Discard analysis, damage and mortality of non-target species in a queen scallop fishery

by

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Declaration & Statements

This work has not previously been accepted in substance for any degree and is not being concurrently submitted for any degree.

This dissertation is being submitted in partial fulfilment of the requirement of M.Sc. Marine Environmental Protection.

This dissertation is the result of my own independent work / investigation, except where otherwise stated.

Other sources are acknowledged by footnotes giving explicit references. A bibliography is appended.

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Abstract

This study describes bycatch abundance and composition in the Isle of Man queen scallop (*Aequipecten opercularis*) fishery as well as the damage and mortality incurred to non-target species. Data were collected on-board three fishing vessels in order to reflect commercial fishing practice: an otter trawl and two toothless dredges with design features consisting mainly of modifications replacing the traditional dredge teeth, namely a tickler chain (in the "skid dredge") and rubber lip (in the "modified dredge"). Bycatch from the dredges was mainly characterised by invertebrates, starfish and molluscs, whilst trawl bycatch had a higher proportion of sea urchins and fish. Landings per unit effort were highest for the modified dredge and lowest for the otter trawl. Catches from the modified dredge also contained the lowest proportion of bycatch. CPUE made with the trawler were the lowest. The skid dredge had the highest proportion of bycatch. A damage assessment was performed on *Asterias rubens* and *Echinus esculentus* which aimed to encompass the towing and sorting elements of the fishing process. This was undertaken by collecting individuals retained in the three gears and hauled on deck as well as individuals collected at the end of the catch sorting process and prior to discarding at sea. Generalized Linear Models (GLMs) were used to predict the damage levels in the discards using vessel and sorting as explanatory variables. Model predictions indicated higher total damage (following towing and sorting) of *A.rubens* and *E.esculentus* in the trawl and least damage in the modified dredge. The lower bycatch damage in the modified dredge may be related to its higher efficiency and lower proportion of stones in the catch. The effects of damage on the survival of *A. rubens* and *E. esculentus* following trawling and skid dredging were also examined. GLMs were used to predict mortality rates across vessels as well as different levels of damage in the discards. Mortality of both *A.rubens* and *E.esculentus* was high (52-100% and 32-95% respectively). There was no difference in mortality across vessels. Mortality increased with the level of injury sustained by both *A.rubens* and *E.esculentus*. The results of this study suggest that due to its higher catch efficiency, lower bycatch and lower damage incurred to bycatch individuals the modified dredge may be the most appropriate gear, in the context of a management target which aims to minimise environmental effects incurred per unit of landings. Further research which addresses the effects of these three different gears on non-captured organisms is however necessary for a comprehensive understanding of their overall environmental effects.

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1. Introduction

Fisheries scientists have traditionally been concerned with the effects of overfishing on target species populations, in particular their loss of productivity and potential extinction (Worm et al. 2009; Goñi 1998; Dayton et al. 1995). However, until recently, little consideration had been given to the wider implications of fishing activities on other (non-target) species and marine ecosystems. Increasing recognition in fisheries science of the undesirable effects that fishing target species has on other living components of marine ecosystems has gained momentum in recent years (Goñi 1998). Consequently, there is a growing body of knowledge on the effects that fishing has on a wider range of ecosystem components as well as on the impacts that fishing operations themselves have on marine habitats (Jennings & Kaiser 1998). Studies of fishing impacts on benthic ecosystems have become a major focus of scientific research (Collie et al. 2000; Hall 1996), in particular the effects of trawling and dredging on the sea bed and benthic communities were recognised early on and have received the most attention (Goñi 1998). Trawling and dredging are known to affect benthic communities and the sea bottom, mostly by increasing mortality of target and non-target species and altering the abiotic environment (Garcia et al. 2006).

As fisheries managers become increasingly aware that target populations cannot be dissociated from their ecosystem, the idea that the ecosystem as a whole should be considered as a management unit has gained strength in recent years (Kaiser et al. 2006; Pikitch et al. 2004; Jennings and Kaiser 1998). The current drive towards an ecosystem approach to fisheries management requires an understanding of the wider ecosystem effects of fishing activities with a view to minimising them and to restoring habitats where possible.

1.1 Bycatch damage and mortality

A particular class of impacts that has received increasing attention during the last decade is the (often unaccounted) indiscriminate capture of non-target organisms, typically referred to as "bycatch" (Davies et al. 2009; Goñi 1998). The role of bycatch in degrading marine ecosystems has made it one of the most significant nature conservation issues of the world today (Davies 2009). Despite increasing public concern and growing literature in the field, a consistent understanding of bycatch is lacking due to issues in its definition, measurement and quantification (reviewed by Davies et al. 2009). Bycatch can sometimes be retained and sold, however it may also be unusable or unwanted for a variety of regulatory or economic reasons, and is subsequently thrown back to sea, often either damaged, dying or dead (Davies et al. 2009). In order to understand the effect that fishing and bycatch can have on marine ecosystems it is important to understand how bycatch organisms are affected by the fishing process, namely their mortality prior to and after discarding, and the parameters most affecting damage and mortality.

This is a particular concern of the Isle of Man queen scallop fishery given their recent attainment of Marine Stewardship Council (MSC) certification for the trawled queen scallop fisheries. Certification requires the exploitation of fisheries resources to be undertaken with adequate knowledge of the ecosystem effects of the fishing activities, which consequently have to be as closely monitored as possible. One component of this research which has so far lacked specific attention is the detailed quantification of bycatch and bycatch mortality associated with the different trawl and dredge gears used by the fishery (Duncan 2009). Such information is necessary to maintain MSC certification for the trawl fishery but also to achieve a potential future certification of the dredge component of the fishery. Most importantly, it allows a preliminary comparison of the environmental effects of different gear types with a view to making informed management decisions with regard to the technology to be used and hence the mitigation of adverse environmental effects.

Studies of bycatch damage and mortality collectively show the influence of a range of species, habitat and fishery specific factors. Fishery-specific factors may include the type of gear, habitat exploited, the number and volume of stones retained in the dredge / net, exposure to air and tow time (i.e. time spent in the dredge/cod end) (Hill et al. 1996; Kaiser & Spencer 1995). Changes in these factors will affect different species to differing extents. In addition, there is a variation in sensitivity to fishing across species (see e.g. review by Alverson et al. 1994) and, in general, large-sized organisms with a slow growth rate and higher and size at maturity are the most sensitive to fishing activities (Greenstreet & Rogers 2000). Survival rates for fishes tend to be low, in particular for those fish whose swim bladders expand and trap them at the surface such as cod, whiting and rock fish (Garcia et al. 2006; van Santbrink & Bergman 2000; Alverson et al. 1994; Evans et al. 1994). In general, invertebrates have higher survival rates than fish, although overall vulnerabilities vary among species. Taxa protected with shells or exoskeletons, or that can regenerate missing limbs, such as starfish, tend to have higher survival rates, whereas long-lived, slow growing epifaunal species often have a more fragile body structure and are especially sensitive to contact with fishing gear (sponges, bryozoans and other sessile organisms) (Garcia et al. 2006; Jenkins et al. 2001; Kaiser & Spencer 1995). Starfish have been found to be fairly resistant to damage by trawling in a number of studies (Ramsay et al. 2000; Kaiser & Spencer 1995), although fishing mortality of echinoderms has been shown to be high in other studies (Gaspar et al. 2001). Overall, high survival rates of echinoderms have been observed in other studies (Bergmann & Moore 2001; Jenkins et al. 2001), suggesting the relative robustness of this taxon to fishing disturbance.

Leitão et al. (2009) observed higher mortalities of bycatch from dredging in bivalves, sea urchins and crabs than all other taxa, and Gaspar et al. (2001) noted that thin shelled bivalves such as *Pharus legumen* and *Ensis ensis* were the most sensitive species. Consistency in mortality data can sometimes be a concern; disparate results observed for the same species are likely to be indicators that species mortality is also driven by a range of factors other than biological "species" factors.

The observations above indicate that better quantitative information would be needed with regard to long-term mortality if more accurate estimations are to be obtained with regard to the fate of discards at sea. In addition, most delayed mortality studies have involved the use of research vessels rather than commercial fishing vessels and/or determination of mortality in the laboratory (tanks). Such studies are unlikely to be entirely realistic as they will not be able to recreate commercial fishing practices or the conditions and multitude of challenges faced by discards at sea (Davis & Ryer 2003; Veale et al. 2001; Kaiser & Spencer 1995). Studies which attempt to estimate discard mortality can make assumptions with the aim of providing approximate estimates of discard mortality based on the level of sub-lethal damage incurred, e.g. the assumption of 100% mortality in damaged animals, or of 100% survivability in seemingly undamaged ones. Existing studies have however indicated such assumptions to be imprecise, in particular when attempting to explain processes involving several species of differing sensitivities to the fishing process (Pranovi et al. 2001). Damaged animals that are discarded and eventually sink to the sea bed have increased vulnerability to stressors such as predation or disease. However, not all animals are equally sensitive to the effects of towing and sorting (Greenstreet & Rogers 2000; Alverson et al. 1994). Therefore, if a realistic estimate of the vulnerability of the species upon discard is to be obtained, it is necessary to evaluate both the extent of damage upon discard as well as delayed mortality in relation to different levels of sub-lethal injury (Pranovi et al. 2001; Ramsay et al. 2001).

Increasing awareness of the impacts of bycatch mortality on ecosystems has led to the implementation of a number of technological or management solutions, such as area restrictions, minimum mesh sizes, or bycatch reduction mechanisms (Leitão et al. 2009). Modifications have been introduced to fishing gear design in order to enhance selectivity and/or decrease the abundance of discards (Leitão et al. 2009; Valdemarsen & Suuronen 2003). The Isle of Man queen scallop fishery uses both trawl and dredge gear; recent management introductions in this fishery have included the imposition of a permanent ban on tooth dredges. In addition, a partial ban on toothless dredges is also in place and consists of a permanent ban from certain areas of the seabed, as well as a temporary ban from all areas within the 12 mile nautical limit in June, July and August (Moody Marine Ltd undated). Two toothless dredge types are thus currently used to catch queen scallops in the Isle of Man: the skid dredge and the recently developed gate gear. A summary description of each dredge type is provided in Table 1 and a more extensive description is available in Moody Marine Ltd (undated).

Direct comparisons of the environmental effects of different fishing gears are highly relevant in the context of a fishery such as the Isle of Man queen scallop fishery, which utilises several gear types to target the same species. However, such studies are not frequently reported in the literature. Furthermore, these studies are not likely to be directly useful in the context of the management of the specific fishery, as impacts differ considerably at the fishery level in accordance with more fishery-specific information such as gear configurations, habitat types and fishing practices (Hinz et al. in revision). More locally specific understanding of the impact of different fishing gears on both target and non-target species and habitats is therefore a pre-requisite of an ecosystem based approach to fisheries management. Such knowledge is necessary to compare the environmental impacts of different fishing gears against target species catch and identify the most environmentally friendly way to achieve an economically viable fishery (Hinz et al. in revision). It is also a requirement for certification by the Marine Stewardship Council (MSC), an environmental certification standard which the queen scallop fishery currently seeks to obtain but has only partly succeed in doing so: a recent independent assessment (2008-2010) of the fishery against MSC Principles and Criteria (Moody Marine Ltd undated) resulted in the trawl fishery being granted MSC certification, however the dredge fishery failed the assessment. The main weakness in the fishery's management was considered to be a lack of understanding of the extent and consequences of the effect of trawls and dredges on marine habitats, including a lack of quantitative knowledge on bycatch species affected by the fishery.

In addition, previous comparisons of damage for different types of fishing gear (e.g. Veale et al. 2001) tended to focus on the effects of the towed gear and the additional damage from the catch sorting process is often left unmeasured. It is expected that the different fishing and sorting gears used in the fishery will differently affect the amount and composition of bycatch, as well as the damage level inflicted upon bycatch individuals. In addition, it is expected that the mortality of individuals upon discard will increase with an increase in the level of damage inflicted upon an individual.

1.2 *Aims of the study*

In this study three Isle of Man queen scallop fishing vessels (using a Skid Dredge, Modified Dredge and Otter Trawl net, as well as a range of different on-deck sorting practices – refer to Table 1) undertook "normal" fishing activities in Isle of Man fishing grounds. The study attempted to investigate:

- Any differences in bycatch quantity and composition across the three fishing gear types; it was hypothesised that bycatch composition and bycatch rates would differ across gears. Given current knowledge it was hypothesised that bycatch rates would be equivalent or lower in the otter trawl than the modified dredge (Hinz in revision), however no prior knowledge existed regarding the skid dredge;

- Any differences in the level of damage exerted upon bycatch across the three vessels; more specifically, it was hypothesised that there would be differences in damage level in (i) the hauled bycatch (pre-sorting) across different fishing gears and (ii) the bycatch after it has undergone sorting as compared to the catch prior to commencement of catch sorting activities; the study aimed to quantify the additional damage incurred to bycatch as a result of sorting processes;

- Any differences in the long-term (> 6 days) mortality of damaged bycatch across damage levels; it was hypothesised that mortality would be significantly different across damage levels and would increase with an increase in damage.

2. Materials and Methods

Three experiments were carried out to investigate the nature and fate of bycatch species routinely caught and discarded during commercial trawling and dredging for queen scallops in Isle of Man fishing grounds. The bycatch was sampled opportunistically from three commercial queen scallop fishing vessels undertaking normal fishing activities in the Isle of Man fishing grounds located SE of Douglas (Figure 1) between 14th June and 26th July 2011.

The fishing vessels used in this study included: Marine Fisheries Vessel (MFV) Maureen Patricia (otter trawler - OT), MFV Q-Varl (skid dredge - SD) and MFV King Challenger (modified toothless dredge - MD). Both MFV Maureen Patricia and Q-Varl belong to the Isle of Man commercial fishing fleet, whilst MFV King Challenger is a Scottish vessel equipped with a modified dredge design (gear specifications of each vessel detailed in Table 1 and illustrated in Figure 2). Given that the aim of the experiment was a comparison of the impacts of different fishing gears and commercial practices, no attempt was made to modify the vessels' normal fishing routines (choice of grounds, tow time, speed and on-deck practices). Tow variables including position (tow start and tow end) and tow duration were recorded.

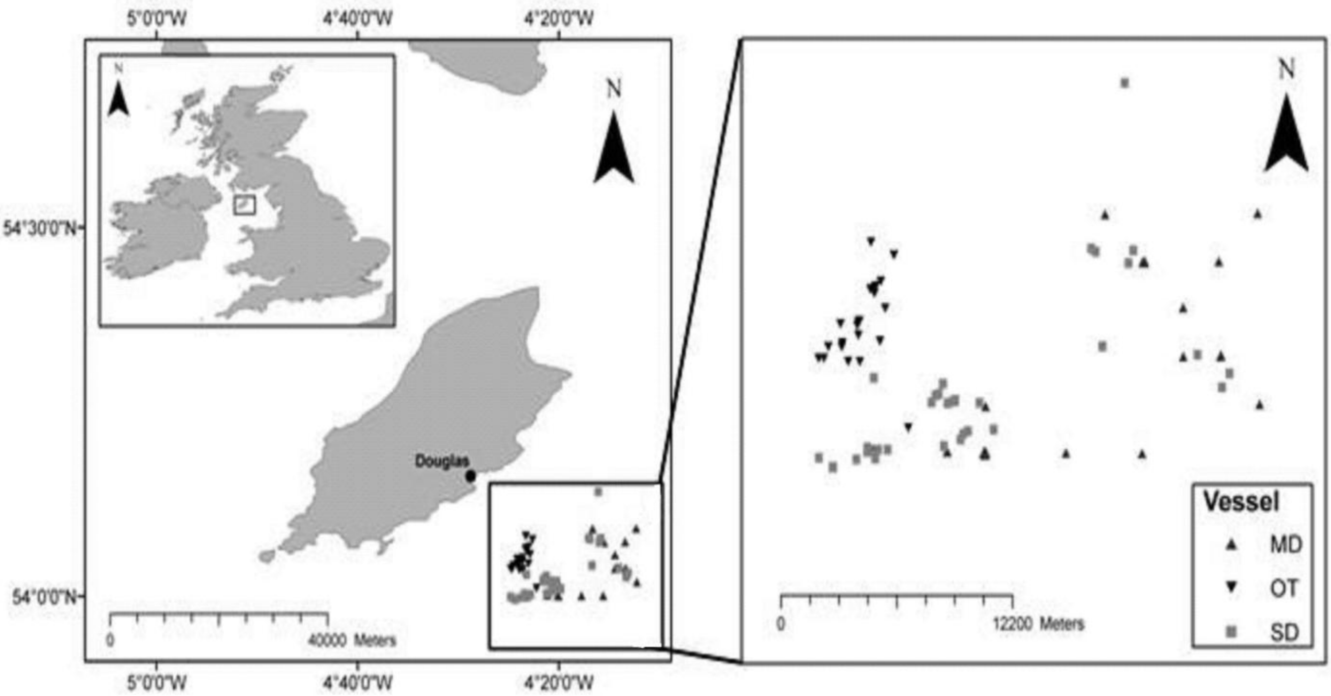


Figure 1: Map of study area. Points indicate end locations of tows.

