , in order to ensure a humane death and to preserve them until the dissection could be carried out. Dissection of the scallops was carried out in a wet laboratory/ specimen preparation room at Dunstaffnage Marine Laboratory. During the dissection the gonad and adductor muscle of the animal were removed. These were placed into individual aluminium foil boats and dried in an oven at 90°C , for 24 hours, to reduce to a constant weight. The dried gonad and adductor muscle of each individual were then weighed to three decimal places. The recorded weights were then used to calculate the Gonadosomatic Index:

$$\text{GSI} = (\text{G}_{\text{dw}} / \text{A}_{\text{dw}}) \times 100$$

GSI = Gonadosomatic Index

G_{dw} = Gonad Dry Weight

A_{dw} = Adductor Muscle Dry Weight

Additionally the length of the right valve of each individual was accurately measured, in millimetres, using Vernier Callipers. The linear relationship between A_{dw} and the valve length was then examined ($S=8.036$ $r^2=26.0\%$).

Data Analysis

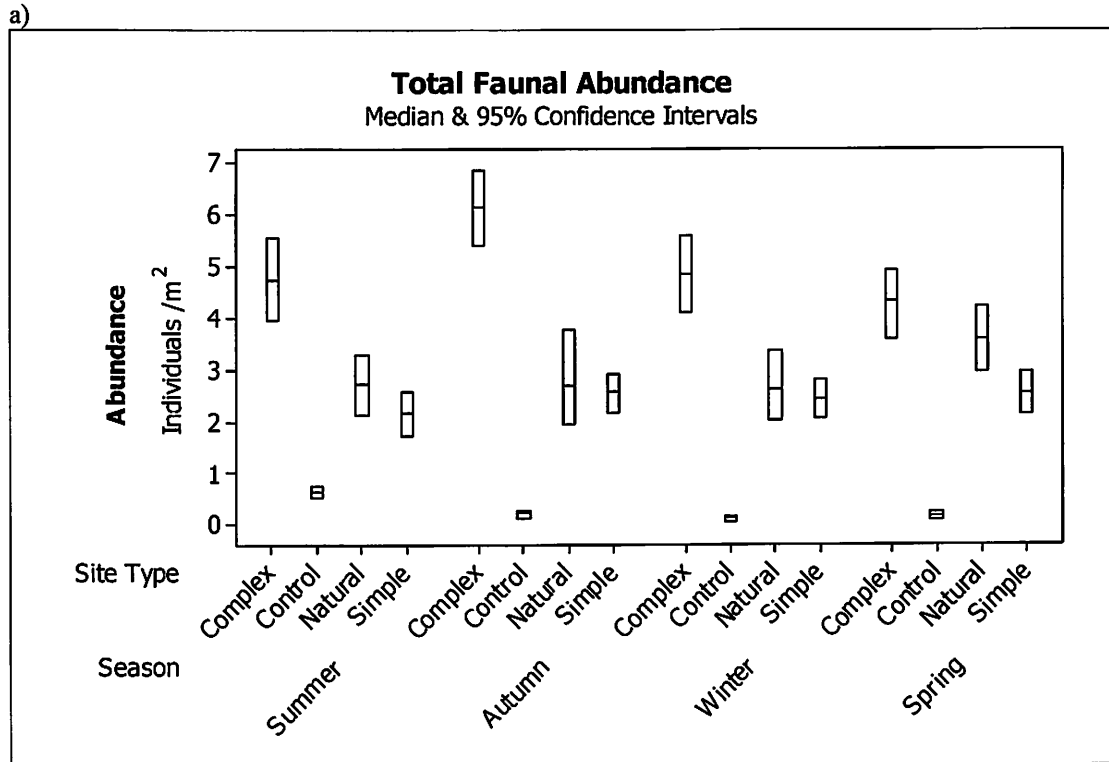
Univariate statistical analysis was used to investigate any differences in the GSI of the individuals and their valve length against the site type. Prior to formal statistical analysis Levene's test was used to test for equal variance to a significance level of 0.05 in the data. The data was then tested for any significant difference using a one-way ANOVA with Fisher's pairwise comparisons.

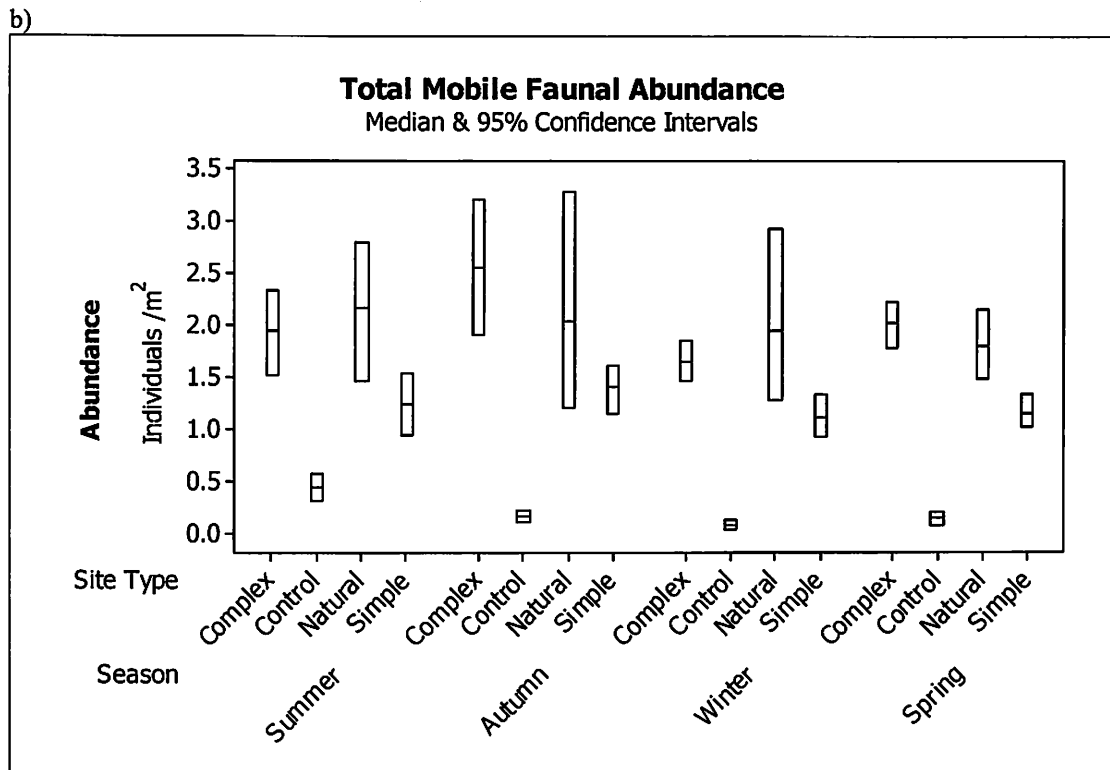
3. RESULTS

3.1 COMPARISON OF ARTIFICIAL & NATURAL REEFS

Total Faunal Aggregations

Fig 3.1 Boxplots showing the a) total abundance of all fauna and the b) total abundance of mobile fauna for each of the four sites, over the four seasonal regimes of Summer, Autumn Winter and Spring.



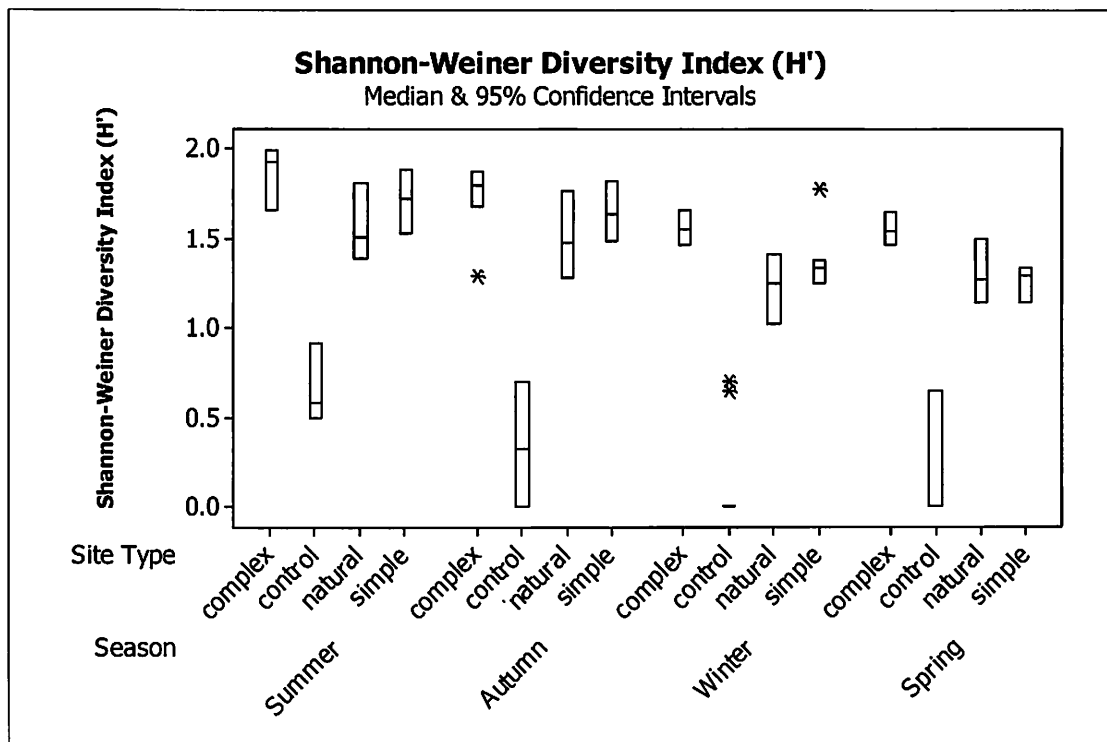


Over the four seasons it was observed that there was relatively little change in the total abundance of animals at each reef site. This was observed both as an overall phenomenon and also within the mobile fauna. It was observed that the total faunal abundance was significantly higher at the complex artificial reef site compared with the other sites during the summer ($p < 0.001$), autumn ($p < 0.001$) and winter periods ($p < 0.001$), with all the reef sites showing significantly higher levels of abundance than the control site. In the spring period no significant difference was observed between the complex artificial reef and natural reef sites, yet both these sites had significantly higher levels ($p < 0.001$) of abundance than either the simple artificial reef or control sites.

On examination of the mobile fauna it was noticed that while numbers recorded at the reef sites were significantly greater than those recorded at the control site ($p < 0.05$), no significant difference was observed between the three reef sites ($p > 0.05$), with the

exception of the spring period where Fisher's pairwise test showed both the complex and natural reef sites to be significantly greater than the simple artificial reef site.

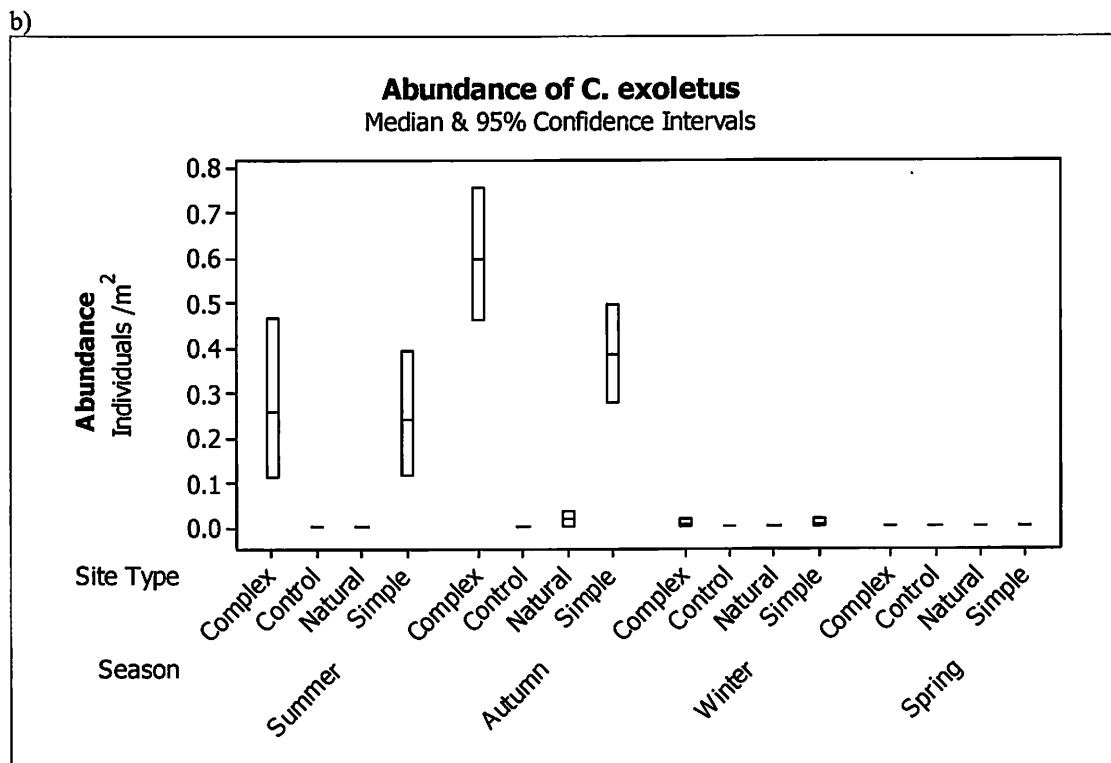
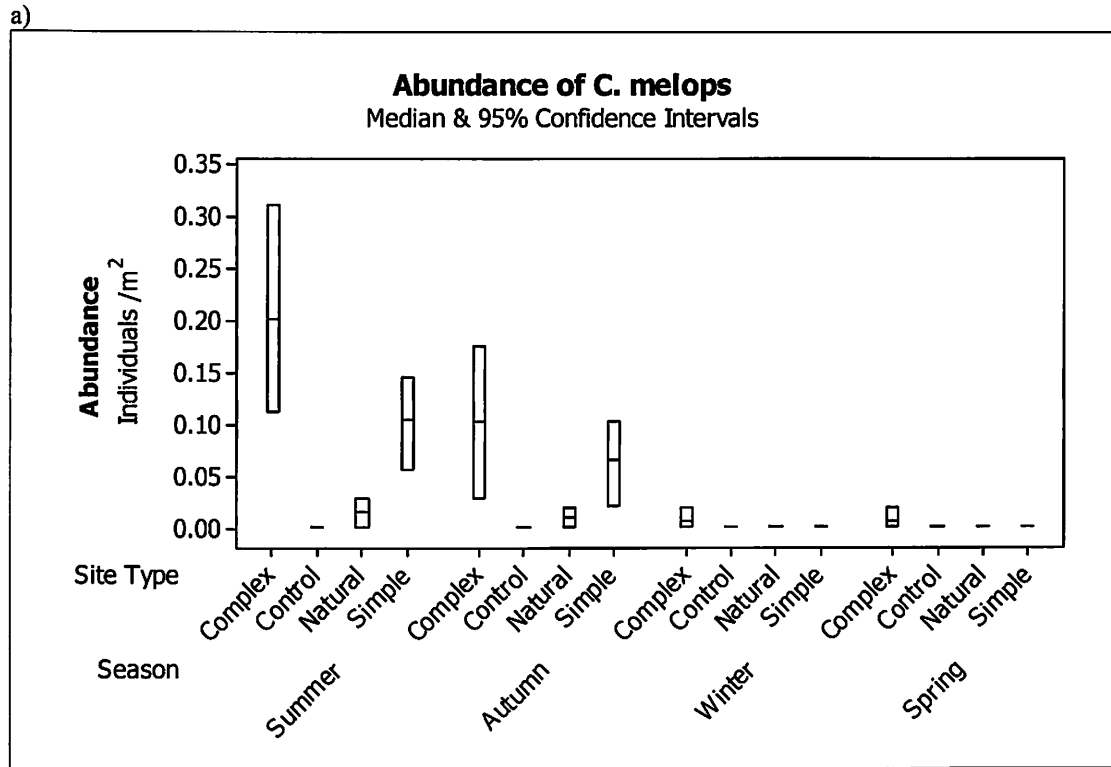
Fig 3.2 Boxplot of the Shannon-Weiner Diversity Index (H') (median & 95% confidence intervals) calculated at each site (complex artificial reef, simple artificial reef, natural reef and control), during each of the summer, autumn, winter and spring periods.