Table 1: Specifications of fishing gear used in the present study

Vessel	No. of dredges/ nets	Gear Dimensions	Fishing Process
Otter Trawl	1 x net	1 x net; mesh size: 85mm; mouth of trawl net: 18.3m wide (wing end to wing end);	**Fishing Gear**: net; **Sorting Gear**: Mechanical riddle; **On-deck sorting process**: undersized queen scallops and smaller bycatch pass through riddle holes into a shoot and discarded at sea; landable queenies and larger bycatch picked up at riddle en by hand and disposed of to a bucket. Bycatch picked up at riddle end and disposed of to a bucket, to be discarded at sea when full or at the end of each tow.
Skid Dredge	16 x skid dregdes	16 x dredges; dredge width: 0.76m; length of dredge (catching bag): 17 belly rings; width of dredge: 10 belly rings. Ring internal diameter: 60mm.	**Fishing Gear:** skid dredge, i.e. modified toothed dredge with each frame mounted on ski-like runners and the typical toothbar replaced by a tickler chain; **Sorting Gear: Gear**: Mechanical shaker; **On-deck sorting process**: smaller bycatch passes through shaker grids into a shoot and is automatically discarded at sea; larger bycatch is picked up by hand at the end of the shaker and left on deck, to then be swept by crew into sea in between tows (however at irregular intervals).
Modified Dredge (“toothless” dredge or “gate gear”)	10 x modified dredges.	10 x dredges; dredge width: 2m; length of dredge (catching bag): 1.5m. Ring internal diameter: 60mm. Traditional metal teeth replaced with a flexible rubber mat mounted along the lower aperture of the net to “flick” scallops into the net – the bow wave produced by the rubber mat provokes a flare response in scallops making them more vulnerable to the moving basket (Malcolm, F. 2009).	**Fishing Gear:** dredge where steel toothbar or tickler chain is replaced by a flexible rubber lip or flappers. **Sorting Gear**: Mechanical shaker; **On-deck sorting process**: smaller bycatch passes through shaker grids into a shoot and is automatically discarded at sea; larger bycatch is picked up by hand at the end of the shaker and disposed of into a basket, which is discarded at sea by the crew in between tows or when full (i.e. at irregular intervals).

a)

b)

c)

Figure 2: a) Modified Dredge; b) Skid dredge; and c) typical Manx queen scallop trawl net (Figure 2c reproduced from Duncan 2009)

2.1 Experiment 1: Bycatch abundance and composition

2.1.1 Experimental Design and Sampling

This experiment investigated one fixed treatment factor (Vessel) across three treatment levels (Skid Dredge or "SD", Modified Dredge or "MD" and Otter Trawl or "OT")in order to establish the role that different gears and associated fishing practices have in determining bycatch composition. 40 replicate tows in total were sampled, including 11 tows on the OT, 17 tows on the SD and 12 tows on the MD.

Given the large commercial catches, a subsample of the catch was taken upon hauling (36.9 kg ± 1.6 mean sub-sample weight). Subsamples were measured for total weight, target species (queen scallop *Aequipecten opercularis*) weight, bycatch weight and stones/pebbles weight. The number of *A.opercularis* in the sub-sample was counted. Bycatch was identified to the lowest possible level of taxonomic resolution and counted. All captured taxa were identified to species level, except a few species that were grouped by genus or class owing to limitations such as the difficulty of rapid field identification, high damage levels resulting from fishing, or the presence of epibionts on crustacean shells.

2.1.2 Data Analysis

This experiment aimed to determine whether the mean biomass and abundance of bycatch subsampled differed across vessels. Secondly, the study also aimed to determine whether the overall assemblage structure (i.e. species composition and the relative abundance of each species) differed across vessels.

In order to address the first question a univariate approach was taken by using a one-factor analysis of variance (ANOVA) to compare the mean number of individuals and biomasses between vessels. Total standardised abundance N (n ha^{-1}) and biomass B (kg ha^{-1}) were calculated and standardised by area swept as:

$$N = n_{landed} + n_{discarded}$$

Where n_{landed} = the number of queenies landed and $n_{discarded}$ = the number of queenies discarded. The number of queenies landed was calculated as:

$$n_{landed} = \frac{\frac{l \times w_b}{w_q}}{a}, \text{ where:} a = \frac{s \times t \times g}{10000}$$

where l = number of landed bags, w_b = weight of one bag (assumed to be 40 kg for all vessels from limited field measurements: average weight of a bag in the SD was 41.5 kg (n=3) and in the OT 39 kg (n=3), no measurements were taken in the MD), w_q = weight of one queen scallop (taken as 55.1 g from field estimates: 55.17 g ± 1.07, n=26), a = area towed (ha), s = tow speed (ms^{-1}), t = tow time (seconds), g = gear width (m). The number of queen scallops discarded per hectare was assumed as 140, 245 and 280 for the OT, SD and MD respectively (refer to Nall 2011 for the methodology used to estimate the number of queen scallops discarded).

Tow time was measured on board vessels and tow start and end location recorded from the vessels' GPS units. Tow distance could not be derived simply from the recorded tow start and end locations, as commercial vessels do not follow a straight line or predictable path as they tow (Lambert et al. in press). It was thus calculated as the product of tow speed and tow time as included in the equation above.

An estimate of tow speed for the MD was obtained from the skipper for each tow whilst on board vessel and the data obtained were used in the above calculation. Tow-specific estimates of speed could not be obtained from the SD and OT for use in the calculation of tow distance; however estimates of overall mean speed were obtained from the skippers, namely 3.5 kn and 2.2 kn for the SD and OT respectively. These were validated by cross-checking with average Vessel Monitoring System (VMS) data for SD and OT available from DEFA's Manx fleet data set for 2010-2011, in which average speeds for the trawl and skid dredge fleet were respectively: 1.4 kn ± 0.4 SD, and 1.2 kn ± 0.4 SD. Vessel speeds for the OT and SD were thus assumed as 1.122 ms^{-1} and 1.785 ms^{-1} in subsequent calculations. Homogeneity of variance was checked using the Levene's Test and when this assumption was not met non-parametric equivalents (Kruskal-Wallis followed by pair-wise Mann-Whitney U tests) were used.

In order to address the second question, the multivariate approach of non-metric multidimensional scaling (MDS) was used to examine similarities in bycatch assemblage structure across vessels. Bycatch composition data were standardised by total in order to account for unequal sample sizes and a unit of sampling (roughly filled and weighed baskets) which could not be tightly controlled (Clarke & Warwick 2001). The community data set was clustered using a Bray Curtis index of similarity on square root transformed data. This was followed by multidimensional scaling (MDS) performed on the resulting similarity matrix in order to identify any resemblance patterns among the samples. MDS is an appropriate technique for ecological data, which often have numerous zeroes (absence of taxon in a sample) (Scrosati & Heaven 2008; Clarke 2001).

Species which contribute to similarity between vessels (species most responsible for observed patterns) were identified through an analysis of the percentage contribution to similarity (SIMPER) made by each taxon within the samples taken at each vessel. The more abundant a species is within a group, the more it contributes to intra-group similarity (Scrosati 2008) and it will typify that group if its abundance is constant throughout (Clarke & Warwick 2001).

Differences in community composition between vessels were tested using an analysis of similarities (ANOSIM). The null hypothesis for the ANOSIM is that there are no significant differences between the different vessels with respect to community composition. The analysis involved generating 9999 random permutations of the data in order to calculate the probability that observed differences in the structure of the bycatch assemblages could arise by chance (Tonks et al. 2008). The ANOSIM test statistic ("global R") is a comparative measure of the degree of separation between groups: $R = 1$ means that all replicates within groups are more similar to each other than to replicates from different groups. $R=0$ means that there is little or no segregation into groups (Scrosati & Heaven 2008). The threshold for acceptance of a significant difference in a pair-wise comparison was set at $P = 0.05$.

Multivariate analysis of community composition used the PRIMER (Plymouth Routines in Multivariate Ecological Research) ecological statistical software package V6.

2.2 Estimation of total bycatch in the Manx queen scallop fishery

In order to obtain a figure of relative discards from the dredge and trawl elements of the queen scallop fishery, estimates of discard biomass per unit weight of landings (weight of discards (tonnes) per weight of landings (tonnes) were calculated. This figure was considered the most appropriate as compared to an annual biomass estimate, which experiences considerable fluctuation year on year (Murray & Kaiser 2011). In addition, few Isle of Man queenie vessels dredge for queen scallops; therefore due to commercial sensitivities there was the concern to avoid including any figures which would provide an indication of individual vessel landings and fishing effort data (Murray, *pers.comm.*[2]).

[2] Murray, L. (2011) *Discussion on data confidentiality*. 20th September 2011.

The average biomass discarded (tonnes) per unit weight (100 tonnes) of landings was calculated as:

$$\frac{\sum_{i=1}^{i=n}\left(\frac{100 b_i}{a_i}\right)}{n}$$

Where b_i = the estimated total bycatch weight in sample i (kg), a_i = the estimated total landed queenie weight in sample i (kg) and n = the total number of tows sampled.

2.3 Experiment 2: Bycatch Damage and Mortality Assessment

2.3.1 Experimental Design and Sampling

Bycatch Damage

The aims of the bycatch damage assessment were to examine variation in the damage caused by the queen scallop fishery across different fishing gears and on-deck sorting processes. The assessment therefore investigated two treatment factors: across Vessel with treatment levels OT, MD and SD and across Sorting with treatment levels PRE and POST defined as:

- "PRE" - damage levels of bycatch following towing and landing of net contents on-deck, but prior to on-deck catchsorting activities; and
- "POST" – damage levels of bycatch following on-deck sorting and prior to discarding.

The response variables were the proportions of undamaged (DL1), lightly damaged (DL2), severely damaged (DL3) and crushed/dead (DL4) individuals in each replicate tow.

A pilot study carried out in May 2011 (on board RV Prince Madog) assessed damage rates in all bycatch species sampled during trawling and dredging activities at several queen scallop fishing grounds across the Isle of Man. This initial exercise indicated that whilst on board fishing vessels an assessment of damage across all species would be unfeasible given the restraints of working on fishing vessels and the time required to process the samples. The damage study was therefore restricted to two species of the same taxon (Echinodermata), regularly encountered across Isle of Man scallop fishing grounds throughout the pilot study. These included the echinoderms *Asterias rubens* and *Echinus esculentus*.

Samples of "pre-sorting" bycatch were obtained from the sub-sampled catch (see Experiment 1). *A.rubens* and *E.esculentus* individuals present in the sub-sample were subjectively assessed for damage using a four point scale adapted to each taxonomic group (as per Veale et al. 2001 – Table 2). "Post-sorting" bycatch was collected at the end of each vessel's on-deck sorting process and assessed for damage using the same four point scale detailed in Table 2. In addition, sub-samples of post-sorting damage-assessed individuals were taken and the size of each individual determined to the nearest mm (length of the longest arm).

Table 2: Damage scores for selected echinoderms retained as bycatch during surveys of commercial queen scallop fishing grounds (reproduced from Veale et al. 2001)

Damage Level	**DL1**	**DL2**	**DL3**	**DL4**
Common Starfish *Asterias rubens*	No visible damage	Arms missing	Worn and arms missing/minor disc damage	Major disc damage/dead
Edible Urchin *Echinus esculentus*	No visible damage	<50% spine loss	>50% spine loss/minor cracks	Crushed / dead

Bycatch mortality

A long-term (10 days) study of post-fishing mortality was undertaken in order to accurately determine *A.rubens* and *E.esculentus* mortality on discard. This experiment investigated two treatment factors in order to quantify their effect on survival: Vessel (treatment levels SD and OT) and the degree of damage to *A.rubens* and *E.esculentus* (treatment levels DL1, DL2 and DL3). Samples of *E.esculentus* and *A.rubens* were collected from vessels SD and OT at the end of the sorting process and prior to discarding at sea. Each sample was composed of individuals from one species randomly selected within the same tow. Samples were stored in perforated plastic bags in tanks with running sea water and subsequently transferred to nephrops creels (one sample per creel) modified to prevent the entry of large epibenthic predators (Figure 3). The creels were deployed at approximately 20 m depth in Douglas Bay (with a 2 m distance between each creel) and hauled after 10 ten days. Upon hauling the number of live animals in each creel was counted. Individuals were considered alive using the following criteria: *A.rubens*: movement of tube feet; *E.esculentus*: movement of spines and tube feet. Since discards often remain on deck exposed to air for 40-60 minutes on board the SD and OT, the selected test animals were exposed to air for 40-50 minutes before deployment at sea.

Figure 3: Plastic nephrops creels employed to house and deploy at sea *Asterias rubens* and *Echinus esculentus* samples used in Experiment 2. Creels measured ca. 56 x 43 x 33 cm and were modified using plastic sheets or mesh to prevent the entrance of large epibenthic predators.

Intact *A.rubens* and *E.esculentus* were either captured in nephrops creels baited with saithe and deployed in Douglas Bay or captured by divers in Port Erin Bay and used as controls for the effects of trawling and on-deck damage. During transfer into creels, air exposure of the control animals was minimised and maintained approximately equal to that experienced by the catch samples upon transfer and hauling (~45 min). Control animals were deployed at sea in Douglas Bay and hauled after 10 days for mortality assessment. It was difficult to obtain control animals by either baiting or using divers, and only two samples of *A.rubens* (n=5 and n=4) and five samples of *E.esculentus* (n=10, n=9, n=2, n=4 and n=4) were deployed at sea (one sample per creel).

2.3.2 Data Analysis

The statistical analysis of the significance of treatment factors and their interactions on damage levels was carried out for each species using generalised linear models (GLMs). GLMs are being increasingly accepted as the most appropriate tool for analysing data with a non-normal error distribution and non-constant variance such as proportional data (Bolker et al. 2009; Crawley 2007). The GLMs were fitted using the “glm()” function within statistical software “R” (R Development Core Team 2011). The binomial distribution family was selected as the most appropriate for proportional data (Crawley 2007). The link function for each model was the canonical function for the binomial distribution family (i.e. logit link function).

The general equation used to predict the proportion of damaged or dead individuals across factors for each species was:

$$Y_i = B(n_i, \pi_i)$$

$$E(Y_i) = \pi_i \times n_i \quad \text{and} \quad var(Y_i) = n_i \times \pi_i \times (1 - \pi_i)$$

$$logit(\pi_i) = \ln\left(\frac{\pi}{1-\pi}\right) = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \ldots + \beta_i x_i$$

Where Y_i is the response variable defined as the number of "successes" in sample i out of n_i individuals (e.g. number of dead animals in a sample of n_i individuals) and π_i is the probability of the response variable on "success". β_i denotes the effect on the logit of classification in category i of variable x (Zuur et al. 2009; Crawley 2007; Agresti 2002). The response variables were, in turn, the proportions of (out of the total sample):

- Damage level (DL) 1 *A.rubens*
- DL 2 *A.rubens*
- DL 3 *A.rubens*
- DL 4 *A.rubens*
- DL 2 *E.esculentus*
- DL 3 *E.esculentus*
- DL4 *E.esculentus*
- Dead *A.rubens*
- Dead *E.esculentus.*

It should be noted that DL1 *E.esculentus* as a response variable were not considered due to the difficulty in obtaining undamaged specimens from the sampled catch.

Table 3 presents the summary of statistics for the data used to fit the models for each species and the notation used for each variable in the present report. Control animals were not included in the models due to the very low sample size.

Table 3: Summary of statistics for damage and mortality data for each study and species. The table gives the number of replicates (tows) (n) within each level of the categorical explanatory variables. * reference level for each categorical variable in the generalised linear models.

	Asterias rubens			*Echinus esculentus*	
Damage Assessment					
Sorting	PRE	POST		PRE	POST
Vessel					
Otter Trawl (OT)	15	23		15	23
Skid Dredge (SD)	17	18		16	17
Modified Dredge (MD)	12	18		12	16
Mortality Assessment					
Damage Level (DL)	DL1	DL2	DL3	DL2	DL3
Vessel					
OT	7	7	5	7	7
SD	7	7	11	7	7

A minimal adequate GLM was fitted to the data sets. In order to achieve this, the model fitting process followed best practice for GLMs involving proportional data proposed by Crawley (2007). A maximal model - i.e. all factors and interactions between factors – was initially fitted. Residual deviance was then checked and in case of overdispersion (taken when the ratio residual deviance / residual degrees of freedom >1) the model was refitted with a quasibinomial error family. Model simplification involved stepwise regression from the saturated model including all explanatory variables and possible interactions through a series of deletion tests until a minimal adequate model was reached. Deletion tests involved the removal of a term from the model (firstly interaction terms, and then single terms) and performance of a χ^2 or F-test (for binomial and quasi-binomial models respectively – Crawley 2007) using the R function "anova()" to assess the significance of the increase in deviance that results (significance threshold taken as $p<0.05$). This function performs a χ^2 test or an F-test to compare models fitted with a binomial or quasi-binomial model respectively (Crawley 2007). If a deletion caused an insignificant increase in deviance, the term deleted was left out of the model. Deletion tests were undertaken until the model contained nothing but significant terms. It is important to note that this was not always possible as a single term, even if non-significant, could not be removed if an interaction in which it is contained was significant and thus retained in the model. For each species and response variable, the fit of the final (minimal adequate) model was assessed visually by plotting Pearson residuals against the predicted values and Quantile-Quantile plots (Ochwada et al. 2008).

The regression coefficients generated are expressed in logits (ln(p/(1-p)). Therefore, back transformation from a logit (x) to a proportion p was undertaken as follows (Crawley 2007):

$$p = \frac{1}{1 + \frac{1}{e^x}}$$

This calculation was automated in R using "predict" as per Crawley (2007) p. 580.