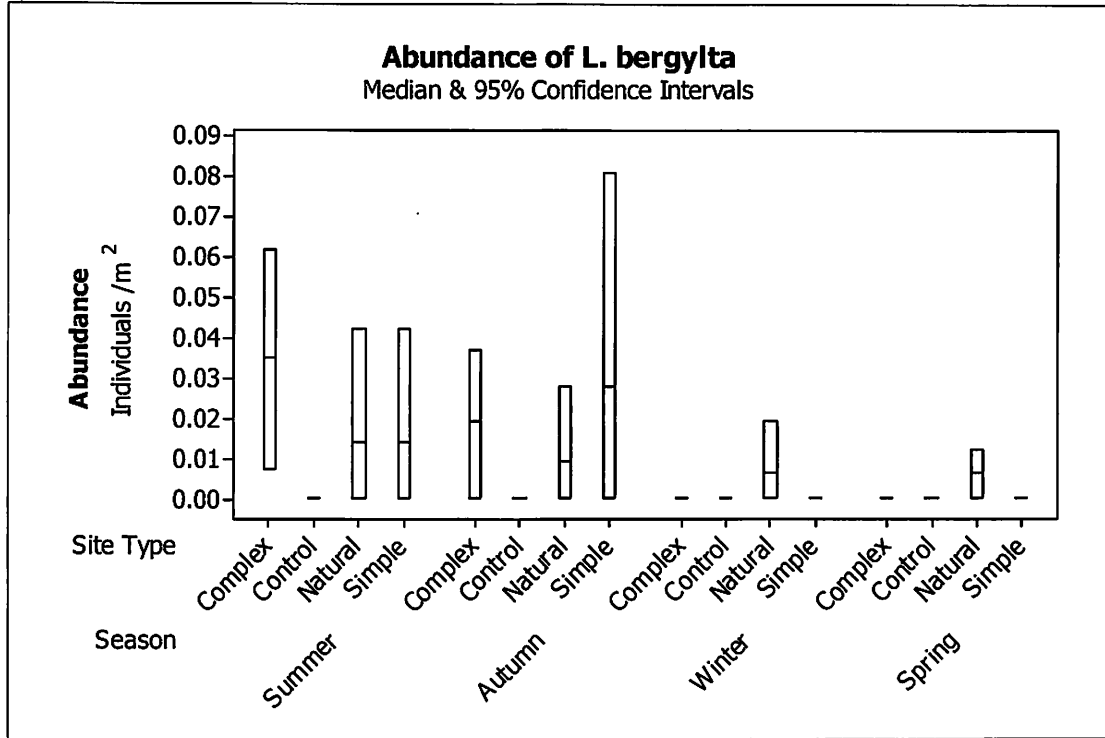
It was observed that there were no major changes in the level of biodiversity with season (Fig 3.2). Analysis of variance showed significant differences in the diversity indices across the four seasons ($p < 0.001$). This was then further investigated by Fisher's pairwise comparisons showing that during the summer and autumn periods the complex artificial reef site tended to have a higher diversity than the natural reef and control sites. The simple artificial reef site fell intermediate between the complex and natural sites, exhibiting no significant difference from either. During the winter and spring periods all reef sites were significantly higher than the control site but the complex artificial reef site had significantly higher diversity than either the simple artificial reef site or natural reef sites.

Labridae

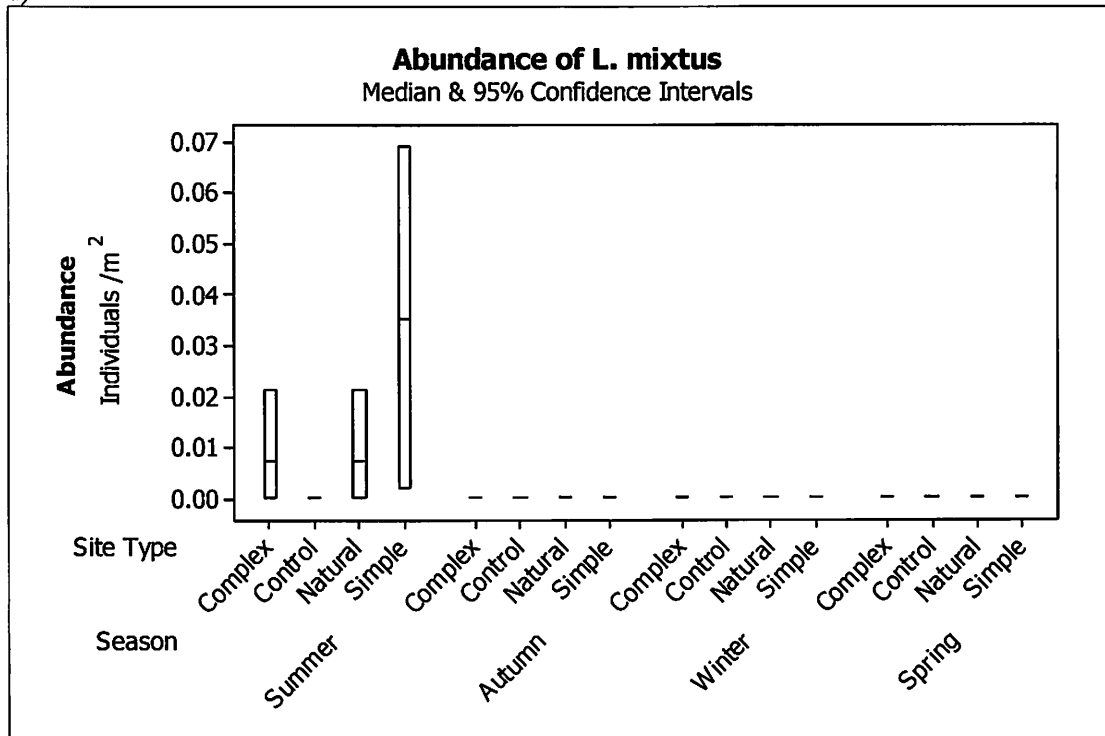
Fig 3.3 Boxplots showing the relative abundance of: a) the corkwing wrasse (*Crenilabrus melops*) b) the rock cook (*Ctenolabrus exoletus*) c) the ballan wrasse (*Labrus bergylta*) d) the cuckoo wrasse (*Labrus mixtus*) e) the goldsinny (*Ctenolabrus rupestris*) over the four seasonal regimes.

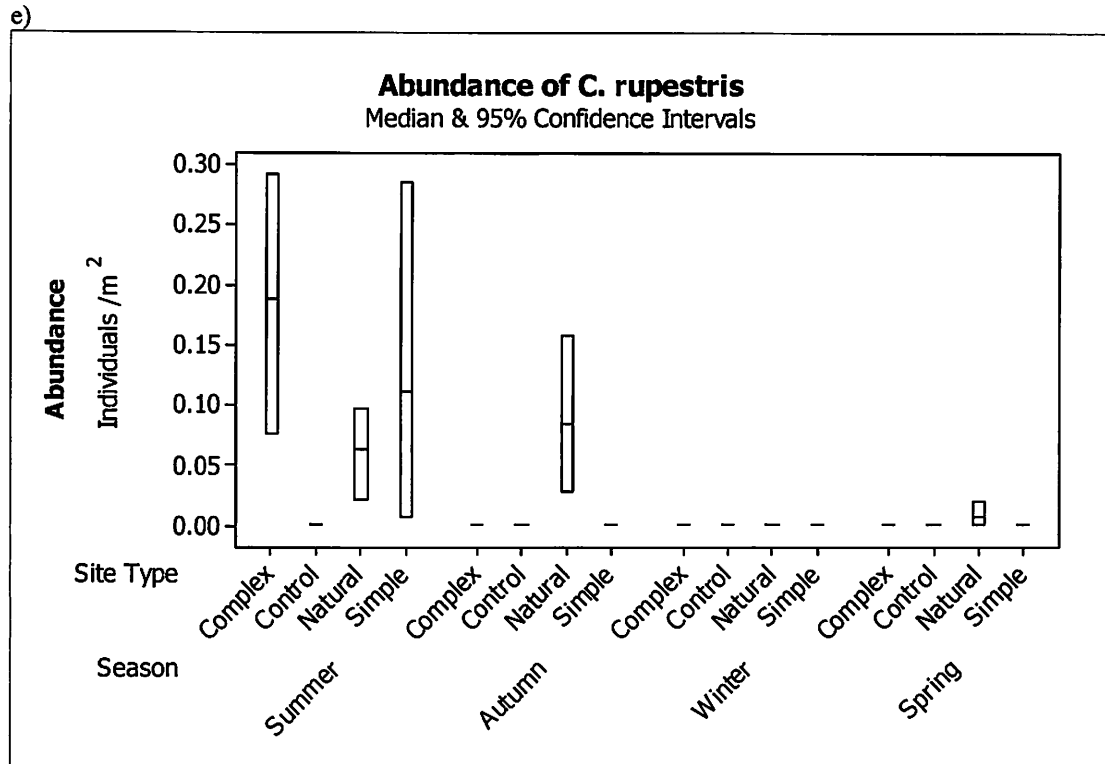


c)



d)





All of the wrasse species investigated during this study only occurred on the hard substrate reef sites. The abundances of ballan wrasse (*Labrus bergylta*) (Fig 3.3c) and cuckoo wrasse (*Labrus mixtus*) (Fig 3.3d) were extremely low and no significant difference was observed between site types for these species.

The corkwing wrasse showed a distinct seasonal trend. The highest levels of abundance were observed during the summer season, declining to an effective zero value in the winter (Fig 3.3a). In the summer period there was a significantly higher level of abundance at the complex artificial reef site ($p=0.006$). During the autumn this had dropped to a level at which no significant difference was observed, which continued during the winter and spring periods.

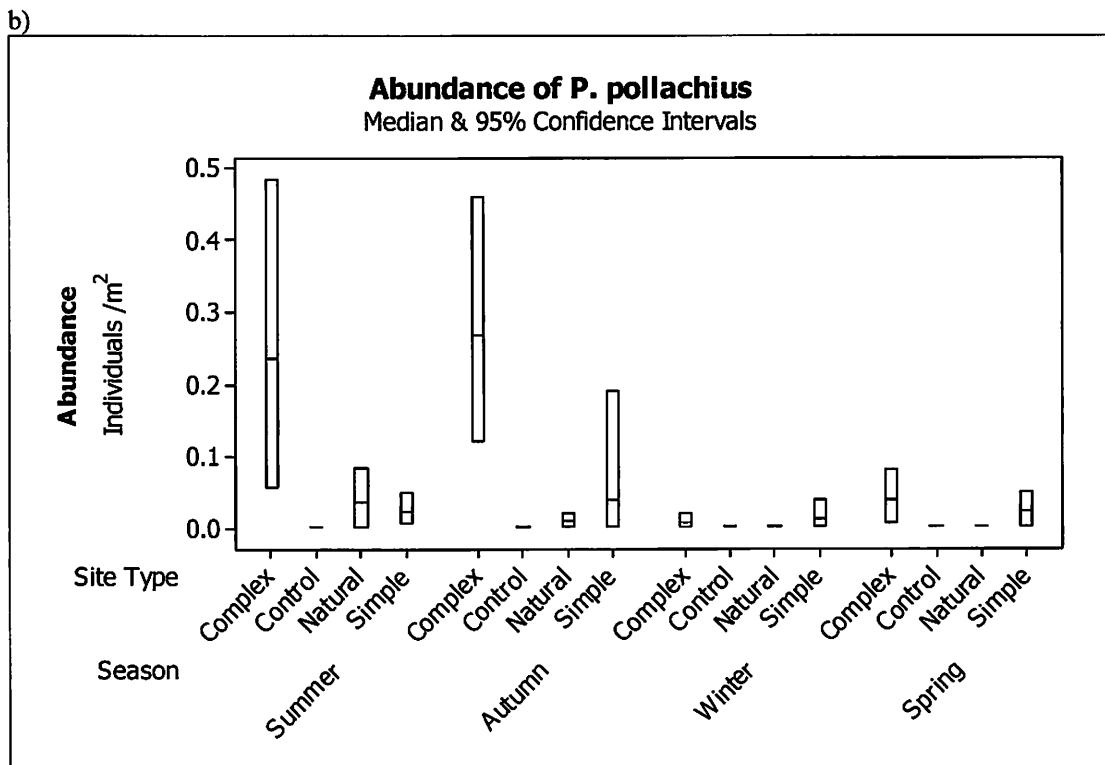
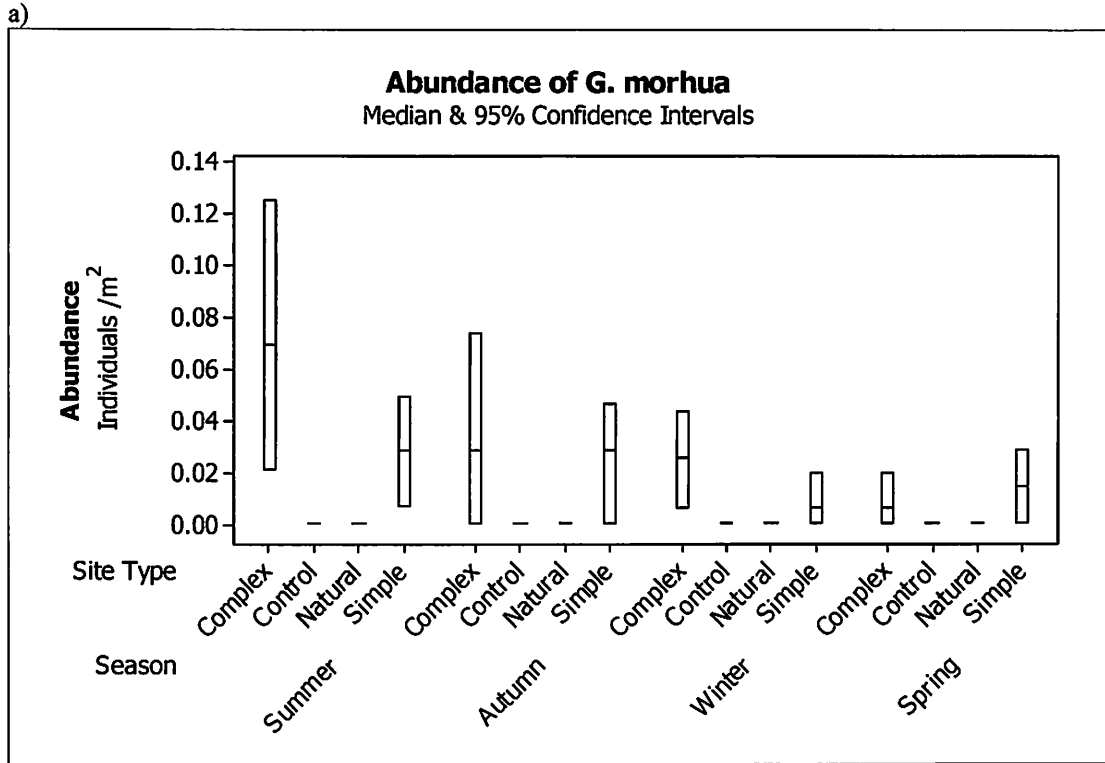
The rock cook (fig3.3b) showed a slightly different trend in seasonal abundance. During the summer period significantly higher levels of abundance were observed on