It was assumed that all animals assessed as DL4 (crushed / dead) would die on discard. In addition, as the mortality of DL1 *E.esculentus* was not assessed in this experiment, it was assumed that all undamaged (DL1) *E.esculentus* would survive. This is likely to lead to an underestimation of mortality and results should thus be interpreted as minimum estimates of discard mortality. The proportion of animals expected to die was calculated as follows (after Veale et al. 2001):

$$\text{Proportion expected to die} = \frac{\sum_{\mathrm{i=1}}^{\mathrm{i=4}} \mathrm{an}}{\mathrm{N}}$$

where a = the proportion of damage level i predicted to die at 10 days using GLM), n = the number of score i in damage-assessed sub-sample, and N = total number of all scores in damage-assessed sub-sample.

2.3.3 *Relationship between Asterias rubens size and damage*

Differences in the length of *A.rubens* across vessels and damage levels were determined using one-way ANOVA followed by post-hoc Fisher's test of Least Significant Difference (LSD). Non-parametric Kruskal-Wallis and post-hoc pair-wise Mann-Whitney U tests were used whenever the condition of homogeneity of variance was not met (Levene's test).

2.3.4 *Relationship between damage in Asterias rubens and Echinus esculentus to the proportion of stones in the catch*

Differences in the proportion of stones across vessels were determined using Mann-Whitney U test as arcsine transformed proportional data did not meet the assumption of homogeneous variance (Levene's test). Relationships between the proportion of undamaged individuals and that of stones in the dredges were tested by non-parametric correlation (Spearman's Rank).

3. Results

3.1 Bycatch Abundance and Composition

Catch was sampled from 44 sites in total, including 12 sites with the MD (average tow time 31.5 minutes), 17 sites with the SD (average tow time 44.8 minutes) and 15 sites with the OT (average tow time 83.2 minutes). The total of 126.3 kg of bycatch processed from the catch samples consisted of 3492 individuals from 70 taxa (excluding queenies). Abundances of *Aequipecten opercularis* (number ha^{-1}) differed significantly across vessels (Kruskal-Wallis $\chi^2 = 29.670$ d.f. = 2 $P < 0.001$). Post-hoc analysis (Mann-Whitney U pairwise tests – Table 4) showed that *A.opercularis* abundances were significantly higher for the MD than the other two vessels, and lowest for the OT. Bycatch abundance was also significantly higher for the MD and lowest for the OT (Kruskal-Wallis $\chi^2 = 19.703$ d.f. 2 $P < 0.001$, Figure 4a). The total average weight of discards per 100 tonnes of landings was estimated at 5.4 tonnes ± 0.8 for the MD, 24.6 tonnes ± 3.1 for the SD and 11.8 tonnes ± 2.7. Table 5 summarises the taxa which were discarded in more than 10% of the sampled hauls. A species-cumulative catch curve shows however that the number of samples collected is unlikely to be representative of the majority of taxa from the fishing grounds sampled (Figure 5).

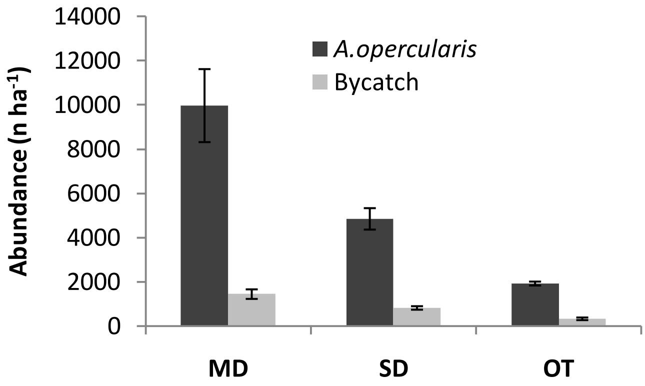
The OT caught a greater number of taxa (44 taxa) than the SD or MD (40 and 37 taxa respectively). Table 5 shows the percentage mean abundance and percentage occurrence of bycatch species encountered in more than 10% of all tows. The dredges caught more invertebrate species, in particular of the phylum Arthropoda, which is poorly represented in the OT. Arthropod abundances were overall higher in the dredges than in the trawl. Echinoderm abundances were also higher in the dredges. Starfish abundance (*Asterias rubens*, *Crossaster papposus*, *Henricia oculata*, however not *Porania pulvillus*) was generally higher in the dredges than in the trawl; abundances of sea urchin *Echinus esculentus* was highest in the trawl, and *Spatangus purpureus* ocurred in a higher proportion of trawl samples, but at a very low abundance. Mollusc abundance was highest in the modified dredge and lowest in the trawl. Percentage occurrence of fish was highest in the trawler.

Table 4: Mann-Whitney U Pair-Wise Comparisons of *Aequipecten opercularis* and bycatch abundances. MD: Modified Dredge (n=12); OT: Otter Trawl (n=15); SD: Skid Dredge (n=17)

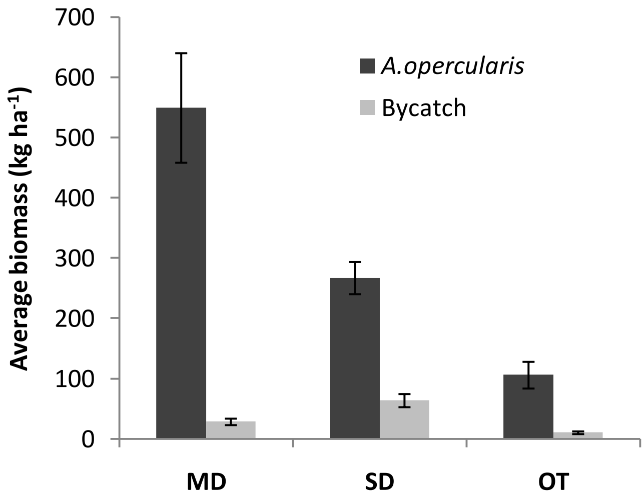
	U	W	P
Abundance of *A.opercularis* (No. individuals ha-1)			
OT,SD	0.0	66.0	0.000
OT,MD	0.0	66.0	0.000
SD,MD	24.0	177.0	0.000
Bycatch abundance (No. individuals ha-1)			
OT,SD	18.0	84.0	0.000
OT,MD	8.0	74.0	0.000
SD,MD	48.0	201.0	0.017

The MD removed the largest biomass per hectare (578.0 kg ha^{-1} ± 95.1) as compared to the SD and OT (331.1 kg ha^{-1} ± 35.3 and 116.6 kg ha^{-1} ± 5.6 respectively, Figure 4b). However, approximately 95% of the MD catch was estimated to be the target species, a higher proportion than for the other two vessels (80.7% and 91.0% for the SD and OT respectively) and suggests that the MD may be comparatively more efficient at catching the target species. The proportion of queenies out of the total catch abundance (n ha^{-1}) was also highest for the MD (87.3%) as compared to the SD (85.5%) and OT (85.3%). The relationship between target catch and bycatch is summarised by a plot of bycatch to target catch ratio (Figure 4c). Abundances of *A.opercularis* (number ha^{-1}) differed significantly across vessels (Kruskal-Wallis χ^2 = 29.670, d.f., = 2, P < 0.001). Post-hoc analysis (Mann-Whitney U pairwise tests – Table 4) showed that *A.opercularis* abundances were significantly higher for the MD than the other two vessels, and lowest for the OT. Bycatch abundance was also significantly higher for the MD and lowest for the OT (Kruskal-Wallis χ^2 = 19.703, d.f. 2, P < 0.001, Figure 4a). The total average weight of discards per 100 tonnes of landings was estimated at 5.4 tonnes ± 0.8 for the MD, 24.6 tonnes ± 3.1 for the SD and 11.8 tonnes ± 2.7.

a)



b)



c)

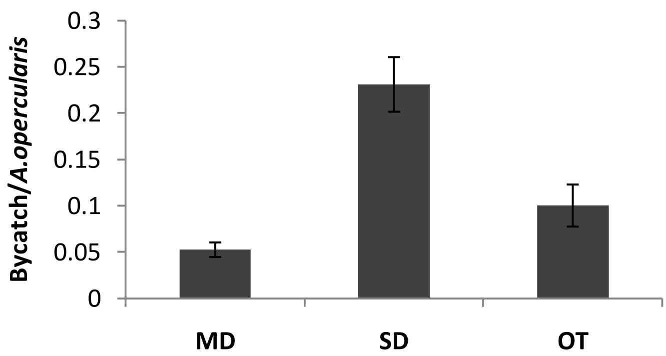


Figure 4: a) Individuals of the target species *Aequipecten opercularis* and by-catch species caught by the three different gears per hectare swept. b) biomass (kg) of the target species *A. opercularis* and by-catch species caught by the three different gears per hectare swept c) Ratio of bycatch to *A. opercularis* catch. MD: Modified Dredge; SD: Skid Dredge; OT: Otter Trawl.

Table 5: Mean catch per unit effort (CPUE) and percentage occurrence (oc.) of the species discarded in more than 10% of all tows.

	MD			SD			OT		
Species	% oc. (n=12)	Mean CPUE (n ha^{-1})	SE	% oc. (n=17)	Mean CPUE (n ha^{-1})	SE	% oc. (n=15)	Mean CPUE (n ha^{-1})	SE
Anthozoa									
Alcyonium digitatum	50.0	103.0	38.9				81.8	64.2	24.2
Ascidiacea									
Ascidiacea	75.0	44.2	17.5				100.0	16.0	2.1
Arthropoda									
Hyas sp				29.4	7.6	4.0			
Inachus sp	25.0	9.1	4.9	35.3	9.7	3.8			
Pagurus prideauxi	41.7	11.9	4.9	47.1	7.6	2.3			
Pagurus sp	75.0	64.0	20.6	100.0	64.5	18.7	90.9	11.1	2.8
Echinodermata									
Asterias rubens	100.0	192.4	76.9	100.0	144.6	19.8	90.9	20.5	6.4
Crossaster papposus	100.0	62.2	23.5	82.4	44.1	8.9	100.0	36.1	7.6
Henricia oculata	58.3	58.2	21.8	64.7	38.0	12.0	100.0	25.5	5.4
Porania pulvillus							36.4	1.6	0.7
Echinus esculentus	66.7	25.4	6.6	94.1	37.5	7.6	100.0	88.3	15.4
Psammechinus miliaris	91.7	273.9	49.8	82.4	125.0	31.2	63.6	3.6	1.2
Spatangus purpureus							27.3	0.9	0.5
Ophiura albida	91.7	253.4	111.2	100.0	94.7	15.9			
Ophiocomina nigra							45.5	4.3	1.9
Ophiothrix fragilis	83.3	119.0	38.2	76.5	127.2	45.5	81.8	19.4	4.4
Ophiura ophiura	25.0	12.2	6.6	41.2	7.1	2.9			
Medusozoa									
Medusozoa							27.3	1.5	0.8
Mollusca									
Anomia sp.							54.5	2.4	0.7
Clausinella fasciata	50.0	24.5	9.4						
Glycymeris glycymeris							27.3	1.2	0.7
Modiolus modiolus							45.5	6.6	4.1
Pecten maximus	58.3	39.4	15.4	70.6	22.8	7.3			
Buccinum undatum	83.3	32.0	8.7	76.5	23.3	4.1	45.5	4.1	1.7
Calliostoma	25.0	11.1	6.9						
Neptunea antiqua	83.3	41.2	12.2	47.1	16.9	6.2			
Chordata									
Red gurnard Aspitrigla cuculus							45.5	8.3	4.3
Scyliorhinus canicula							36.4	2.1	1.0
Polychaeta									
Polychaeta sp.	25.0	8.7	5.2						

The cluster dendrogram and MDS plot shown in Figure 6 were generated from the calculated Bray-Curtis similarity matrix (Clarke & Warwick 2001). The cluster dendrogram (Figure 6a) revealed a difference in community structure and abundance patterns between the OT and the two dredges. The first major dichotomy of the dendrogram splits the samples into two distinct groups separating the OT samples from other two groups. The MDS plot illustrates the similarity between samples in a two-dimensional form where the degree of similarity is represented by the distance between points (Kaiser et al. 1994). It is apparent that the OT samples cluster in a distinct group. However, although most SD samples cluster together, there was an overlap in similarity between the SD and MD samples. The samples collected with the SD appear overall to be more tightly clustered than those collected with the MD, suggesting a lower variation between samples collected with the SD gear.

Significant dissimilarities in bycatch abundance and composition ocurred between gears (ANOSIM, $R=0.621$, $P < 0.001$). Multivariate pair-wise ANOSIM tests conducted on the bycatch data showed that all pairwise comparisons of vessels showed a statistically significant difference in species composition (Table 6). ANOSIM R statistics suggest that dissimilarities between the OT and SD were higher than those between OT and MD (Clarke & Gorley 2006). The species that most contributed to the similarity in bycatches within each vessel were identified by the SIMPER analysis and are presented in Table 7.

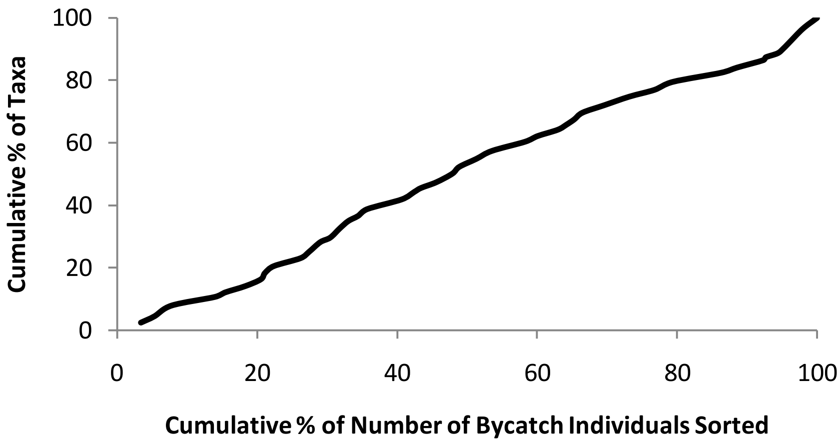
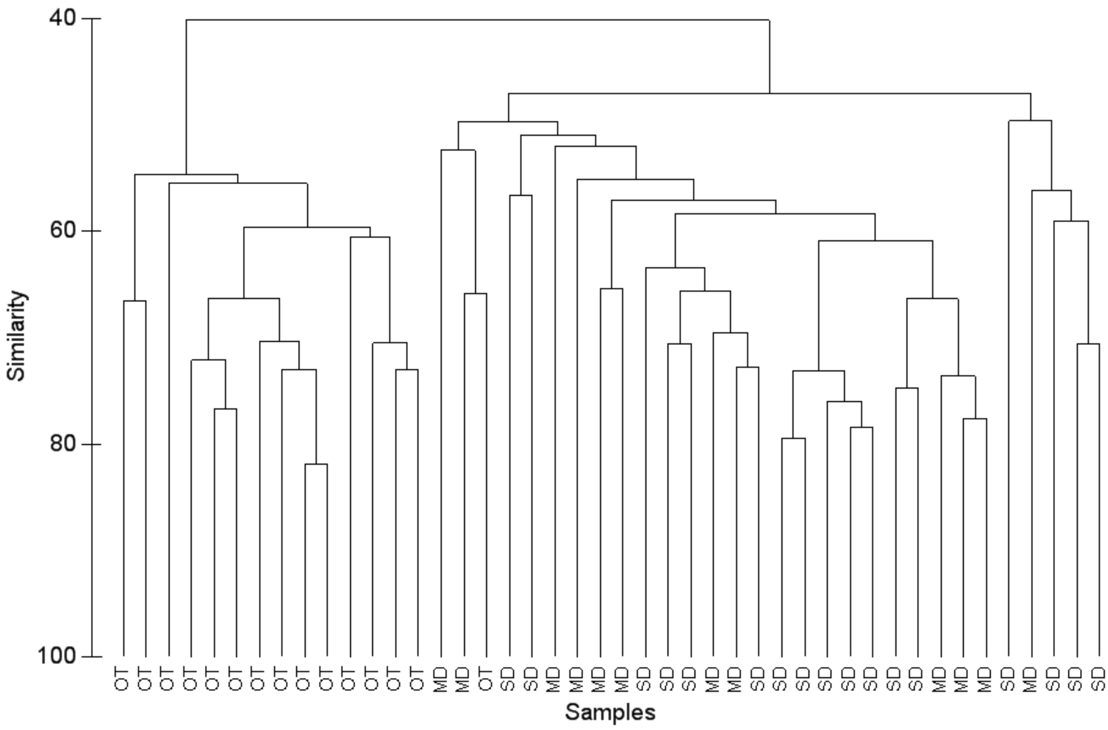


Figure 5: Cumulative percentage of taxa identified plotted against the cumulative number of bycatch individuals processed for bycatch caught during sampled tows.

a)



b)

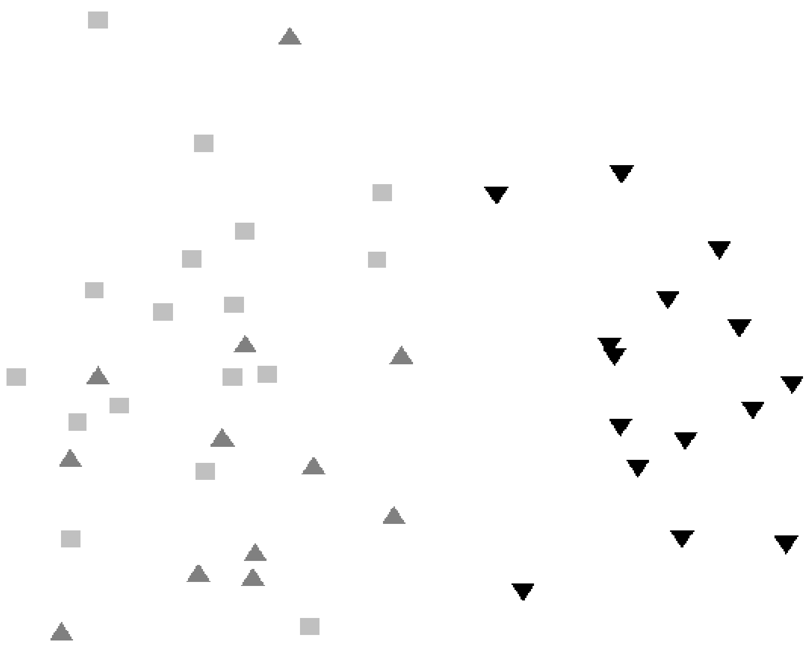


Figure 6: (a) Cluster analysis and (b) multidimensional scaling (MDS) of the bycatch in samples (standardised data) taken with the Modified Dredge (MD, ▲), Otter Trawl (OT, ▼) and Skid Dredge (SD, ■).

Table 6: ANOSIM pair-wise comparisons of bycatch species compositions. MD: Modified dredge; OT: Otter Trawl; SD: Skid dredge.

	R Statistic	**P**
MD,OT	0.798	0.001
MD,SD	0.133	0.016
OT,SD	0.856	0.001

Echinoderms were overall the most important taxa contributing to intra-group similarity. OT catches were primarily characterised by a small number of echinoderm species, in particular *E.esculentus* and *Crossaster papposus*, as well the anthozoan *Alcyonium digitatum*. Similarities within each of the two dredges were mostly due to the relative abundance of echinoderm species, namely *A.rubens*, *Ophiura albida* and *Psammechinus miliaris*, the latter two species being relatively small and less likely to be caught in the trawl net.

Table 7: Summary of results of SIMPER analysis showing the average similarity and percentage contribution of species to the similarity matrix of bycatch within each vessel. Only the species that contributed to 80% of the overall similarity for each gear are shown.

	MD		**SD**		**OT**	
	Average similarity	**% Contribution**	**Average similarity**	**% Contribution**	**Average similarity**	**% Contribution**
Echinodermata						
Asterias rubens	7.39	13.79	11.52	20.39	5.01	8.45
Crossaster papposus	4.31	8.04	4.01	7.11	8.35	14.08
Echinus esculentus			4.74	8.39	13.35	22.5
Henricia oculata					5.91	9.96
Ophiothrix fragilis	5.21	9.73	4.41	7.8		
Ophiura albida	7.07	13.19	8.41	14.89		
Psammechinus miliaris	9.3	17.35	6.16	10.91		
Crustacea						
Pagurus sp	3.02	5.63	6.5	11.51		
Mollusca						
Buccinum undatum	2.72	5.08				
Neptunea antiqua	2.76	5.14				
Pecten maximus						
Anthozoa						
Alcyonium digitatum					7.41	12.48
Ascidiacea						
Ascidiacea					5.81	9.8

3.2 Predicting damage and mortality with generalised linear models

Final GLM models for *A.rubens* and *E.esculentus* damage scores and mortality rates are presented in Table 8 and include different numbers of variables. Examination of model residuals exhibited minimal divergence or patterns (Appendix 1).

3.2.1 Damage Assessment

Observed proportions of individuals in each damage level are given in Table 9. The proportions of undamaged (DL1) and slightly damaged (DL2) *E.esculentus* were higher before sorting. A higher proportion of *E.esculentus* sampled post-sorting was highly damaged or crushed/dead (DL3 and DL4). *A.rubens* sampled before sorting exhibit overall higher damage levels than the individuals sampled following sorting (higher proportion of DL3 and DL4 individuals and lower proportion of DL1 individuals before sorting). Such an observation goes against common sense and may be a result of a number of methodological limitations in the fieldwork, namely: (i) reduced sample sizes pre-sorting may not have allowed for the full variability in damage within the samples to be recorded (Table 2); (ii) tows were not systematically sampled both before and after sorting, i.e. the pre-sorting tows do not entirely overlap with post-sorting tows, therefore the samples do not reflect actual differences between damage levels before and after sorting. Despite such limitations it is notable that the proportions of undamaged individuals were consistently highest in the MD and lowest in the OT for both species and both before and after sorting.

The results of the F-tests performed on the fixed effects added to the *A.rubens* and *E.esculentus* damage models are shown in Appendix 1. Stepwise deletion and F-tests performed on the fixed effects and interactions showed that Vessel and Sorting were significant factors throughout the four damage levels, with the exception of Sorting in the model fitted to DL4.

Table 8: Results of the analysis used to predict a) the proportion of damage individuals in each damage score and b) the proportion of dead individuals at ten days for *Asterias rubens* and *Echinus esculentus*. These generalised linear models assumed a binomial distribution. The table gives each model's intercept (β0) and its standard error (S.E.); the partial regression coefficient for each model's categorical independent variables and their standard errors (β1-5); and the probability that βi =0 for the intercept and each categorical variable (P(>|t|)).

8a)

***Asterias rubens* (n=103)**			***Echinus esculentus* (n=100)**		
Damage Level 1	**β (S.E.)**	**P(>$\|t\|$)**	**Damage Level 1**	**β (S.E.)**	**P(>$\|t\|$)**
Intercept	β_0 = 1.24 (0.12)	< 0.0001	Intercept	β_0 =-3.79 (0.28)	< 0.0001
$Vessel_{MP}$	β_1 =-1.42 (0.15)	< 0.0001	$Vessel_{MP}$	β_1 = -1.89 (0.46)	< 0.0001
$Vessel_{SD}$	β_2 =-0.98 (0.15)	< 0.0001	$Vessel_{SD}$	β_2 =-0.45(0.42)	> 0.05
$SORT_{PRE}$	β_3 =-0.84 (0.17)	< 0.0001	$SORT_{PRE}$	β_3 = 2.22 (0.39)	< 0.0001
Damage Level 2	**β (S.E.)**	**P(>$\|t\|$)**	**Damage Level 2**	**β (S.E.)**	**P(>$\|t\|$)**
Intercept	β_0 =-1.44 (0.11)	< 0.0001	Intercept	β_0 =-0.30 (0.12)	< 0.05
$Vessel_{MP}$	β_1 = 0.74 (0.15)	< 0.0001	$Vessel_{MP}$	β_1 = -1.06 (0.16)	< 0.0001
$Vessel_{SD}$	β_2 = 0.75 (0.14)	< 0.0001	$Vessel_{SD}$	β_2 = 0.03 (0.18)	> 0.05
$SORT_{PRE}$	β_3 = 0.42 (0.29)	> 0.05	$SORT_{PRE}$	β_3 =-0.05 (0.75)	> 0.05
$Vessel_{MP}$: $SORT_{PRE}$	β_4 =-0.69 (0.47)	> 0.05	$Vessel_{MP}$: $SORT_{PRE}$	β_4 = 1.51 (0.79)	> 0.05
$Vessel_{SD}$: $SORT_{PRE}$	β_4 =-1.2 (0.37)	< 0.01	$Vessel_{SD}$: $SORT_{PRE}$	β_5 =-2.50 (1.18)	< 0.05
Damage Level 3	**β (S.E.)**	**P(>$\|t\|$)**	**Damage Level 3**	**β (S.E.)**	**P(>$\|t\|$)**
Intercept	β_0 =-3.48 (0.23)	< 0.0001	Intercept	β_0 =-0.45 (0.11)	< 0.0001
$Vessel_{MP}$	β_1 = 2.08 (0.25)	< 0.0001	$Vessel_{MP}$	β_1 = 0.56 (0.13)	< 0.0001
$Vessel_{SD}$	β_2 = 1.10 (0.27)	< 0.0001	$Vessel_{SD}$	β_2 =-0.08 (0.16)	> 0.05
$SORT_{PRE}$	β_3 = 1.11 (0.44)	< 0.05	$SORT_{PRE}$	β_3 =-0.73 (0.78)	> 0.05
$Vessel_{MP}$: $SORT_{PRE}$	β_4 =-0.95 (0.57)	> 0.05	$Vessel_{MP}$: $SORT_{PRE}$	β_4 =-0.43 (0.80)	> 0.05
$Vessel_{SD}$: $SORT_{PRE}$	β_5 =-1.55 (0.57)	< 0.01	$Vessel_{SD}$: $SORT_{PRE}$	β_5 = 0.76 (0.87)	> 0.05
Damage Level 4	**β (S.E.)**	**P(>$\|Chi\|$)**	**Damage Level 4**	**β (S.E.)**	**P(>$\|t\|$)**
Intercept	β_0 =-5.25 (0.45)	< 0.0001	Intercept	β_0 =-1.63 (0.15)	< 0.0001
$Vessel_{MP}$	β_1 = 1.64 (0.49)	< 0.001	$Vessel_{MP}$	β_1 = 0.53 (0.17)	< 0.01
$Vessel_{SD}$	β_2 =-0.14 (0.63)	> 0.05	$Vessel_{SD}$	β_2 = 0.11 (0.21)	> 0.05

8b)

***Asterias rubens* (n=58)**			***Echinus esculentus* (n=54)**		
Mortality	**β (S.E.)**	**P(>$\|t\|$)**	**Mortality**	**β (S.E.)**	**P(>$\|t\|$)**
Intercept	β_0 = 3.64 (1.05)	<0.01	Intercept	β_0 = 3.01 (0.36)	< 0.0001
$Vessel_{SD}$	β_1 = 17.27 (2329.0)	> 0.05	$Vessel_{SD}$	β_1 =-1.49 (0.31)	< 0.0001
Damage $Level_{DL1}$	β_2 =-2.27 (1.10)	< 0.05	Damage $Level_{DL2}$	β_2 =-2.29 (0.34)	< 0.0001
Damage $Level_{DL2}$	β_3 =-2.11 (1.11)	> 0.05			
$Vessel_{SD}$:Damage $Level_{DL1}$	β_4 =-18.55 (2329.0)	> 0.05			
$Vessel_{SD}$:Damage $Level_{DL2}$	β_5 =-17.22 (2329.0)	> 0.05			

Table 9: a) Individuals assigned to each damage category (DL1 to DL4) expressed as % of total individuals scored across the tows. *n* represents the number of tows sampled b) Mortality at 10 days expressed as a percentage of dead individuals in each sample. *n* represents the number of samples (creels) OT: Otter Trawl; MD: Modified Dredge; SD: Skid Dredge; CL: Control

a)

		Asterias rubens						*Echinus esculentus*					
Sorting	**Response variable**	**OT**	**n**	**SD**	**n**	**MD**	**n**	**OT**	**n**	**SD**	**n**	**MD**	**n**
PRE	% DL1	41.7	15	55.0	17	64.0	12	3.9	15	7.9	17	23.5	12
	% DL2	28.3	15	32.8	17	27.2	12	52.3	15	7.9	17	41.2	12
	% DL3	23.3	15	9.9	17	8.8	12	26.0	15	52.6	17	23.5	12
	% DL4	6.7	15	2.3	17	0.0	12	17.8	15	31.6	17	11.8	12
POST	% DL1	44.7	23	57.9	18	77.3	18	0.2	23	2.0	17	23.5	16
	% DL2	33.1	23	33.4	18	19.1	18	20.4	23	43.5	17	42.5	16
	% DL3	19.8	23	8.4	18	3.0	18	52.8	23	36.9	17	38.9	16
	% DL4	2.4	23	0.2	18	0.6	18	26.6	23	17.6	17	16.6	16

b)

	OT	**n**	**SD**	**n**	**MD**	**n**	**CL**	**n**	**OT**	**n**	**SD**	**n**	**MD**	**N**	**CL**	**n**
Response variable																
% Dead (DL1)	79.7	7	52.2	7	N/A				N/A	N/A	N/A	N/A	N/A	N/A		
% Dead (DL2)	82.1	7	82.9	7	N/A				65.3	7	33.8	7				
% Dead (DL3)	97.4	5	100.0	11	N/A				97.3	7	80.0	7				
Control							0.0	2	6.9						0.04	5

The minimal adequate models fitted to the proportions of DL1 to DL4 *E.esculentus* are presented in Table 8. The P values indicate significance whereas the regression coefficients provide information on the nature of the relationship. For example, in the model fitted to the proportion of DL1 individuals, the first three coefficients relate to the overall damage rate in the three vessels. The significant coefficient for $SORT_{PRE}$ indicates that logit(odds) is significantly increased by 2.22 pre-sorting relative to post-sorting. The significant $Vessel_{SD}$:$SORT_{PRE}$ interaction in the model fitted to the proportion of DL2 *E.esculentus* indicates that in the SD the logit(odds) is reduced by 2.50 pre-sorting relative to post-sorting (Maindonald & Braun 2003). The model for DL1 *E.esculentus* incorporated both Vessel and Sorting, but the Vessel*Sorting interaction was not significant and was thus removed from the model. It seems therefore that in this model the response variable is affected by Vessel and Sorting. The significance values for each term in the model (Table 8) indicate that there are no differences in the way the SD and MD affect logit (odds), however the difference was significant between MP and MD ($P < 0.001$). The partial regression coefficient ($\beta_2 = -1.89$)

indicates that logit (odds) is significantly decreased by 1.89 for MP in relation to SD. In the models for *E.esculentus* DL2 and DL3 the Vessel*Sorting interaction remained in the model, indicating that sorting affects damage differently in different vessels. The coefficients β_4 and β_5 for both DL2 and DL3 models indicate that sorting in the SD seems to have a stronger effect on the response variable than on the MP.

In order to illustrate the model results, the model was used to generate damage predictions (Table 10). Predicted values indicate an increase in the proportion of highly damaged and dead (DL3 and DL4 respectively) individuals following sorting. In addition, the proportions of undamaged (DL1) individuals pre-sorting were lowest in the OT suggesting a higher sensitivity of the species to trawling as compared to dredging.

Table 10: Mean generalised linear model estimates of measured parameters. a) Mean estimates of proportions of individuals assigned to each damage level (DL) category across Vessel and Sorting (pre-sorting and post-sorting) factors. b) Mean estimates of the proportions of dead individuals across Vessel and Damage Level (DL) factors. Vessel treatment levels include Modified Dredge (MD), Otter Trawl (OT) and Skid Dredge (SD).

a)

Vessel	**Pre-Sorting**				**Post-Sorting**			
Asterias rubens								
	DL1	**DL2**	**DL3**	**DL4**	**DL1**	**DL2**	**DL3**	**DL4**
MD	0.60	0.26	0.09	0.01	0.78	0.19	0.03	0.01
OT	0.27	0.27	0.23	0.03	0.46	0.33	0.20	0.03
SD	0.36	0.19	0.06	0.00	0.57	0.33	0.08	0.00
Echinus esculentus								
	DL1	**DL2**	**DL3**	**DL4**	**DL1**	**DL2**	**DL3**	**DL4**
MD	0.17	0.41	0.24	0.16	0.22	0.43	0.39	0.16
OT	0.03	0.52	0.26	0.25	0.00	0.20	0.53	0.25
SD	0.12	0.06	0.38	0.18	0.01	0.43	0.37	0.18

b)

	Damage Level		
Asterias rubens			
Vessel	**DL1**	**DL2**	**DL3**
OT	0.797	0.821	0.974
SD	0.522	0.829	1.000
Echinus esculentus			
Vessel	**DL1**	**DL2**	**DL3**
OT	N/A	0.673	0.953
SD	N/A	0.317	0.821

3.2.2 Mortality Assessment

The results of the F-tests performed on the fixed effects of the *A.rubens* model are shown in Appendix 2. Stepwise deletion and F-tests performed on the fixed effects and interactions of the *A.rubens* model revealed a significant “Vessel*DL” interaction (at $\alpha=0.05$) which was therefore retained in the model. Damage Level (DL) was also a significant factor. Vessel was not a significant factor in the model; however it could not be deleted since the interaction was retained.

The final models fitted to the proportion of dead *A.rubens* and *E.esculentus* are shown in Table 8. For *A.rubens*, the model indicates that the logit (odds) is significantly decreased by 2.27 for DL1 as compared to DL3 in vessel MP. However, no significant effect of damage level is found in DL2 relative to DL3 for vessel MP. In the SD, there is no significant effect on logit (odds) of DL1 or DL2 as compared to DL3. The model fitted for *E.esculentus* indicates a highly significant reduction in logit (odds) for DL2 as compared to DL3, and a reduction of logit (odds) of 1.49 in SD as compared to OT. The proportion of dead creel and diver caught controls was 0 in both *A.rubens* samples (no mortality) and averaged 0.04 ± 0.03 for *E.esculentus*.

Estimates of the proportions expected to die were calculated over the three vessels as an index of sensitivity to capture in the three different vessels separately (Table 11). These should be considered minimum values that serve the purpose of comparing damage across vessels, as injured animals returned to the seabed would be prone to increased predation. The calculated estimates indicate higher mortality in the OT as compared to the SD.

Table 11: Sensitivity scales for *Asterias rubens* and *Echinus esculentus*. Values are the arithmetic means of the proportion expected to die within 10 days of capture.