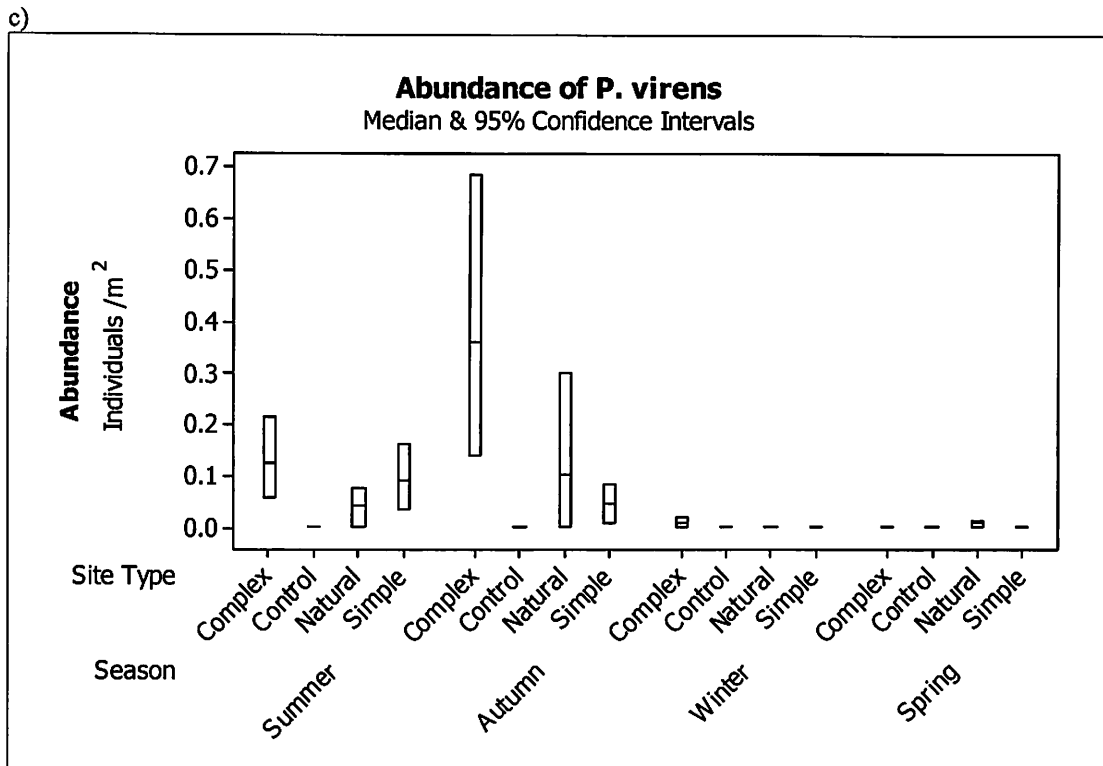
the complex and simple artificial reef sites ($p=0.024$) compared with a very low level associated with the natural reef site and a zero value at the control site. The abundance at the artificial reef sites then increased significantly in the autumn period ($p<0.001$). Pairwise testing showed the complex artificial reef site to support a higher level of abundance than all other sites, with the simple artificial reef site showing higher levels of abundance than either the natural or control site. In winter and spring periods observations of the wrasse were scarce and so no significant difference between sites was observed.

The goldsinny wrasse (fig 3.3e) showed a comparatively low level of abundance overall. However, significantly higher levels of abundance were observed on the natural reef site ($p=0.022$) during the autumn period, when compared with the artificial reef sites.

Gadidae

Fig 3.4 Boxplots showing the relative abundance of: a) juvenile cod (*Gadus morhua*) b) juvenile pollack (*Pollachius pollachius*) c) juvenile saithe (*Pollachius virens*) at each site type, over the four seasonal regimes.





Juvenile cod (fig 3.4a) showed low levels of abundance across all the site types and seasons. It was observed that there were significantly greater numbers on the complex artificial reef site compared with other sites during the summer period ($p=0.042$).

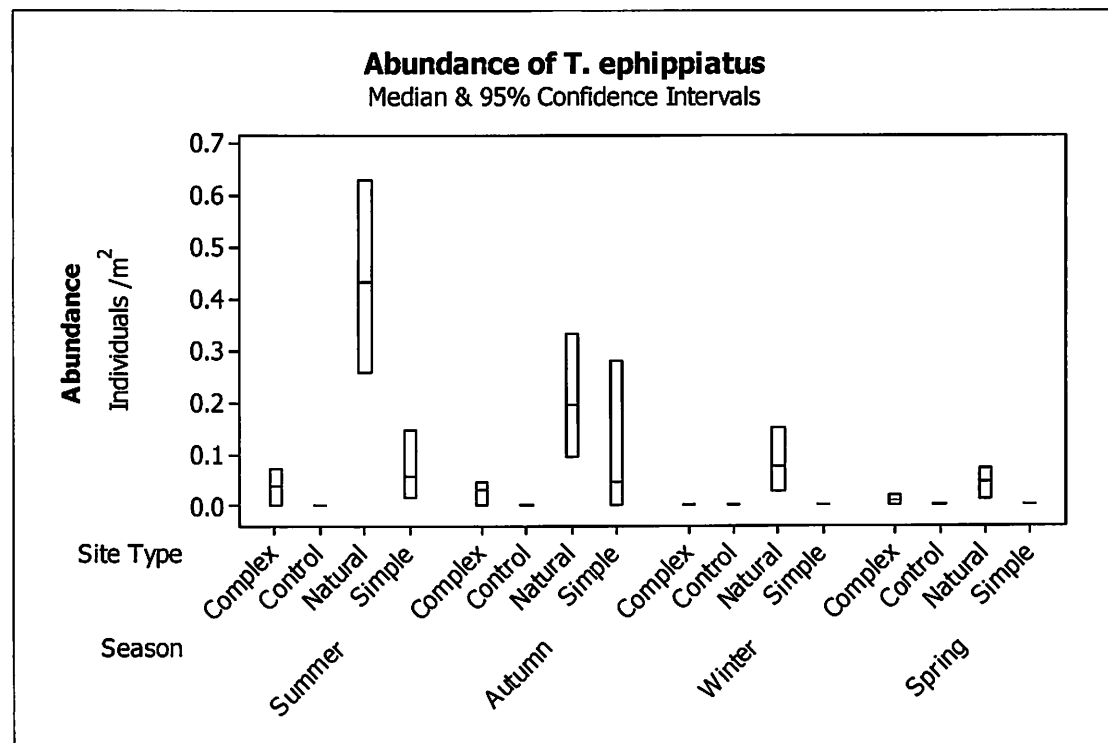
There were no significant differences were observed in cod abundance between sites in the autumn, winter and spring periods.

Juvenile pollack (Fig 3.4b) showed a similar seasonal trend to the cod, but there was no significant difference in abundance between the site types during the summer period. During the autumn, however, it was observed that there were significantly higher numbers of juvenile pollock on the complex artificial reef site than the other site types ($p=0.022$). No significant difference was observed between the site types in winter and spring.

No significant differences were observed in the abundance of juvenile saithe (Fig 3.4c) between the sites types during any of the seasonal periods.

Gobioidea

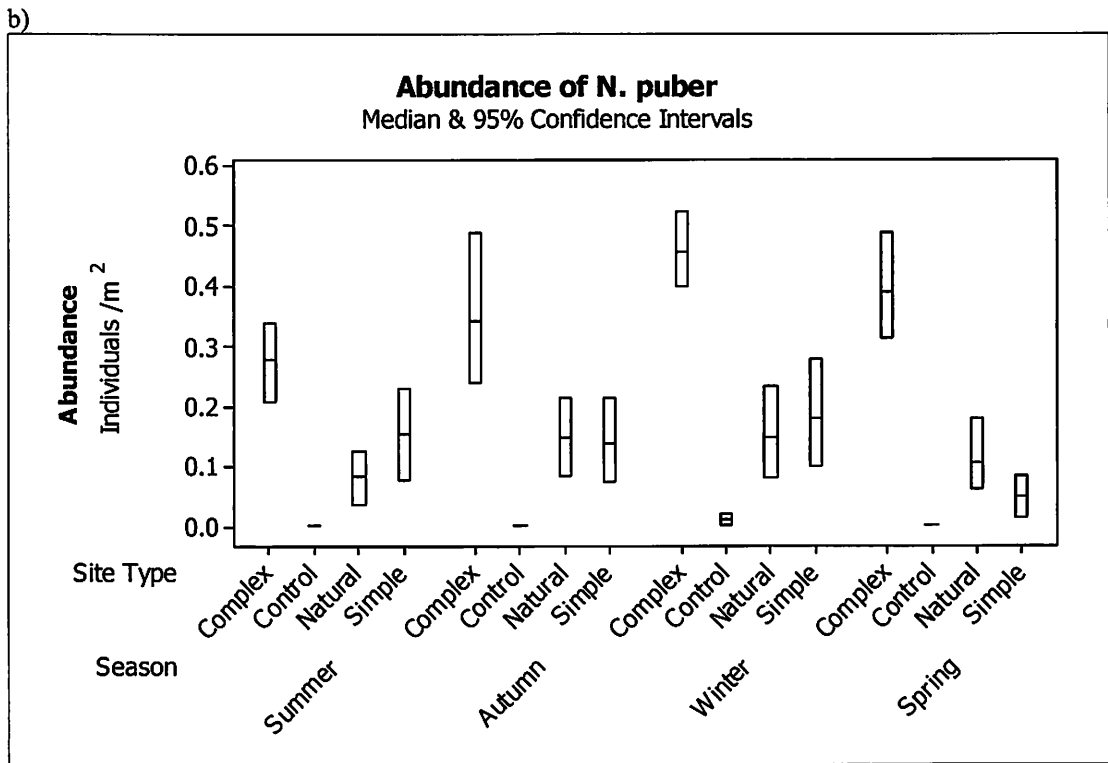
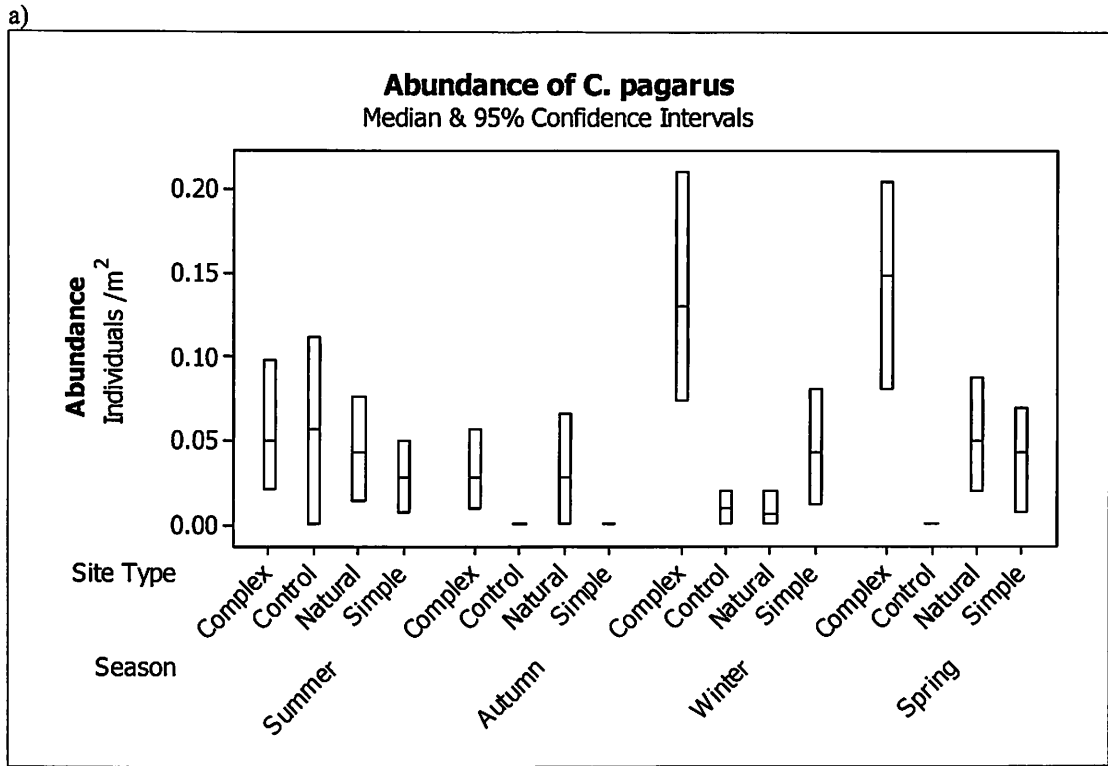
Fig 3.5 Boxplot showing the relative abundance of the leopard spotted goby (*Thorogobius ephippiatus*) at each of the our site types, during the seasonal regimes.

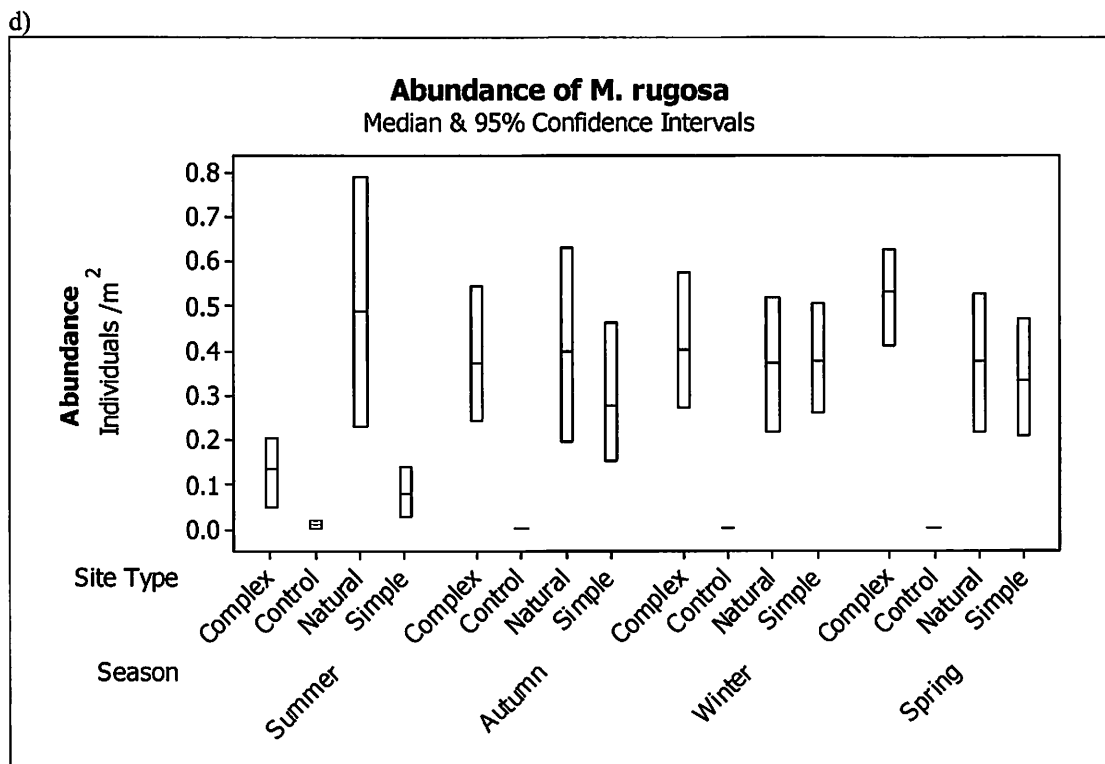
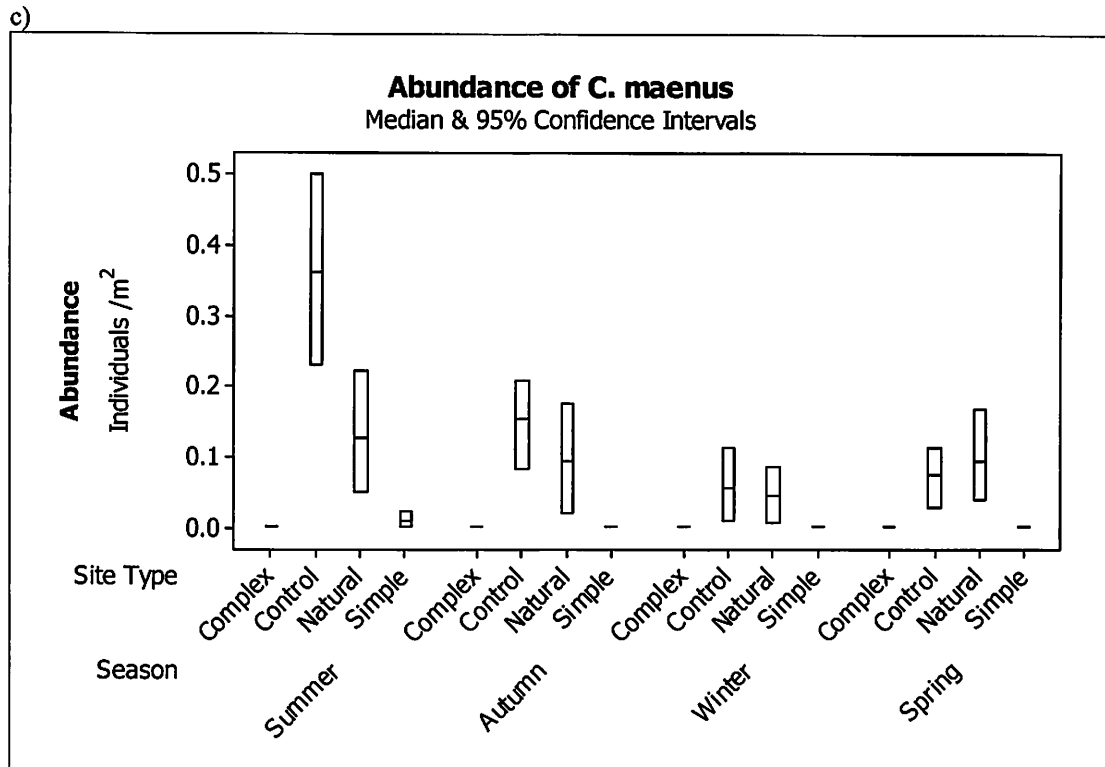


The leopard spotted goby showed a similar trend to the other fish species observed as recorded numbers were highest during the summer period and declined during the autumn to a low baseline in the winter (Fig 3.5). It was observed for this species that a significantly higher level of abundance was present on the natural reef site during the summer period ($p=0.002$) and spring period ($p=0.031$). No significant differences were observed between the four site types during the autumn or winter.

Crustacea

Fig 3.6 Boxplots showing the relative abundance of: a) the edible crab (*Cancer pagarus*) b) velvet swimming crab (*Necora puber*) c) shore crab (*Carcinus maenus*) d) long clawed squat lobster (*Munida rugosa*) at each site type over the four seasonal periods.





The edible crab (Fig 3.6a) showed low but fairly consistent levels of abundance across all sites in the summer period of the study with the numbers declining at the simple artificial reef and control sites in the autumn. No significant difference was observed

between the four site types during either the summer or autumn periods. During the winter however there was a significantly higher level of abundance recorded at the complex artificial reef site ($p=0.011$), compared with the other site types. This continued in the spring period, with the complex artificial reef showing significantly higher levels of abundance ($p=0.011$) than the other three site types.

The velvet swimming crab showed a relatively consistent level of abundance at each of the three reef sites during all four seasonal periods (fig 3.6b). The complex artificial reef site supported a significantly higher level of abundance than the simple or natural reef sites during the summer ($p=0.002$), autumn ($p=0.004$), winter ($p<0.001$) and spring ($p<0.001$). Additionally it was observed by Fisher's pairwise test that during the spring period the natural reef exhibited a significantly higher level of abundance than the control site.

The shore crab (fig 3.6c) exhibited significantly higher levels of abundance at the control site during the summer ($p=0.002$) and autumn ($p=0.016$) before declining to low levels in winter where no significant difference was observed. In the spring period however the natural reef site showed a significantly greater abundance ($p=0.029$) than the other three site types.

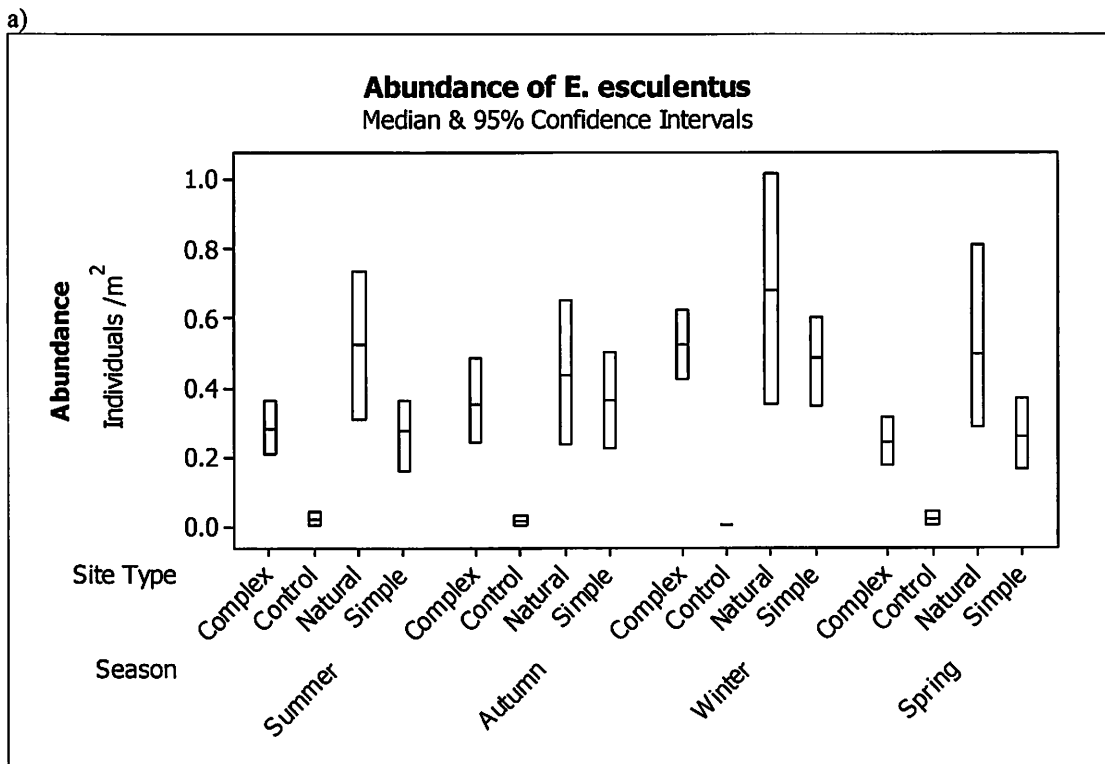
The abundance of the long clawed squat lobster was significantly higher ($p=0.015$) during the summer period on the natural reef sites, when compared with the other three site types. However, during the autumn ($p=0.041$), winter ($p=0.011$) and spring ($p=0.003$) periods although there was a significantly higher level of abundance upon the reef sites compared with the control site. There was no significant difference

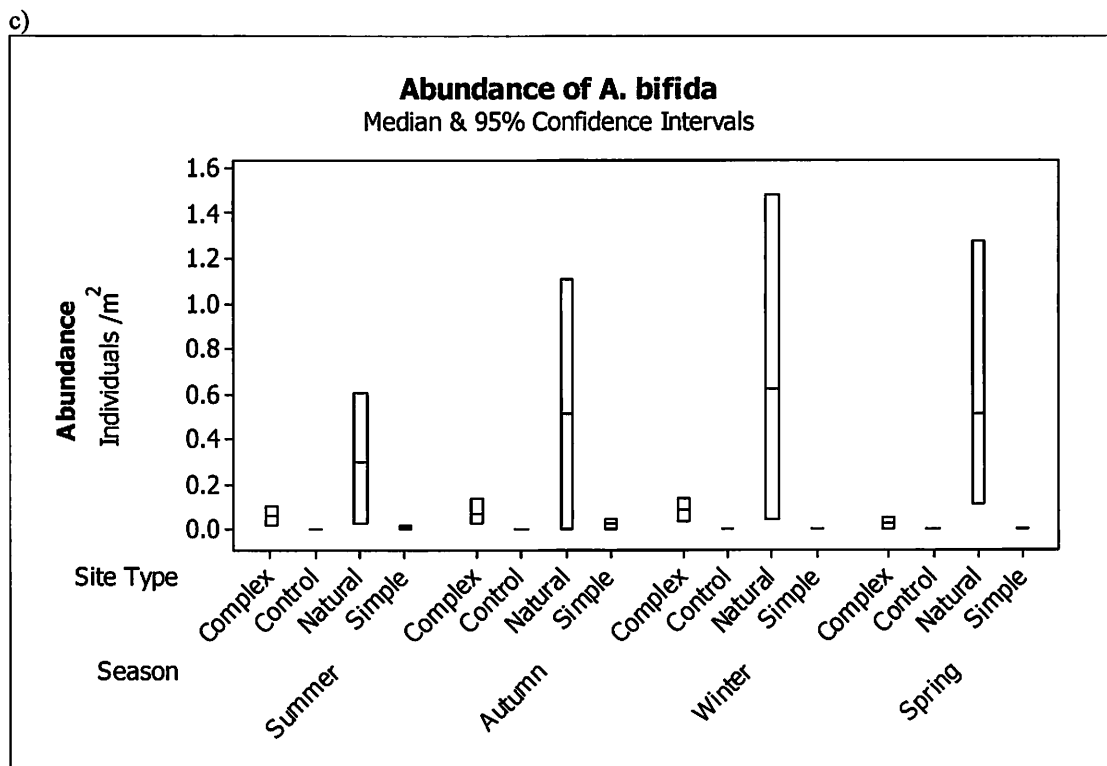
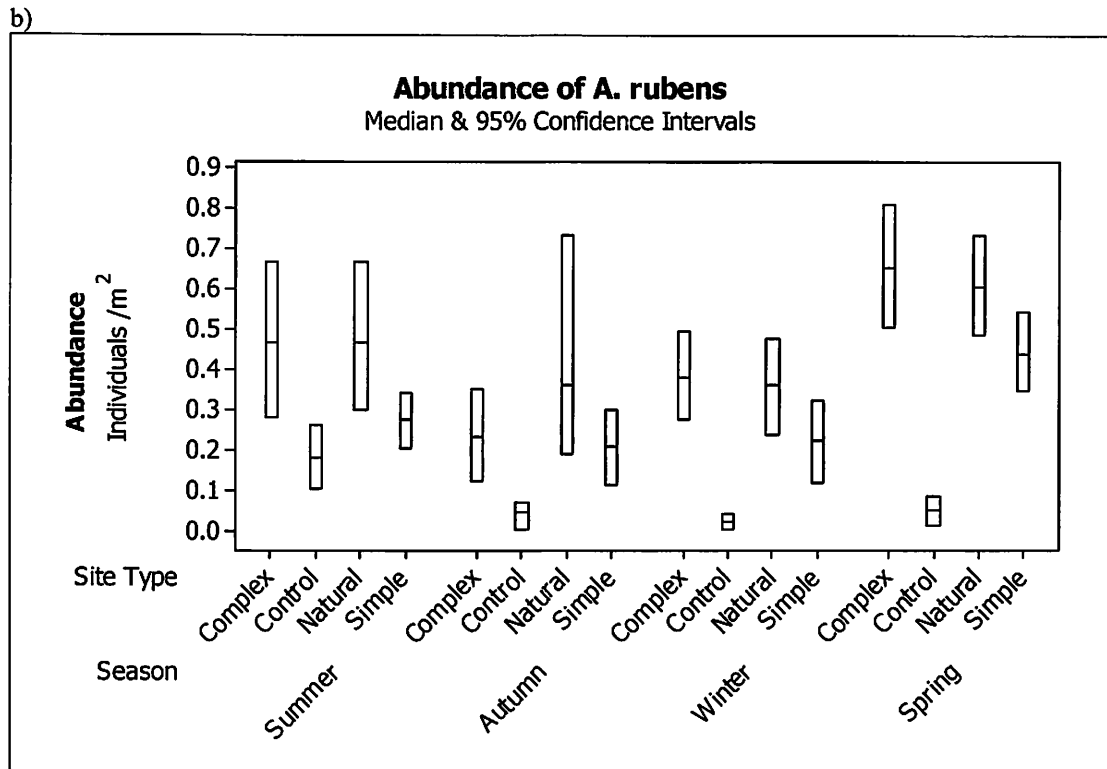
between the three reef sites investigated. This species showed no seasonal trends in abundance levels. (Fig 3.6d).

Very small numbers of individuals of European lobster (*Hommarus gammurus*) only occurred at the natural reef site station Eilean Mhor. Thus no significant difference was observed between site types for this species and no meaningful graphical display of the data could be made.

Echinodermata

Fig 3.7 Boxplots showing the trends and distribution of the abundance of a) the sea urchin (*Echinus esculentus*) b) the starfish (*Asterias rubens*) c) the crinoid (*Antedon bifida*) at each of the four sites, across the summer, autumn, winter and spring periods.