	Skid Dredge		**Otter Trawl**
Asterias rubens (n=18)	0.67 ± 0.008	*Asterias rubens* (n=23)	0.84 ± 0.003
Echinus esculentus (n=17)	0.62 ± 0.012	*Echinus esculentus* (n=23)	0.91 ± 0.005

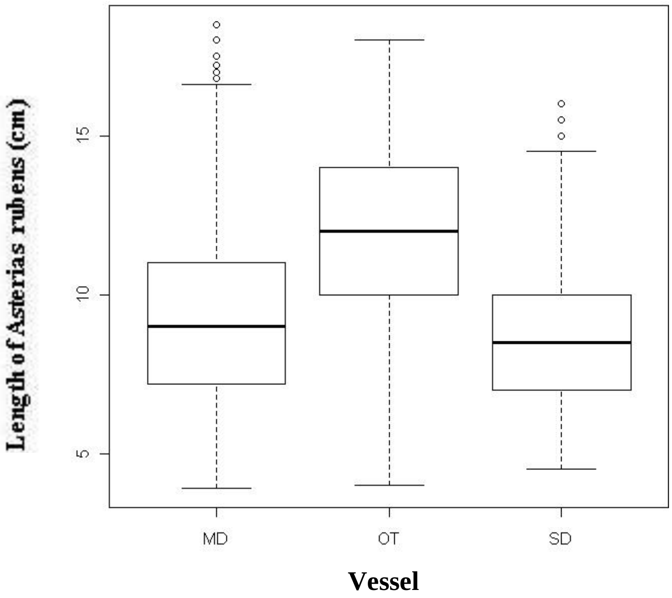


Figure 7: Box plots of the length of *Asterias rubens* (longest arm length, cm) sampled post sorting from the discards of each vessel. MD: Modified Dredge; OT: Otter Trawl; SD: Skid Dredge.

3.2.3 Relationship between size and damage in Asterias rubens

The size of *A.rubens* sampled post-sorting was significantly different across vessels (ANOVA P < 0.001, F = 65.793, d.f. = 2, Figure 7). Post-hoc analysis revealed a significant difference at all levels of the pairwise tests (LSD pairwise tests across vessels: MD-OT P < 0.001, MD-SD P < 0.05, SD-OT P < 0.001) indicating that the mean length of *A.rubens* was highest in the OT and lowest in the SD (mean lengths of *A.rubens* in the OT, MD and SD respectively: 11.75 cm ± 0.18; 9.46 ± 0.12; 8.8 cm ± 0.22).

Length was significantly different across damage levels (Kruskal-Wallis χ^2 = 46.251, d.f. = 3, P < 0.001). Damage increased with an increase in the mean length of *A.rubens* (Figure 8). Lengths of damage categories DL1, DL2 and DL3 were significantly different, however lengths did not differ between DL3 and DL4 (Figure 8, Table 12).

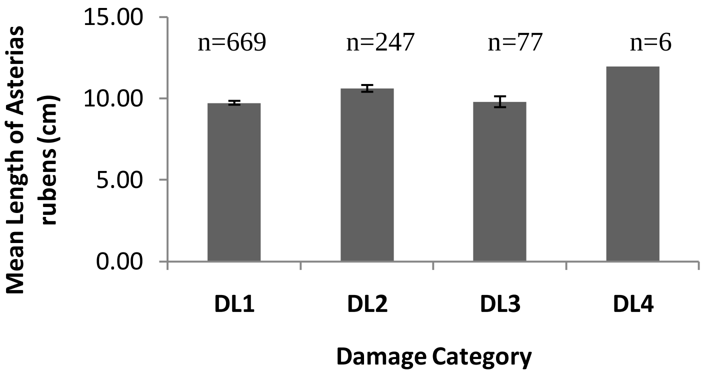


Figure 8: Mean length (and standard error bars) of *Asterias rubens* across damage levels. Data were pooled across all tows

3.2.4 Relationship of damage to the proportion of stones in the catch

Predictably only the two dredges contained stones as a proportion of the total catch. The proportion of stones in the catch was significantly higher for the SD (38.4% ± 4.6 and 14.8% ± 5.4 for the SD and MD respectively, ANOVA $F = 9.360$, d.f. = 1, $P < 0.01$). There was a significant negative correlation between the percentage of undamaged *A.rubens* and the percentage of stones in the catch of the MD (Spearman's Rank, $P < 0.05$, $\rho = -0.697$), however no significant correlation was found between the percentage of undamaged *A.rubens* and the percentage stones in the SD (Spearman's Rank, $P > 0.05$, $\rho = -0.123$). There was no significant correlation between the percentages of undamaged *E.esculentus* and the proportions of stones in the MD (Spearman's Rank, $P > 0.05$, $\rho = -0.395$) and SD (Spearman's Rank, $P > 0.05$, $\rho = 0.065$).

Table 12: Results of Pair-wise Mann-Whitney U tests performed to compare the lengths of *Asterias rubens* (longest arm length, cm) across damage levels (DL).

	P	**Z**	**U**
DL1-DL2	< 0.01	-2.802	72672.5
DL1-DL3	<0.001	-6.005	15013.0
DL1-DL4	< 0.01	-2.981	590.5
DL2-DL3	<0.001	-3.691	6863.0
DL2-DL4	< 0.05	-2.435	310.0
DL3-DL4	> 0.05	-1.568	142.0

4. Discussion

The Isle of Man queen scallop fishery has been prosecuted since the late 1960s. Initially the fishery exploited a reduced number of grounds, however the introduction of spring-toothbar dredges (currently under a total ban[3]) and smaller dredges capable of operating on rougher grounds led the fishery to expand to a higher number of grounds (see review by Moody Marine Ltd undated). Habitat type (e.g. the restriction of trawlers to soft sediment grounds) and management considerations are thus key factors determining vessel movement in the fishery. At present there is no harvest strategy and catches are limited by market demand (Murray et al. 2009). Current management measures in the fishery include areas where dredging is prohibited, a closed season and a minimum landing size[3] (Murray & Kaiser 2011).

This study aimed to provide a comparative assessment of bycatch in the Isle of Man queen scallop fishery encompassing three commercial fishing vessels commonly used in the fishery. Opportunistic sampling was undertaken on-board the fishing vessels. Working on commercial fishing vessels meant accurate data collection was more difficult than in controlled experiments using research vessels. It was therefore not possible to undertake experiments where tow parameters known to affect damage and mortality were kept constant (e.g. tow depth, tow time, air exposure of the catch on-deck, total catch volume / biomass and air temperature - Bergmann et al. 2001). However, our study had the advantage over a research vessel of following regular commercial fishing practice, hardly achievable in a research vessel, in particular with regard to the catch sorting process. The obtained results may therefore provide more useful management information.

4.1 Bycatch Abundance and Diversity

The bycatch from the queen scallop fishery examined was characterised by a large proportion of invertebrates, in particular of the phyla Arthropoda and Echinodermata. Whilst the catch of the dredges was mainly characterised by invertebrates, starfish and molluscs, the catch in the trawl had a higher proportion of sea urchins and fish, a pattern observed in previous studies of this fishery (Hinz et al. in revision; Kaiser et al. 1996). Differences in bycatch species composition were more marked between the trawl and the two dredges than between the SD and MD. This is in line with recent bycatch composition studies of the fishery (Hinz et al. in revision) and reflects how the selective properties of the gear affect different components of the benthic ecosystem.

[3] Sea Fisheries Act 1971. Isle of Man Sea Fisheries (queen scallop fishing) Bye Laws 2010. Statutory Document No. 668/10

The MD caught the highest total biomass per hectare towed, including the highest queenie biomass and the lowest bycatch biomass. The estimated proportion of bycatch of the total catch weight was also lowest for the MD and highest for the SD (5%, 10% and 22% of total catch weight for the MD, OT and SD respectively). This result is broadly in line with a previous MD and OT comparison by Hinz et al. (in revision) which also observed higher catches of queen scallop by the MD, however lowest bycatch abundance was found for the OT (although, similarly to the present study, both bycatch/catch ratios were found to be well below 1). It should be noted however that the abundance and composition of bycatch is also dependent on environmental factors, such as habitat type and abiotic environmental factors (see e.g. Kaiser et al. 2006; Alverson et al. 1994). Available studies on tooth dredge efficiency (as arguably one of the better studied bottom towed gears) reveal a wide range of reported percentage target species in the catch, with values ranging between 1% and 41% (Gedamke et al. 2005; Jenkins et al. 2001; Currie & Parry 1999).

The lower bycatch proportions (both in terms of abundance and biomass) in the MD suggest that it is the most efficient of the gears tested – estimated catch rates indicate that in order to catch the same mean biomass of queenies as that caught by the MD in one hectare towed, the OT and SD would respectively need to tow 4.98 ha and 1.74 ha of seabed. Sampling was undertaken in the summer months when queen scallops are most active and most likely to be caught by trawling as their active upwards escape response brings them to the net mouth, but often allows them to swim over the dredge mouth (Jenkins et al. 2003). It is therefore reasonable to speculate that the relative efficiency of the MD in comparison with the OT is likely to be even higher in the winter months. Conversely, in winter months the OT is likely to become less efficient as compared with the SD when the swimming activity of the queen scallops is reduced.

The improved efficiency of the MD in relation to the SD may be due to the increased height of the gear above seabed (30 cm for the MD as opposed to 20 cm for the SD (Hatton and White, *pers.comm.*[4]) allowing more queenies to be caught as they swim upwards. In addition, it should also be noted that a common reason for reduced dredge efficiency is the clogging of the dredge mouth with sediment, blocking more queenies from entering the dredge (Leitão et al. 2009). A reduction in the amount of sediment caught by the dredge may help explain the higher efficiency and lower proportion of bycatch in the MD, as a quicker fill up of the dredges with the target species would reduce the opportunity for bycatch to be stored in the dredge. It must be noted however that the efficiency and catch composition of a vessel are

[4] Hatton, S. and White, D. (2011) Discussions with Steve Hatton (Skipper of Q-Varl) and Dougie White (Skipper of King Challenger) re. gear dimensions. August 2011.

determined not only by fishery characteristics (selectivity of the fishing gear and fishing behaviour) but also the community composition of the habitat towed (Currie 1999; Pranovi 2001). Sediment type in the Isle of Man fishing grounds exhibits high levels heterogeneity (Jenkins 2001) and consequently variation in community types (Veale 2001). The abundance and composition of the bycatch in the present study are thus not necessarily representative of that caught in other fishing ground types and by the different gears; it follows that if effective ecosystem-led management and/or monitoring decisions are to be made regarding the intensity and spatial distribution of fishing effort the results of the present study would need to be refined with regard to habitat-specific variations in vessel efficiency and the nature and intensity of fishing impacts.

4.2 Bycatch damage

In this study a damage scale was applied in relation to the physical damage incurred to bycatch individuals. External damage to individuals as a result of trawling has been found to be highly species specific (e.g. Pranovi et al. 2001; Kaiser & Spencer 1995). Sartor et al. (2006) noted that damage as a result of towed gears is more frequently found in certain groups such as echinoderms and crabs rather than hermit crabs and gastropods, different species having differing degrees of sensitivity. The present study found more *Echinus esculentus* than *Asterias rubens* to show the highest level of damage, i.e. crushed / dead (0.2 to 2.4% for *A.rubens* and 16.6% to 26.6% for *E.esculentus*), a difference which is related to the anatomical characteristics of each species, allowing them differing degrees of flexibility: the test of *A.rubens*, constructed of interlinked plates, would allow it more flexibility than the fused plates of sea urchins (Kaiser & Spencer 1995). The results of our study appear to be broadly in line with, although slightly higher than, previous findings by Kaiser & Spencer (1995) and Ramsay et al. (1998) where *A.rubens* was found to be fairly resistant to the effects of trawling. Kaiser & Spencer (1995) found 71-73% of *A.rubens* to be undamaged following beam trawling. Veale et al. (2001) found *E.esculentus* to be significantly more sensitive than *A.rubens* to tooth dredges; the study found 7% of *A.rubens* and 34% of *E.esculentus* to be highly damaged or crushed (DL3 and DL4) after towing, as compared to 8.8-30% of *A.rubens* and 35.3-84.2% of *E.esculentus* in the present study. However, the higher damage observed in the present study may be due in part to seasonality. Our study was undertaken in the summer months where longer tows are associated with larger catches, which may signify higher levels of physical injury (Bergmann et al. 2001).

The comparison between total damage (i.e. post-sorting damage encompassing towing and sorting) suffered by the same species caught in the three vessels indicated that animals were more severely injured in the trawl than in the other two gears, whereas the modified dredge showed the lowest level of injury. Significant differences in total damage were found across vessels for both species. Differences in damage across vessels may be related to the characteristics of the catch, namely the higher efficiency and lower proportion of stones in the modified dredge. Damage in bycatch has been previously found to be inversely proportional to catch efficiency in a bivalve dredge fishery (Gaspar et al. 2003; Gaspar et al. 2001) and correlated with the proportion of stones in the dredge of a scallop fishery (Veale et al. 2001; Hill et al. 1996). However, the available body of knowledge indicates overall that the importance of these factors as contributors to damage and mortality is likely to be highly species specific (refer to Parker et al. 2003, Kaiser & Spencer 1995 and Van Beek et al. 1990 as cited in Kaiser & Spencer 1995). In this context a previous study of damage in a beam trawl (Kaiser & Spencer 1995) suggested that damage to asteroid starfish did not seem to be related to tow time (and subsequent heavier catch weight) and that observed injuries were more likely to be due to a specific part of the gear. Whilst direct comparisons of damage in the bycatch across different gears are lacking in the literature, the relationship of damage to gear fullness and catch volume has been previously reported. Large, heavy catches, in particular if the contribution of "hard" material is high, increase the probability of injury both during hauling as well as whilst on deck (Sartor et al. 2006; Oddsson et al. 1994). Although catch volume and net/dredge fullness were not recorded in this experiment, on-deck observations have suggested the OT to be consistently fuller upon hauling (with bycatch visibly compressed against net), possibly as a result of longer tow times. This may have contributed in part towards higher damage levels in the catch of the trawl.

The results of our study showed that sorting had an effect on damage, although due to the limitations of our data set it was not possible to quantify the contribution that sorting makes to total discard damage. Results of the GLMs for *E.esculentus* indicate however that most damage to *E.esculentus* occurs as a result of towing. The relative effects of towing as compared to sorting differ across species and fishing practices, as a species which may be more resilient to injury in the towed gear may on the other hand be seriously affected by sorting times or the shaking action of the dredges' sorting trays (e.g. Oddsson et al. 1994). Few studies have focussed on the specific effects of on-deck sorting upon damage levels in non-target species. Pranovi et al. (2001) detected significant differences between pre and post sorting bycatch in the western Adriatic Sea queen scallop trawl fishery and noted that sorting – in that fishery consisting of manual handling and a certain degree of trampling – produced similar levels of injury to those of the gear itself. Importantly, such results indicate that assessing catch and bycatch damage as it is hauled on-deck (either in a research or fishing vessel) undoubtedly leads to an underestimation of damage levels in discards (Pranovi et al. 2001) and thus of any subsequent mortality estimates.

4.3 Bycatch Mortality

The key implication of fishing induced damage to benthic organisms is the increased mortality of damaged organisms, either directly upon contact with the gear, or through increased predation (Jenkins et al. 2001). In the long term, high discard mortality rates can have significant knock-on effects on ecosystems through the differential mortality of different species and consequent modification of food webs (Stobutzki et al. 2001; Gaspar et al. 2001). If discarded, the fate of non-target animals can vary considerably, as they are exposed to a range of stresses. A proportion of discards are consumed by sea birds and this has been estimated as high as 70% for a North Sea Nephrops trawl fishery (Evans et al. 1994); otherwise, they sink through the water column becoming available to predation by pelagic organisms. The remainder which sink to the sea bed become available to benthic predators and scavengers (Ramsay et al. 2000). If damaged, their vulnerability to predation and disease is likely to increase, and an evaluation of delayed mortality in relation to sub-lethal injury has been proposed as necessary to accurately evaluate the relative fragility of non-target species upon discard (Pranovi et al. 2001; Ramsay et al. 2000).

We had therefore hypothesised that long-term (10 day) discards mortality would increase with damage level. This held true for *E.esculentus*; however, for *A.rubens* mortality of damaged (DL2 and DL3) individuals was significantly higher than that of undamaged (DL1) individuals, but the very high predicted mortality of damaged organisms (ranging between 82% and 100%) did not differ across different levels of damage. This suggests that increased mortality of damaged *A.rubens* in relation to undamaged individuals appears to occur independently of the level of damage induced, i.e. even minimal damage leads to increased mortality. In addition, mortality of undamaged *A.rubens* was also high (52-80%). Previous mortality studies, undertaken in controlled conditions (i.e. sea water holding tanks) have estimated high survivability: Kaiser & Spencer (1995) observed 99% survivability of *A.rubens* at two days following trawling, although the short time scale may underestimate long-term mortality; Ramsay et al. (2000) observed otter trawl mortalities of 0% at 8 days, and dredge mortalities of 7% at 14 days). However, the controlled (holding tanks) setting of these experiments is unlikely to reflect the multitude of challenges faced by animals upon discard, such as the increased risk of disease and predation (see e.g. Jenkins et al. 2001). Predicted estimates of mortality from damage score assessments have also yielded lower mortality estimates than those observed in our study. Bergman et al. (1990) estimated almost 100%

survivability of *A.rubens* on discard following beam trawling, and Veale et al. (2001) estimated 10% mortality of *A.rubens* and 31% mortality of *E.esculentus* following queen scallop dredging. The results of our study, however, clearly indicate previous studies are likely to have underestimated long-term mortality for these species. Upon exposure to the marine environment, mortality was high for both undamaged and damaged animals, and would likely have been even higher had the animals also been exposed to predation by larger members of the benthos.