The sea urchin (fig 3.7a) showed significantly higher levels of abundance at the reef sites compared with the control site during the summer ($p=0.009$), autumn ($p=0.023$), winter ($p=0.010$) and spring ($p=0.019$). It was observed by Fisher's pairwise test that

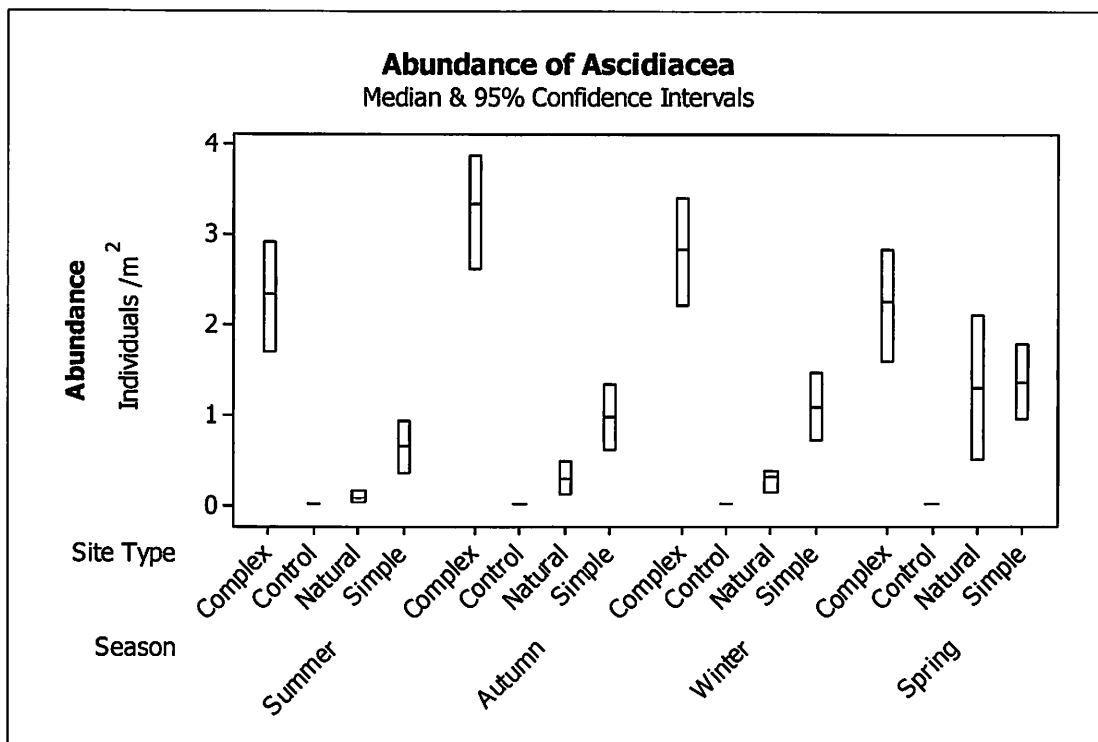
sea urchin abundance was significantly higher on the natural reef site in the summer and spring periods, with no significant difference between the two artificial sites. During the autumn and winter periods there was no significant difference between the levels of abundance observed on the three reef types.

The common starfish (Fig 3.7b) showed no significant differences in abundance between the three reef sites and the control site during the summer and autumn periods. During the winter and spring period the three reef sites exhibited significantly higher levels of abundance ($p=0.006$ & $p=0.001$) than the control site. In the spring period Fisher's pairwise test showed the abundance on complex artificial reef sites to be significantly greater than the simple artificial reef sites.

The crinoid (Fig 3.7c) was found in large numbers only on the natural reef site station Eilean Mhor. Small numbers of individuals were also recorded throughout the year on the complex artificial reef site. No seasonal trends were observed for this species and no significant differences between the four site types were measured.

Tunicates (Ascidiacea)

Fig 3.8 Boxplot showing the levels of abundance of the tunicates (Class- Ascidiacea) on the four site types, during the summer, autumn, winter and spring periods.

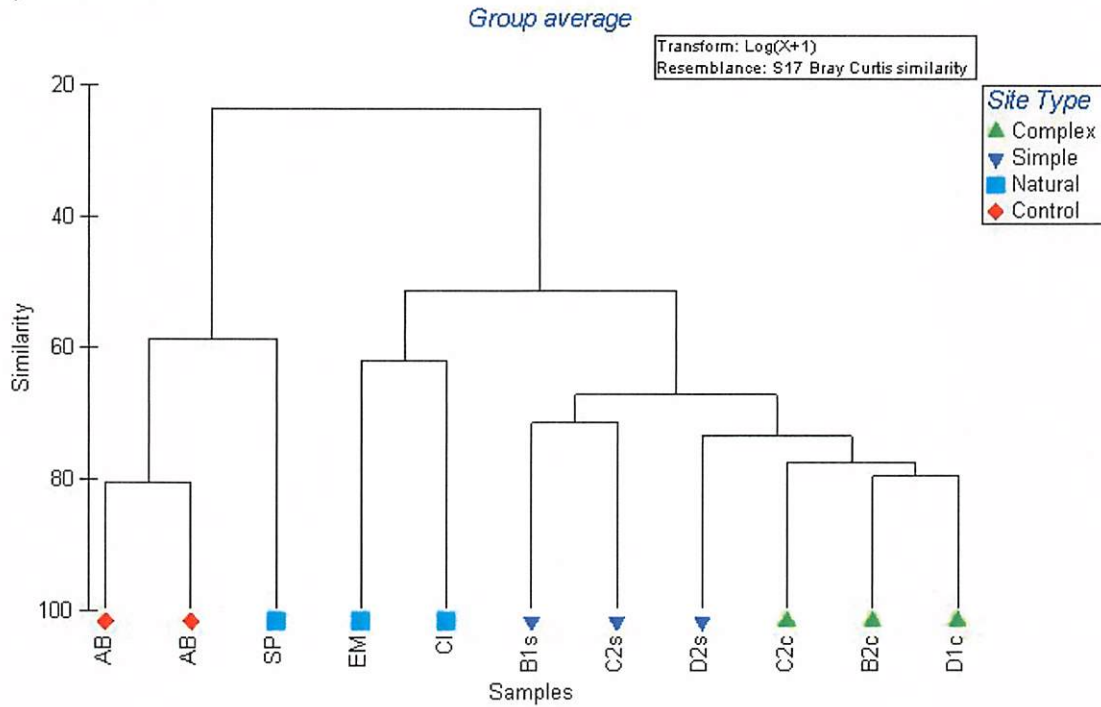


It was observed that over the four seasons the abundance of tunicates was significantly higher on the complex artificial reef site ($p < 0.001$). No significant seasonal trends were apparent for this animal group (Fig 3.8).

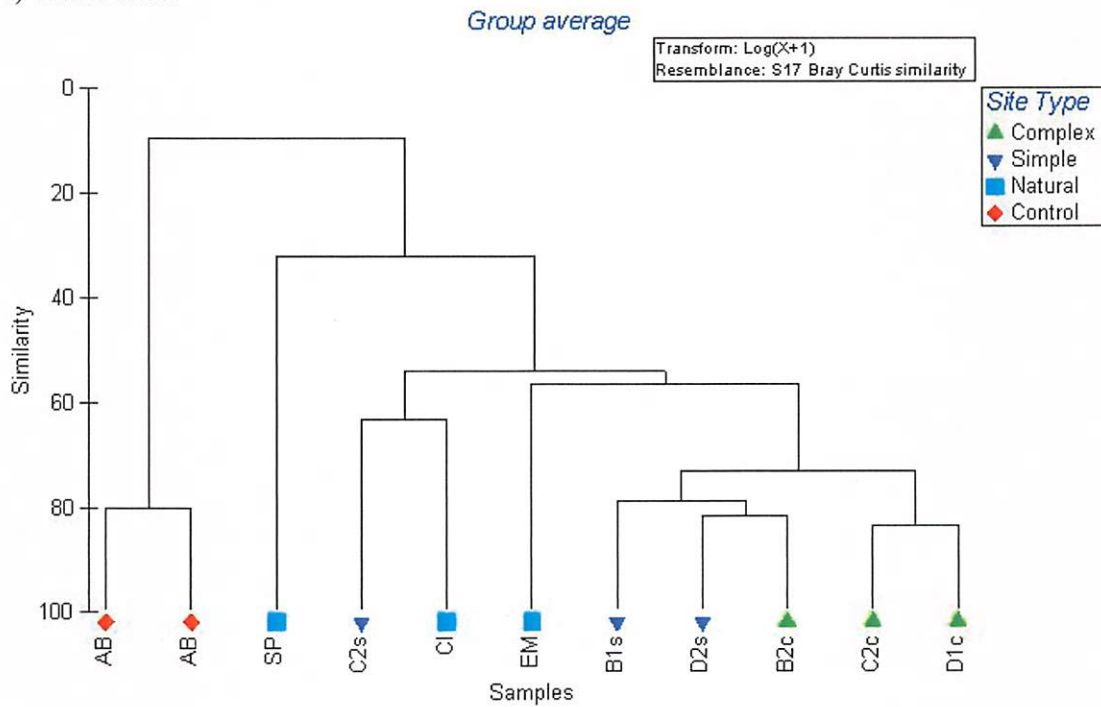
Cluster Analysis

Fig 3.9 Cluster Analysis diagrams showing the level of similarity between the stations of each site type (Complex artificial reef, simple artificial reef, natural reef and control) for each of the four seasons.

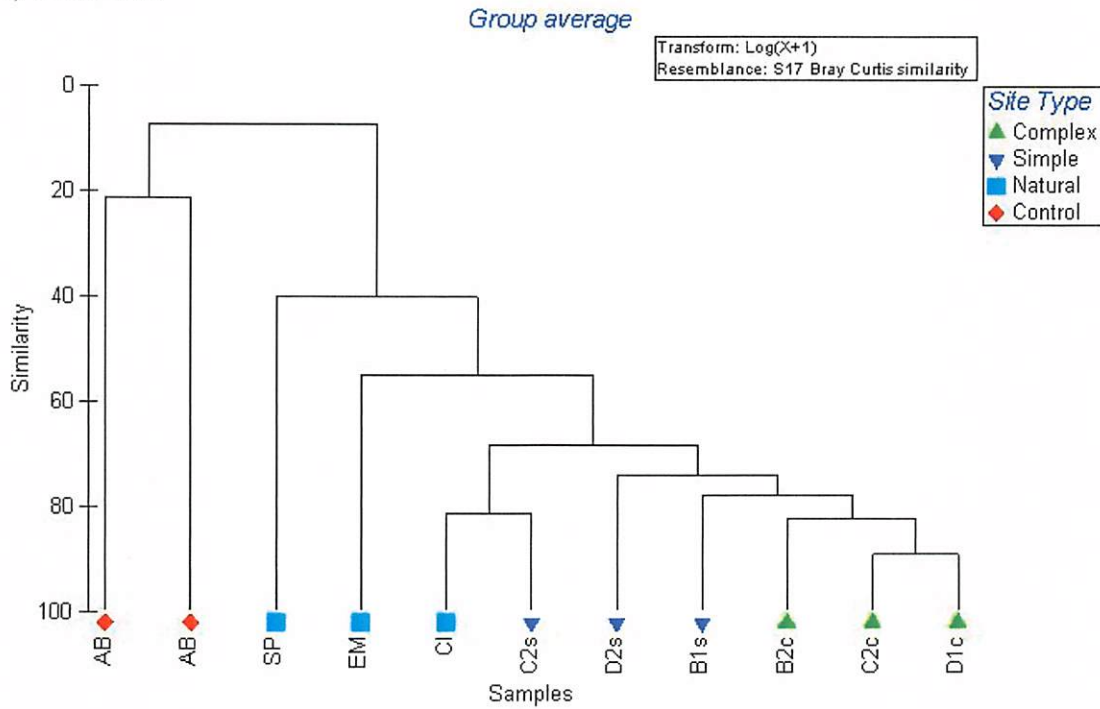
a) Summer 2005



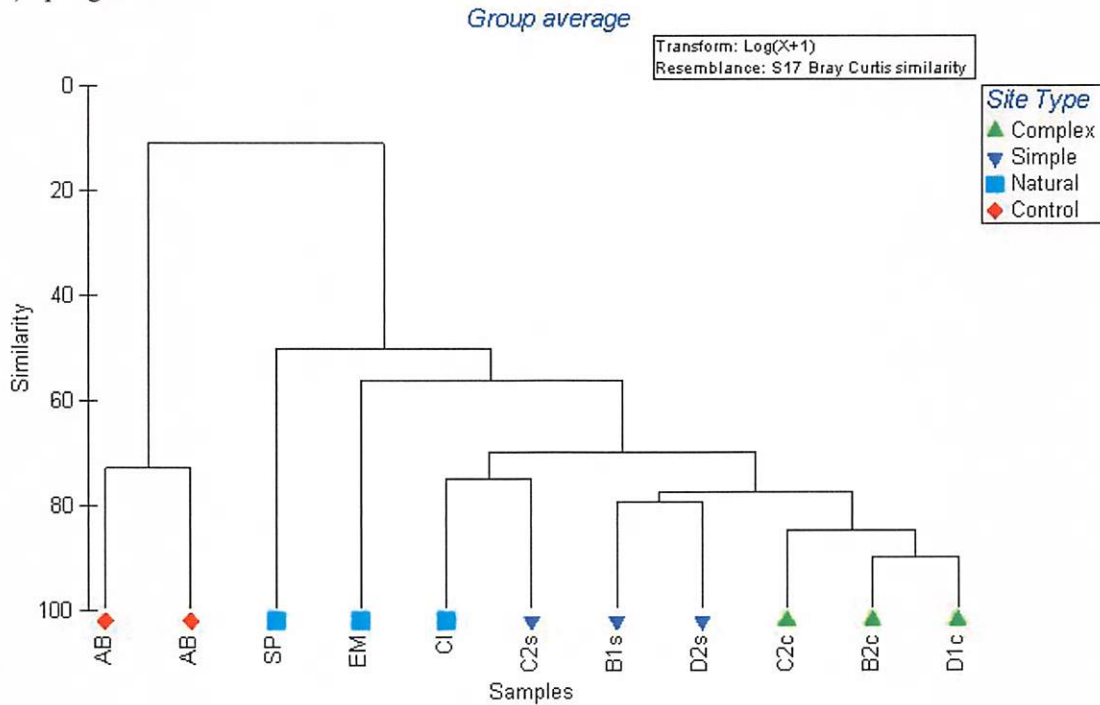
b) Autumn 2005



c) Winter 2006



d) Spring 2006



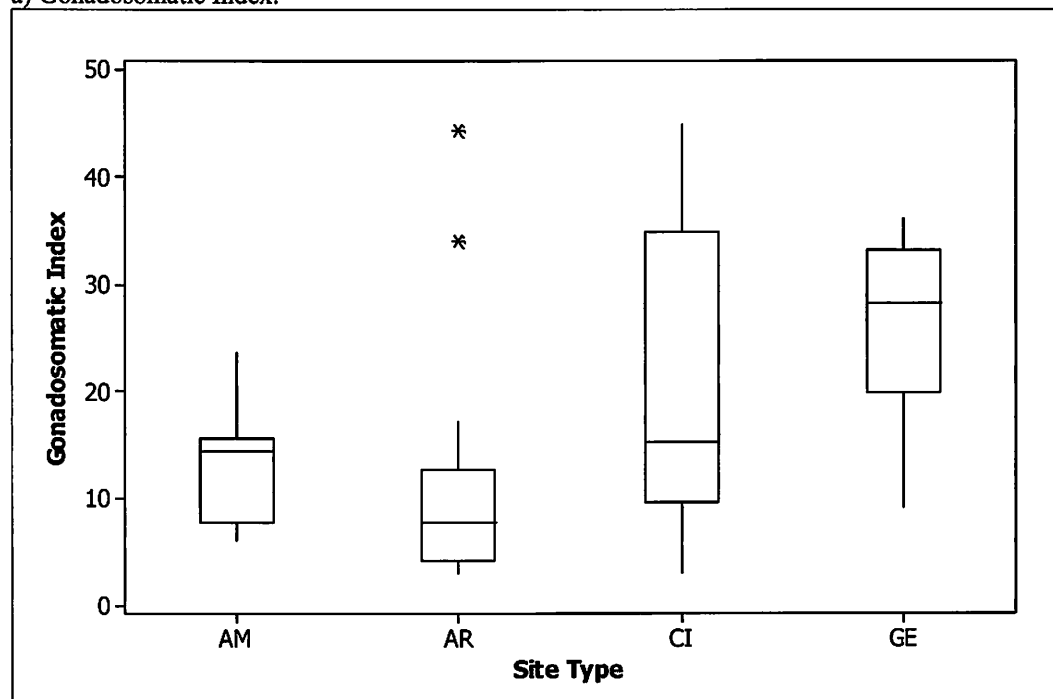
Cluster analysis was carried in order to assess the level of similarity between the four site types. Figure 3.9 shows the dendrograms formed from this cluster analysis. These show that the faunal aggregations observed on the three reef sites exhibit higher levels of similarity to each other, typically between 40 & 50%, than they do with the control

site. It was observed that in the summer (Fig 3.9a), winter (fig 3.9c) and spring (fig 3.9d) periods, the complex artificial reef sites tend to cluster into a single homogenous group, with around 80% similarity. Also, it was observed that overall the simple artificial reef sites tend to fall as intermediates in similarity between complex artificial reef sites and the natural reef sites. The natural reef sites tend to form the most heterogeneous group, with the lowest similarity of 20% observed in summer (Fig 3.9a) and highest level of similarity observed in spring (Fig 3.9d). The highest level of heterogeneity is exhibited by the control sites during the winter period (Fig 3.9c), however the control sites showed similarity between 70 & 80% in the summer, autumn and spring periods.

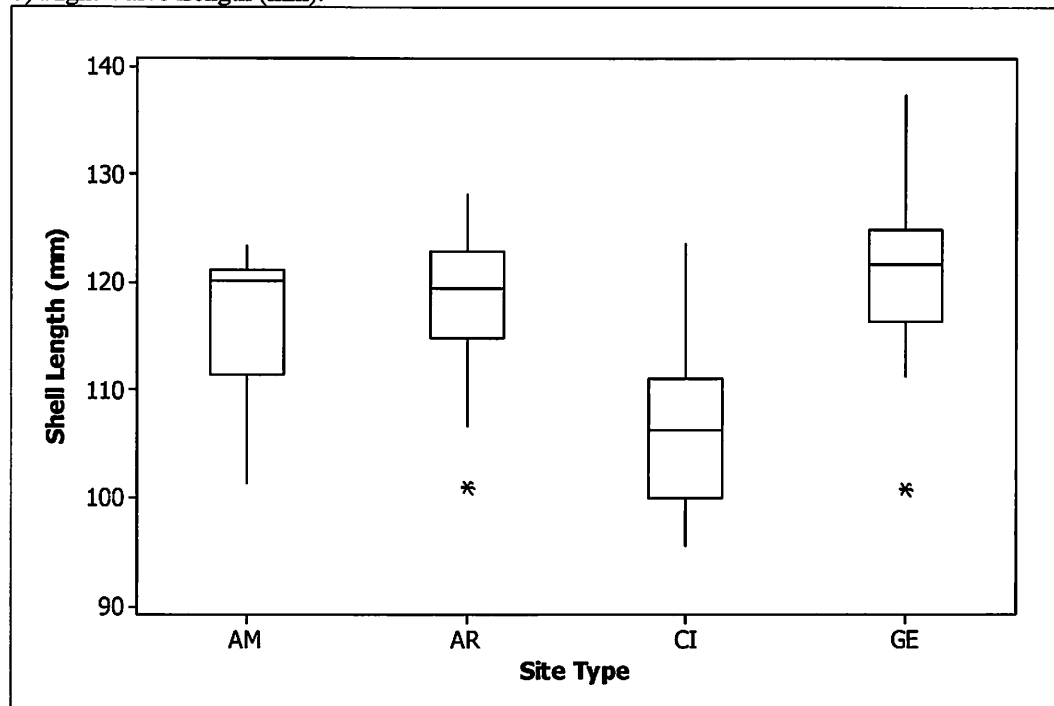
3.2 REPRODUCTIVE INVESTMENT IN SCALLOPS

Fig 3.10 Boxplot showing the mean and inter-quartile range for: a) the Gonadosomatic Index and b) the right valve length (mm) of specimens of the scallop (*Pectens maximus*) collected at Loch Linnhe Artificial Reef site (AR), which is protected from fishing, and three unprotected sites at the Creag Isles (CI), Ardnamucknish Bay (AB), and Glas Eilean (GE).

a) Gonadosomatic Index.



b) Right Valve Length (mm).



The GSI (Fig 3.10a) and right valve length (Fig 3.10b) were recorded and plotted for specimens collected from each site. One way ANOVA and Fisher's pairwise tests were used to examine the data and it was observed that the scallops collected at the Glas Eilean site had a significantly higher level of Gonadosomatic Index than those collected at either the artificial reef, Creag Isles or Ardnamucknish Bay sites ($p < 0.001$). Additionally it was noticed that the average valve length of the scallops collected at the Creag Isles was significantly less than at the other sites ($p < 0.001$), with no significant differences between the other sites.

4. DISCUSSION

The present study has shown that geometric habitat complexity has a significant impact on faunal abundance and diversity in the rocky subtidal. Overall the faunal

abundance on high complexity artificial reefs was 2-3 times higher than that on natural reefs or low complexity artificial reefs, in the Lynn of Lorne. However, for mobile fauna as a single group, it was observed there was no difference between the artificial and natural reef sites, yet abundance of individual species differed greatly. Also the present study has shown that although the Loch Linnhe Artificial Reef area has been closed to scallop fishing for over five years there has been no obvious improvement of the condition of scallops in that area. Scallops sampled from unprotected sites were in better condition than those from the reef site, suggesting that the artificial reef may not be located in an area where conditions are optimal for scallop growth.

4.1 COMPARISON OF ARTIFICIAL & NATURAL REEFS

Non-destructive ecological sampling presents a number of problems in the sub-littoral environment because many external factors can affect the survey, e.g. tidal conditions, underwater visibility, weather etc. These multifactorial influences have led to a large amount of research has been devoted to the development of effective survey techniques. The present study utilised the belt transect method of visual census (Brock 1954). This method has come to be one of the most common methods of underwater visual census as it provides a reliable method to obtain quantitative abundance data (Kimmel, 1985) by restricting the researcher to a distinct set of boundaries. Census was carried out *in situ* as previous research suggests this provides a better estimate of abundance than video-recording and playback (Tessier, *et al.*, 2005).

Two major experimental design issues were encountered during the present study.

The first of these was attempting to prevent excessive pseudoreplication, as has been observed in previous work (Kock, 1982; Bortone, *et al.*, 1994; Jensen, *et al.*, 1994).

Secondly, dealing with heterogeneity of variance within the sample data was essential to allow inferential statistical tests to be carried out.

Some pseudoreplication was encountered within the present study, even though much was done to minimise its effect. The Loch Linnhe artificial reef essentially provides a pseudoreplicated experimental matrix, with different treatments replicated a number of times within each reef group, and the overall reef covering 146 hectares of seabed. Experimental sites were chosen so that only one replicate of both the complex and simple artificial reef treatments were located in any specific reef group. Additionally the control site provided a pseudoreplicated site as all four transects were located at the same station. This was caused by problems finding sites which were suitable in terms of depth and also provide a sheltered and easily found location.

Heterogeneity of variance within the data was found to be an inevitable consequence of sampling mobile fauna. As a statistical test, the analysis of variance is reasonably robust against data that are not normal, but heterogeneous variance remains a major problem (Underwood, 1997). Increasing the size of the sampled area has been found to increase precision (Sayer & Poonian, 2006); however, area size has always to be matched against logistical considerations. As such it was found that bootstrap resampling (Diaconis & Efron, 1983) was the most effective way to obtain data with homogeneous variance.

The levels of abundance recorded during any ecological sampling can also be problematic. These values can only be considered in relative terms for each study. This is because the observer/sampler will have to contend with a number of behavioural and environmental factors which can make determining true abundance difficult. In animal based experiments the most obvious of these is activities. Different species will exhibit different behavioural patterns ranging from cryptic to curious, which can skew the data in either a negative or positive manner (Nash, *et al.*, 1984). Additionally within species, individuals of different age-classes or sex can behave in different ways, which can make them easier or harder to sample. Factors such as seasonal activity and migration can also have profound effects. Therefore, the present study was only able to record relative estimates of abundance values at each site, based on animal activity.

It has been shown that biological communities will change with time (Perkol-Finkel *et al.*, 2005). Thus in comparing the developing communities at the artificial reef sites with much longer established communities on the natural reef sites, the present study is subject to a potential confounding factor. It is often assumed that biological diversity will increase as the initial pioneer community changes and is succeeded by subsequent communities over time (Dickenson & Murphy, 1998). The present study examined both biological diversity and used a Cluster analysis to examine the level of similarity between the artificial and natural reefs, to illustrate the dynamic nature of the ecosystems in their present state of development and further inform the interpretation of the results.

Previous research has shown increased habitat complexity to have a positive effect upon the abundance of fish (Fujita, *et al.*, 1996; Charbonnel, *et al.*, 2002), species richness (Fabi & Fiorentini, 1994; Charbonnel, *et al.*, 2002, Gatwick & Speight, 2005) and species diversity (Roberts & Ormond, 1987, Fabi & Fiorentini, 1994). However, these studies only target a variety of fish species and as such do not adequately represent the whole communities which develop upon rocky subtidal habitats.

The present study provides a more holistic examination of macrofaunal assemblages that are found upon artificial and natural reefs, examining a cross-section of fish and benthic macrofauna, including both mobile and sedentary organisms. The complex artificial reef sites showed the highest level of geometric habitat complexity compared with the other four sites (Rose 2005). Species diversity (H') was observed to be highest at these sites. However, when the mobile organisms were examined, no differences in diversity were observed between the artificial and natural reef sites. Not all species were observed to exhibit their highest levels of abundance at the complex artificial reef sites. As such, each species can be categorised individually as to whether it was successful on either the artificial or natural reef sites.

The fish species observed in the present study showed the highest levels of abundance during the summer and autumn periods, with only occasional observations during winter and spring periods. This is probably because of the behavioural responses of most of the fish to low water temperatures (Sayer *et al.*, 1996). It was observed that the corkwing wrasse (*Crenilabrus melops*) and rock cook (*Ctenolabrus exoletus*), showed higher levels of abundance on the complex artificial reef site compared with the simple artificial reef, natural reef and control sites. Similar observations were

made of the juveniles of the gadoids, cod (*Gadus morhua*) and pollack (*Pollachius pollachius*). These species show similar responses to habitat complexity as the fish assemblages examined in previous studies (Fujita *et al.*, 1996, Tupper & Boutilier, 1997, Charbonnel *et al.*, 2002, Gatwick & Speight, 2005), with the complex artificial reef sites supporting higher levels of abundance.

Interestingly the goldsinny wrasse (*Ctenolabrus rupestris*) and leopard spotted goby (*Thorogobius ephippiatus*) showed higher levels of abundance upon the natural reef sites. The goldsinny wrasse tends to be a more cryptic species than either the rock cook or corkwing wrasse. However, the goldsinny is a benthic-pelagic spawner and its eggs are among the most abundant fish eggs within the plankton (Darwall *et al.*, 1992). Thus the goldsinny would have been expected to be a primary settler on the artificial reefs. The leopard spotted goby is an extremely cryptic species, which was thought to be rare (Wheeler, 1969) until diving surveys showed it to be highly abundant on rocky reefs in the UK (Miller *et al.*, 1973). The present study suggests that although this species is hard substrate specific, habitat complexity is not the determining factor dictating leopard-spotted goby abundance.

The fish species are likely to be very sensitive to inter-annual variation, as fluctuations in ocean climate can have profound and complex effect on juvenile fish survival and recruitment. Magill and Sayer (2002) found winter seawater temperature to possibly act as an indicator of these variations, and further annual datasets are required to place the present data into a medium term context.

The present study provides a unique piece of research by using the reef to carry out *in situ* investigations into the effect habitat complexity has upon the abundance of a variety of crustacean species. The study showed habitat complexity had a profound effect upon the abundance of the edible crab (*Cancer pagarus*) and velvet swimming crab (*Necora puber*). It was observed that the abundance of the long clawed squat lobster (*Munida rugosa*) did not differ between the three rocky reef site types, and the shore crab (*Carcinus maenus*) was most abundant at the control site.

The edible crab showed interesting patterns of abundance across the four seasons. Abundance was two times higher upon the complex artificial reef sites during the winter and spring periods, compared with the other site types. The edible crab is a highly nomadic species (Bennett & Brown, 1983) and the females require soft sediment during the autumn, in which to burrow when spawning (Edwards, 1979). After this period the females seek out shelter where they remain for six to nine months (Howard, 1982). This corresponds with the winter and spring periods with the crabs observed to shelter in the cavities and crevices of the complex artificial reef stations.

The long clawed squat lobster (*Munida rugosa*) showed higher levels of abundance during the summer period upon the natural reef stations. However, during the rest of the year no significant differences were observed in abundance between the three reef site type types. This species tends to be cryptic in behaviour and are observed to actively takes refuge when approached by a diver (Hunter, pers obs.). Therefore, any observed differences are likely to be caused by behavioural responses.

The present study also provides a direct investigation into the effect of habitat complexity upon three echinoderm species; the edible sea urchin (*Echinus esculentus*), the common starfish (*Asterias rubens*) and the crinoid (*Antedon bifida*). Little research has been undertaken to illustrate how these species respond to habitat complexity but a recent study comparing epifaunal communities upon the Loch Linnhe artificial reef with natural reef sites (Beaumont, 2006) inferred that the increased grazing pressure upon epifauna on the artificial reef sites provided evidence of higher levels of abundance of the grazing sea urchin and starfish. The current study found no significant differences between the abundance of either the edible sea urchin or common starfish at the three reef sites. This suggests that these species are not strongly affected by habitat complexity but are simply more successful on hard substrates. The crinoid was only observed at a small numbers of the stations under survey. Thus no meaningful conclusions can be drawn about its abundance and distribution from the present study.

The abundance of ascidians was generally between two and three orders of magnitude higher at the complex artificial reef site than at the other sites. However, in the spring period their abundance upon the natural reef sites increased markedly. This would seem to indicate noise caused by inter-annual variation, as there was little change in abundance on the other site types. As a group, the ascidians provide an effective biotic indicator of habitat complexity, attaching to hard substrate where available. Also as suspension feeders they provide one of the important importers of nutrients into the reef ecosystem from the pelagic environment.

The Cluster analysis showed that the faunal assemblages upon the complex artificial reef sites tend to be grouped tightly together. This contrasts dramatically with the heterogeneous group formed from the simple artificial reef and natural reef sites. This in turn suggests that reefs constructed from the complex blocks tend to exhibit greater stability in community structure than the other site types and as such, are more likely to maintain productive animal assemblages. However, the evidence also suggests that although habitat complexity does have a significant effect on the faunal assemblages of a rocky reef, many other variables can mask this. In particular the behavioural responses of many species to intrusion by the investigator can lead to over- or under-estimation of abundance (Nash *et al.*, 1984; Lincoln Smith, 1989).