It should also be noted that the effect of sampling and deployment of samples at sea may have led to increased mortality; however this remains unclear due mostly to the difficulty in obtaining control animals. Sampling of discards onboard fishing vessels and their storage and transport to the creel deployment site may have contributed to mortality through increased stress and air exposure. Due to logistical limitations during our sampling it was not possible to establish controls for this portion of the experiment. In addition, maintaining individuals in the creels, in closer proximity than that experienced naturally at sea (Murray, *pers.comm.*[5]) may contribute to increased stress and/or disease. Although control animals exhibited high survivability (0% mortality in *A.rubens* controls and 4% mortality in *E.esculentus* controls), the density of individuals in each sample (creel) was much lower and thus any consequent effects of the high-density conditions experienced by discard samples not accounted for.

There were no significant differences in mortality of *A.rubens* across vessels for the same damage level, which suggests that the likelihood of an additional element of internal damage which differs across vessels is low, and that mortality can be inferred directly from the level of damage incurred to an individual. Mortality of *E.esculentus* was higher for the trawl as compared to the skid dredge, although it is not possible to conclude as to whether this is solely an effect of the gear, or also the result of the longer tows as well as longer sorting time and consequent air exposure experienced in the otter trawl as compared to the dredge (e.g. van Beek et al. 1990; Davis and Ryer 2003). The results of the mortality study thus suggest that as mortality increased with damage level, lower damage levels in the discards are likely to result in reduced mortality. It follows that given the lower damage levels in the modified dredge, the latter is likely to incur the least discard mortality.

[5] Murray, L (2011) Discussion on the density of *Asterias rubens* and *Echinus esculentus* observed during photographic seabed surveys of the Isle of Man queen scallop fishing grounds undertaken by Bangor University in 2010/2011. June 2011.

5. Conclusions and recommendations for further study

The analysis of bycatch rates and composition and physical damage sustained by discards in the Isle of Man queen scallop fishery provided information on the impact of different phases of the fishing process. Catch composition differed across the gears, which reflects potential different selective pressures on non-target species. In addition, opportunistic sampling on-board commercial vessels aimed to assess the impacts of sorting on damage levels incurred to bycatch species, often disregarded in previous studies.

The results of our experiment indicated the modified dredge to be the most efficient at catching queenies per unit of area towed, with minimum bycatch. In addition, it led to lower bycatch damage which is likely to translate into lowest mortality upon discard. This suggests that the use of the modified dredge could be the most environmentally friendly option in the context of a gear-based solution aimed at restricting effort to achieve a given landings target, whilst minimising both bycatch and the area of sea bed towed. However, in the absence of a cap on landings dredging effort should be carefully restricted. The present study did not assess the damage and mortality incurred to uncaught non-target organisms which were not hauled on deck but which came into contact with the gear, i.e. by direct physical impact of the gear, or by entering and subsequently escaping the dredge or net. Previous studies (see Jenkins et al. 2001) have indicated that damage to non-captured organisms is a significant contributor to total damage level caused by the fishing process and in some species may be the main source of impact. In this regard Hinz et al. (in revision) found the modified dredge to be significantly more damaging than the otter trawl, although a comparison with the skid dredge was not undertaken. If management decisions are to be made with regard to the total mortality of non-target species incurred as a result of fishing, a quantitative estimate of total mortality (i.e. of both caught and uncaught organisms) should be obtained for each gear.

A practicable strategy for a multi-gear fishery such as the Isle of Man queen scallop fishery is however unlikely to involve the adoption of a single preferred gear for the whole fishery. The solution is more likely to involve the temporal and spatial management of fishing effort for each different vessel type. In this context it is important to note that the effects of a certain gear on benthic organisms are strongly habitat specific (for a review see Kaiser et al. 2006) as well as affected by seasonality (Hinz et al. in revision; Jenkins et al. 2003). However, we are yet to establish the total impact to individual bycatch organisms and wider ecosystems incurred by each gear in different habitats. Future surveys should be designed to address habitat differences and seasonality when attempting to characterise bycatch composition, abundance and damage.

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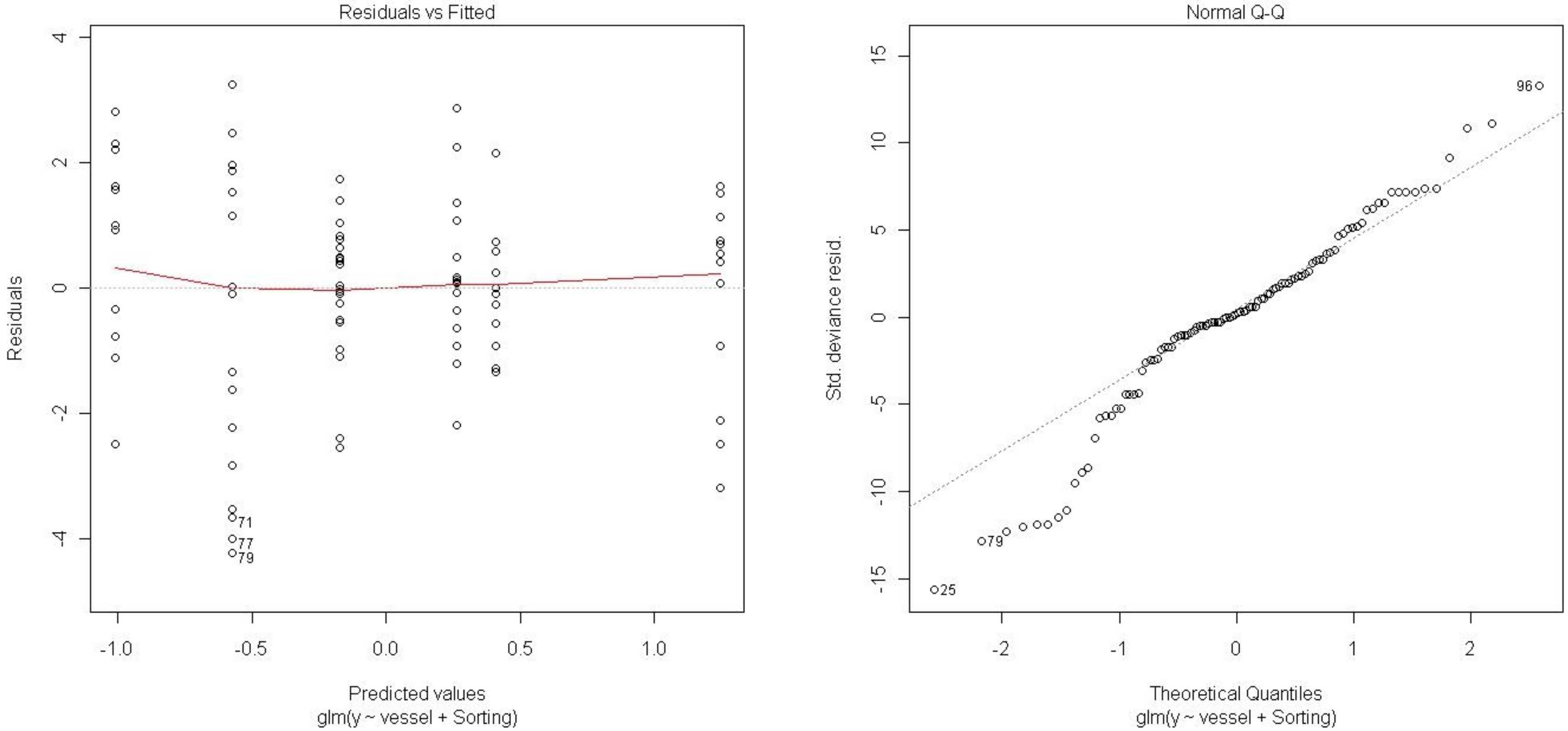
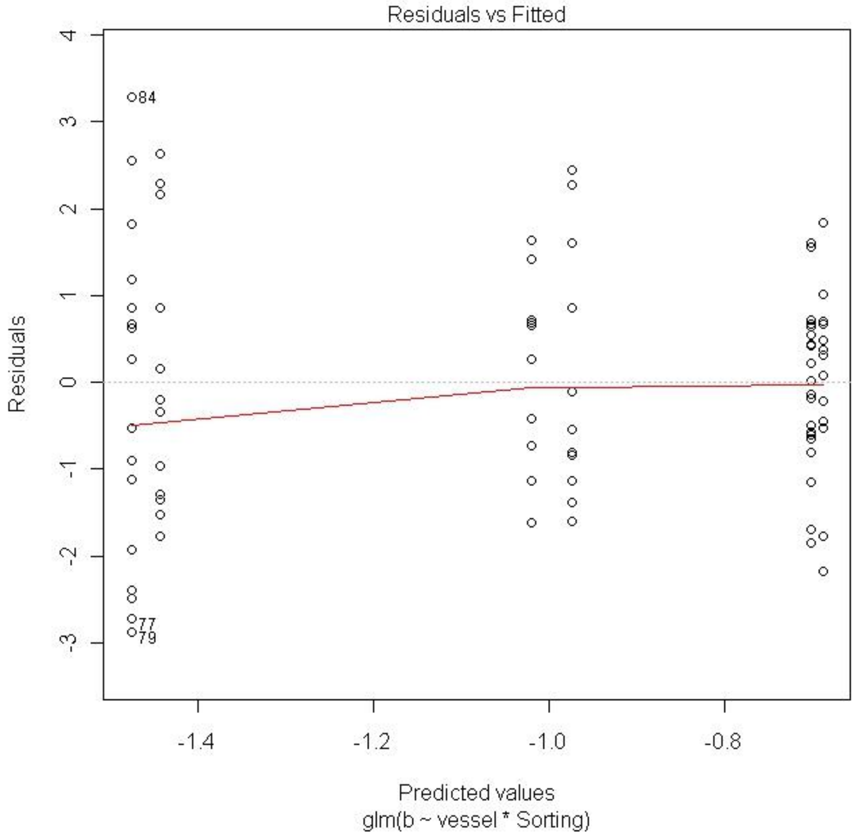
Appendices

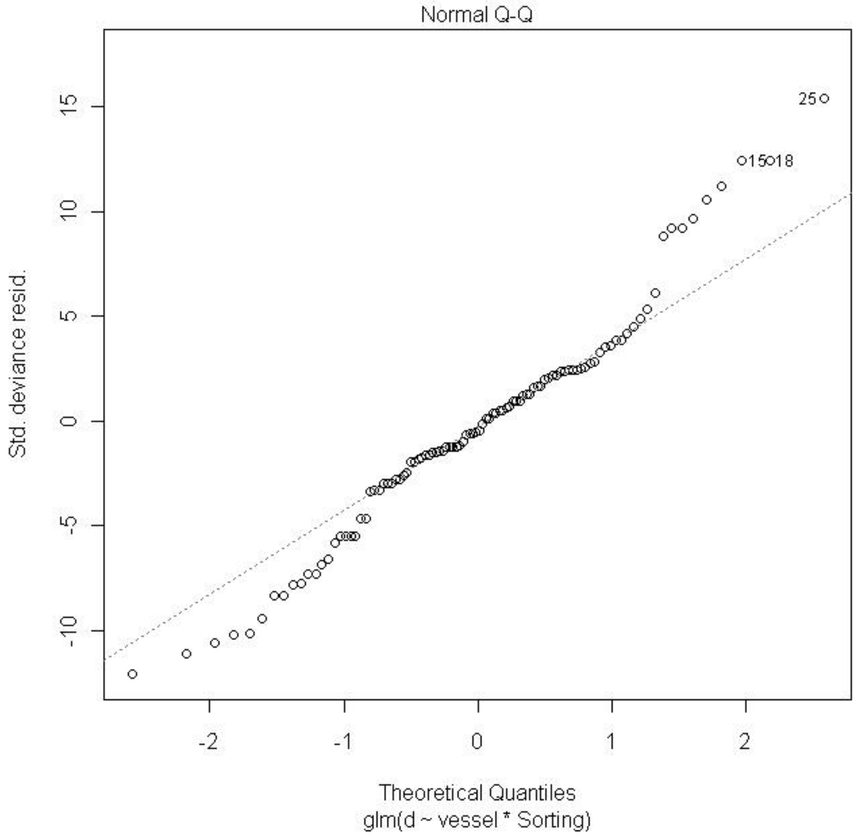
Appendix 1: Model checking plots used to assess the Generalised Linear Models fitted to the Damage and Mortality data sets in order to predict damage and mortality levels. Subtitles indicate species and response variable in each model.

Asterias rubens, proportion of Damage Level 1 individuals:

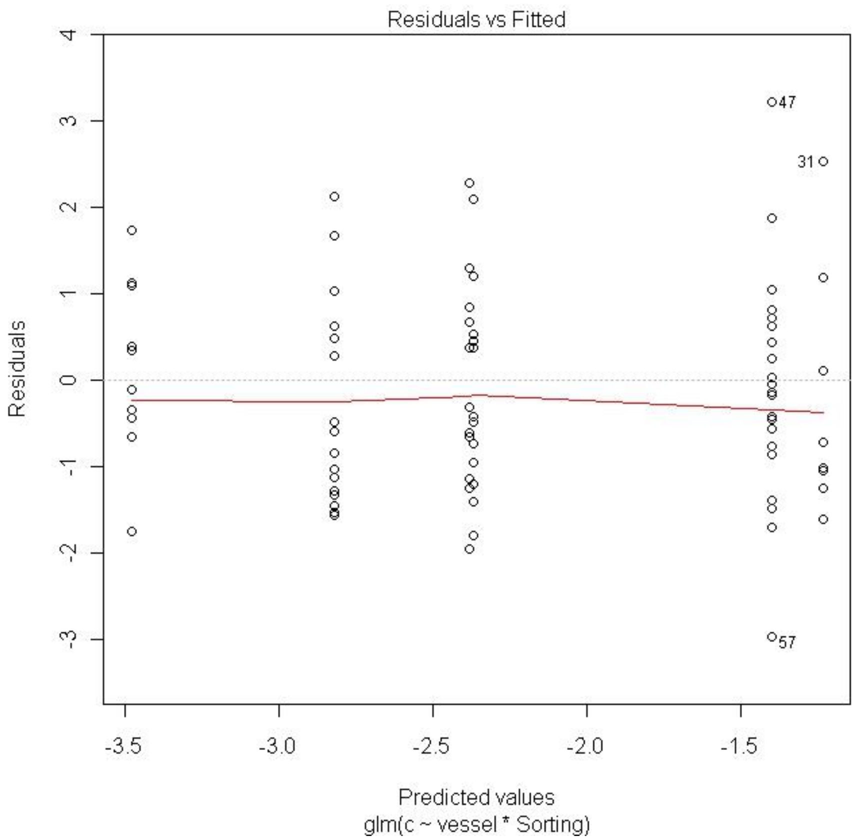


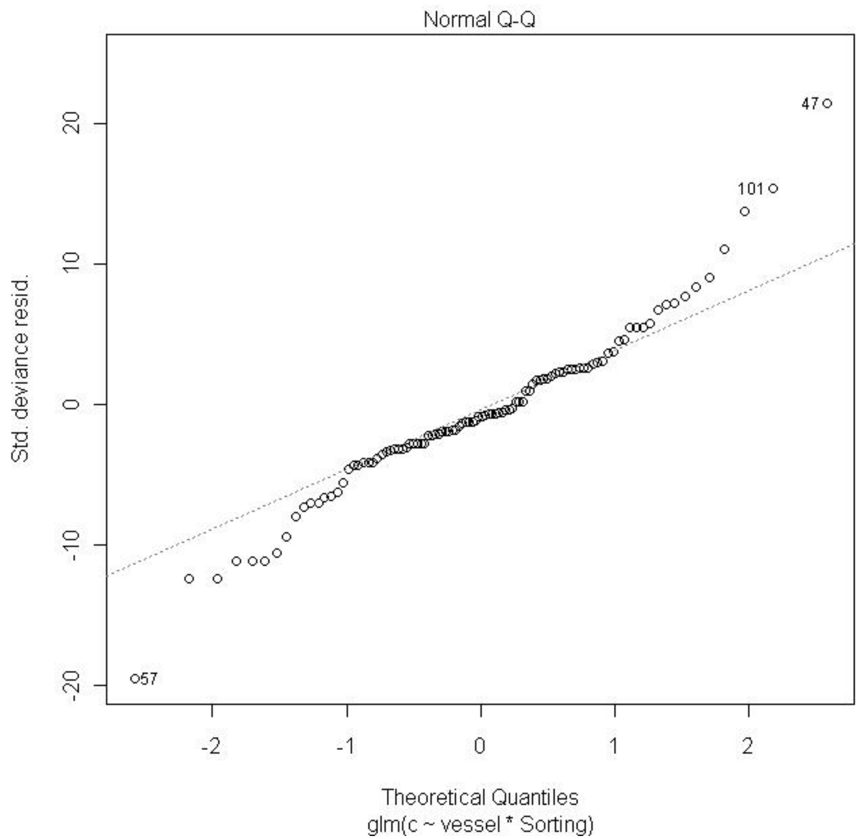
Asterias rubens, proportion of Damage Level 2 individuals:



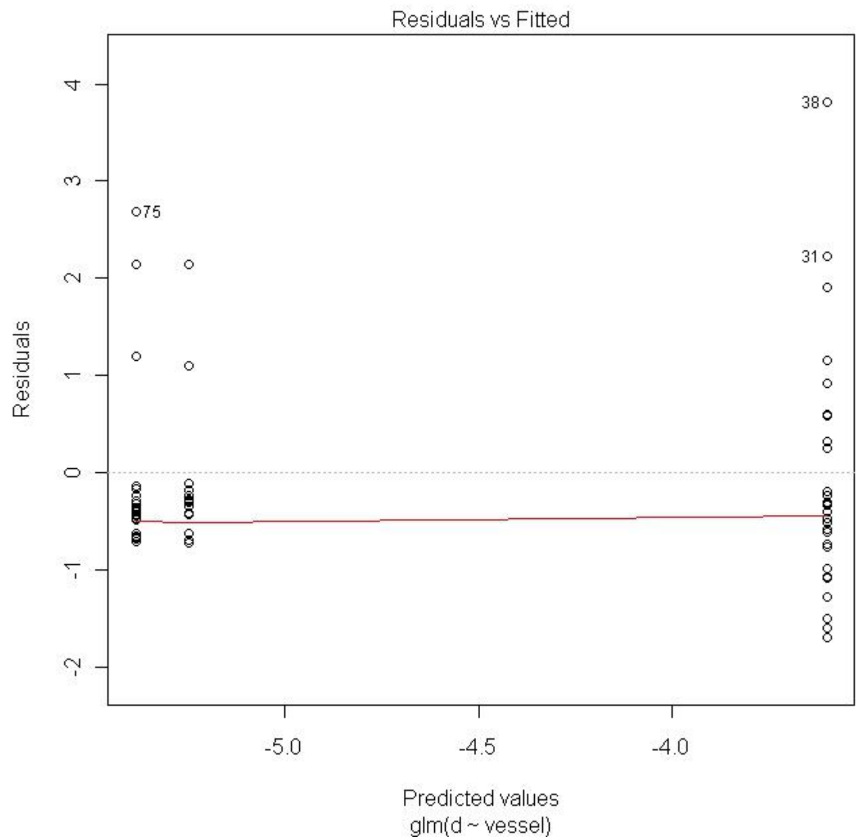


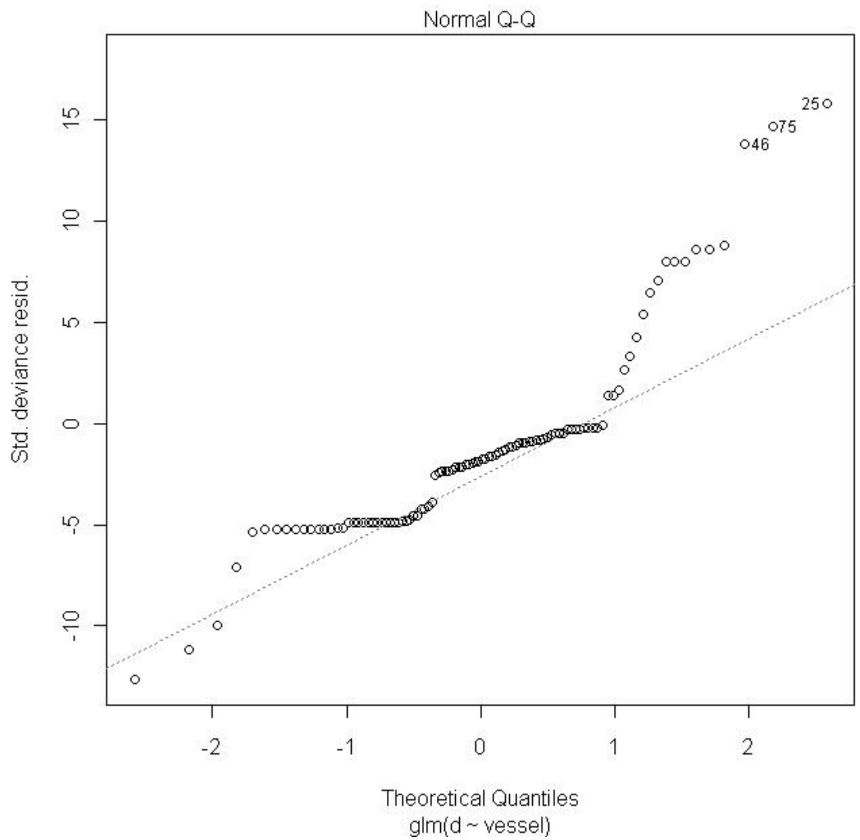
Asterias rubens, proportion of Damage Level 3 individuals:



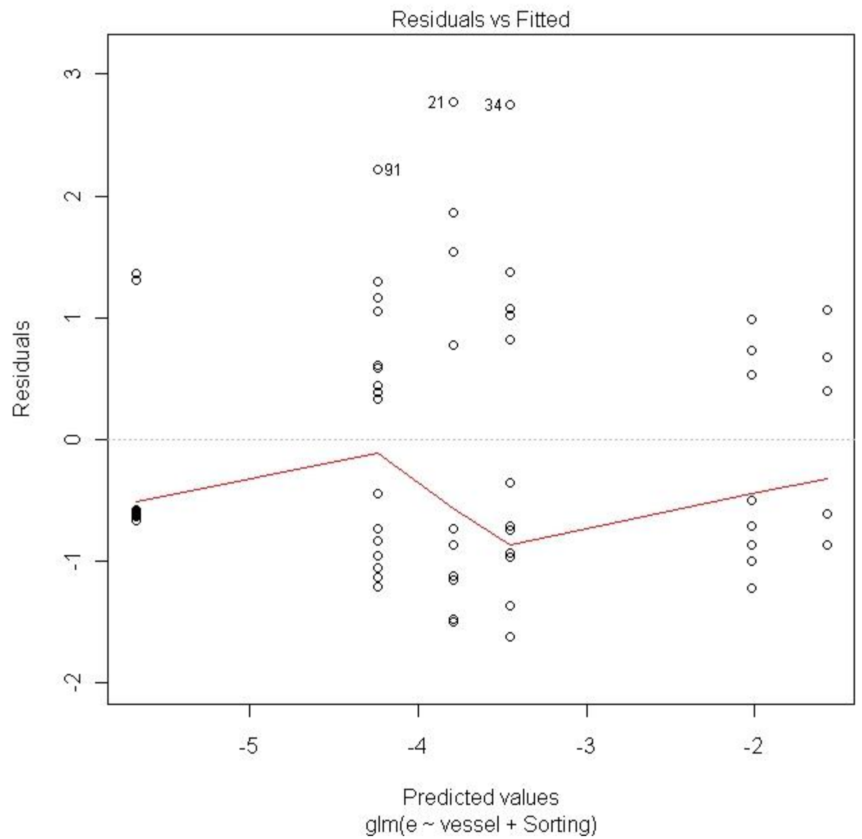


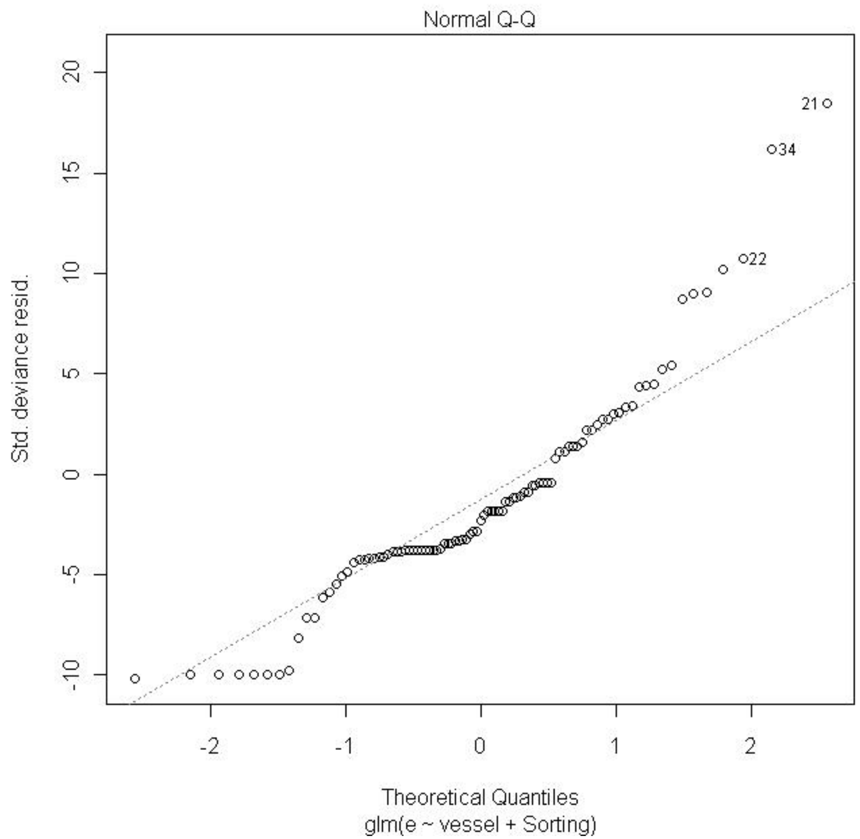
Asterias rubens, proportion of Damage Level 4 individuals:



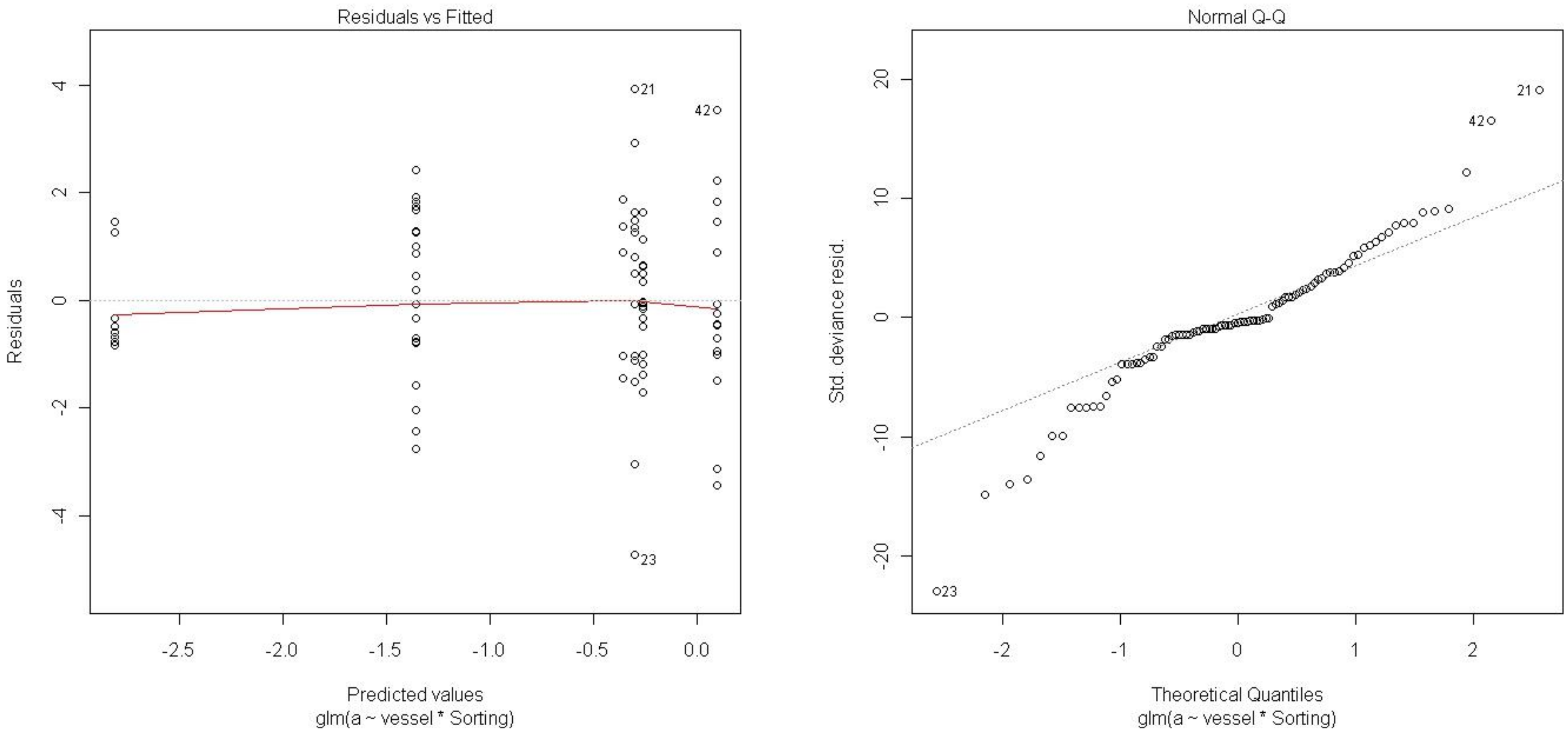


Echinus esculentus, proportion of Damage Level 1 individuals:

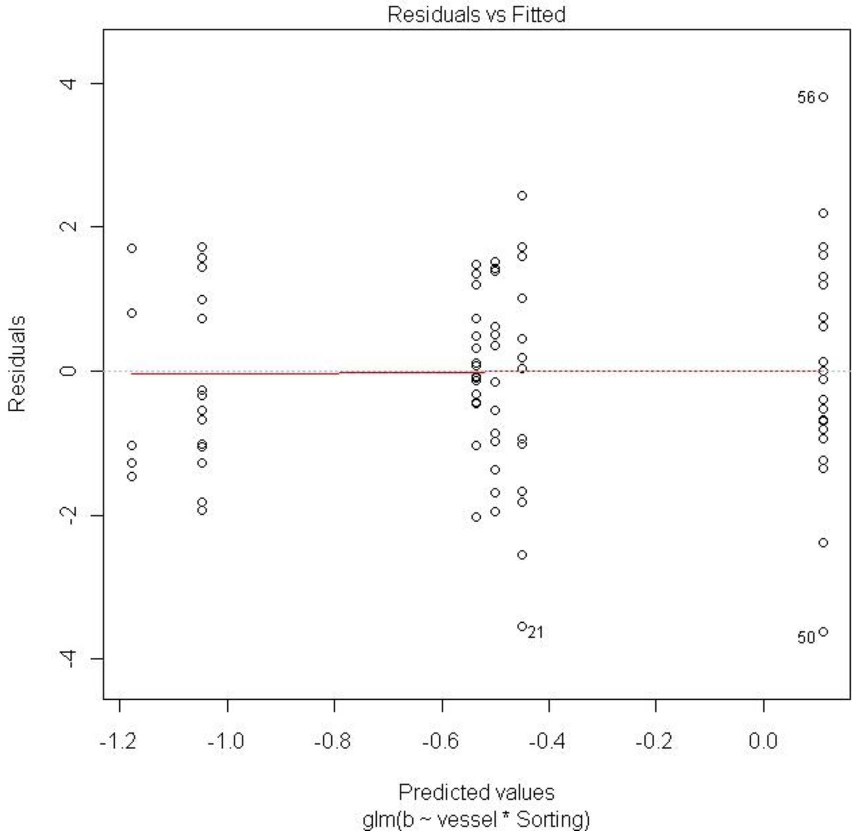


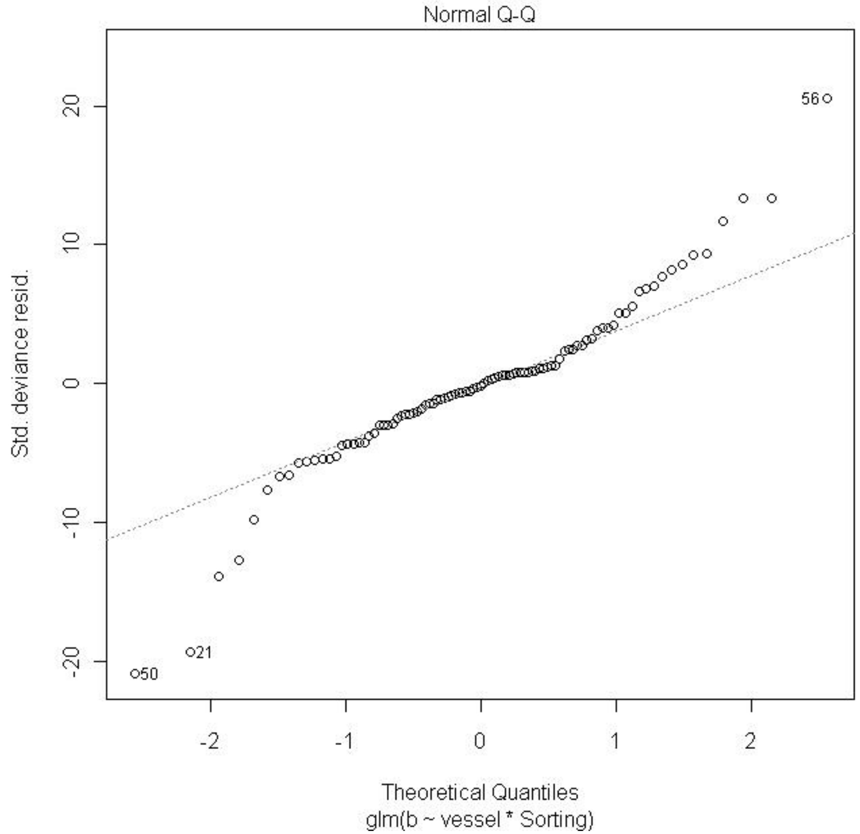


Echinus esculentus, proportion of Damage Level 2 individuals:

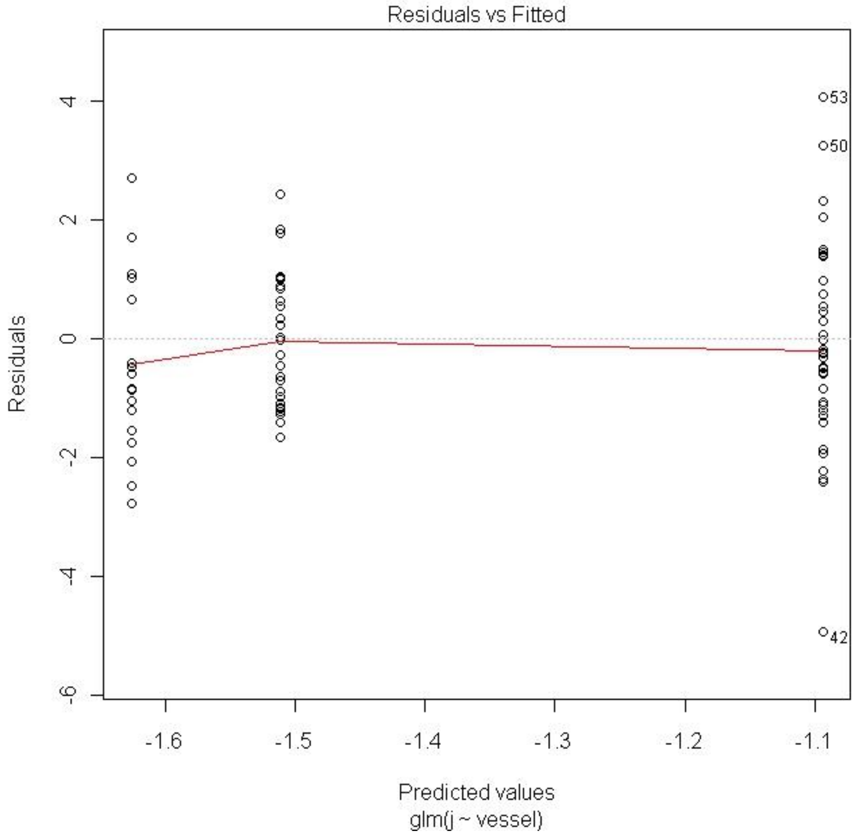


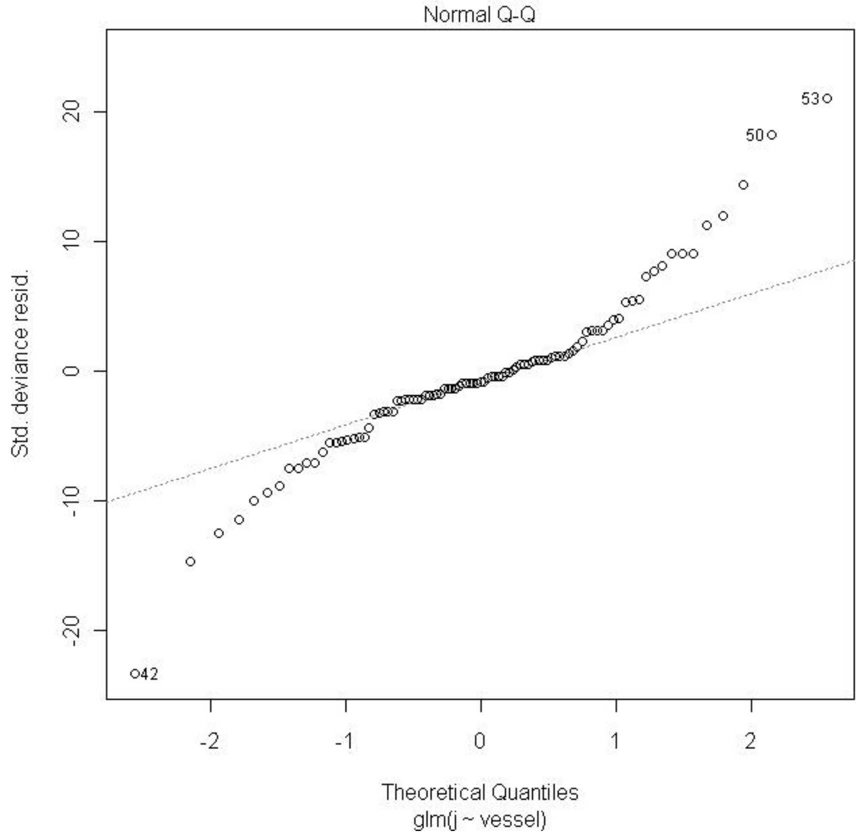
Echinus esculentus, proportion of Damage Level 3 individuals:



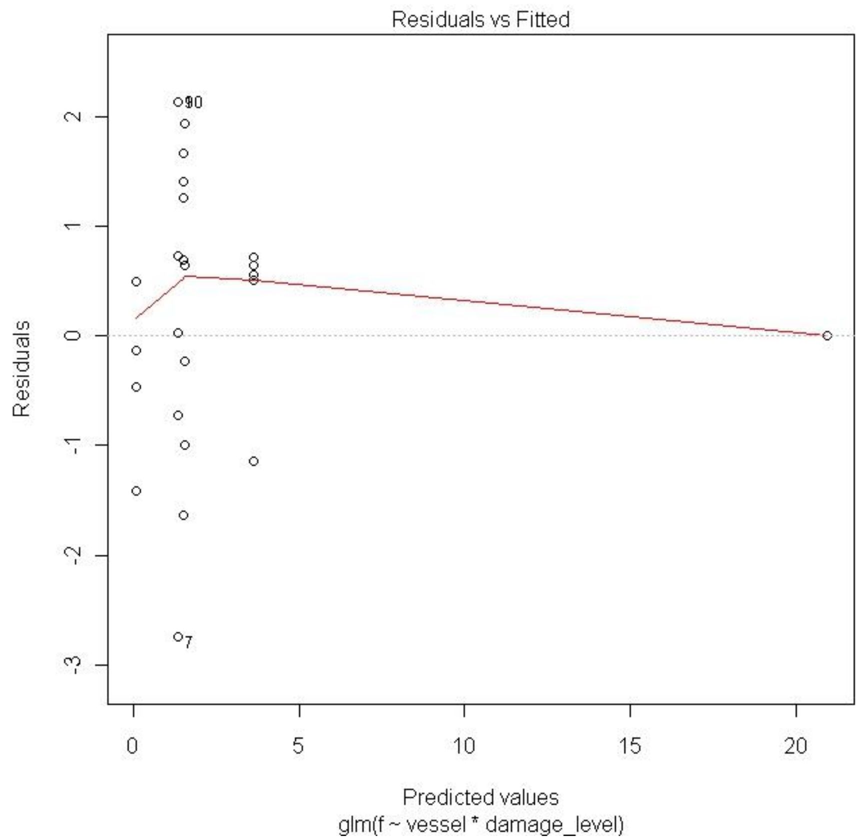


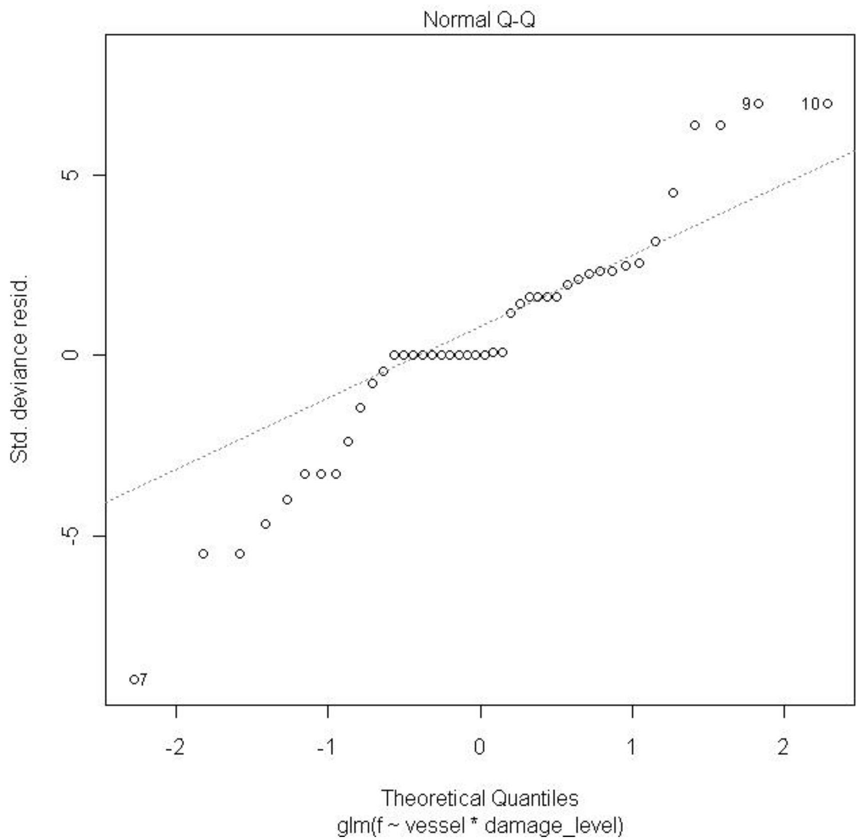
Echinus esculentus, proportion of Damage Level 4 individuals:



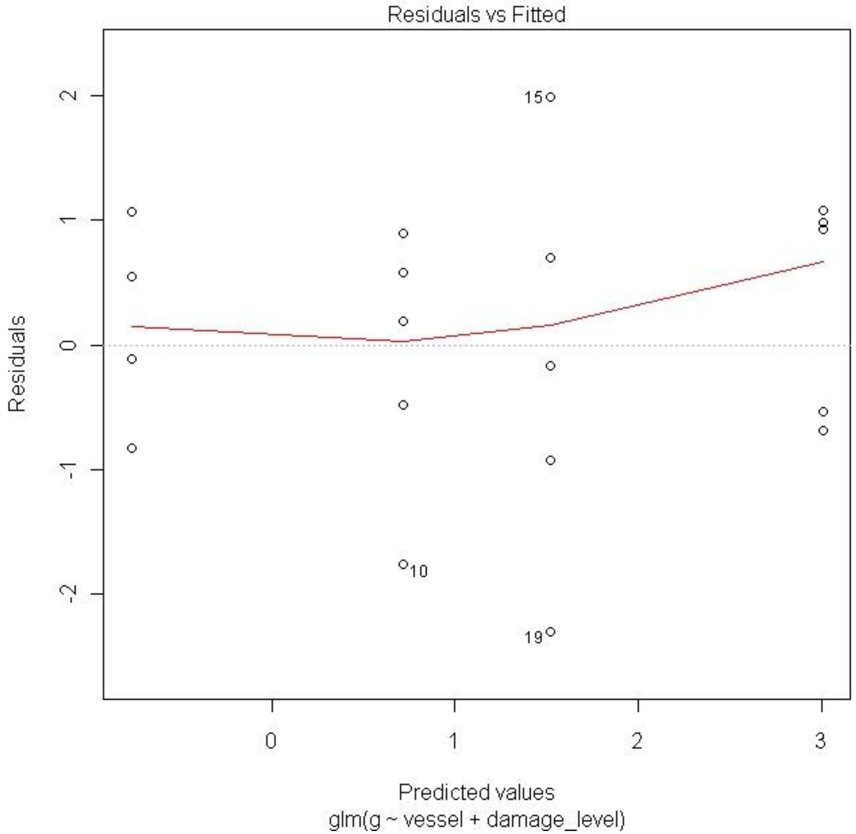


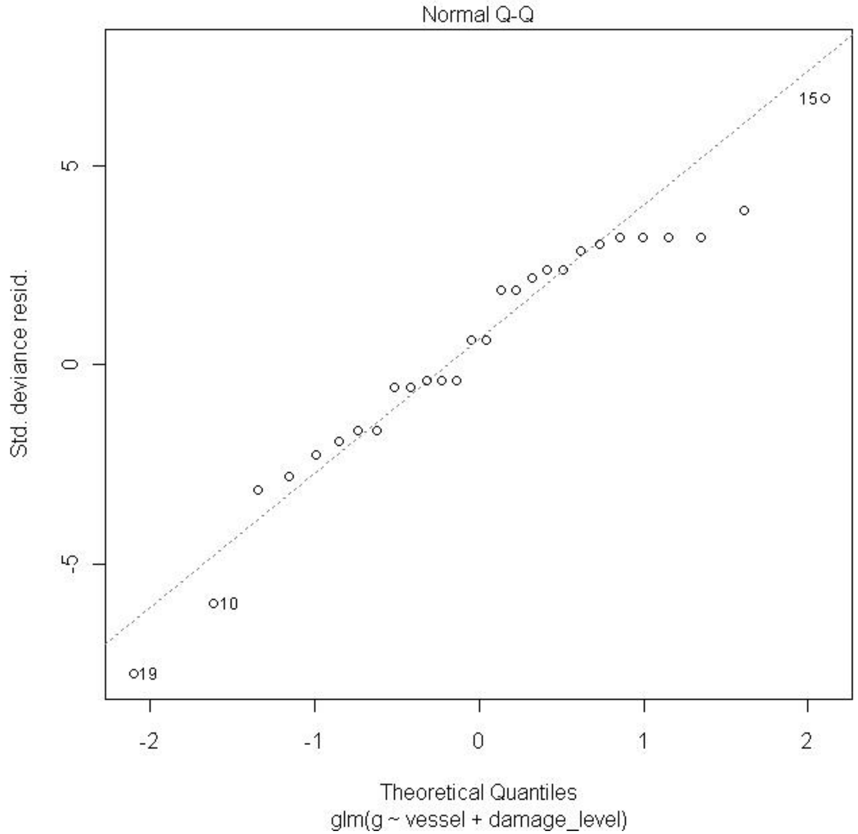
Asterias rubens, proportion of dead individuals:





Echinus esculentus, proportion of dead individuals:





Appendix 2: Results of F-tests and Chi-Square tests performed during the Generalized Linear Model fitting process. D.F.: Degrees of freedom; Res.DF: Residual degrees of freedom.

***Asterias rubens* – F-tests on models fitted to damage data**

Damage Level 1	DF	Deviance	Res. DF	Res. Dev.	F	Pr(>F)
Vessel	2	209.6	99	311.2	45.7	9.8e-15
Sorting	1	55.6	98	255.7	24.2	3.5e-06
Vessel:Sorting	2	9.1	96	246.5	2.0	0.137
Damage Level 2	DF	Deviance	Res. DF	Res. Dev.	F	Pr(>F)
Vessel	2	44.8	99	190.0	13.9	5.0e-06
Sorting	1	7.9	98	182.1	4.9	0.029
Vessel:Sorting	2	16.6	96	165.5	5.1	0.008
Damage Level 3	DF	Deviance	Res. DF	Res. Dev.	F	Pr(>F)
Vessel	2	141.2	99	142.3	54.4	<2.0e-16
Sorting	1	0.2	98	142.1	0.1	0.714
Vessel:Sorting	2	9.2	96	132.9	3.5	0.033
Damage Level 4	DF	Deviance	Res. DF	Res. Dev.		Pr(>Chi)
Vessel	2	23.3	99	83.3		8.8e-06
Sorting	1	3.8	98	79.5		0.052
Vessel:Sorting	2	3.9	96	75.7		0.145

***Echinus esculentus* – F-tests on models fitted to damage data**

Damage Level 1	DF	Deviance	Res. DF	Res. Dev.	F	Pr(>F)
Vessel	2	12.2	92	135.1	5.1	0.008
Sorting	1	33.5	91	101.5	27.9	8.7e-07
Vessel:Sorting	2	5.9	89	95.7	2.4	0.098
Damage Level 2	DF	Deviance	Res. DF	Res. Dev.	F	Pr(>F)
Vessel	2	72.6	92	354.2	15.9	1.3e-06
Sorting	1	40.6	91	313.6	17.7	6.0e-05
Vessel:Sorting	2	96.8	89	216.8	21.1	3.0e-08
Damage Level 3	DF	Deviance	Res. DF	Res. Dev.	F	Pr(>F)
Vessel	2	29.7	92	238.0	8.17	0.001
Sorting	1	53.3	91	184.6	29.4	5.0e-07
Vessel:Sorting	2	12.1	89	172.5	3.3	0.041
Damage Level 4	DF	Deviance	Res. DF	Res. Dev.	F	Pr(>F)
Vessel	2	25.8	92	196.4	6.5	0.002
Sorting	1	6.2	91	190.2	3.1	0.080
Vessel:Sorting	-2	-4.2	91	190.2	1.1	0.341

***Asterias rubens* – F-tests on models fitted to mortality data**

Mortality	DF	Deviance	Res. DF	Res. Dev.	F	Pr(>F)
Vessel	1	1.38	42	122.3	1.27	0.266
Damage Level	2	66.70	40	55.6	30.83	1.1e-08
Vessel:Damage Level	2	8.27	38	47.3	3.82	0.031

***Echinus esculentus* – F-tests on models fitted to mortality data**

Mortality	DF	Deviance	Res. DF	Res. Dev.	P(>$\|Chi\|$)
Vessel	1	19.80	26	82.32	8.6e-06
Damage Level	1	58.20	25	24.12	2.4e-14
Vessel:Damage Level	1	1.2	24	22.92	0.273