4.2 REPRODUCTIVE INVESTMENT IN SCALLOPS

The comparison of how the fishing exclusion afforded by the Loch Linnhe artificial reef site affects the condition of the scallops (*Pectens maximus*) on the surrounding soft sediment was a relatively limited study. Sample numbers were small at each site because of low scallop abundance at the Loch Linnhe artificial reef, which led to highly labour intensive specimen collection. Unlike studies by Kaiser *et al.* (2000, 2002), dredging could not be used to gather samples because of the risk posed by the artificial reef. Therefore, all specimen collection was undertaken using scientific diving techniques.

The results of the scallop study show gonadal investment to be significantly higher at one of the unprotected sample sites, Glas Eilean. Specimens collected at the reef site showed similar levels of gonadal investment to the other unprotected sites. Qualitative

observations indicate scallop abundance at the artificial reef site to be lower than the other sites investigated. This indicated the Loch Linnhe artificial reef site provides sub-optimal habitat for scallops as a much healthier population would have been expected. However, more research is required to allow firm conclusions to be drawn in relation to the protective effects of artificial reefs.

4.3 CONCLUSIONS

The present study captures the state of animal assemblages on and around the Loch Linnhe artificial reef. However, it is limited because it could not account for inter-annual variation. The present study has shown that habitat complexity will affect different species in different ways, because of the spatial scales at which they interact with their environment (Rose 2005) and the inter-specific competition within the community. The increased habitat complexity provided by an artificial reef will tend to have a positive impact on the biodiversity in the local area, with animal abundance at least matching that found on natural reefs for most species investigated. The study has also shown that individual species will exhibit different responses to artificial reef habitat and, as such, the careful consideration of how a target organism might respond needs to be included in artificial reef design.

It was observed that the closure of the Loch Linnhe artificial reef site has resulted in no significant improvement in the condition of scallops in that area. This indicates that careful planning, in order to identify optimal habitat for the target species, needs to be undertaken during any fisheries management or environmental mitigation project. However, it cannot be discounted that optimal habitat for one target species

may conflict with another. In addition, under present legislation, the sites for artificial reef development have to be agreed by stakeholders, such as fishermen (Sayer & Wilding 2002).

The present study opens a number of avenues for future research. The first of these is a continuation of the present study over a number of years, so as to mitigate against inter-annual variation within the data. The second avenue is further examination of habitat complexity upon a particular species in the laboratory. Possible directions this may take include examining the effect habitat complexity has upon fish growth when abundance and food availability are constant. This may be examined by observing if the increase in abundance of small invertebrates with increased habitat complexity (Atilla *et al.*, 2005) would affect habitat selection in a small carnivorous grazing fish, for example, the rock cook or goldsinny (Darwall *et al.*, 1992).

In examining the impact of the Loch Linnhe artificial reef upon the surrounding benthic fauna, a more detailed investigation into the reproductive investment of scallops could be undertaken. Potentially scallops could be transplanted between the protected and un-protected sites, to control for natural variation between sites (Gosling & Burnell, 1988). Additionally larger sample sizes would be required, involving a significantly higher investment of labour. A further development in the study could utilise cage experiments to investigate predation upon scallops at the artificial reef site. This would be similar to the use of cages to investigate the impact of limpet grazing on rocky shores (Jenkins *et al.*, 1998) with newly settled scallops placed into the cages and specimens removed after a specific period of time.

4.4 SUMMARY

Based on the findings of this study, the introduction of complex artificial habitats would appear to offer possibilities for protecting, enhancing and augmenting populations of some commercially important species (Sayer, 2001). However, the scales of intervention and the costs versus benefits of such interventions remain to be quantified (Sayer *et al.*, 2005).

5. REFERENCES

- Atilla, N.A., Fleeger, J.W., Finelli, C.M. 2005. Effects of habitat complexity and hydrodynamics on the abundance and diversity of small invertebrates colonizing artificial substrates. *Journal of Marine Research*. **63**, 1151-1172.
- Baxter, J.H.S. 2000. The enhancement of marine fish stocks. *Advances in Marine Biology*. **38**, 1-54.
- Beaumont, J. 2006. Quantifying biotic interactions with inshore subtidal structures: comparisons between artificial and natural reefs. Ph.D Thesis. Scottish Association for Marine Science.
- Bennett, D.B. & Brown, C.G. 1983. Crab (*Cancer pagurus*) migrations in the English Channel. *Journal of the Marine Biological Association of the United Kingdom*, **63**, 371-398.
- Bortone, S.A. 2006. A perspective of artificial reef research: the past, present, and future. *Bulletin of Marine Science*, **78** (1), 1-8.
- Bortone, S.A., Martin, T. & Bundrick, C.M. 1994. Factors affecting fish assemblage development on a modular artificial reef in a northern Gulf of Mexico estuary. *Bulletin of Marine Science*, **55** (2-3), 319-332.
- Brickhill, M.J., Lee, S.Y., Connolly, R.M. 2005. Fishes associated with artificial reefs: attributing changes to attraction or production using novel approaches. *Journal of Fish Biology*. **67** (Supplement B), 53-71.
- Brock, V.E. 1954. A preliminary report on a method of estimating reef fish populations. *Journal of wildlife management*. **18** (3), 297-308.

- Buxton C.D. & Smale M.J. 1989. Abundance and distribution patterns of three temperate marine reef fish (Teleosti: Sparidae) in exploited and unexploited areas off the Southern Cape coast. *Journal of Applied Ecology*. **26**, 441-451.
- Caddy, J.F. 1986. Modelling stock recruitment processes in Crustacea- some practical and theoretical perspectives. *Canadian Journal of Fisheries and Aquatic Sciences*. **43**, 2330-2344.
- Charbonnel, E., Serre, C., Ruitton, S., Harmelin, J.-G. & Jensen, A. 2002. Effects of increased habitat complexity on fish assemblages associated with large artificial reef units (French Mediterranean coast). *ICES Journal of Marine Science*, **59**, Supplement, 208-213.
- Cripps, S.J. & Aabel, J.P. 2002. Environmental and socio-economic impact assessment of Ekoreef, a multiple platform rigs-to-reefs development. *ICES Journal of Marine Science*. **59**, Supplement, 300-308.
- Darwall. W.R.T., Costello, M.J., Donnelly, R., Lysaght, S. 1992. Implication of life-history strategies for a new wrasse fishery. *Journal of Fish Biology*. **41** (Supplement B), 111-123.
- Deysher, L.E., Dean, T.A., Grove, R.S. & Jahn, A. 2002. Design consideration for an artificial reef to grow giant kelp (*Macrocystis pyrifera*) in Southern California. *ICES Journal of Marine Science*. **59**, Supplement, 201-207.
- Diaconis, P. & Efron, B. 1983. Computer Intensive Methods in Statistics. *Scientific American* **248** (5), 116-130.
- Dickenson, G. & Murphy, K. 1998. The role of disturbance and succession in ecosystem functioning. In *Ecosystems*. pp. 99-113. Routledge.
- Duinker, A. & Nylund, A. 2002. Seasonal Variations in the ovaries of the great scallop (*Pectens maximus*) from western Norway. *Journal of the Marine Biological Association of the United Kingdom*. **82**, 477-482.
- Edwards, E. 1979. The Edible Crab and its fishery in British Waters. pp 142. Fishing News Books Ltd.
- Efron, B. 1981. Nonparametric estimates of standard error- the jackknife, the bootstrap and other methods. *Biometrika* **68** (3), 589-599.
- Fabi, G. & Fiorentini, L. 1994. Comparison between an artificial reef and a control site in the Adriatic Sea: analysis of four years of monitoring. *Bulletin of Marine Science*. **55** (2-3), 538-558.
- Fujita, T., Kitagawa, D., Okuyama, Y., Jin, Y., Ishito, Y. & Inada, T. 1996. Comparison of fish assemblages among an artificial reef, a natural reef and a sandy-mud bottom site on the shelf off Iwate, northern Japan. *Environmental Biology of Fishes*. **46**, 351-364.

- Gatwick, B. & Speight, M.R. 2005. The relationship between fish species richness, abundance and habitat complexity in a range of shallow tropical marine habitats. *Journal of Fish Biology*. **66**, 650-667.
- Gosling, E.M., Burnell, G.M. 1988. Evidence for Selective mortality in *Chlamys varia* (L.) transplant experiments. *Journal of the Marine Biological Association of the United Kingdom*. **68** (2), 251-258.
- Grant, J.J., Wilson, K.C., Grover, A. & Togstad, H. 1982. Early development of Pendleton artificial reef. *Marine Fisheries Review*. **44**, 53-60.
- Hem, S. & Avit, J.L.B. 1994. First Results on "Acadja-Enclos" as an extensive aquaculture system (West Africa). *Bulletin of Marine Science*. **55** (2-3), 1038-1049.
- Henriques, M. & Almada, V.C. 1998. Juveniles of non-resident fish found in sheltered rocky subtidal areas. *Journal of Fish Biology*. **52**, 1301-1304.
- Howard, A.E. 1982. The distribution and behaviour of ovigerous edible crabs (*Cancer pagarus*) and consequent sampling bias. *Journal do Conseil*. **40**, 259-261.
- Hurlbert, S.A. 1984. Pseudoreplication and the design of ecological field experiments. *Ecological Monographs*. **54** (2), 187-211
- Jenkins, S.R., Hawkins, S.J., Norton, T.A. 1998. Interaction between a fucoid canopy and limpet grazing in structuring a low shore intertidal community. *Journal of Experimental Marine Biology and Ecology*. **233**, 41-63.
- Jensen, A.C. 2002. Artificial Reefs of Europe: perspective and future. *ICES Journal of Marine Science*. **59**, Supplement, 3-13
- Jensen, A.C., Collins, K.J., Lockwood, A.P.M., Mallinson, J.J., Turnpenny, W.H. 1994. Colonisation and fishery potential of a coal-ash artificial reef, Poole Bay, United Kingdom. *Bulletin of Marine Science*. **55** (2-3), 1263-1276.
- JNCC. 1993. Summary of data from the coastal resources database. *Joint Nature Conservation Committee Coastal Conservation*. Branch Information Note. 7/93
- Kaiser, M.J., Collie, J.S., Hall, S.J., Jennings, S., Poiner, I.R. 2002. Modification of marine habitats by trawling activities: prognosis and solutions. *Fish and Fisheries*. **3** (2), 114-136.
- Kaiser, M.J., Spence, F.E., Hart, P.J. 2000. Fishing-gear restrictions and conservation of benthic habitat complexity. *Conservation Biology*. **14** (5), 1512-1525.
- Kimmel, J.J. 1985. A new species-time method for visual assessment of fishes and its comparison with established methods. *Environmental Biology of Fishes*. **12** (1), 23-32.
- Kock, R.L. 1982. Patterns of abundance variation in reef fishes near an artificial reef at Guam. *Environmental Biology of Fish*. **7** (2), 121-136.

- Lippmann, J. & Mitchell, S. 2005. Bühlmann tables. In *Deeper into Diving (2nd edition)*. pp. 225-240. J.L. Publications.
- Lincoln Smith, M.P. 1989. Improving multispecies rocky reef fish censuses by counting different groups using different procedures. *Environmental Biology of Fishes*. **26**, 29-37.
- Magill, S.H.; Sayer, M.D.J. 2004. Abundance of juvenile Atlantic cod (*Gadus morhua*) in the shallow rocky subtidal and the relationship to winter seawater temperature. *Journal of the Marine Biological Association of the UK*. **84**, 439-442.
- Miller, P.J., Rice, A.L., Johnstone, A.D.F. 1973. A western Scottish population of the leopard-spotted goby, *Thorogobius ephippiatus* (Lowe) (Teleostei: Gobioidae). *Journal of Fish Biology*. **5**, 233-239.
- Nash, R.D.M., Chapman, C.J., Atkinson, R.J.A., Morgan, P.J. 1984. Observations on the burrows and burrowing behaviour of *Calocaris machandreae* (Crustacea: Decapoda: Thalassinoidae). *Journal of Zoology*. **202** (3), 425-439.
- Nybakken, J.W. 1993. Rocky Subtidal Communities. In *Marine Biology: An Ecological Approach (3rd edition)*. pp. 181-186. Harper Collins College Publishing.
- Ormond, RFG (2000) *Reef Fish Watch 2000 Protocol for Reef Fish Underwater Visual Census* <http://www.gla.ac.uk/centres/marinestation/fishwatch/protocol.rtf>
- Osenberg, G.W., Mittelbach, G.G., Wainwright, P.C. 1992. Two-stage life histories in fish: the interaction between juvenile competition and adult performance. *Ecology*. **73**, 255-267.
- Pazos, A.J., Román, G., Acosta, C.P., Abad, M., Sánchez, J.L. 1996. Stereological studies on the gametogenic cycle of the scallop, *Pectens maximus*, in suspended culture in Ria de Arousa (Galicia, NW Spain). *Aquaculture*. **142**, 119-135.
- Perkol-Finkel, S.N., Shashar, O., Barneah, R., Ben-David-Zazlow, U., Oren, T., Reichart, T., Yacobovich, G., Yahel, R., Benayahu, Y. 2005. Fouling reefal communities on artificial reefs: Does age matter? *Biofouling*. **21** (2), 127-140.
- Pitcher, T.J., Buchary, E.A. & Hutton, T. 2002. Forecasting the benefits of no-take human-made reefs using spatial ecosystem simulation. *ICES Journal of Marine Science*. **59**, Supplement, 17-26.
- Roberts, C.M. & Ormond, R.F.G. 1987. Habitat complexity and coral reef fish diversity and abundance on Ref Sea fringing reefs. *Marine Ecology Progress Series*. **41**, 1-8.
- Rose, C.A. 2005. Modelling and measuring the habitat complexity of artificial reefs. Ph.D Thesis. University of Newcastle Upon Tyne.

Sayer, M.D.J. 2001. Fishery manipulation through stock enhancement or restoration. In *Encyclopaedia of Ocean Sciences* (eds. Steele, J.H., Turekian, K.K., Thorpe, S.A.). 2 (D-H), pp. 1029-1035. Academic Press Inc.

Sayer, M.D.J. & Baine, M.S.P. 2002. Rigs to Reefs: A Critical Evaluation of the potential for Reef Development Using Decommissioned Rigs. *Journal of the Society for Underwater Technology*. 25 (2), 93-97.

Sayer, M.D.J., Gibson, R.N., Atkinson, R.J.A. 1995. Growth, diet and condition of goldsinny on the west coast of Scotland. *Journal of Fish Biology*. 46, 317-340.

Sayer, M.D.J., Gibson, R.N., Atkinson, R.J.A. 1996. Growth, diet and condition of corkwing wrasse and rock cook on the west coast of Scotland. *Journal of Fish Biology*. 49, 76-94.

Sayer, M.D.J., Magill, S.H., Pitcher, T.J., Morissette, L., Ainsworth, C. 2005. Simulation-based investigations of fishery changes as affected by the scale and design of artificial habitats. *Journal of Fish Biology*. 67 (Supplement B), 218-243.

Sayer, M.D.J. & Poonian, C. 2006. The influences of census technique on the quantification of macrofaunal activity on natural and artificial reefs. *Journal of the Society for Underwater Technology*. (in press).

Sayer, M.D.J. & Wilding, T.A. 2002. Planning, licensing, and stakeholder consultation in an artificial reef development: the Loch Linnhe reef, a case study. Short Communication. *ICES Journal of Marine Science*. 59, Supplement, 178-185.

Seaman Jr., W. 2002. Unifying trends and opportunities in global artificial reef research, including evaluation. Short Communication. *ICES Journal of Marine Science*, 59, Supplement, 14-16.

Seaman Jr., W. & Jensen, A. C. 2000. Purposes and Practices of Artificial Reef Evaluation. In *Artificial Reef Evaluation with Application to Natural Marine Habitats* (ed. W. Seaman). pp. 1-19. CRC Press.

Tessier, E.; Chabanet, P.; Pothin, K.; Soria, M.; Lasserre, G. 2005. Visual censuses of tropical fish aggregations on artificial reefs: slate versus video recording techniques. *Journal of Experimental Marine Biology and Ecology*. 315, 17-30.

Tupper, M. & Boutilier, R.G. 1997. Effects of habitat on settlement, growth, predation risk and survival of a temperate reef fish. *Marine Ecology Progress Series*. 151, 225-236.

Underwood, A.J. 1997. *Experiments in Ecology: Their logical design and interpretation using analysis of variance*. Cambridge University Press.

Williamson, C.E. 1993. Linking predation risk models with behavioural mechanisms: identifying population bottlenecks. *Ecology*. 74, 320-331.

Wheeler, A. 1969. Fish-life and pollution in the lower Thames: a review and preliminary report. *Biological Conservation*. 2 (1), 25-30.

6. ELECTRONIC REFERENCES

Hong Kong Artificial Reef Project 2003. *Agriculture, Fisheries and Conservation Department, The Government of the Hong Kong Special Administration Area*. Available at www.artificial-reef.net

APPENDIX 1: PLACEMENT YEAR SUMMARY

The placement year was carried out at the UK National Facility for Scientific Diving, based at the Scottish Association for Marine Science. The National Facility is a NERC funded facility which aim is to support scientific diving and provide the necessary training to ensure it is safe and effective. The placement year took place between the 4th July 2005 and 30th June 2006 and was based around a research project upon the Loch Linnhe artificial reef. This project accounted for around two thirds of the placement year, with sampling and survey work on a monthly basis. Additionally work was undertaken as a diver in support of a number of other scientific projects, including work for a large multi national research group, the Coastal Observatory of Benthic Organisms.

The student gave a presentation about the study at the annual meeting of the artificial reef project management committee and was involved in the Scottish Association for Marine Science open evening. In addition, the student attended the first SUT conference on Diving in Science and Archaeology, and the Oceanography International exhibition at EdEXEL, London.

Overall the placement year has provided a valuable experience for the student and has provided both academic and vocational training, which included HSE SCUBA and RYA Level 2 boat handling. Finally the project carried out should provide the basis for a publication within the peer-reviewed literature.

APPENDIX 2: BALANCE SHEET

	Income	Outgoings
British Sub-Aqua Jubilee Trust Grant	£2,000.00	
Project AWARE Foundation Grant	£2,000.00	
NFSD Science Budget	£20,231.45	
Student Loans Compant Student Loan	£4,000.00	
Boat Support (60 days @ £200 per day)		£12,000.00
Diving Support (60 days @ £100 per day)		£6,000.00
Student subsistance		£6,000.00
SAMS Bench Fee		£3,500.00
Dry Suit		£330.00
HSE Medical		£150.00
Leaded Rope (100m @ £50.50 per 50m)		£101.00
Plastic Aquarium Piping		£100.00
Vernier Calipers		£35.00
MINITAB software licence		£15.45
Total	£28,231.45	£28,231.45

Additional Funding sought but not secured

EU Leader+ scheme
 Society for Underwater Technology Educational Support Fund
 Argyll & Islands Enterprise Grant

APPENDIX 3: MEAN AND 95% CONFIDENCE INTERVALS FROM BOOTSTRAP FOR EACH OF THE FOUR SITE TYPES DURING THE SUMMER PERIOD.

Site Type	Complex			Simple			Natural			Control		
	Mean	95% C.I.		Mean	95% C.I.		Mean	95% C.I.		Mean	95% C.I.	
<i>C.melops</i>	0.201	0.111	0.312	0.104	0.056	0.146	0.014	0.000	0.028	0.000	0.000	0.000
<i>C.exoleus</i>	0.257	0.111	0.470	0.243	0.118	0.396	0.000	0.000	0.000	0.000	0.000	0.000
<i>L.bergylta</i>	0.035	0.007	0.062	0.014	0.000	0.042	0.014	0.000	0.042	0.000	0.000	0.000
<i>L.mixtus</i>	0.007	0.000	0.021	0.035	0.002	0.069	0.007	0.000	0.021	0.000	0.000	0.000
<i>C.rupestris</i>	0.187	0.076	0.292	0.111	0.007	0.285	0.062	0.021	0.097	0.000	0.000	0.000
<i>G.morhua</i>	0.069	0.021	0.125	0.028	0.007	0.049	0.000	0.000	0.000	0.000	0.000	0.000
<i>P.pollachius</i>	0.236	0.056	0.485	0.021	0.005	0.049	0.035	0.000	0.083	0.000	0.000	0.000
<i>P.virens</i>	0.125	0.056	0.215	0.090	0.035	0.160	0.042	0.000	0.076	0.000	0.000	0.000
<i>T.ephippiatus</i>	0.035	0.000	0.069	0.056	0.014	0.146	0.431	0.257	0.632	0.000	0.000	0.000
<i>C.pagarus</i>	0.049	0.021	0.097	0.028	0.007	0.049	0.042	0.014	0.076	0.056	0.000	0.111
<i>N.puber</i>	0.278	0.209	0.340	0.153	0.076	0.229	0.083	0.035	0.125	0.000	0.000	0.000
<i>C.maenus</i>	0.000	0.000	0.000	0.007	0.000	0.021	0.125	0.049	0.222	0.361	0.229	0.500
<i>H.gammarus</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.021	0.000	0.035	0.000	0.000	0.000
<i>M.rugosa</i>	0.132	0.049	0.201	0.076	0.028	0.138	0.486	0.229	0.789	0.009	0.000	0.019
<i>E.esculentus</i>	0.278	0.208	0.364	0.271	0.155	0.361	0.521	0.306	0.736	0.019	0.000	0.037
<i>A.rubens</i>	0.465	0.279	0.668	0.271	0.201	0.342	0.465	0.299	0.667	0.176	0.102	0.259
<i>A.bifida</i>	0.056	0.014	0.097	0.007	0.000	0.014	0.292	0.021	0.603	0.000	0.000	0.000
Ascidiacea	2.326	1.687	2.923	0.653	0.354	0.931	0.076	0.028	0.146	0.000	0.000	0.000
Total												
Abundance	4.736	3.937	5.529	2.167	1.720	2.542	2.715	2.125	3.286	0.620	0.519	0.722
Total Mobile	1.944	1.508	2.329	1.243	0.938	1.544	2.174	1.468	2.799	0.444	0.313	0.574

APPENDIX 4: MEAN AND 95% CONFIDENCE INTERVALS FROM BOOTSTRAP FOR EACH OF THE FOUR SITE TYPES DURING THE AUTUMN PERIOD.

Site Type	Complex			Simple			Natural			Control		
	Mean	95% C.I.		Mean	95% C.I.		Mean	95% C.I.		Mean	95% C.I.	
<i>C.melops</i>	0.102	0.028	0.176	0.065	0.021	0.102	0.009	0.000	0.019	0.000	0.000	0.000
<i>C.exoleus</i>	0.602	0.464	0.759	0.389	0.278	0.500	0.019	0.000	0.037	0.000	0.000	0.000
<i>L.bergylta</i>	0.019	0.000	0.037	0.028	0.000	0.081	0.009	0.000	0.028	0.000	0.000	0.000
<i>L.mixtus</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
<i>C.rupestris</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.083	0.028	0.157	0.000	0.000	0.000
<i>G.morhua</i>	0.028	0.000	0.074	0.028	0.000	0.046	0.000	0.000	0.000	0.000	0.000	0.000
<i>P.pollachius</i>	0.269	0.120	0.461	0.190	0.000	0.037	0.009	0.000	0.019	0.000	0.000	0.000
<i>P.virens</i>	0.361	0.139	0.685	0.046	0.009	0.083	0.102	0.000	0.300	0.000	0.000	0.000
<i>T.ephippiatus</i>	0.028	0.000	0.046	0.280	0.000	0.046	0.194	0.093	0.332	0.000	0.000	0.000
<i>C.pagarus</i>	0.028	0.009	0.056	0.000	0.000	0.000	0.028	0.000	0.065	0.000	0.000	0.000
<i>N.puber</i>	0.343	0.241	0.488	0.139	0.074	0.213	0.148	0.083	0.213	0.000	0.000	0.000
<i>C.maenus</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.093	0.019	0.176	0.153	0.083	0.208
<i>H.gammarus</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
<i>M.rugosa</i>	0.370	0.241	0.542	0.278	0.149	0.463	0.398	0.194	0.630	0.000	0.000	0.000
<i>E.esculentus</i>	0.352	0.241	0.481	0.361	0.222	0.499	0.435	0.231	0.648	0.014	0.000	0.028
<i>A.rubens</i>	0.231	0.120	0.352	0.204	0.111	0.296	0.361	0.185	0.732	0.042	0.000	0.069
<i>A.bifida</i>	0.065	0.019	0.130	0.019	0.000	0.037	0.509	0.000	1.111	0.000	0.000	0.000
Ascidiacea	3.352	2.611	3.880	0.963	0.611	1.324	0.287	0.117	0.472	0.000	0.000	0.000
Total Abundance	6.148	5.387	6.848	2.565	2.157	2.910	2.685	1.926	3.750	0.208	0.097	0.250
Total Mobile	2.565	1.907	3.213	1.398	1.147	1.602	2.037	1.194	3.287	0.167	0.097	0.222

APPENDIX 5: MEAN AND 95% CONFIDENCE INTERVALS FROM BOOTSTRAP FOR EACH OF THE FOUR SITE TYPES DURING THE WINTER PERIOD.

Site Type	Complex			Simple			Natural			Control		
	Mean	95% C.I.		Mean	95% C.I.		Mean	95% C.I.		Mean	95% C.I.	
<i>C.melops</i>	0.006	0.000	0.019	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
<i>C.exoleus</i>	0.006	0.000	0.019	0.006	0.000	0.019	0.000	0.000	0.000	0.000	0.000	0.000
<i>L.bergylta</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.006	0.000	0.019	0.000	0.000	0.000
<i>L.mixtus</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
<i>C.rupestris</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
<i>G.morhua</i>	0.025	0.006	0.043	0.006	0.000	0.019	0.000	0.000	0.000	0.000	0.000	0.000
<i>P.pollachius</i>	0.006	0.000	0.019	0.012	0.000	0.037	0.000	0.000	0.000	0.000	0.000	0.000
<i>P.virens</i>	0.006	0.000	0.019	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
<i>T.ephippiatus</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.074	0.025	0.148	0.000	0.000	0.000
<i>C.pagarus</i>	0.130	0.074	0.210	0.043	0.012	0.080	0.006	0.000	0.019	0.009	0.000	0.019
<i>N.puber</i>	0.457	0.401	0.525	0.179	0.099	0.279	0.148	0.080	0.235	0.009	0.000	0.019
<i>C.maenus</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.043	0.006	0.086	0.056	0.009	0.111
<i>H.gammarus</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.006	0.000	0.019	0.000	0.000	0.000
<i>M.rugosa</i>	0.401	0.270	0.572	0.377	0.259	0.506	0.370	0.216	0.519	0.000	0.000	0.000
<i>E.esculentus</i>	0.525	0.423	0.623	0.481	0.346	0.599	0.679	0.350	1.017	0.000	0.000	0.000
<i>A.rubens</i>	0.377	0.272	0.493	0.222	0.117	0.321	0.358	0.237	0.475	0.019	0.000	0.037
<i>A.bifida</i>	0.080	0.031	0.130	0.000	0.000	0.000	0.617	0.037	1.488	0.000	0.000	0.000
Ascidiacea	2.833	2.207	3.398	1.068	0.709	1.457	0.302	0.141	0.357	0.000	0.000	0.000
Total Abundance	4.852	4.074	5.568	2.395	2.038	2.767	2.611	2.004	3.359	0.093	0.023	0.148
Total Mobile	1.642	1.469	1.846	1.105	0.922	1.322	1.951	1.272	2.931	0.074	0.019	0.120

APPENDIX 6: MEAN AND 95% CONFIDENCE INTERVALS FROM BOOTSTRAP FOR EACH OF THE FOUR SITE TYPES DURING THE SPRING PERIOD.

Site Type	Complex			Simple			Natural			Control		
	Mean	95% C.I.	95% C.I.	Mean	95% C.I.	95% C.I.	Mean	95% C.I.	95% C.I.	Mean	95% C.I.	95% C.I.
<i>C.melops</i>	0.006	0.000	0.019	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
<i>C.exoleus</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
<i>L.bergylta</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.006	0.000	0.012	0.000	0.000	0.000
<i>L.mixtus</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
<i>C.rupestris</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.006	0.000	0.019	0.000	0.000	0.000
<i>G.morhua</i>	0.006	0.000	0.019	0.014	0.000	0.028	0.000	0.000	0.000	0.000	0.000	0.000
<i>P.pollachius</i>	0.037	0.005	0.080	0.021	0.000	0.049	0.000	0.000	0.000	0.000	0.000	0.000
<i>P.virens</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.006	0.000	0.012	0.000	0.000	0.000
<i>T.ephippiatus</i>	0.006	0.000	0.019	0.000	0.000	0.000	0.043	0.012	0.071	0.000	0.000	0.000
<i>C.pagarus</i>	0.148	0.080	0.204	0.042	0.007	0.069	0.049	0.019	0.087	0.000	0.000	0.000
<i>N.puber</i>	0.389	0.315	0.488	0.049	0.014	0.083	0.105	0.062	0.179	0.000	0.000	0.000
<i>C.maenus</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.093	0.037	0.167	0.074	0.028	0.111
<i>H.gammarus</i>	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
<i>M.rugosa</i>	0.531	0.410	0.627	0.333	0.208	0.472	0.377	0.216	0.525	0.000	0.000	0.000
<i>E.esculentus</i>	0.241	0.173	0.313	0.257	0.160	0.365	0.494	0.284	0.814	0.019	0.000	0.037
<i>A.rubens</i>	0.654	0.502	0.809	0.437	0.347	0.542	0.605	0.486	0.734	0.046	0.009	0.083
<i>A.bifida</i>	0.025	0.000	0.049	0.000	0.000	0.000	0.506	0.111	1.278	0.000	0.000	0.000
Ascidiacea	2.259	1.586	2.833	1.354	0.947	1.773	1.278	0.500	2.090	0.000	0.000	0.000
Total												
Abundance	4.302	3.558	4.915	2.507	2.118	2.935	3.574	2.944	4.185	0.139	0.065	0.196
Total Mobile	2.019	1.778	2.218	1.153	1.014	1.329	1.790	1.488	2.147	0.139	0.065	0.196